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ENVIRONMENTAL SCIENCES DIVISION

FIRST ANNUAL REPORT ON THE BIOLOGICAL MONITORING AND ABATEMENT PROGRAM AT OAK RIDGE NATIONAL LABORATORY

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ABBREVIATIONS

ACD	ORNL Analytical Chemistry Division
ANOVA	analysis of variance
ATDL	Atmospheric Turbulence and Diffusion Laboratory
ATP	adenosine triphosphate
BaP	benzo[a]pyrene
BFK	Brushy Fork kilometer
BMAP	Biological Monitoring and Abatement Program
CRK	Clinch River kilometer
C.V.	coefficient of variation
DEM	ORNL Department of Environmental Management (*now* the Environmental Management and Compliance Section)
D.F.	degrees of freedom
DMSO	dimethyl sulfoxide
DNA	deoxyribonucleic acid
DOC	dissolved organic carbon
DOE	U.S. Department of Energy
DTT	dithiothreitol
EDTA	ethylenediaminetetraacetic acid
EFPC	East Fork Poplar Creek
EPA	U.S. Environmental Protection Agency
EROD	7-ethoxyresorufin O-deethylase
ESD	ORNL Environmental Sciences Division
FCK	First Creek kilometer
FDA	U.S. Department of Agriculture Food and Drug Administration
FFK	Fifth Creek kilometer
GLM	general linear model
HFIR	High Flux Isotope Reactor
ICRP	International Commission on Radiological Protection
LSI	liver-somatic index
MEK	Melton Branch kilometer
MFO	mixed function oxydase
MSL	mean sea level
NADH	nicotinamide-adenine dinucleotide (reduced form)
NADPH	nicotinamide-adenine dinucleotide phosphate (reduced form)
NPDES	National Pollutant Discharge Elimination System
NRWTF	Nonradiological Wastewater Treatment Facility
NTK	Northwest Tributary kilometer
ORGDP	Oak Ridge Gaseous Diffusion Plant (*now* the Oak Ridge K-25 Site)
ORNL	Oak Ridge National Laboratory
PAHs	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls
ppm	parts per million
RAP	Remedial Action Program
RCRA	Resource Conservation and Recovery Act

ABBREVIATIONS (continued)

RNA	ribonucleic acid
s.d.	standard deviation
s.e.	standard error
SGOT	serum glutamic oxaloacetic transaminase
SREL	Savannah River Ecology Laboratory
SRP	soluble reactive phosphorus
SWSA	solid radioactive waste disposal/storage area
TCMP	Toxicity Control and Monitoring Program
TDS	total dissolved solids
TRK	Tennessee River kilometer
TRU	Transuranium Processing Facility
TVA	Tennessee Valley Authority
USGS	U.S. Geological Survey
VSI	visceral-somatic index
WCK	White Oak Creek kilometer
WOC	White Oak Creek
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 (currently the Tennessee Department of Environment and Conservation). The plan, which is referred to as the ORNL Biological Monitoring and Abatement Program (BMAP), describes characterization and monitoring studies to be conducted for the 5-year duration of the permit. This report (Sects. 3–9) is organized according to the seven major tasks that make up BMAP.

This document, the first of a series of annual reports presenting the results of BMAP for White Oak Creek watershed and the Clinch River, describes studies that were conducted from March through December 1986. However, the actual period covered by each of the seven BMAP tasks varied, depending upon when the task was initiated and the turnaround time required for analysis of samples and review of data. In addition to presenting results of the various investigations, these annual reports will address any significant modifications in the scope of work from that presented in BMAP, as described 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* (ORNL/TM-10370).

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EXECUTIVE SUMMARY

As 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; and the Clinch River. BMAP consists of seven 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; (6) radioecology of WOC and White Oak Lake (WOL); and (7) contaminant transport, distribution, and fate in the WOC embayment-Clinch River-Watts Bar Reservoir system. This document, the first of a series of annual reports presenting the results of BMAP, describes studies that were conducted from March through December 1986.

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

Ambient (instream) toxicity was evaluated at 15 sites on 5 streams in WOC with 7-d static-renewal toxicity tests based on the survival and growth of fathead minnow (*Pimephales promelas*) larvae and the survival and reproduction of a small crustacean (*Ceriodaphnia dubia/affinis*). Based on the variation in survival in ten ambient tests and the relatively low toxicity of effluents evaluated in the Toxicity Control and Monitoring Program (four tests each on the effluent from the Process Waste Treatment Plant, the Sewage Treatment Plant, and the Coal Yard Runoff Treatment Facility), it is hypothesized that ambient toxicity patterns in WOC watershed may be determined more by episodic or periodic events, such as intermittent discharges or inadvertent spills, than by wastewater treatment facilities that discharge effluents continuously. Information to date suggests that episodic discharges of chlorine dominate the ambient toxicity patterns observed at sites on Fifth Creek and the middle reaches of WOC adjacent to the main ORNL complex. To identify toxic sources, continuous instream monitoring of chlorine levels by automated instrumentation will be initiated at these and other sites in 1987.

The greatest impact of plant operations on the benthic invertebrate and fish communities was also observed in Fifth Creek and middle WOC as well as in Melton Branch below the confluence of a small tributary that receives discharges from the High Flux Isotope Reactor area. At these sites, diversity and mean density of benthic species were substantially reduced compared with those found in upstream reference sites. No fish were collected at the Fifth Creek and Melton Branch sites, and densities in a 160-m reach of WOC adjacent to the plant complex were more than an order of magnitude lower than the densities at any other site on WOC. Possible toxic effects on periphyton biomass (i.e., chlorophyll *a* per unit area) and production were also observed at this site on WOC and on lower Melton Branch. Concentrations of chlorophyll *a* in WOC below ORNL were well above the concentrations found in upstream reference sites and were similar to those observed in other area streams with elevated nutrient levels. Moderate impacts on the benthic communities were also observed on the lower reaches of First Creek, Northwest Tributary, and WOC. Potential causes of these adverse effects include process water discharges, elevated temperatures, nutrient enrichment, modified flow regimes, and siltation from recent construction activities.

Although community studies (periphyton, benthic invertebrates, and fish) and ambient toxicity monitoring demonstrated substantial adverse ecological impacts of current plant operations on most of Fifth Creek and the middle reaches of Melton Branch and WOC, these same studies also provided evidence of downstream recovery in WOC and, to some extent, in Melton Branch. Improvement was noted in all communities, especially the periphyton, at the downstream site on WOC that was located ~300 m below the confluence with Melton Branch. Also, the magnitude of the ecological impacts decreased with increasing distance downstream of the main plant area. The only exception to this general trend occurred at a site on WOC below a small tributary that drains solid radioactive waste disposal/storage area (SWSA) 4, where diversity and richness of benthos and fish species were lower than in sites ~0.5 km upstream and downstream.

Even at the downstream site on WOC, however, some impacts of plant operations were evident. For example, many taxa that are sensitive to water quality degradation and habitat alteration and that inhabit upstream reference areas were absent from the lower reaches of WOC. A major factor affecting the downstream recovery of the fish communities in both WOC and Melton Branch is the isolation of these areas from the Clinch River system by several weirs and by White Oak Dam. Such structures may effectively block recolonization from downstream areas, and poor water quality and elevated temperatures in the middle reaches of these streams prevent recolonization from unimpacted upstream areas. Without access, more-sensitive, pollution-intolerant species can never recolonize the WOC area regardless of improvements in water quality. Because the adult stage of most stream insects is terrestrial, the benthic community is likely to show a more rapid response to improving conditions than the fish community.

Biological indicator studies provided additional evidence that downstream fish populations were impacted by upstream perturbations. Two measures of nutritional status, serum triglyceride levels and the liver-somatic index (which also reflects short-term metabolic energy demands), were significantly lower in redbreast sunfish from each of two sites on lower WOC than in those found in the reference site on Brushy Fork. In addition, levels of three liver detoxification enzymes, which regulate the biotransformation or metabolism of organic compounds, were significantly higher in fish from WOC than in those from Brushy Fork.

Future studies will utilize water quality sampling and continuous monitoring of stream temperatures at selected locations, in addition to biological monitoring, to evaluate the importance of modified nutrient and thermal regimes as factors impacting the periphyton, benthic invertebrate, and fish communities in WOC and tributaries. Monitoring of these communities will continue in order to assess the ecological recovery of receiving streams as various pollution abatement projects, such as the Wastewater Piping Replacement Plan and the Cooling Tower Pollution Elimination Plan, are implemented.

CONTAMINANT TRANSPORT AND BIOACCUMULATION (TASKS 2 AND 5–7)

Levels of mercury in fish collected from WOL and Melton Branch were well below the U.S. Department of Agriculture Food and Drug Administration (FDA) action limit of 1 ppm and differed little from levels in fish from uncontaminated reference streams. Elevated levels observed in fish from Northwest Tributary and WOC above the lake were also well below the FDA limit but indicate the existence of an active source or sources of mercury within the watershed. Future studies will focus on identification of the contaminant source(s) and will be coordinated with the studies outlined in the NPDES Mercury Monitoring Plan.

Some channel catfish caught by fishermen in a 7-km reach of the Clinch River downstream from Melton Hill Dam are likely to contain polychlorinated biphenyls (PCBs) exceeding the FDA tolerance limit of 2 ppm. The proportion of fish containing such levels, however, is low. (Only 5%, or 2 of the 39 channel catfish collected from this reach of the Clinch River in 1986, exceeded the limit.) Although a significant portion of the PCB burden in catfish from WOC embayment and nearby locations in the Clinch River may result from recent or ongoing releases from sources within the embayment or upstream, only a small fraction of the total PCB content of fish from other sites in the Clinch River can be attributed to these sources. Monitoring of PCB contamination in channel catfish will be conducted annually, and the role of the food-chain pathway in determining PCB levels in Clinch River catfish will be evaluated. Results of the annual screening of metals and organic contaminants in fish collected in the winter 1986–1987 from seven sites in WOC watershed are incomplete and will be presented in the next annual BMAP report.

Monitoring of contaminants in the terrestrial environs at ORNL focused initially on radionuclides; mercury; and benzo(a)pyrene (BaP), a representative polycyclic aromatic hydrocarbon (PAH). A survey of tritium (^3H) in surface waters and shallow groundwaters showed that SWSA 4 and SWSA 5 are major contributors of ^3H to WOC watershed. Tritium concentrations in soil water and atmospheric moisture were greater in areas immediately south of SWSA 5 than in areas immediately south of SWSA 4, and a strong positive correlation was observed between the concentrations in air moisture and surface soils (0–10 cm). Strong seasonal changes in the ^3H concentration in soil water were also identified in the floodplain soils below SWSA 5. Patterns of ^3H in air moisture, surface waters, and pine tree cores indicate a major area of ^3H seepage from SWSA 5 near the middle tributary drainage. Future studies will quantify ^3H fluxes in this area.

Concentrations of ^{137}Cs and ^{60}Co in muscle tissue from groundhogs collected at the 3524 Equalization Basin at ORNL ranged from 3.7 to 4.2×10^3 Bq/kg dry wt and from 15.5 to 26.2 Bq/kg dry wt respectively. Because the groundhogs at this site probably represent the highest radiation exposure of feral mammals at ORNL due to the high radionuclide levels in the basin [5.6 TBq (150 Ci), of which 68% is ^{137}Cs], they are excellent candidates for evaluating the use of biochemical indicators of genotoxic exposure in future studies. Mercury concentrations in kidneys of small mammals collected near WOL were 4 to 40 times lower than concentrations found in the same species from the floodplain of East Fork Poplar Creek, but the WOL levels were generally higher than those of small mammals at the Bull Run Steam Plant. Levels of BaP metabolites in blood from 7 mammal species (total of 31 individuals) from 3 sites at ORNL were all below the detection limit of 10 pg/mg hemoglobin. Radionuclides that predominate in WOL sediment and/or water (^{137}Cs, ^{60}Co, and ^{90}Sr) were also found in turtles, although considerable differences in body burdens were noted among species. These initial studies indicated that PAHs in terrestrial species at ORNL are unlikely to exceed background levels, while radionuclides are elevated in several wildlife species. Radionuclide analyses of small mar mals will be used to identify sources/areas that contribute to the uptake of radionuclides by dt :r. Additional sampling will be conducted to evaluate mercury as a contaminant of terrestrial biota at WOL.

A screening analysis was conducted for radionuclides in WOL and environs to identify potential critical pathways for future human exposure and problem radionuclides. Three remedial action scenarios and four radionuclides (^{137}Cs, ^{60}Co, ^{3}H, and ^{90}Sr) were included in the screening. For most aquatic and terrestrial food-chain pathways, ^{137}Cs was the major dose contributor (57%), followed by ^{90}Sr (38%), ^{60}Co (3%), and ^{3}H (2%). The potential food-chain pathways producing the highest total estimated doses were beef and milk from cows grazing on the WOL bed and vegetables grown in the sediments of the lake bed. The ratio of the estimated dose for a given pathway to a 1.0-mSv/year (100-mrem/year) screening limit provides a method for estimating the importance of various food-chain pathways. When this method was used, the following pathways had a ratio equal to or greater than 1: vegetables, milk, beef, aquatic plants, waterfowl, and poultry. Individuals engaged in activities in which they would be exposed to radio ıctivity in the drained lake bed would receive the highest estimated doses of all exposure pathways; external exposure to humans from gamma radiation exceeded the food-chain pathway doses by about a factor of 2. Draining the lake that has free access to the area would eliminate the aquatic food-chain pathways but would result in an exposed lake bed with a potential for higher external radiation doses and would create a larger area for radionuclide accumulation via components of terrestrial food chains. Based on the initial screening analysis, additional information is needed on WOL sediments, aquatic macrophytes, small game animals, and waterfowl.

Studies were initiated in July 1986 to assess the extent and spatial distribution of off-site contamination of the aquatic environment and to evaluate the hydrodynamic, geochemical, and ecological factors that determine the transport, distribution, and ecological fate of contaminants in the Clinch River and Watts Bar Reservoir system downstream of WOL derived from U.S. Department of Energy facilities and operations. Based on 35 sediment cores collected from the Clinch River and the upper half of Watts Bar Reservoir, ^{137}Cs concentrations ranged from 200 to 2000 mBq/g, with the highest value (2640 mBq/g) occurring at a sediment depth of 40 to 44 cm in a core collected offshore from the City of Kingston, near the mouth of the Clinch River. Concentrations of ^{137}Cs were also measured in 85 surface-sediment grab samples and ranged from 4 to 395 mBq/g. Concentrations of ^{60}Co in reservoir surface sediments ranged from nondetectable (<2 mBq/g) to 70 mBq/g. Data from sediment cores and surface sediment samples will be used to provide an inventory of ^{137}Cs accumulation in the river-reservoir sediments and to develop a sediment-type distribution map of the river-reservoir system that identifies zones of ^{137}Cs accumulation. A working simulation model of the WOC embayment–Clinch River–Watts Bar Reservoir system was formulated to assist in identifying and quantifying the various factors that determine the transport, distribution, and fate of contaminants in this system. Following an initial run of the model in spring 1987, historical data sets will be prepared to calibrate it. The model will ultimately be used to predict effects of ORNL remedial actions on off-site contaminant transport and fate.

ABATEMENT PROGRAMS

Abatement efforts at ORNL are directed toward providing both short- and long-term management and technical solutions to water quality problems. As the biological monitoring and toxicity testing progressed through the first year, a number of abatement projects or programs were initiated to address problem areas within the storm sewer system that were identified in the early testing stages. Concomitantly, the water pollution control effort, a major task within the ORNL Environmental Restoration and Facilities Upgrade Program, has the objective of establishing a sound basis for proceeding with the significant long-term commitments that will be required to give ORNL the capability to achieve and maintain environmental compliance with state and federal water quality regulations. The majority of these projects and programs addressed the areas of Fifth Creek and WOC identified in this report as being of particular significance, because survival of aquatic test organisms at these locations was greatly reduced. Abatement projects include (1) the cooling tower pollution elimination plan, (2) the water supply system characterization plan, (3) continuous instream chlorine monitoring, (4) the wastewater piping replacement project, (5) the process waste system inflow/infiltration and upgrade process waste collection system, (6) the wastewater/storm drain isolation project, (7) the Clean Water Act compliance study, and (8) the PCB and mercury monitoring plans.

1. INTRODUCTION

As 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; and the Clinch River (Loar et al. 1991). BMAP consists of seven 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; (6) radioecology of WOC and White Oak Lake; and (7) contaminant transport, distribution, and fate in the WOC embayment–Clinch River–Watts Bar Reservoir system.

BMAP was developed to meet several objectives. First, studies (tasks) were designed to provide sufficient data to demonstrate that the effluent limits established for ORNL protect and maintain the classified uses of WOC, Melton Branch, Northwest Tributary, First Creek, and Fifth Creek. These streams have been classified for (1) growth and propagation of fish and aquatic life, (2) irrigation, and (3) livestock watering and wildlife (EPA 1986).

Second, BMAP will provide ecological characterizations of WOC and tributaries and of White Oak Lake (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 eventual development and assessment of remedial action alternatives as part of the Resource Conservation and Recovery Act (RCRA) planning process within the ORNL Remedial Action Program (RAP). It can also be used to develop various Remedial Investigation/Feasibility Study plans, as required.

Third, BMAP will document the effects on stream biota from implementation of RAP and the Water Pollution Control Program (WPCP) at ORNL. The major remedial action included in the latter program is completion of a new Nonradiological Wastewater Treatment Plant in 1989. The ecological characterization in Objective 2 will provide baseline data that can be used to document the ecological effects of WPCP and 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.

*Abbreviations used in the text of this report are defined in the list on pp. xxi and xxii.

2. DESCRIPTION OF WHITE OAK CREEK WATERSHED

WOC watershed is located near the southern boundary of the 150-km^2 U.S. Department of Energy (DOE) Oak Ridge Reservation (Fig. 2.1). With a drainage area of 16.9 km^2 at its mouth at Clinch River kilometer (CRK) 33.5,* WOC watershed is similar in size to Bear Creek watershed (20.1 km^2) near the Oak Ridge 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. 2.2). 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 Reservation (McMaster 1963, McMaster and Waller 1965).

Because of dam construction, three distinct environments can be identified within WOC watershed: (1) WOL, (2) WOC embayment below the lake, and (3) WOC and tributaries above the lake. WOL 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. 2.2). It is a shallow impoundment that extends ~0.7 km upstream from the dam and has a surface area of ~6 ha at a lake elevation of 227.1 m MSL.

The water level in WOC embayment is controlled by operation of the Melton Hill Dam at CRK 37.2 and Watts Bar Dam, which is located at Tennessee River kilometer (TRK) 852.6, ~61 km below the confluence of the Clinch and Tennessee rivers. When Watts Bar Reservoir is maintained at or near full pool (approximately April to October) and discharges occur at Melton Hill Dam, the subsequent rise in water level in the Clinch River creates an embayment extending from the mouth of the creek to White Oak Dam. Because of this regulated condition, WOC watershed is generally considered to be limited to the 15.5-km^2 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, 1981b), Boyle et al. (1982), Oakes et al. (1982), and Sherwood and Loar (1987) and in Sects. 4 and 7–9 of this report.

2.1 GEOHYDROLOGY

The headwaters of WOC originate on the southeast slope of Chestnut Ridge (Fig. 2.2). The belt of Knox Dolomite underlying the ridge is the principal water-bearing formation, and the springs that occur along the base of Chestnut Ridge and in its valleys are the chief source of the base flow of 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. 2.3), ~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

*CRK 0.0 is located at the confluence of the Clinch and Tennessee rivers.

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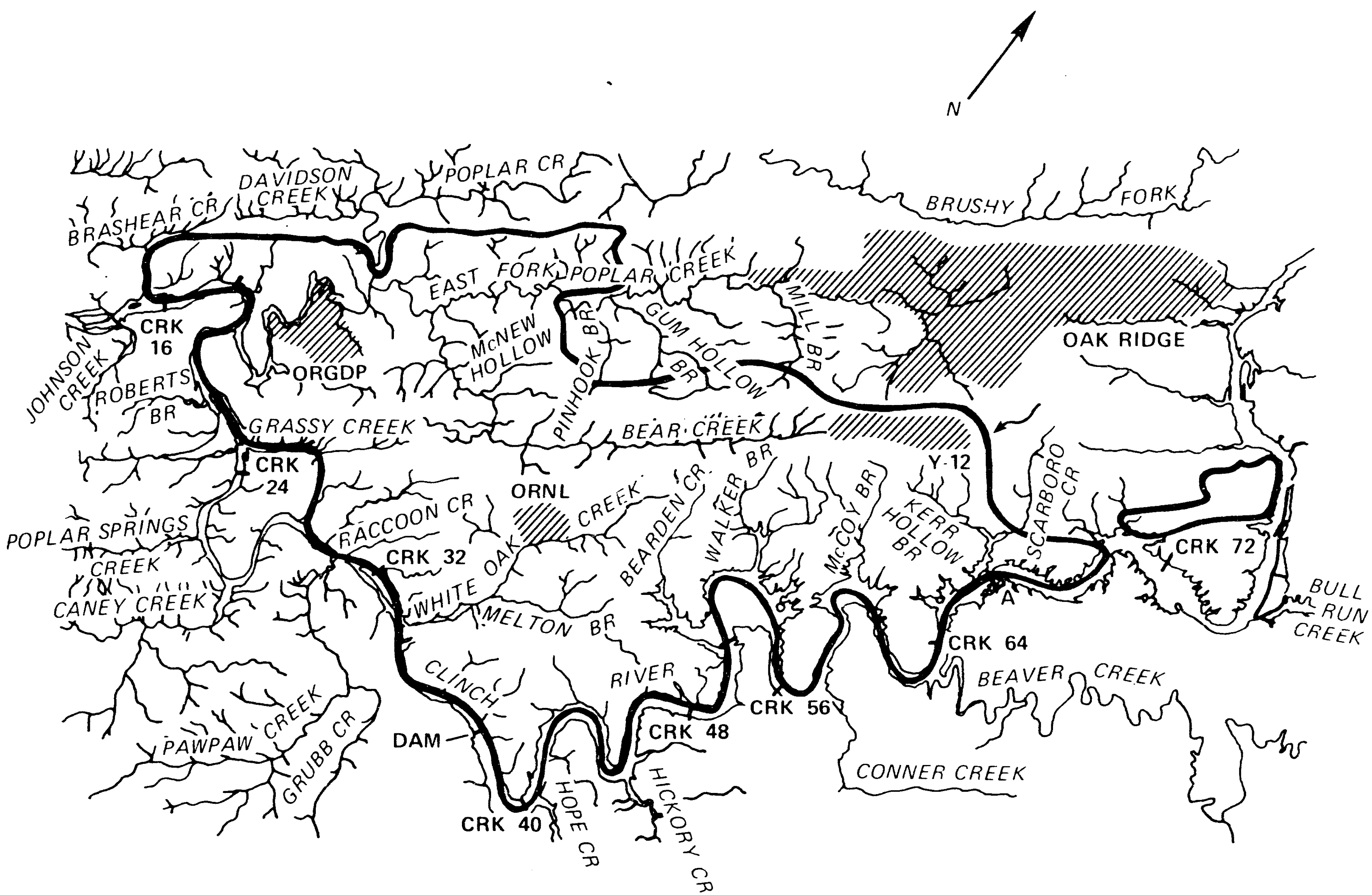

Fig. 2.1. Major streams on and contiguous with the U.S. Department of Energy Oak Ridge Reservation.

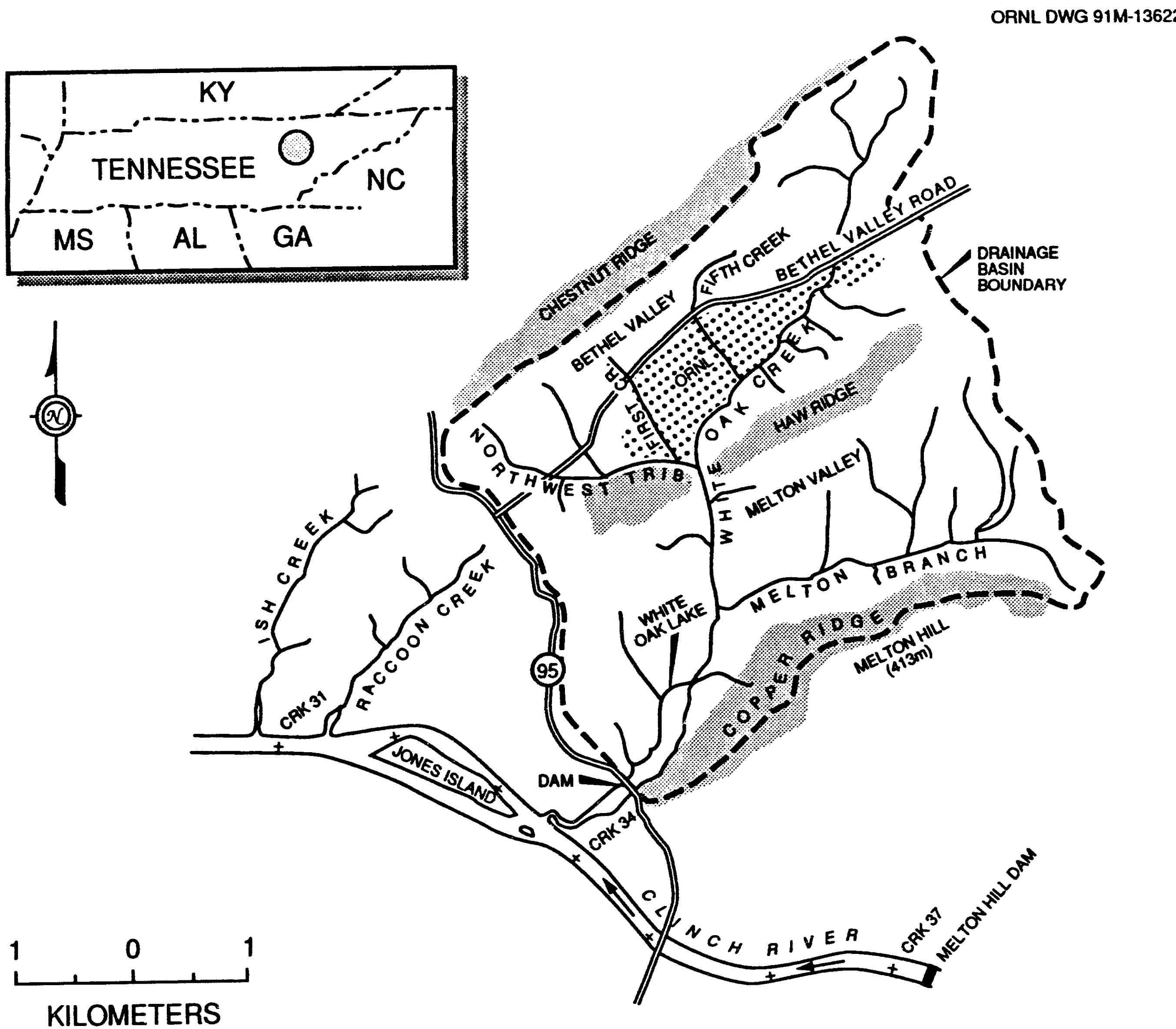

Fig. 2.2. Map of White Oak Creek watershed above White Oak Dam.

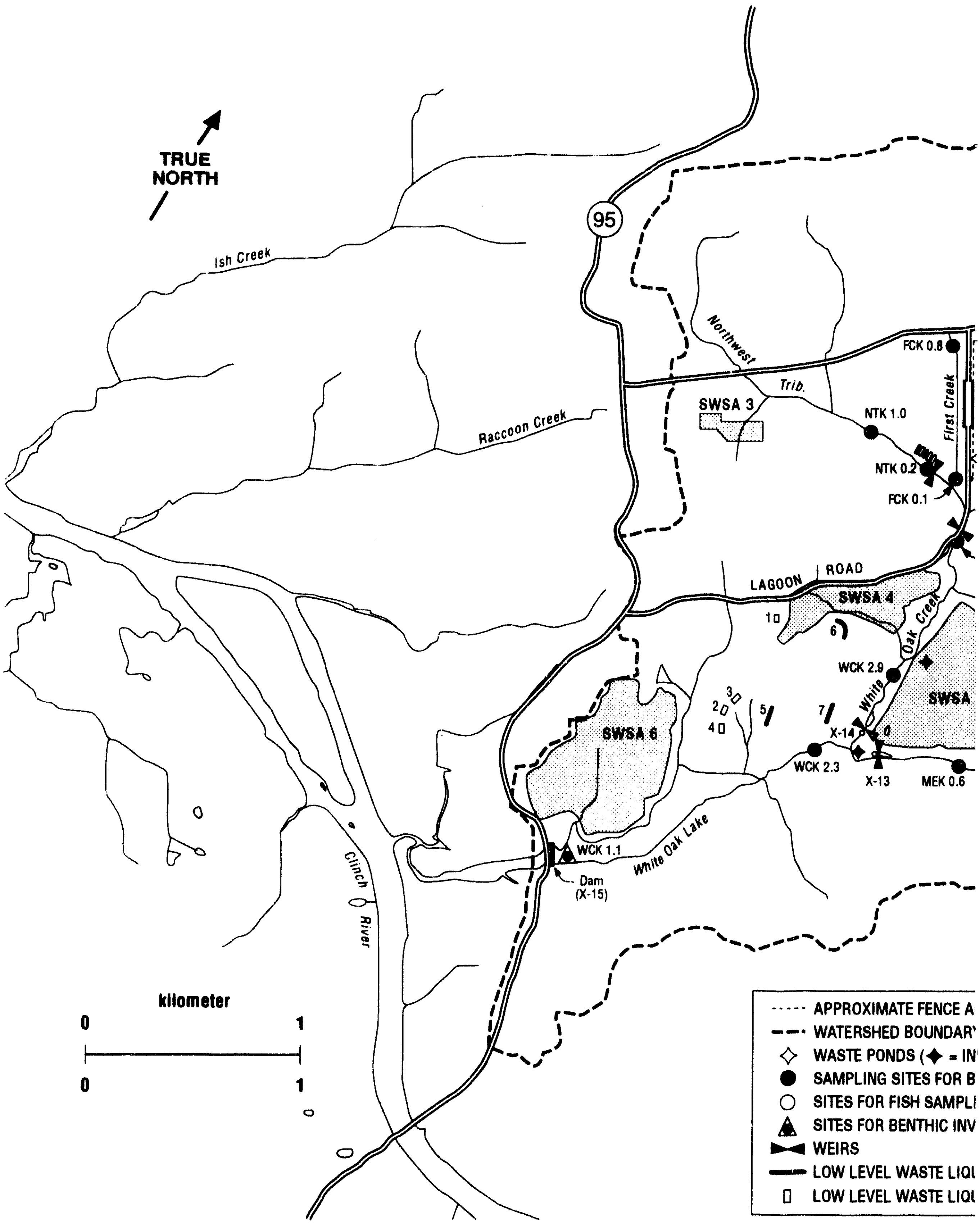

Fig. 2.3. Location of liquid and solid radioactive waste disposal/storage areas; National Pollutant Discharge I Dam (X15); and sampling sites for benthic invertebrates and fish in White Oak Creek watershed. FCK = First Branch kilometer; STP = Sewage Treatment Plant; SWSA = solid radioactive waste disposal/storage area; WCK

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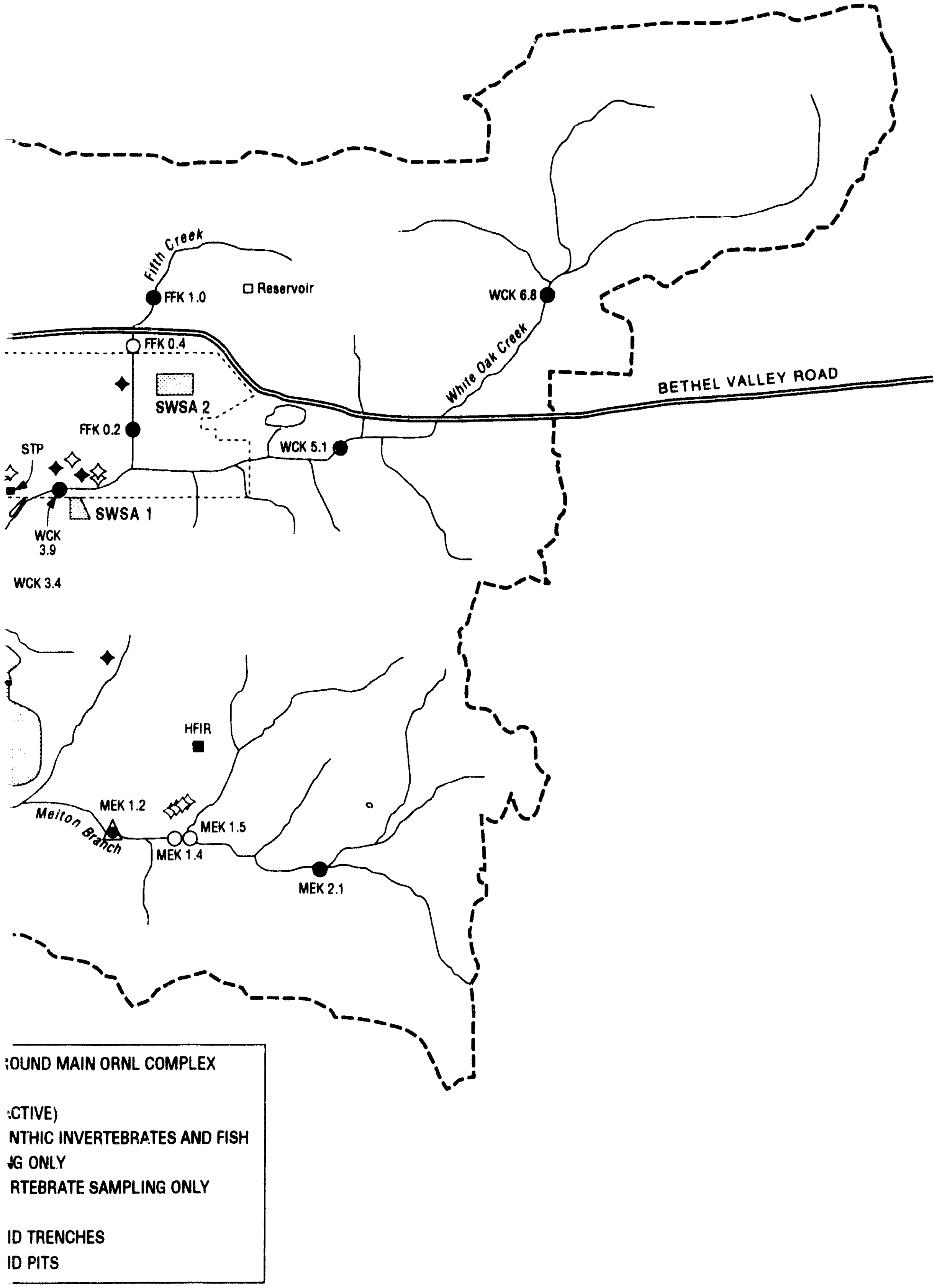

limination System monitoring sites on Melton Branch (X13), White Oak Creek (X14), and White Oak
Creek kilometer; FFK = Fifth Creek kilometer; HFIR = High Flux Isotope Reactor; MEK = Melton
= White Oak Creek kilometer.

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, McMaster 1967); both are poor water-bearing formations (McMaster and Waller 1965). Together, the two formations compose 95% of the surface area of Melton Branch watershed, whereas all of upper WOC watershed north of Bethel Valley Road is underlain by Knox Dolomite (McMaster 1967, Table 10).

The hydrology of Melton Branch reflects the underlying geology of the watershed. For example, base-flow discharge is low, with periods of no flow at times (McMaster 1967). Zero flow was observed in upper Melton Branch on 7 d in 1985 and 100 d in 1986 [U.S. Geological Survey (USGS), provisional data for gage 03537100]. Excluding September 2–6, when daily flow was 0.3 L/s, there were 83 consecutive days of no flow in 1986 (from July 17 through October 12; Fig. 2.4a). Zero flow was also observed in 1986 in upper Northwest Tributary, where a portion of the watershed drains the north slope of Haw Ridge. Total precipitation measured at the Atmospheric Turbulence and Diffusion Laboratory (ATDL) in Oak Ridge was 118.2 cm in 1985 and 98.6 cm in 1986, or 85% and 71% of normal, which is based on the 1951–80 recording period (NOAA 1986a, 1986b).

Streamflow in lower Melton Branch, on the other hand, is augmented by periodic discharges of several process waste basins and cooling tower blowdown (see Sect. 2.2.1). The discharges, which enter the stream near Melton Branch kilometer (MEK) 1.4 via an unnamed tributary (Fig. 2.3), are a significant fraction of the flow in Melton Branch and so reduce the probability of zero flow below MEK 1.4 (Figs. 2.4a, 2.4b).

Although gaging records for WOC above ORNL are limited to periodic measurements (n = 10) in 1961–64 (McMaster 1967, Table 5), the hydrology of this headwater region could be expected to differ from that of upper Melton Branch due to differences in the geologies of the 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). No periods of zero flow were observed in 1986 during studies conducted at 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 (Fig. 2.2). The watershed area at WCK 6.8 is 2.07 km^2 and is similar in size to that of upper Melton Branch at the USGS gaging station (1.35 km^2). A period of no flow was observed in 1979 in a reach of WOC just north of Bethel Valley Road near WCK 6.3 (Loar et al. 1981a).

As in lower Melton Branch, streamflow in WOC below ORNL is augmented by discharges from various facilities (Sect. 2.2.1). Low-flow measurements have shown that ~90% of the dry-weather discharge of the creek originates as groundwater discharge from the Knox Dolomite of Chestnut Ridge, from the Chickamauga Limestone of Bethel Valley, and from ORNL plant effluent (McMaster 1967). Although ORNL effluent provides a substantial portion of the flow in lower WOC, the magnitude of this contribution cannot be accurately assessed without data on streamflows in upper WOC.

Whereas low flow upstream and flow augmentation downstream dominated the hydrographs for Melton Branch and WOC in 1985 and 1986, high flows occurred infrequently. There were four major storms during this 2-year period (Fig. 2.4), but only one had a recurrence interval greater than 1 year. The maximum 24-h rainfall in 1985 occurred on

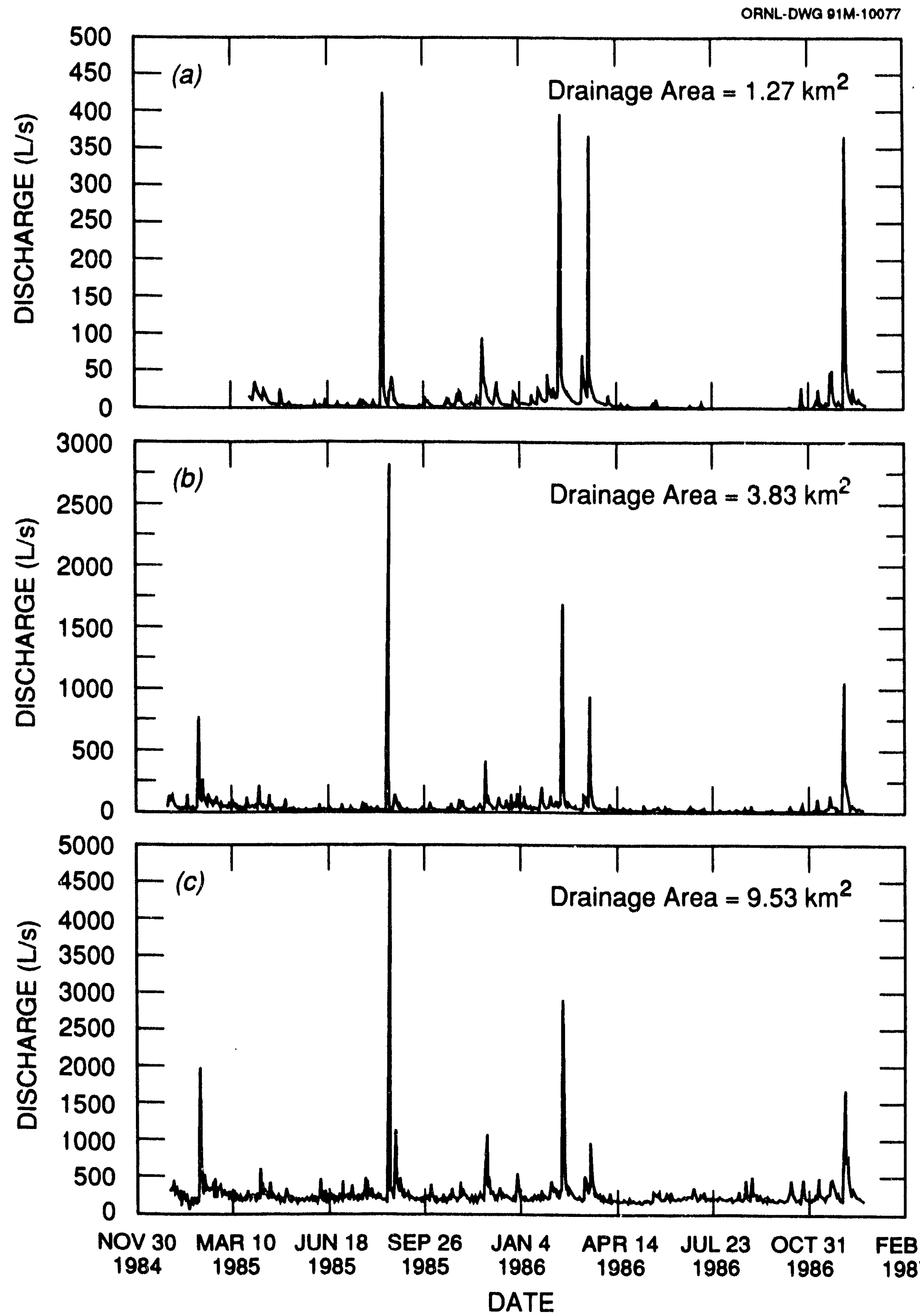

Fig. 2.4. Daily streamflow in (a) upper Melton Branch at U.S. Geological Survey gage 03537100, (b) lower Melton Branch at National Pollutant Discharge Elimination System (NPDES) Station X13 (Melton Branch kilometer 0.16), and (c) White Oak Creek at NPDES Station X14 (White Oak Creek kilometer 2.65) for January 1985–December 1986. Sites are shown in Fig. 2.3. *Source:* ORNL Department of Environmental Management.

August 16–17, when 10.9 cm (4.30 in.) of rain was recorded at ATDL in Oak Ridge (NOAA 1986b). A storm of this magnitude has a recurrence interval of 3 years (Sheppard 1974, Fig. 2).

2.2 WATER QUALITY

ORNL is centrally located in upper WOC watershed (Fig. 2.2). Although most of the ORNL complex is situated in Bethel Valley, some facilities are located in Melton Valley (Fig. 2.3). Both WOC and numerous tributaries, including First Creek, Fifth Creek, and Northwest Tributary, are located within or adjacent to the main plant area and receive effluents from various ORNL operations.

2.2.1 Description of ORNL Effluent Discharges

Wastewater discharges at ORNL are generated by the operation of nuclear reactors, chemical pilot plants, research laboratories, radioisotope production laboratories, and various support facilities. Such discharges 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 1986). Of the estimated total effluent discharge volume of 0.46 m^3/s (16.4 ft^3/s), ~30% and 36% 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 ten major effluent discharges that enter WOC either directly or indirectly via several tributaries (Table 2.1). Direct discharges to WOC, as listed in Table 2.1 and shown in Fig. 2.5, are largely restricted to a 1.5-km reach of the creek that flows west along the southern perimeter of ORNL (Fig. 2.3). Effluents are discharged to Melton Branch via a small tributary near MEK 1.4, whereas discharges to Northwest Tributary from the 1500 area and to Fifth Creek from the Oak Ridge Reactor are located above Northwest Tributary kilometer (NTK) 0.3 and just below Fifth Creek kilometer (FFK) 0.4, respectively (Figs. 2.3 and 2.5).

In addition to these major waste streams, there are 127 outfalls that also discharge to streams in WOC watershed, including Fifth Creek, First Creek, Melton Branch, and WOC. These include 34 noncontaminated storm drains (identified as Category 1 outfalls in the NPDES permit); 61 drains that have been contaminated by ORNL operations (Category II outfalls), including drains of roofs and parking lots, storage/spill areas, once-through cooling water, cooling tower blowdown and condensate; and 32 untreated process drains (Category III outfalls) that are contaminated by pollutants because of inflow/infiltration, cross-connections, or improper disposal of chemicals (DEM 1986a, EPA 1986).

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). Seven

Table 2.1. Description of the ten major effluent discharges regulated under the ORNL National Pollutant Discharge Elimination System permit that was issued April 1, 1986[a]

Receiving stream	Source of effluent discharge	NPDES outfall[b]	Average flow rate (L/s)[c]
Fifth Creek	ORR[d] resin regeneration facility	X10	0.4 (0.01)[e]
Melton Branch	TRU[d] process waste basins	X08	2.2 (0.08)[e]
	HFIR[d] process waste basins	X09	7.0 (0.25)[e]
Northwest Tributary	1500 area	X03	0.3 (0.01)
White Oak Creek	Sewage Treatment Plant	X01	10.1 (0.36)[f]
	Coal Yard Runoff Treatment Facility	X02	1.0 (0.04)[f]
	2000 area	X04	0.6 (0.02)
	3539 and 3540 ponds	X06	5.9 (0.21)[e]
	3544 Process Waste Treatment Plant	X07	7.9 (0.28)[f]
	3518 Acid Neutralization Facility	X11	1.8 (0.06)[e]

[a]Discharge locations are shown in Fig. 2.5.

[b]No outfall X05 exists. Outfall X12 is the planned discharge from the Nonradiological Wastewater Treatment Facility scheduled for completion in 1989, with a March 1990 date for compliance and an estimated average flow rate of 22 L/s (0.8 ft^3/s).

[c]Discharge in cubic feet per second in parentheses. For batch operations, average flow rate is based on days when waste is discharged.

[d]ORR = Oak Ridge Reactor; TRU = Transuranium Processing Facility; HFIR = High Flux Isotope Reactor.

[e]Batch discharge with frequencies of once every 5 d (X08), three times per month (X09), once every 5–8 d (X10), and three batches per day (X11). Discharge X06 is batch if radioactivity is below predetermined levels.

[f]Maximum flow rates are 32.9 L/s (1.16 ft^3/s) at X01, 9.6 L/s (0.34 ft^3/s) at X02, and 18.8 L/s (0.67 ft^3/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.

mechanical-draft cooling towers discharge the principal heat burden generated by the operation of ORNL facilities, 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 is discharged to Melton Branch from operation of the High Flux Isotope Reactor (HFIR), 3.3 L/s is discharged to Fifth Creek from operation of the Oak Ridge Reactor, and 3.8 L/s is 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 Facility prior to discharge. Normally, however, cooling tower blowdown is discharged directly to area streams or indirectly via the storm sewer system.

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 (Sect. 10). The sewage treatment plant was upgraded in August 1985 to include a new extended aeration–activated sludge plant to reduce periodic hydraulic overloading, and defective sections of sewer pipes were relined to reduce groundwater infiltration (Kasten 1986). In early March 1986, a new coal yard

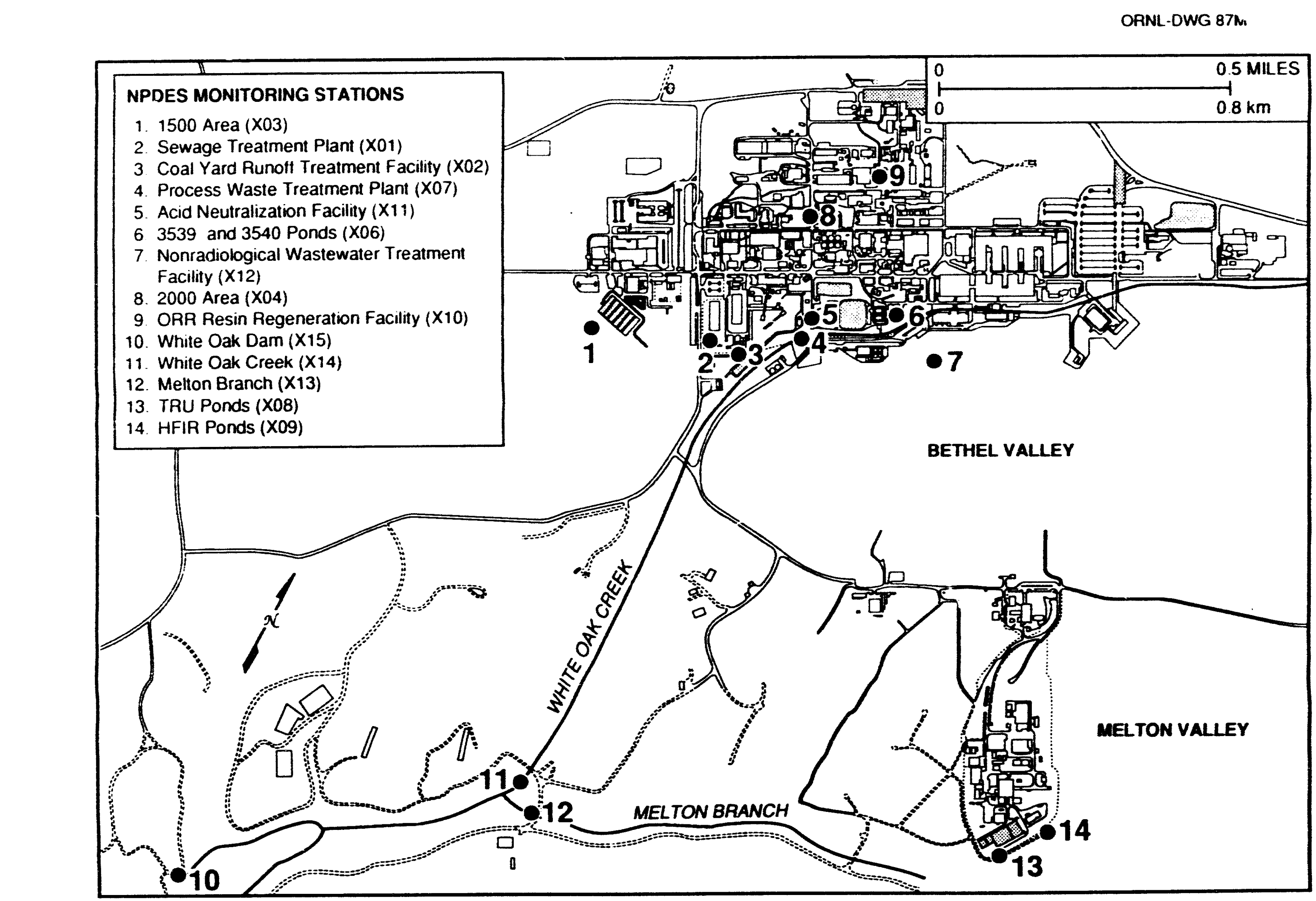

Fig. 2.5. Location of National Pollutant Discharge Elimination System (NPDES) effluent (Sites 1–9 and 13–14) and instream (Sites 10–12) water quality monitoring stations. Station numbers on legend do not correspond to serial identification of the discharges in the NPDES permit. ORR = Oak Ridge Reservation; TRU = Transuranium Processing Facility; HFIR = High Flux Isotope Reactor. *Source:* Adapted from ORNL Department of Environmental Management, *Environmental Surveillance Data Report for the Second Quarter of 1986*, ORNL/M-173, Oak Ridge National Laboratory, Oak Ridge, Tenn., 1986, Fig. 8.

runoff treatment facility became operational. This facility may also be used to treat boiler blowdown and demineralizer regeneration waste from the steam plant, wastewaters that are currently treated at the 3518 Acid Neutralization Facility. In August 1986, new demineralizer systems were installed at the HFIR and Oak Ridge Reactor to reduce nitrate discharges to Melton Branch and Fifth Creek, respectively (Martin Marietta Energy Systems 1986b). One of the most significant components of the WPCP is construction of the Nonradiological Wastewater Treatment Facility (NRWTF). Scheduled for completion in 1989, the facility will treat effluents that currently are (1) untreated and discharged directly to area streams, (2) untreated but discharged to process waste basins prior to release, or (3) treated by existing facilities. NRWTF will collect and treat NPDES serial discharges X03 through X10; discharge X11 will be treated by either NRWTF or the Coal Yard Runoff Treatment Facility (Table 2.1). Additional pollution abatement projects and programs are described in Sect. 10.

2.2.3 NPDES 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 radioactive waste disposal/storage areas (SWSAs), liquid radioactive waste disposal/storage areas (pits and trenches), and inactive process waste basins (Fig. 2.3). The discharge to area streams of leachates from waste disposal areas has been documented by Stueber et al. (1981) for Northwest Tributary (from SWSA 3) and by Cerling and Spalding (1982) for WOC (SWSA 4), Melton Branch (SWSA 5), and WOL (SWSA 6). The following characterization of ambient water quality is based on routine NPDES monitoring conducted at sites MEK 0.16 and WCK 2.65 (NPDES sites X13 and X14, respectively), both of which are located downstream of all point-source discharges and most major area sources (Fig. 2.3); data collected from a site above ORNL and at White Oak Dam (X15) are also included for comparative purposes.

Routine radiological monitoring is also conducted at these sites and others, including Melton Branch just below the HFIR, First Creek, Fifth Creek, Northwest Tributary, Raccoon Creek, and WOC at WCK 3.41 (Fig. 2.3). These data are collected either daily or weekly and are published quarterly by the ORNL Department of Environmental Management (DEM 1986a, 1986b).

Ambient water quality data collected by DEM during the period January 1985 through December 1986 are summarized in Tables 2.2 and 2.3. Data were available on 12 parameters for 1985 and 1986 (Table 2.2); routine monitoring of an additional 20 parameters was initiated in 1986 in compliance with the conditions stipulated in the new NPDES permit that was issued on April 1, 1986 (Table 2.3). Construction of new weirs and monitoring stations at these three NPDES sites (X13–X15) was completed in 1984 (DEM 1986a). Each site was again upgraded in July 1986 (September 1986 at WCK 2.65) to provide real-time monitoring of conductivity, dissolved oxygen, pH, temperature, and streamflow (DEM 1986b).

Water quality in both 1985 and 1986 was characterized by periods of low dissolved oxygen (below 5 mg/L); high conductivities and total dissolved solids (TDS), especially in Melton Branch; and high turbidity levels that did not always coincide with major storms (Table 2.2). Temperatures in lower Melton Branch in 1985 were higher and more variable than in lower WOC (Fig. 2.6). Based upon better records that were available after July 1986 (i.e., daily grab samples in 1985 vs continuous records in 1986), average daily temperatures

Table 2.2. Median concentration (range in parentheses) of 12 water quality parameters monitored at Melton Branch kilometer (MEK) 0.16, at White Oak Creek kilometer (WCK) 2.65, and at White Oak Dam (WOD) under the old and new (after April 1, 1986) National Pollutant Discharge Elimination System (NPDES) Permit

(Values in milligrams per liter unless otherwise noted)

	Sample type[a]	Sampling frequency[b]	MEK 0.16[c]		WCK 2.65[d]		WOD	
			1985	1986	1985	1986	1985	1986
Ammonia	1	M/M	NS[e]	0.06 (0.03–2.00)	0.26 (<0.10–1.60)	0.07 (0.03–0.14)	NS	0.08 (0.05–0.47)
Biological oxygen demand, 5 d	1	W/M	<5 (5–6)	<5 (<5–13)	<5 (<5–52)	<5 (<5–10)	NS	<5 (<5–11)
Chemical oxygen demand	1	W/0	6 (0–34)	8.5 (<5–66)	5 (0–102)	2.5 (<1–21)	NS	NS
Conductivity (μS/cm)	2	W/M	500 (270–900)	455 (100–1000)	370 (200–600)	365 (200–480)	NS	390 (200–410)
Chromium (μg/L)	1	W/M	<10 (<10–10)	<10 (<10–12)	<10 (<10–10)	<10 (<4–38)	NS	<24 (<20–4)[f]
Dissolved oxygen	2	D/W	8.0 (4.9–13.2)	8.0 (4.0–12.9)	7.8 (2.3–10.0)	7.8 (5.0–12.3)	8.4 (2.7–17.0)	7.4 (2.0–15.0)
Dissolved solids	2	M/M	358 (145–759)	464 (120–1000)	243 (158–300)	254 (168–324)	NS	270 (136–358)
Oil and grease	2	M/W	2 (<2–5)	2 (<2–69)	<2 (<2–6)	<2 (<2–107)	NS	2 (<2–27)
pH	2	D/M	7.8 (6.0–9.0)	8.0[g] (7.1–8.9)	7.8 (6.5–9.1)	7.7[g] (7.0–8.6)	7.8 (6.2–9.2)	7.9[g] (7.2–8.5)
Suspended solids	1	W/M	5 (<5–165)	<5 (<5–85)	<5 (<5–33)	<5 (<5–87)	NS	5 (<5–52)

Table 2.2 (continued)

	Sample type[a]	Sampling frequency[b]	MEK 0.16[c]		WCK 2.65[d]		WOD	
			1985	1986	1985	1986	1985	1986
Temperature (°C)	2	D/M	19 (2–29)	9[g] (0–17)	19 (3–26)	11[g] (6–16)	19 (2–29)	9[g] (2–18)
Turbidity (JTU)[h]	2	W/M	22 (5–150)	10 (0–82)	109 (5–240)	11 (0–240)	NS	78 (35–240)

[a]1 = 24-h composite; 2 = grab.

[b]Numerator is frequency under old NPDES permit (January 1, 1985–March 31, 1986), and denominator is frequency under new NPDES permit (April 1–December 31, 1986). D = daily; W = weekly; M = monthly; 0 = discontinued.

[c]Average background concentrations were 159 mg/L for total dissolved solids and 0.5 μg/L for chromium, based on grab samples collected weekly between April 1979 and January 1980 in upper Melton Branch near MEK 1.8 (Boyle et al. 1982, Fig. 3.11 and Table 4.16). Complete citations to references are included in Sect. 11 of this report.

[d]Average background concentrations above ORNL near WCK 6.3 were 101 mg/L for total dissolved solids and 0.5 μg/L for chromium (see footnote *c* and Boyle et al. 1982, Fig. 3.11 and Table 4.15); near WCK 6.8, average pH and conductivity were 7.8 and 209 μS/cm respectively (McMaster 1967, Table 11).

[e]NS = not sampled.

[f]Data for this parameter were 7 observations of <24, 2 observations of <20, and 1 observation of 4.

[g]January through March only.

[h]Jackson turbidity unit.

Source: ORNL Department of Environmental Management.

Table 2.3. Median concentration (range in parentheses) of 20 water quality parameters measured monthly at Melton Branch kilometer (MEK) 0.16, at White Oak Creek kilometer (WCK) 2.65, and at White Oak Dam (WOD) during the period April 1–December 31, 1986[a]

Parameter	Concentration (μg/L, unless noted otherwise)			
	Above ORNL[b,c]	MEK 0.16	WCK 2.65	WOD
Aluminum (mg/L)	0.06[d]	<0.12 (<0.02–0.84)	<0.12 (<0.02–0.41)	0.20 (<0.12–1.30)
Arsenic[e]	NS	<60 (<10–<60)	<60 (<10–<60)	<60 (<10–<60)
Cadmium	0.12	<2 (all <2)	<2 (all <2)	<2 (all <2)
Chlorine, residual (mg/L)[f,g]	NS	0.00 (0.00–0.16)	0.00 (all 0.00)	0.00 (0.00–0.10)
Chloroform[g]	NS	<1.6 (0–3.0)	6.8 (0–8.0)	2.8 (0–3.4)
Copper	0.9	<12 (<2–<12)[e]	<12 (<2–13)	<12 (<2–15)
Fluoride (mg/L)	0.1[d]	2.2 (<1.0–25.0)	1.0 (<1.0–1.0)	1.0 (<1.0–1.0)
Iron	65	178 (50–650)	94 (22–490)	285 (86–1300)
Lead	0.9	<4 (<4–4)	<4 (<4–5)	<4 (<4–5)
Manganese	12	94 (31–450)	29 (23–43)	82 (28–1500)
Mercury	0.02	<0.05 (<0.05–0.10)	0.10 (<0.05–0.20)	<0.05 (<0.05–0.10)
Nickel[e]	4	<36 (<6–<36)	<36 (<6–<36)	<36 (<6–<36)
Nitrate (mg/L)	0.3[d]	<5.0 (<2.0–14.0)	<5.0 (<2.0–5.3)	<5.0 (<2.0–5.0)
Organic carbon, total (mg/L)[g]	NS	4.0 (1.8–5.0)	3.0 (2.2–4.6)	3.6 (1.9–5.7)
Phenols, total[g]	2	<1 (<1–2)	<1 (<1–3)	NS
Phosphorus (mg/L)	0.02	0.88 (0.11–1.70)	0.63 (0.13–0.80)	0.36 (0.17–0.76)

Table 2.3 (continued)

Parameter	Concentration (μg/L, unless noted otherwise)			
	Above ORNL[b,c]	MEK 0.16	WCK 2.65	WOD
Silver[e]	NS	<5 (<5–<30)	<25 (all <25)	<25 (all <25)
Sulfate (mg/L)	2.7[d]	306 (21–1065)	60 (24–299)	62 (25–305)
Trichloroethylene[g]	NS	<1.9 (0.0–<5.0)	<1.9 (0.0–<10.0)	<1.9 (0.0–<10.0)
Zinc	3	41 (15–150)	32 (27–40)	14 (<10–61)

[a]Values represent 24-h composite samples unless noted otherwise. Most parameters were also sampled in January prior to issuance of the new National Pollutant Discharge Elimination System permit. NS = not sampled.

[b]North of Bethel Valley Road near WCK 6.3.

[c]Values represent the mean concentration of grab samples collected weekly between April 1979 and January 1980; $n = 37$ (see Boyle et al. 1982, Sect. 3.2.3.2 and Table 4.15). Sin..ar sampling was conducted in upper Melton Branch near MEK 1.8 over the same time period, and average concentrations were the same as those for upper White Oak Creek above ORNL except for Fe (130 μg/L), Hg (0.05 μg/L), Mn (32 μg/L), and Ni (5 μg/L) (Boyle et al. 1982, Table 4.16). Complete citations to references are included in Sect. 11 of this report.

[d]North of Bethel Valley Road near WCK 6.8; values represent the mean concentration of samples collected in 1961–1964; $n = 6$ (McMaster 1967, Table 11).

[e]All values were below the detection limits.

[f]Sampled weekly.

[g]Grab sample.

Source: ORNL Department of Environmental Management.

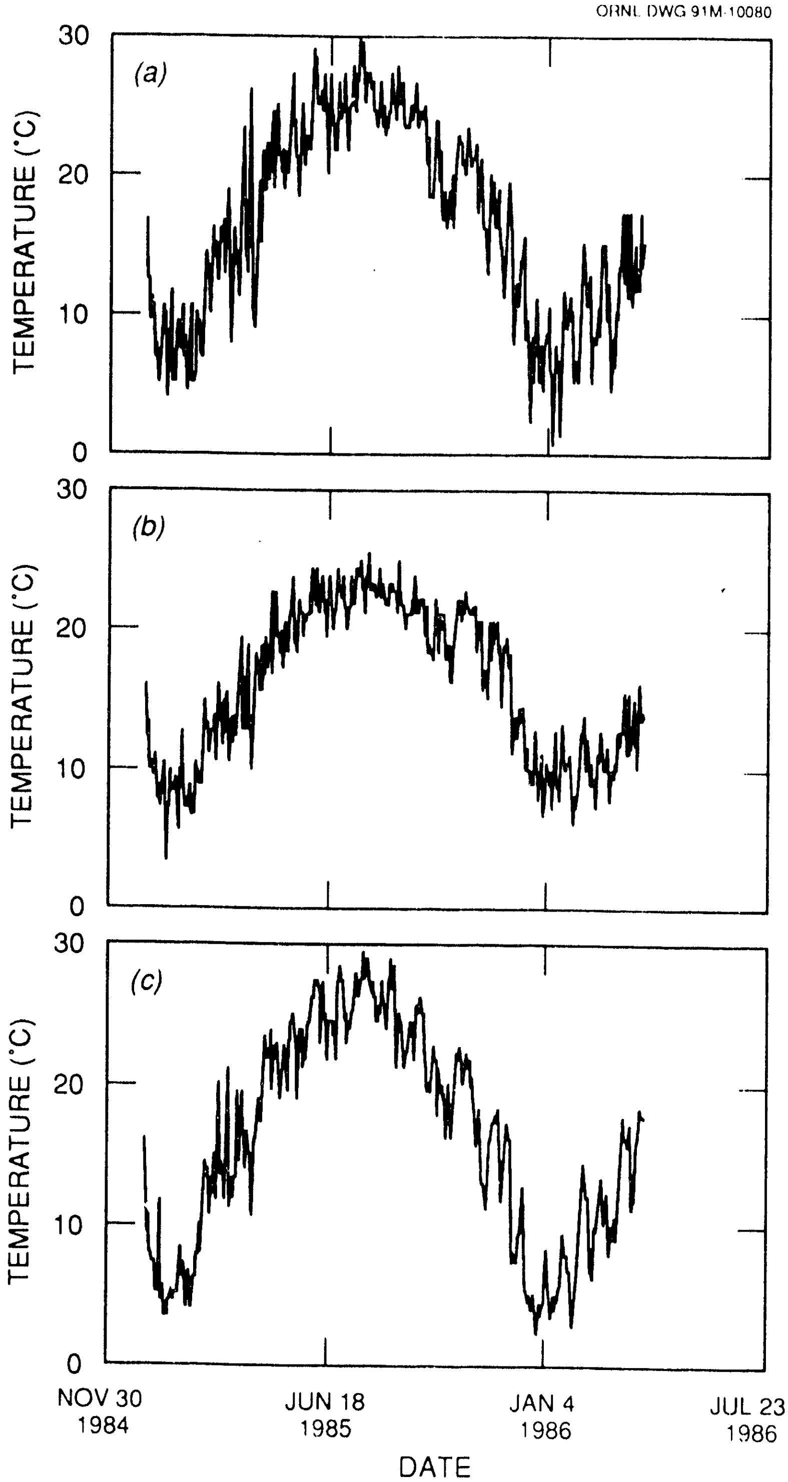

Fig. 2.6. Daily water temperatures at National Pollutant Discharge Elimination System monitoring sites, January 1985 through March 30, 1987: (a) site X13 at Melton Branch kilometer 0.16, (b) site X14 at White Oak Creek kilometer 2.65, and (c) site X15 at White Oak Dam. Data are equivalent to grab samples collected in the morning at each location. *Source:* ORNL Department of Environmental Management.

in Melton Branch during summer 1986 occasionally exceeded 30°C, and maximum temperatures approached 38°C (Fig. 2.7). As measured by continuous strip-chart recorders, maximum temperatures in Brushy Fork, a reference stream north of Oak Ridge (Fig. 2.1), never exceeded 25°C over the same period (J. M. Loar, unpublished data).

Inclusion of 20 additional parameters in the NPDES monitoring program in 1986 provides a more complete characterization of water quality in the two streams (Table 2.3). For example, concentrations of trace elements and total phenols in lower WOC and lower Melton Branch exceeded the levels observed in the unimpacted upper reaches of these two streams. Most, however, did not exceed the U.S. Environmental Protection Agency (EPA) water quality criteria for freshwater aquatic life (EPA 1976, 1980a). The maximum level of iron observed at White Oak Dam exceeded the criterion of 1000 μg/L (EPA 1976), and the maximum mercury concentration in WOC exceeded the criterion of 0.14 μg/L for protection of human health from ingestion of organisms and water (EPA 1980a). The detection limits for Cd, Cu, Ni, and Ag 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. 3.1.3.1), however, it is likely that concentrations of these elements are usually below levels that would be toxic to biota.

The lower reaches of both Melton Branch and WOC are enriched by nutrients, especially phosphorus. The average concentration of 0.02 mg/L total phosphorus reported by Boyle et al. (1982) for WOC north of Bethel Valley Road is lower than the concentrations in Melton Branch and lower WOC by more than an order of magnitude (Table 2.3). Some nitrate loading to the streams is also evident, although high detection limits preclude an accurate assessment of the degree of enrichment. (At the three sites 85% of the values were below the detection limits of either 2 or 5 mg/L.) Additional information on nutrient enrichment in WOC watershed is provided in Sect. 2.2.4.

Differences in the water quality of Melton Branch and lower WOC are related to the composition, frequency, and volume of the discharges from the Transuranium Processing Facility (TRU) and HFIR process waste basins. The batch discharges have a high total dissolved solids (TDS) content composed primarily of sulfates (Table 2.3). The two TRU basins, each with a capacity of 189 m^3 (50,000 gal), are emptied, on average, every 5 d. The single HFIR waste basin has a capacity of 946 m^3 (250,000 gal) and is emptied when two-thirds of the capacity is reached, or approximately three times per month (EPA 1986). Streamflow in Melton Branch above the HFIR/TRU tributary was generally low or zero during much of 1986 (Sect. 2.1 and Fig. 2.4a). Consequently, dilution of these discharges was minimal between the point of discharge and the monitoring site ~1.2 km downstream. Maximum values for conductivity, TDS, and sulfates in 1986 occurred in October at both MEK 0.16 and White Oak Dam, suggesting that HFIR/TRU discharges can be observed downstream of Melton Branch. The batch discharge mode also contributed to the high variability observed in several parameters, including conductivity, dissolved oxygen, and pH (Fig. 2.8). When the HFIR was shut down on November 14, 1986 (J. D. Story, ORNL Department of Environmental Management, personal communication, 1986), this variability was reduced compared both with the pre-HFIR period and with WOC (Fig. 2.9). Minimum values for conductivity, TDS, and sulfates in 1986 were observed the following month at all three sites, due both to the absence of HFIR discharges and to dilution resulting from high streamflows (Fig. 2.4). Although discharge from the waste basins contributes significantly to short-term fluctuations in water quality, more than 90% of the TDS loading is from cooling

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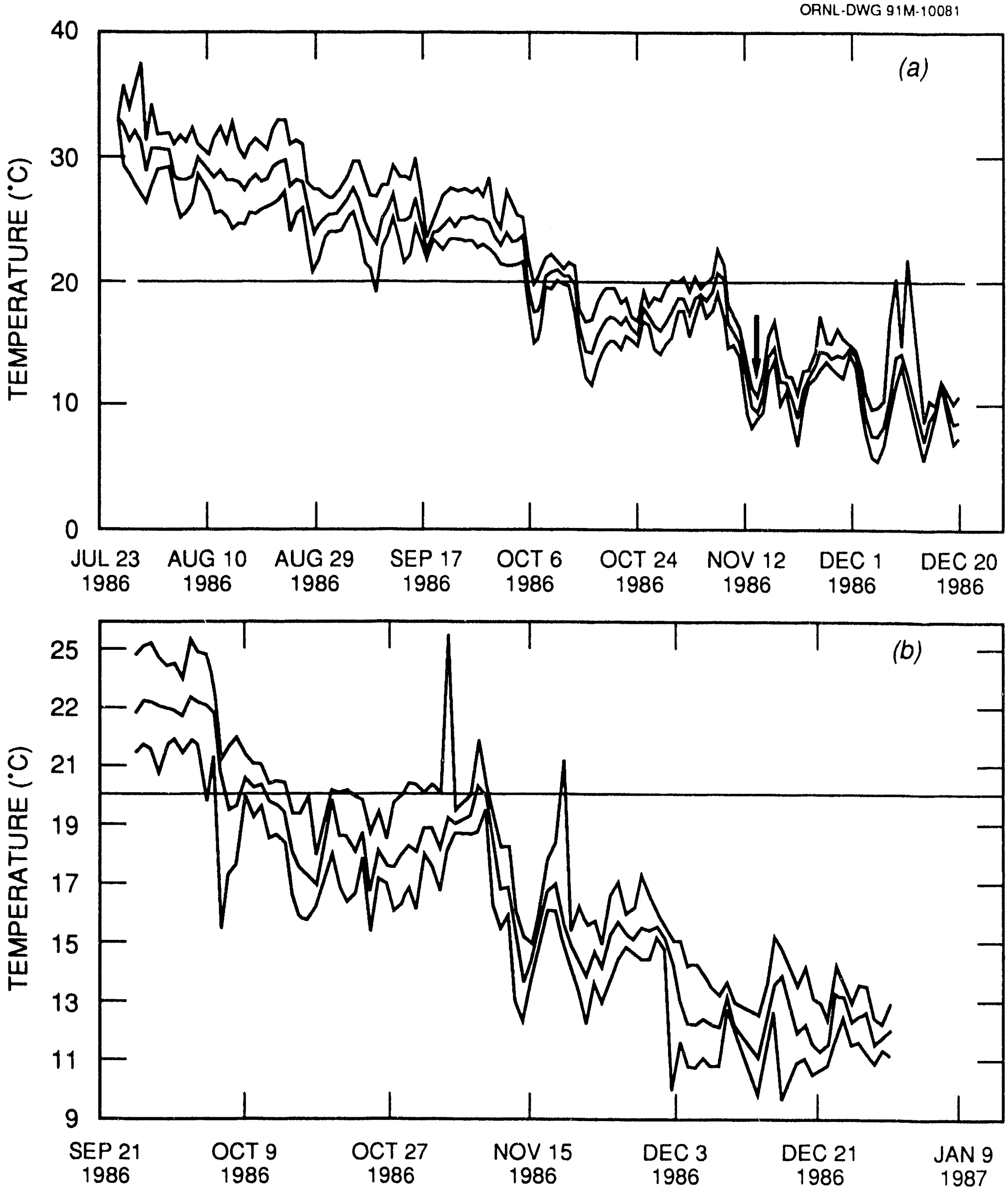

Fig. 2.7. Maximum, average, and minimum daily temperatures at National Pollutant Discharge Elimination System monitoring sites (a) X13 on lower Melton Branch and (b) X14 on White Oak Creek. These sites are located at Melton Branch kilometer 0.16 and White Oak Creek kilometer 2.65 respectively. Data were collected at 10-min intervals by real-time monitoring systems installed in 1986. Arrow marks November 14, 1986, when the High Flux Isotope Reactor was shut down. *Source:* ORNL Department of Environmental Management.

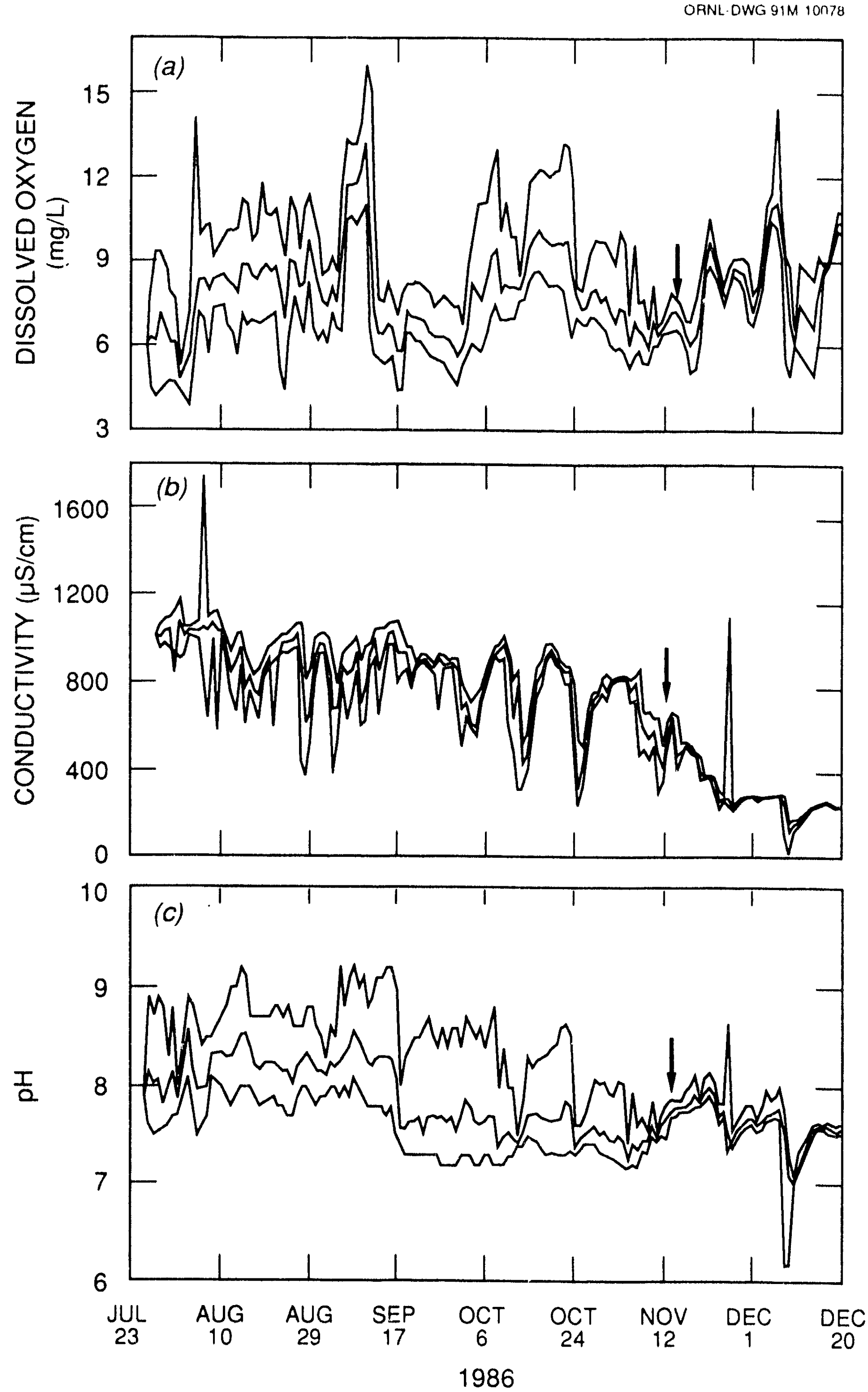

Fig. 2.8. Daily maximum, average, and minimum values for (a) dissolved oxygen, (b) conductivity, and (c) pH in lower Melton Branch at National Pollutant Discharge Elimination System monitoring station X13 (Melton Branch kilometer 0.16). Data were collected at 10-min intervals by real-time monitoring systems installed in July 1986. Arrows mark November 14, 1986, when the High Flux Isotope Reactor was shut down. *Source:* ORNL Department of Environmental Management.

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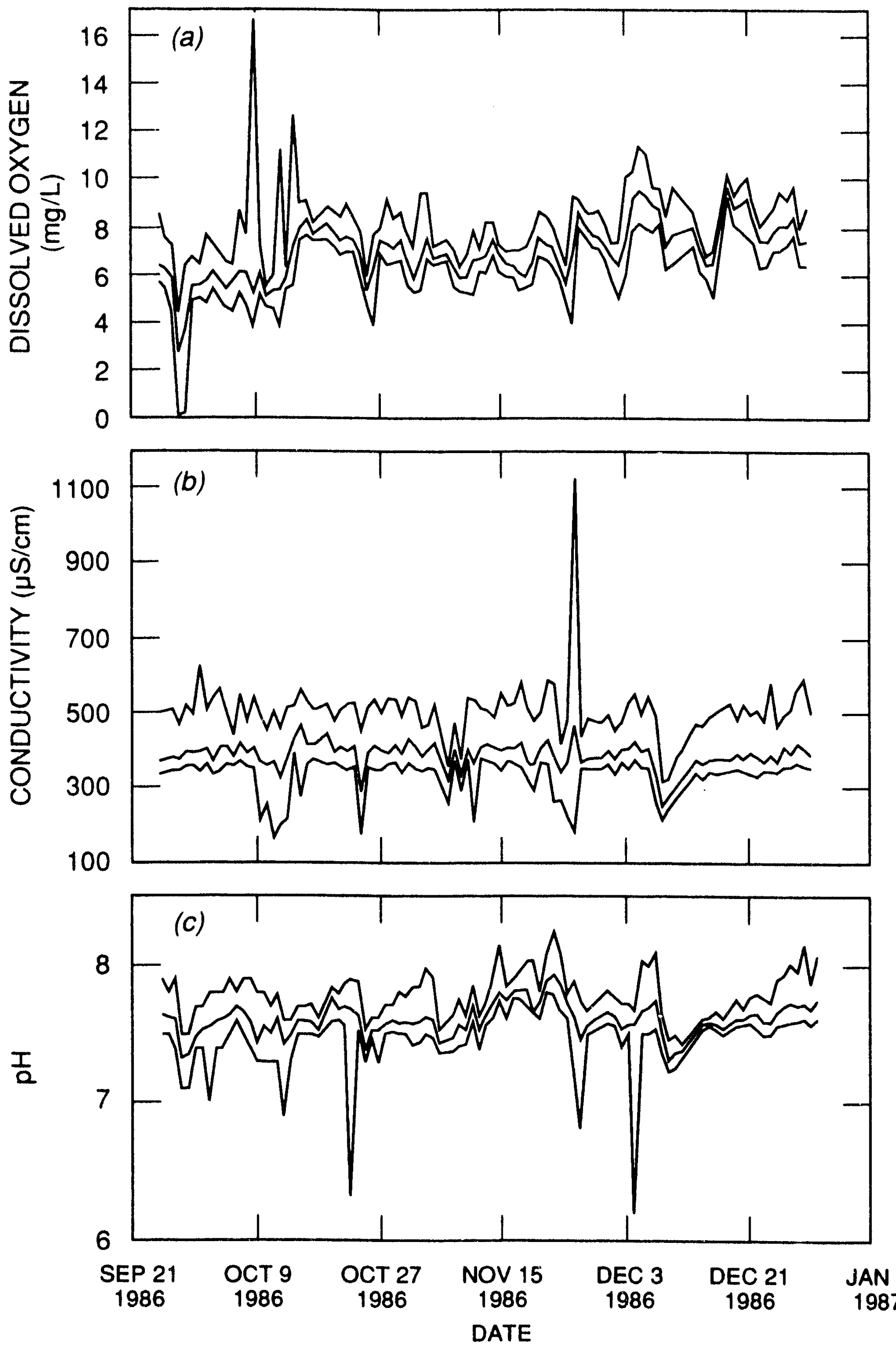

Fig. 2.9. Daily maximum, average, and minimum values for (a) dissolved oxygen, (b) conductivity, and (c) pH in White Oak Creek at National Pollutant Discharge Elimination System monitoring station X14 located at White Oak Creek kilometer (WCK) 2.65. (The confluence with Melton Branch is downstream at WCK 2.49.) Data were collected at 10-min intervals by real-time monitoring systems installed in September 1986. The low concentration of dissolved oxygen occurred on September 28–29, 1986, and was associated with a release of ethylene glycol from a ruptured chilled waterline in the Building 6010 area (near WCK 4.9) on September 27, 1986. *Source:* ORNL Department of Environmental Management.

tower blowdown, which has an average flow rate that is more than ten times greater than that of the waste basins combined (Boyle et al. 1982, Table 2.12).

2.2.4 Periphyton-Related Discrete Water Quality Sampling Program

In addition to the water quality monitoring conducted by DEM on lower WOC, on Melton Branch, and at White Oak Dam, water quality analyses were conducted as a component of the periphyton/microbial community studies described in Subtask 1c of BMAP (see Sect. 3.2). Water samples were collected monthly and coincided with the collection of periphyton samples. This sampling program provides information on environmental parameters (inorganic carbon, plant nutrients, etc.) that can be used to evaluate the condition and activity of the biological communities. Rates of algal photosynthesis and accrual of primary-producer biomass are affected by the availability of nutrients, such as phosphorus and nitrogen. Other aspects of water quality (such as available metals) may adversely influence the primary producers and either directly or indirectly (via the food chain) affect other biotic communities. Although grab samples only provide information on the conditions at the time of sampling, such samples document spatial variations in water quality, information that existing monitoring programs could not provide. These data were used to interpret the characteristics and seasonal dynamics of the periphytic algal communities at the various stream sites (Sect. 3.2). This information further contributes both to the characterization of the ecology of the streams in WOC watershed and to the capability for predicting ecosystem responses to remedial actions.

2.2.4.1 Methods

Two 1-L grab samples of stream water were collected from each site 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 a subsample; for the remainder, the DOC was partitioned into hydrophilic and hydrophobic fractions and was partitioned by molecular weight. (Samples collected in September were not fractionated.) On each sampling date, each site was also evaluated for pH, alkalinity, conductivity, hardness, and soluble metals from single measurements; analyses of the other parameters were performed in duplicate. Measurements of pH, alkalinity, hardness, and conductivity were made within 2 h after collection. Samples for other analyses were preserved and/or frozen, according to approved methods (EPA 1983), until analysis. The parameters determined for the discrete water samples are listed in Table 2.4.

2.2.4.2 Results

Although the discrete water quality sampling program was initiated in August 1986, the first complete data set for this sampling program was available for September 1986 (Table 2.5). Compared with sites farther downstream, the upstream reference sites generally had lower concentrations of soluble nitrogen and phosphorus and were lower in conductivity. The decrease in alkalinity and the increase in conductivity and soluble nutrients at the downstream sites are attributable to discharges from ORNL. The greatly increased concentrations of soluble reactive phosphorus (SRP)—>100 micrograms of phosphorus per liter—and NO^-_2-N + NO^-_3-N at WCK 3.9 were similar to those downstream of the discharge of treated sewage effluents (e.g., WCK 3.4). Site MEK 0.6 on Melton Branch below the

Table 2.4. Water quality parameters determined for discrete samples collected monthly at ten sites in streams in White Oak Creek watershed

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 titration[b]	APHA (1985)
Phosphorus	Ascorbic acid method	APHA (1985)
Total P	Persulfate digestion	
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)
Particulate carbon and nitrogen	Elemental analyzer	
Dissolved metals	Filter and induction coupled plasma emission spectros	APHA (1985)
Dissolved organic C	Persulfate oxidation and infrared detection	Dohrmann Co. (1984)

[a]Complete citations to references are included in Sect. 11 of this report.
[b]EDTA = ethylenediamine tetracetic acid.

HFIR/TRU tributary was greatly enriched with nitrogen and phosphorus and was elevated in conductivity and hardness. It also had less alkalinity than the upstream site. DOC concentrations were lowest at the upstream sites, especially FFK 1.1 and near First Creek kilometer (FCK) 1.0; these two reference sites are only a short distance from their spring origins. Concentrations of DOC at other sites were similar to those for other streams in this region (H. L. Boston, unpublished data). Although samples collected later in the year were fractionated according to hydrophobicity and molecular weight, these analyses are not yet complete. Concentrations of total suspended solids were lowest for the reference sites near their spring sources (FCK 1.0 and FFK 1.1) and were higher for those upstream reference sites that had substantial watershed areas (MEK 1.8, NTK 1.0, and WCK 6.8). The concentrations of total suspended solids for the sites corresponded to patterns of water velocity and drainage area. Concentrations of most soluble metals were not markedly different between upstream and downstream sites, although the alkaline earth metals (especially calcium) tended to be elevated downstream. Data for basic cations and alkalinity show the influences of ORNL discharges (e.g., sulfates from the HFIR) at the downstream sites (see Sect. 2.2.3). Data for other elements are presented in Appendix A.

Table 2.5. Discrete water quality monitoring results for September 1986

	WCK[a] 6.8	WCK 3.9	WCK 3.4	WCK 2.9	WCK 2.3	MEK[a,b] 1.8	MEK 0.6	FCK[a] 1.0	FFK[a] 1.1	NTK[a,b] 1.0
pH	8.19	8.12	8.03	8.05	8.06	7.9	7.75	8.23	7.84	7.89
Alkalinity (meq/L)	3.03	2.5	2.3	2.28	2.2	2.24	0.99	3.14	2.98	4.56
Conductivity (μmhos/cm)	266	361	334	342	391	226	878	259	269	390
Hardness (mg/L as $CaCO_3$)	166	180	172	166	194	138	460	162	142	190
Total phosphorus (mg P/L)	19.1	595.1	673.3	680.2	748.1	20.6	1568	43	27.8	15
Soluble unreactive P (μg P/L)	15.1	69.6	40.2	90.2	31	12.1	99.6	28.6	11.8	10.9
Soluble reactive P (μg P/L)	0.7	127.4	275.6	292.8	350	7.3	858.9	3.5	7.7	1.9
$NO_2^- + NO_2^-$ (mg N/L)	0.05	0.58	1.29	1.4	1.34	0.01	1.97	0.04	0.15	0.02
Ammonia-N (μg N/L)	24	25.3	26.5	22.8	26.5	2.9	31.5	8.4	12.7	2.4
Dissolved organic C (mg C/L)	2.24	3.54	3.34	3.36	3.11	3.21	3.49	0.96	0.76	2.24
Total suspended solids (mg/L)	3.3	1.7	2.9	2.5	3.2	5.5	1.2	2.8	2.3	5.7
Temperature (°C)	17.1	22.5	21.6	21.5	21.8	10.4	25.7	18	14.2	12.5
Soluble Ca (mg/L)	12	44	40	44	47	NA[c]	140	13	12	NA

Table 2.5 (continued)

	WCK[a] 6.8	WCK 3.9	WCK 3.4	WCK 2.9	WCK 2.3	MEK[a,b] 1.8	MEK 0.6	FCK[a] 1.0	FFK[a] 1.1	NTK[a,b] 1.0
Soluble Cu (mg/L)	<0.2	<0.2	<0.2	<0.2	<0.2	NA	<0.2	<0.2	<0.2	NA
Soluble Cr (mg/L)	<0.04	<0.04	<0.04	<0.04	<0.04	NA	<0.04	<0.04	<0.04	NA
Soluble Cd (mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	NA	<0.005	<0.005	<0.005	NA
Soluble Fe (mg/L)	<0.03	<0.03	<0.03	<0.03	<0.03	NA	<0.03	<0.03	<0.03	NA
Soluble Mg (mg/L)	17	13	11	11	13	NA	31	16	16	NA
Soluble Mn (mg/L)	<0.005	0.019	0.018	0.015	0.023	NA	0.052	<0.005	<0.005	NA
Soluble Na (mg/L)	0.63	9.5	12	13	15	NA	28	0.98	1.6	NA
Soluble Pb (mg/L)	<0.2	<0.2	<0.2	<0.2	<0.2	NA	<0.2	<0.2	<0.2	NA
Soluble Si (mg/L)	3.9	3	2.5	2.6	2.9	NA	7.1	4	3.5	NA
Soluble Zn (mg/L)	<0.02	<0.02	<0.02	<0.02	<0.02	NA	<0.02	<0.02	<0.02	NA

[a]Upstream reference sites: White Oak Creek kilometer 6.8 and Northwest Tributary kilometer 1.0 are shown in Fig. 2.3: site Melton Branch kilometer 1.8 is located ~50 m below the U.S. Geological Survey gage shown in Fig. 2.3; First Creek kilometer (FKC) 1.0 is located above the small pond on upper First Creek, ~200 m above site FCK 0.8 (Fig. 2.3); and site Fifth Creek kilometer (FFK) 1.1 is located on upper Fifth Creek ~100 m above site FFK 1.0 in Fig. 2.3.

[b]Site is dry in September; data shown are for November.

[c]NA = not available for September.

Based on the results of the discrete water sampling to date, the analyses of soluble metals will be conducted quarterly during 1987. The results of the analyses of suspended particulate matter for carbon and nitrogen have not yet been completed. For other parameters, the results for samples collected during the first several months will be used to determine the frequency with which a given parameter will be evaluated.

2.3 DESCRIPTION OF BIOLOGICAL STUDY SITES

Fifteen primary sites were chosen on WOC and selected tributaries above WOL (Fig. 2.3) for routine sampling of benthic invertebrates and fishes (Task 4 of BMAP, as described in Sect. 6). Criteria used in the selection of these sites included (1) point-source discharge locations; (2) area sources with confirmed seepage to surface waters; (3) known areas of impact, as determined from the 1985 synoptic survey (Loar et al. 1991); and (4) previous sampling locations (Loar et al. 1981a). Upstream reference (control) sites were located on WOC and each of the tributaries that receives effluent discharges, including Fifth Creek, First Creek, Melton Branch, and Northwest Tributary (Fig. 2.3).

For some tasks, an additional reference site was selected on Brushy Fork at Brushy Fork kilometer (BFK) 7.6 to provide a basis for comparison with the downstream sites on WOC. Brushy Fork is a fourth-order (4°) stream that drains a predominantly agricultural watershed just north of Oak Ridge (Fig. 2.1). No industrial development and only limited residential and commercial development occur in the watershed above the site. In addition to its size and land-use characteristics, Brushy Fork has relatively good water quality (McMaster 1967, Table 11; H. L. Boston, unpublished data) and a diverse assemblage of both benthic invertebrates and fish (J. M. Loar, unpublished data). Consequently, Brushy Fork is considered to be representative of preindustrial streams in this area.

The physical characteristics of the primary sampling sites on WOC and tributaries are summarized in Sect. 6.2.2.1. With the exception of First Creek, which is a large 1° stream, and the portion of WOC below the confluence with Melton Branch that becomes a 4° stream, all sites are located on either 2° or 3° streams. Fifth Creek, Northwest Tributary, upper Melton Branch, and upper WOC are all 2° streams: mean depths range from 5 to 12 cm, and mean widths, from 1.1 to 2.4 m. Melton Branch below the HFIR/TRU tributary and WOC from WCK 3.9 downstream to the confluence with Melton Branch are 3° streams. WOC is wider and deeper below ORNL than above it; channel modifications have been made to increase the flow rate through the central site area (Boyle et al. 1982). Moreover, from WCK 6.8 downstream to the confluence with Melton Branch, the substrate shifts from medium (fist-size) rubble, gravel, and sand with bedrock outcrops to smaller rubble and gravel (Loar et al. 1981a). The substrate in lower Melton Branch, on the other hand, is similar to that at WCK 6.8.

The rationale of including both downstream sites and an upstream reference site on WOC and each of the tributaries was followed in other tasks of BMAP (Loar et al. 1991). Subtask 1b, involving ambient toxicity testing, for example, includes 15 sites in WOC watershed (Sect. 3.1.3), and most of these overlap with the sites shown in Fig. 2.3. Likewise, Subtask 1c, periphyton monitoring, includes Brushy Fork and several sites on WOC and tributaries that are located in riffle areas near those that are sampled for benthic invertebrates (Sect. 3.2). Sampling sites included in the bioaccumulation monitoring in WOC watershed above WOL (Subtask 2a) coincided with the sampling sites in Task 3 (biological indicator studies). Because of the specific nature and objectives of Tasks 5–7 of BMAP, coordination of sampling locations between these predominantly radioecological tasks and those described in Tasks 1–4 was not required. Coordination of Tasks 5–7, however, was emphasized.

3. TOXICITY MONITORING

The toxicity monitoring task (Task 1) described in the BMAP for WOC watershed (Loar et al. 1991) outlined three goals, each of which was to be addressed by separate subtasks. The three goals were to (1) identify sources of toxicity in WOC watershed (Subtask 1a); (2) monitor toxicity of water and sediments in WOC and its tributaries and assess the usefulness of the toxicity test systems in detecting ambient toxicity (Subtask 1b); and (3) monitor periphyton/microbial communities and test, by manipulative field experiments, relationships between ambient toxicity and processes regulating energy flow in streams within WOC watershed (Subtask 1c). From March 1986 through January 1987, progress was made toward completing each of these subtasks. Results of Subtasks 1a and 1b are discussed in Sect. 3.1, and results of Subtask 1c are discussed in Sect. 3.2.

3.1 POINT-SOURCE AND AREA-SOURCE CONTRIBUTIONS TO AMBIENT TOXICITY

Point- and area-source contributions to ambient toxicity in streams within WOC watershed were evaluated with 7-d static-renewal toxicity tests based on the survival and growth of fathead minnow (*Pimephales promelas*) larvae and on the survival and reproduction of the daphnid *Ceriodaphnia dubia/affinis*. These two "minichronic" tests are described in detail in Horning and Weber (1985), Norberg and Mount (1985), and Mount and Norberg (1984).

Point-source discharges to WOC that were systematically evaluated for toxicity under the ORNL Toxicity Control and Monitoring Program (TCMP), as stipulated in the NPDES permit (EPA 1986, Part V), included those from the Sewage Treatment Plant, Process Waste Treatment Plant, and Coal Yard Runoff Treatment Facility (Fig. 2.5). Effluents from these three treatment facilities were tested according to the schedule shown in Table 3.1; 24-h composite water samples were normally used for these tests.

Contributions to ambient toxicity from area sources were evaluated by monthly tests of water from 15 sites on 5 streams (Fig. 3.1). Four of the 15 sites (upstream sites on First Creek, Fifth Creek, WOC, and Melton Branch) served as reference sites because they were presumed to lack contaminants in toxic concentrations. Samples to be evaluated for ambient toxicity in each 7-d test were normally collected as daily grabs.

Ambient toxicity monitoring sites 6 and 7 (Fig. 3.1) are also listed as sites to be evaluated under the TCMP section of the NPDES permit. To make testing under the TCMP portion of the permit as consistent as possible, tests in June and August for sites 6 and 7 (NPDES sites X13 and X14, respectively) were performed on 24-h composite water samples instead of on grab samples (A. J. Stewart and L. F. Wicker, ORNL, personal communication to J. B. Murphy, ORNL, 1987).

Tests with the two species were always conducted concurrently. On each day of every test, subsamples of each effluent or water sample were routinely analyzed for pH, conductivity, alkalinity, and water hardness. Most samples collected for toxicity testing were

Table 3.1. Toxicity test results for effluents discharged from the ORNL Process Waste Treatment Plant (PWTP), Sewage Treatment Plant (STP), and Coal Yard Runoff Treatment Facility (CYRTF)[a]

Facility	Test period	IWC (%)	NOEC (% of full strength)	
			Fathead minnows	*Ceriodaphnia*
PWTP	10–17 July	21.9	80	80
PWTP	4–11 Sept.	21.9	80	80
PWTP	13–20 Nov.	21.9	80	80
PWTP	22–29 Jan.	21.9	≥100	80
STP	25 May–5 June	25.7	≥100	20
STP	20–26 June	25.7	NT[b]	≥100
STP	22–29 Aug.	25.7	≥100	≥100
STP	16–23 Oct.	25.7	≥100	≥100
CYRTF	10–17 July	3.6	20	<10
CYRTF	4–11 Sept.	3.6	≥60	5
CYRTF	30 Oct.–6 Nov.	3.6	≥80	NT[b]
CYRTF	22–29 Jan.	3.6	60	3

[a]The Instream Waste Concentration (IWC) for effluent from each facility is the percent of the total streamflow at the point of discharge that is composed of effluent. The no-observed-effect concentration (NOEC) designates the highest tested concentration (in percent of full strength) of the effluent causing no significant reduction in survival or growth of fathead minnow larvae or in survival or reproduction of *Ceriodaphnia*. Tests of effluent from PWTP and CYRTF in January were done in 1987; all other tests were done in 1986.

[b]NT = not tested.

also routinely analyzed for free and total residual chlorine. The temperature of the water at each site was recorded at the time of sample collection.

Summary statistical computations [means, standard deviations, coefficient of variation (C.V.) and analysis of variance (ANOVA) general linear model (GLM) procedures] were accomplished with the use of SAS®. Correlations (Spearman rank order and product-moment) were determined with the use of StatSoft® software and an IBM-AT computer. In all cases, survival values for fathead minnow larvae and for *Ceriodaphnia* were transformed (arcsine square root; Steel and Torrie 1960) before being analyzed statistically.

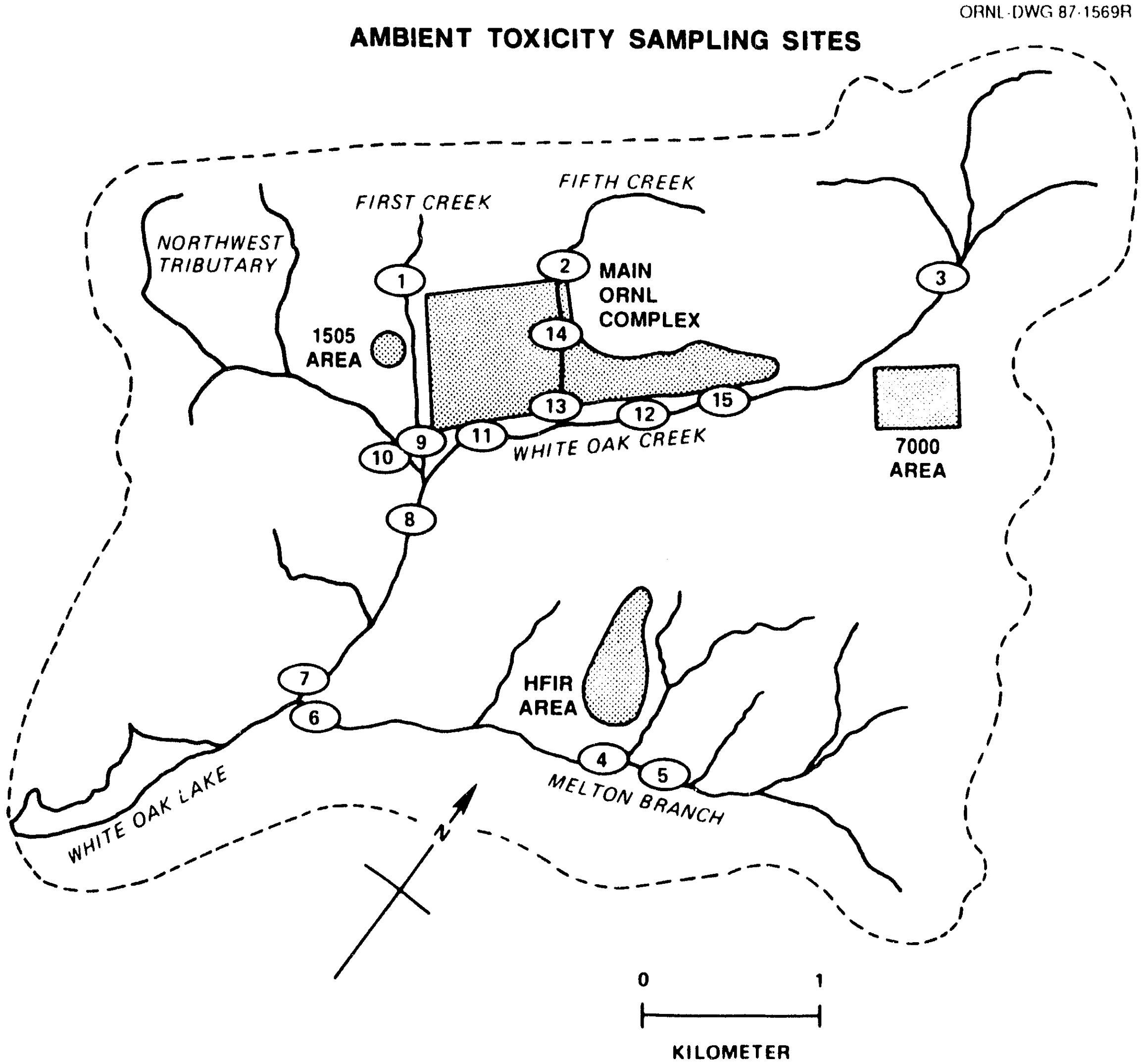

Fig. 3.1. Schematic of streams within White Oak Creek watershed, showing the locations of the 15 monitoring sites from which samples were taken for assessments of ambient toxicity. Stippling designates developed areas, which are major regions of point- and area-source discharges to streams. The dashed line shows the approximate boundaries of the watershed. HFIR = High Flux Isotope Reactor; ORNL = Oak Ridge National Laboratory.

3.1.1 Toxicity of Point-Source Discharges to White Oak Creek

Discharges from the ORNL Sewage Treatment Plant and Process Waste Treatment Plant contribute significantly to the flow of WOC (Table 2.1), but generally did not appear very toxic to *Ceriodaphnia* or to fathead minnow larvae (Table 3.1). Fathead minnow larvae and *Ceriodaphnia* reared in full-strength effluent from the Sewage Treatment Plant, for example, almost always performed as well as and, in some cases, better than the controls, which were consistently acceptable (Horning and Weber 1985). Effluent from the Process Waste Treatment Plant was consistently toxic at full strength to both species, but was not toxic to either species at a concentration of 80% full strength. Effluent from the Coal Yard Runoff Treatment Facility, in contrast, is a smaller fraction of the flow in WOC (Table 2.1), but was generally more toxic to *Ceriodaphnia* and fathead larvae (Table 3.1). *Ceriodaphnia* appeared to be more sensitive than fathead minnow larvae to effluent from this facility.

In general, water quality measurements showed that effluents from the three treatment facilities were chemically distinct and that major chemical constituents of effluent from each facility did not vary much through time (Table 3.2). Most notably, compared with water in WOC, effluent from the Process Waste Treatment Plant had elevated conductivity and very low levels of hardness, whereas effluent from the Sewage Treatment Plant had intermediate levels of conductivity, alkalinity, and hardness (Sect 3.1.3.4). The Coal Yard Runoff Treatment Facility generates effluent that is very high in conductivity and hardness but that contains negligible amounts of alkalinity (Table 3.2).

3.1.2 Point-Source Inputs of Chlorine and Phosphorus

Chemical analyses performed in conjunction with toxicity tests used to evaluate area-source contributions to ambient toxicity also identified a previously unknown point source of chlorine in upper reaches of Fifth Creek and a point-source input of phosphorus from the 7000 area. The chlorine inputs to upper Fifth Creek, first identified in April 1986, are presumed to originate from a leaking water main (R. K. Owenby, ORNL Department of Environmental Management, personal communication, 1986). Concentrations of total residual and free chlorine in upper reaches of Fifth Creek in April averaged about 0.13 and 0.10 mg/L respectively.

Measurements of SRP at various sites in WOC during April 25–30, 1986, showed that a small tributary with headwaters near Building 7001 was highly enriched with this nutrient (SRP concentrations ≥ 500 μg/L). In WOC upstream from Bethel Valley Road, SRP was less than 10 μg/L; in WOC east of Haw Ridge Road just downstream from the phosphorus-enriched tributary, the concentration of SRP was 184 μg/L. These data suggest that inputs of SRP from the 7000 area may be a significant source of nutrients to downstream reaches of WOC.

3.1.3 Toxicity of Area-Source Inputs to ORNL Streams

Samples from most of the 15 monitoring sites on 5 streams in WOC watershed (Fig. 3.1) were tested 10 times for toxicity according to the schedule shown in Table 3.3. The monitoring site farthest upstream on Melton Branch was dry from July through October (Sect. 2.1) and consequently was tested only six times. Additionally, sites 1, 10, and 15 (Fig. 3.1) were not tested with *Ceriodaphnia* in November due to inadequate numbers of

Table 3.2. Chemical characteristics of effluents from the ORNL Process Waste Treatment Plant (PWTP), Sewage Treatment Plant (STP), and Coal Yard Runoff Treatment Facility (CYRTF)[a]

Facility	Test period	pH	Conductivity[b]	Alkalinity[c]	Hardness[c]
PWTP	10-17 July	7.81	828	80.0	19.6
PWTP	4-11 Sept.	7.88	827	65.3	20.3
PWTP	13-20 Nov.	7.47	711	51.4	0.0
PWTP	22-29 Jan.	7.47	544	40.9	2.6
STP	25 May-5 June	7.91	366	90.1	151.1
STP	20-26 June	8.02	385	82.9	152.7
STP	22-29 Aug.	7.74	381	74.1	174.0
STP	16-23 Oct.	8.13	403	108.7	166.1
CYRTF	10-17 July	7.19	1810	37.1	1160.0
CYRTF	4-11 Sept.	7.19	2883	9.0	2500.0
CYRTF	30 Oct.-6 Nov.	7.23	1468	8.3	1023.1
CYRTF	22-29 Jan.	6.70	1404	5.6	910.0

[a]Data for each parameter are expressed as means for daily samples taken during the indicated 7-d periods. Analyses of effluents from PWTF and CYRTF in January were done in 1987; all other analyses were done in 1986.

[b]In microsiemens per centimeter, corrected to 25°C.

[c]In milligrams of $CaCO_3$ per liter.

animals of the proper age. The 15 monitoring sites used to determine patterns of ambient toxicity were selected to reflect characteristics of major reaches of the 5 streams, with emphasis (6 of 15 sites) on WOC.

3.1.3.1 Results of ambient toxicity tests: Fathead minnow larvae

The results of ambient toxicity tests, based on growth of fathead minnow larvae, are summarized in Fig. 3.2. This figure shows growth of the larvae at the 15 sites, relative to controls, averaged across the 10 test periods. On average, growth of the larvae was at least 90% that of controls for 10 of the 15 sites. A considerable amount of variation was present, however, particularly between tests (Table 3.3). This variation obscured possible trends in growth reductions attributable to the presence of sublethal concentrations of toxic materials. Fish reared in water from the reference site in First Creek (site 1; Fig. 3.1), from sites 6 and 4 on Melton Branch, from site 9 on First Creek, and from site 13 on lower Fifth Creek all tended to grow less than fish in the controls, but it remains unclear whether these divergences are of biological significance. Over sites for which data from nine tests were available (no

Table 3.3. Growth (change in dry weight over a 7-d period) and survival of fathead minnow larvae in controls used in ambient toxicity tests of stream water from 15 sites in White Oak Creek, First Creek, Fifth Creek, Northwest Tributary, and Melton Branch[a]

Test	Date	Growth (mean ± 1 s.d.)	Survival (%)
1	27 March 1986	0.640 ± 0.085	97.5
2	24 April 1986	0.654 ± 0.105	95.0
3	29 May 1986	0.447 ± 0.025	100.0
4	26 June 1986	0.277 ± 0.006	92.5
5	1 August 1986	—	95.0
6	29 August 1986	0.586 ± 0.077	100.0
7	2 October 1986	0.371 ± 0.059	92.5
8	23 October 1986	0.357 ± 0.056	97.5
9	20 November 1986	0.498 ± 0.093	97.5
10	16 January 1987	0.410 ± 0.060	100.0

[a]Date indicates the date that a given 7-d test was started. Growth values are milligrams per surviving fish (mean ± 1 s.d.); survival values are mean percentages. In each test, percent survival is the geometric mean of four replicates, each of which contained ten larvae.

weight data are available for any sites tested in July), the total range in mean growth (corrected for test-to-test differences in weight of larvae at the start of a test) tended to be small (from 0.37 mg per fish for site 6 to 0.45 mg per fish for site 11) and not much different from the average growth for the controls (0.47 mg per fish). These data do not argue strongly for the usefulness of growth data as definitive endpoints for the fathead minnow test in determining chronic toxicity of streams in WOC watershed.

Compared with the growth endpoint, the survival endpoint of the fathead minnow test was more definitive (Fig. 3.3). The overall range in survival was fairly large (from 60.7 to 98.1%), and survival in controls was, on average, high (99.3%) and had a coefficient of variation of only 4.7%. Based on all ten tests, the amount of variation in survival (expressed as the coefficient of variation) for arcsine square root-transformed values at each site was negatively correlated with mean survival ($r = -0.758$, $n = 15$; $p = 0.0013$).

Frequency distributions of survival for the six sites where the coefficient of variation of minnow survival was highest (range = 24.1 to 61.5%; mean = 41.6%) were compared with those for the six sites where it was lowest (range = 8.9 to 16.8%; mean = 11.6%). This comparison showed that only about 10% of all observations in the "low-C.V." sites (sites 1, 3, 7, 8, 9 and 11; Fig. 3.1) had survival values that were less than or equal to 50%, whereas 30% of all observations in the "high-C.V." sites (sites 2, 4, 6, 12, 13, and 14; Fig. 3.1) had survival values that were less than or equal to 50%. Hence, sites having lower average

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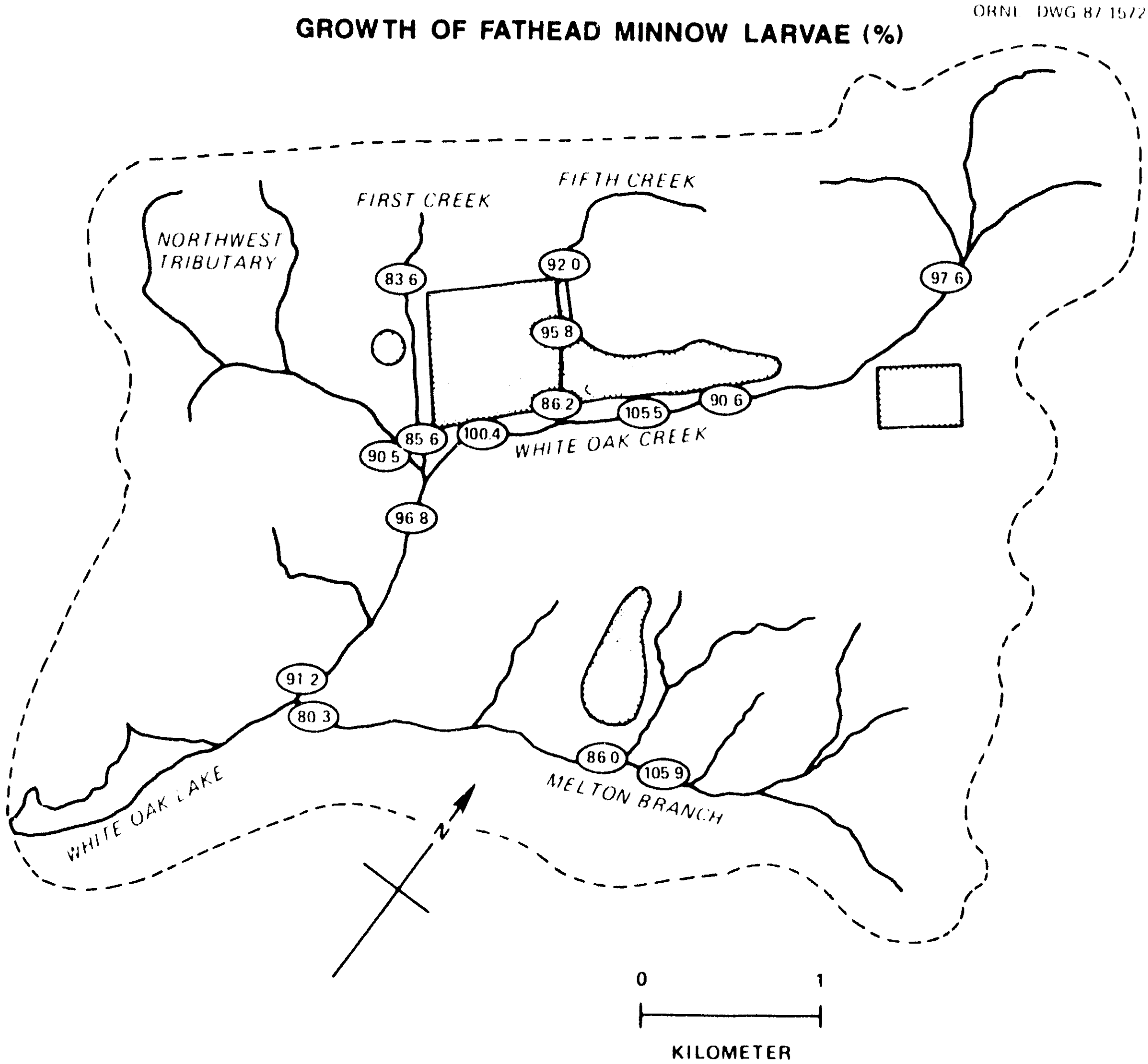

Fig. 3.2. Mean growth of fathead minnow larvae (as a percent of the controls), averaged for all dates at sites monitored for ambient toxicity.

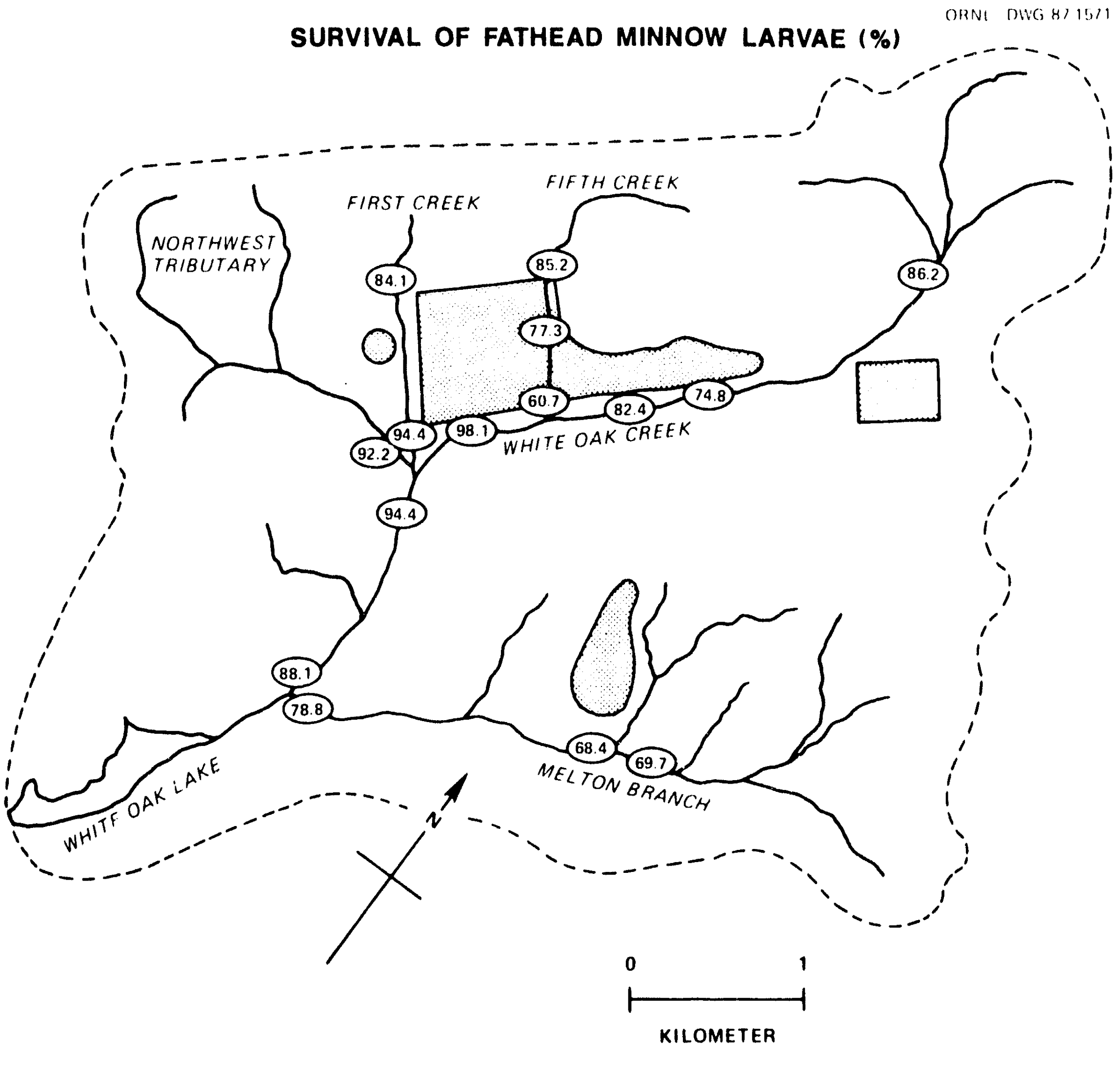

Fig. 3.3. Survival of fathead minnow larvae (as percent, mean for all tests) in samples collected from sites monitored for ambient toxicity.

survival values for fathead minnow larvae tended to have water that was sometimes highly toxic, and sometimes not, whereas sites with high average survival values were much more consistently of higher water quality.

Over the 15 sites, mean percentage survival of the fathead minnow larvae was poorly correlated with growth ($r = 0.159$, $p = 0.577$). Additionally, the relationship was not improved by ranking the sites with respect to these two endpoints (Spearman $R = 0.143$, $p = 0.599$).

3.1.3.2 Results of ambient toxicity tests: *Ceriodaphnia*

Results of the *Ceriodaphnia* tests showed that animals in tests 5, 6, and 9 tended to have low fecundity (the number of offspring per female surviving all 7 d of a test) relative to other tests (Table 3.4). Fecundity was low for all animals in tests 5 and 6 (9.0 and 13.0 offspring per female in controls, respectively), suggesting that insufficient food or suboptimal incubation conditions affected the outcomes of these tests. Fecundity of animals in the control was low in test 9 (6.7 offspring per female), but was high for animals reared in water from nearly all of the monitoring sites. These data suggest that, in test 9, the reconstituted hard water used for rearing the control animals was problematic but that the incubation and food regimes were otherwise acceptable. This interpretation served as justification for excluding data obtained in *Ceriodaphnia* tests 5 and 6, and not those obtained in test 9, from all subsequent analyses.

The range in mean survival of *Ceriodaphnia* over the 15 sites was slightly greater than that for the minnows (43.8 to 98.1% for *Ceriodaphnia* vs 60.7 to 98.1% for the minnows) (Figs. 3.3 and 3.4). Mean survival of *Ceriodaphnia* at sites 4, 11, 12, 13 and 14 was low relative to survival at other sites, and ANOVA (SAS-GLM) showed that differences in survival of *Ceriodaphnia* between sites were substantially greater than differences between tests ($p = 0.0002$ vs $p = 0.1343$, respectively; Table 3.5). As with the minnow test, the range of within-site variability of *Ceriodaphnia* survival (expressed as coefficient of variation) was large (from 13.5 to 87.0%) and, across the 15 sites, was related significantly and inversely to mean survival ($r = -0.969$; $p < 0.0001$).

Averaged over the eight valid tests, mean fecundity of *Ceriodaphnia* ranged from 18.7 (site 11) to 27.2 offspring per female (site 10) (Table 3.6). An evaluation of the pattern of *Ceriodaphnia* reproduction over all site-date combinations by ANOVA, however, showed that within-test variability was high and that, on average, variation in fecundity of animals between tests exceeded the variation in fecundity between sites (Table 3.5). Because 10 of the 15 sites appeared so similar in their mean fecundity values (between 22.0 and 24.0 offspring per female; Table 3.6), statistical analyses based on site rankings for fecundity seemed inappropriate. However, based on the fecundity endpoint, two sites were noteworthy: site 10 (downstream on Northwest Tributary), which supported on average the highest levels of fecundity, and site 11 (WOC 3.9; Fig. 3.1 and Table 3.6), where average fecundity was lowest.

3.1.3.3 Results of ambient toxicity tests: Endpoint comparisons

The correlation between the survival endpoints of the fathead minnow and the *Ceriodaphnia* tests across the 15 sites was low ($r = 0.212$; $p = 0.454$). When the sites were retested after being ranked according to their mean survival values for fathead minnows and

Table 3.4. Fecundity of *Ceriodaphnia* in 7-d chronic toxicity tests of water from 15 stream sites in White Oak Creek watershed[a]

Test	Sampling sites: Fecundity (mean ± 1 s.e.)	Sampling sites: C.V. (%)	Controls: Fecundity (mean ± 1 s.e.)	Controls: C.V. (%)
1	17.35 ± 0.44 (101)	25.3	22.89 ± 1.55 (9)	20.3
2	25.56 ± 0.73 (95)	28.0	25.44 ± 3.60 (9)	42.4
3	20.18 ± 0.61 (77)	26.6	22.13 ± 1.63 (8)	20.8
4	22.37 ± 0.89 (73)	34.1	24.83 ± 1.58 (6)	15.6
5	5.80 ± 0.33 (93)	54.6	9.00 ± 2.73 (7)	80.1
6	9.30 ± 0.37 (103)	40.4	13.00 ± 0.71 (8)	15.4
7	21.02 ± 0.80 (103)	38.7	24.75 ± 1.33 (8)	15.2
8	22.97 ± 0.63 (104)	28.0	27.71 ± 1.58 (7)	15.1
9	28.13 ± 0.57 (101)	20.5	6.70 ± 1.54 (10)	72.4
10	26.38 ± 0.51 (122)	21.3	26.80 ± 0.89 (10)	10.5

[a]Data are for tests conducted from March 1986 through January 1987; test numbers coincide with dates indicated in Table 3.3. Fecundity values (the number of offspring per female that survived all 7 d of a test) are means for all sites combined (left column) and for controls (right column). The numbers in parentheses are the numbers of females used in the computation of each fecundity value. The coefficient of variation (C.V.) provides an index of the variability of fecundity from test to test.

Ceriodaphnia, the correspondence was not improved (Spearman $R = 0.261, p = 0.331$). This finding suggested that one or several sites were relatively unfavorable to one species but not to the other. To identify such sites, the overall mean percentage survival for each site was plotted with respect to both species (Fig. 3.5). This graph revealed a cluster of seven sites that supported good (greater than ~85%) survival for both species; a group of four sites that had low (less than ~85%) survivorship for both species (sites 4, 12, 13, and 14; Fig. 3.1); three sites where survivorship of *Ceriodaphnia* was high (>90%) but survivorship of fathead minnow larvae was relatively low (<80%); and one site (11) where survivorship of the fish was high but survival of *Ceriodaphnia* was low. The reduction in water quality of WOC, Melton Branch, and Fifth Creek, and the subsequent recovery of this water in lower Melton Branch and in WOC below the confluence with Northwest Tributary and First Creek (Fig. 3.1), is shown by the two-species toxicity trajectories for these streams (Fig. 3.5).

Although correspondence in patterns of percent survival for the 2 species across the 15 monitoring sites was not significant, the distributions of *variation* in survival (as coefficient of variation) of the minnow larvae and the *Ceriodaphnia* for the sites were in good agreement when site rankings were used ($r = 0.670; p = 0.006$). This finding suggests that, in general, ambient toxicity patterns in streams in WOC watershed may be determined more by episodic or periodic events, such as intermittent discharges or inadvertent spills, than by wastewater treatment facilities that discharge effluent continuously. The relatively low toxicities of

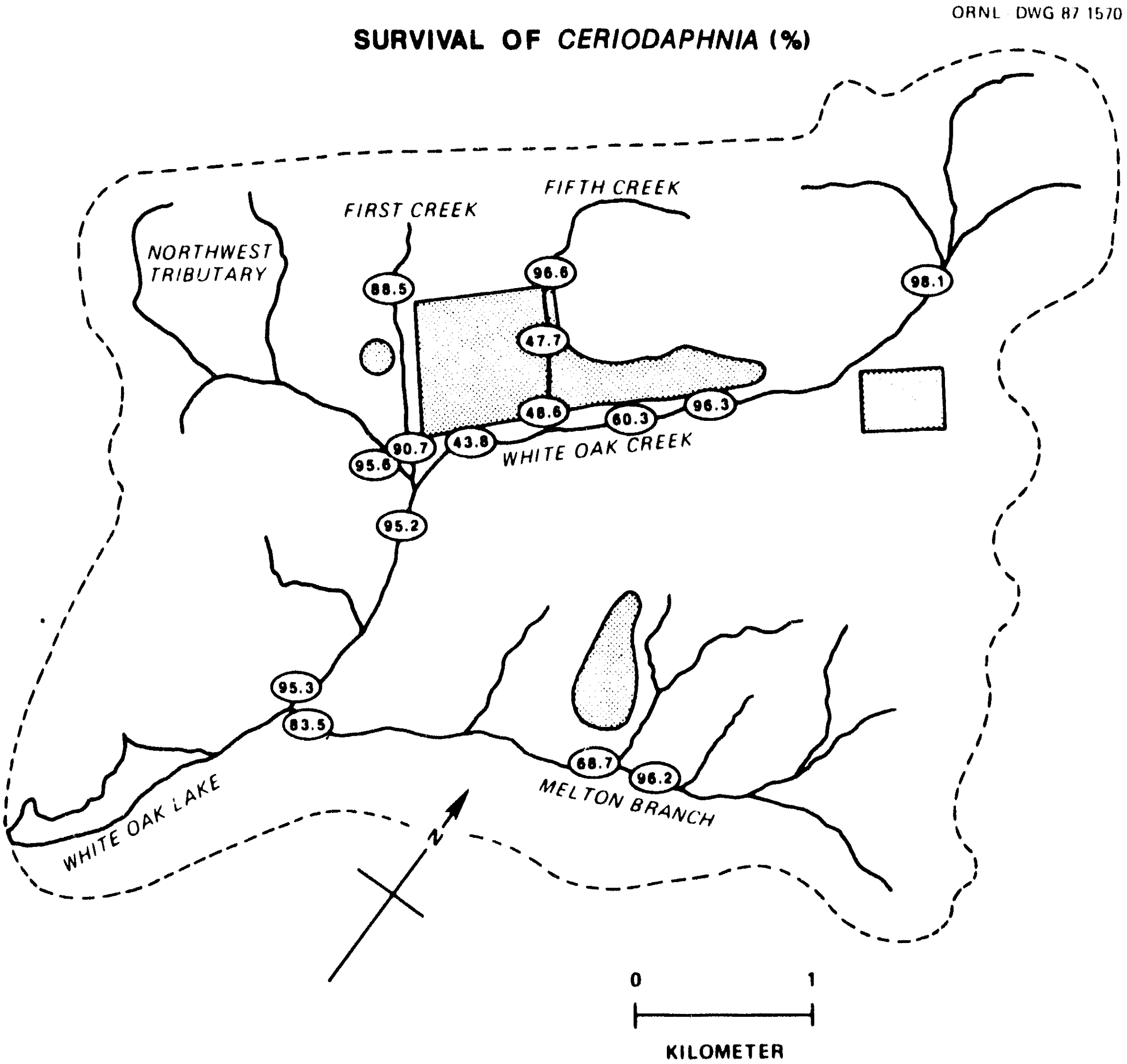

Fig. 3.4. Survival of *Ceriodaphnia* (as percent, mean for all tests) in samples collected from sites monitored for ambient toxicity.

Table 3.5. Analysis of variance of *Ceriodaphnia* survival and fecundity[a]

Source	D.F.	Sum of squares	Mean square	F-ratio	Level of significance (p value)
			Survival		
Model	21	30,485.99	1,451.71	2.70	0.0006
Error	91	49,013.09	538.61	—	—
Site	14	25,463.82	—	3.38	0.0002
Test	7	6,181.90	—	1.64	0.1343
Total	112	79,499.08	—	—	—
			Fecundity		
Model	21	10,194.21	485.44	13.20	0.0001
Error	652	23,972.40	36.77	—	—
Site	14	2,750.09	—	5.34	0.0001
Test	7	8,194.88	—	31.84	0.0001
Total	673	34,166.61	—	—	—

[a]Data from tests 5 and 6 (in August and September) are not included in these analyses due to unacceptably low fecundity of animals in controls and at all sites (note Table 3.4). Site and test subcategories indicate type III sums of squares.

effluents tested under the ORNL TCMP (A. J. Stewart and L. F. Wicker, ORNL, personal communication to J. B. Murphy, ORNL, 1987; Table 3.1) tend to substantiate this interpretation. An exception to this generalization occurred at site 2, which, despite its location upstream from most ORNL operations, had a moderate amount of variability in fathead minnow survival (C.V. = 28.2%).

3.1.3.4 Results of ambient toxicity tests: Chemical analyses

The frequency of sampling (daily for 7 d for each of 10 tests); the number of monitoring sites (15); and the number of water quality parameters measured (temperature, pH, conductivity, alkalinity, hardness, total residual chlorine, and free chlorine) generated a large data set. These data ultimately will be used to characterize the monitoring sites by (1) parameters measured (e.g., ratio of alkalinity to hardness and ratio of conductivity to alkalinity) and (2) variability characteristics for combinations of the different parameters (e.g., site-to-site differences in the degree of skewness of each factor's frequency distribution). In this report, only the results of summary statistical analyses of the chemical data are provided; more-detailed analyses of the chemical data will be given in subsequent reports.

Mean values for pH, alkalinity, conductivity, and hardness from March through November 1986 are given in Table 3.7 for the monitoring sites shown in Fig. 3.1. Even cursory inspection of these data reveals a number of noteworthy points. First, downstream sites on Melton Branch were atypical with respect to pH, conductivity, alkalinity, and hardness. The high conductivity and hardness values for these sites, together with low values

Table 3.6. Fecundity of *Ceriodaphnia* in 7-d chronic toxicity tests of water from 15 stream sites in White Oak Creek watershed[a]

Site	Number of tests	Offspring per female (mean ± 1 s.e.)	C.V. (%)
1	7	22.67 ± 1.17 (48)	38.5
2	8	21.85 ± 0.74 (68)	28.0
3	8	22.88 ± 0.82 (58)	27.3
4	8	22.98 ± 0.89 (47)	26.5
5	4	23.06 ± 1.21 (33)	30.3
6	8	23.81 ± 0.94 (62)	30.9
7	8	24.80 ± 0.74 (64)	24.0
8	8	23.03 ± 0.67 (69)	24.2
9	8	22.52 ± 0.85 (62)	29.6
10	7	27.18 ± 1.11 (57)	30.8
11	8	18.68 ± 1.33 (34)	41.4
12	8	23.87 ± 1.63 (39)	42.6
13	8	20.61 ± 1.44 (38)	43.0
14	8	23.42 ± 0.86 (42)	23.7
15	7	23.78 ± 0.92 (55)	28.6
Control	8	22.21 ± 1.05 (67)	38.7

[a]Data are for tests conducted from March 1986 through January 1987. Site numbers correspond to those shown in Fig. 3.1. Fecundity values (the number of offspring per female that survived all 7 d of a test) are means for all tests. The values in parentheses are the numbers of females used in the computation of fecundity. The coefficient of variation (C.V.) provides an index of the variability of fecundity from site to site. Control values are for animals reared in hard reconstituted water.

for alkalinity and pH, suggested that acidification and inputs of calcium and/or magnesium sulfates from operations at the HFIR site controlled major aspects of water quality in this stream. This supposition was substantiated, in part, by water quality analyses conducted in September 1986 (Sect. 2.2.4). The concentrations of calcium (140 mg/L) and magnesium (31 mg/L) in lower Melton Branch (MEK 0.6) were much higher than the concentrations in upstream reference sites (Table 2.5). Second, comparisons of changes in major water quality factors over similar distances of various streams can provide information about possible toxicity sources and the extent of water quality degradation. For example, on average, over an ~1-km reach of First Creek, pH remained unchanged, alkalinity declined by 1.5 mg/L, conductivity increased by 60 μS, and hardness increased by 10.3 mg/L. Over a similar distance on Fifth Creek, average pH increased by 0.36 units, alkalinity declined by 11.2 mg/L, conductivity increased by 112 μS, and hardness increased by 35.9 mg/L. Large shifts in ionic

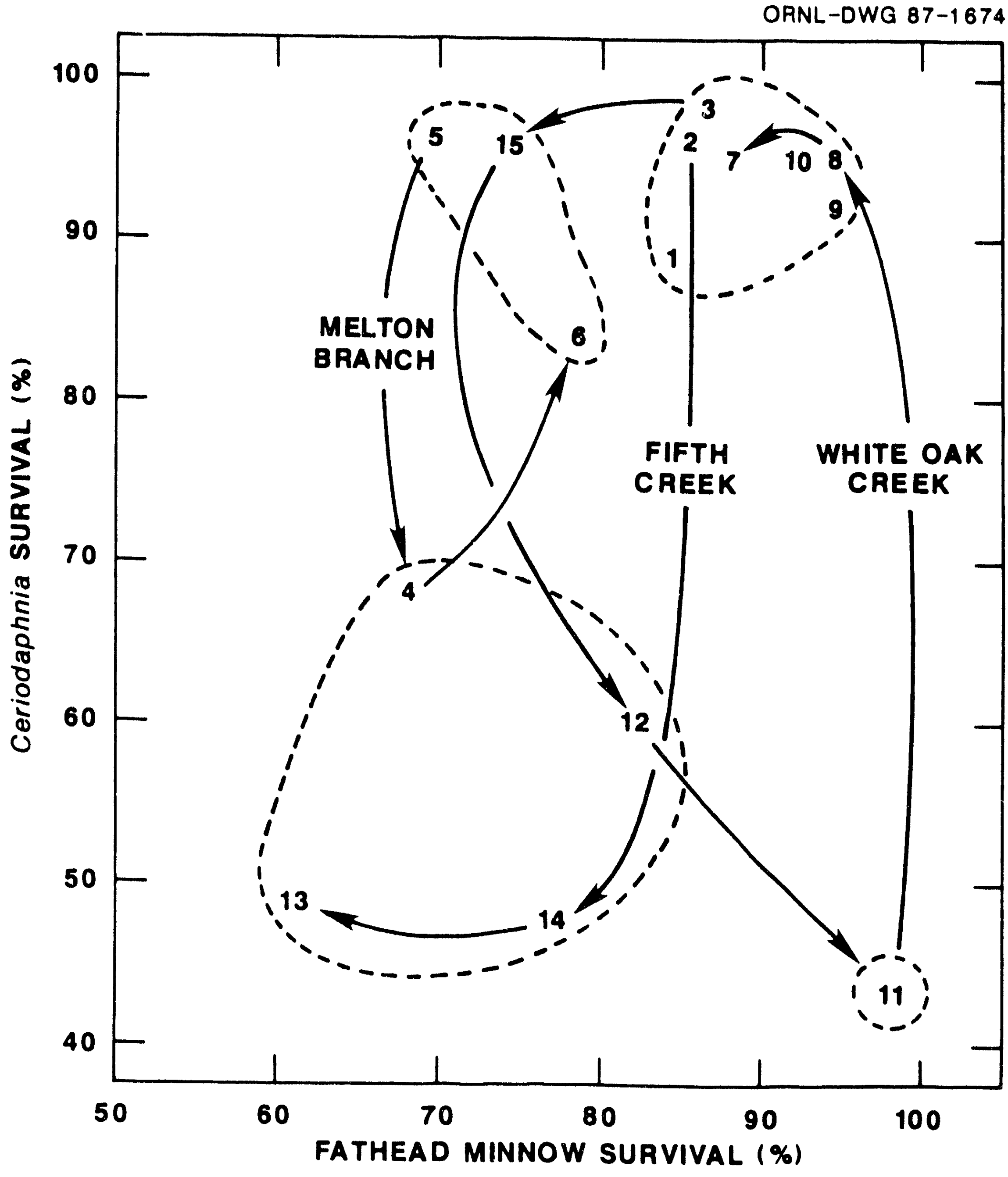

Fig. 3.5. Two-species toxicity trajectories for sampling sites in Fifth Creek, Melton Branch, and White Oak Creek. Numbers indicate sampling sites (as for Fig. 3.1). Arrows indicate direction of water flow (upstream to downstream) in each stream. Dashed lines around groups of sites are used to show relative similarity of some sites (e.g., sites 5, 15, and 6) with respect to mean toxicity to fathead minnow larvae (horizontal axis) and *Ceriodaphnia* (vertical axis).

Table 3.7. Mean (± 1 s.d.) pH, alkalinity, conductivity, and hardness of water from monitoring sites in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary[a]

Site	N	pH	Alkalinity[b]	Conductivity[c]	Hardness[b]
1	60–63	8.21 ± 0.16	139.8 ± 21.8	242 ± 38	148.9 ± 24.1
2	60–63	7.94 ± 0.23	132.2 ± 14.7	247 ± 28	143.5 ± 19.1
3	59–63	8.23 ± 0.16	129.2 ± 28.0	234 ± 47	137.4 ± 31.6
4	57–62	7.76 ± 0.29	45.1 ± 28.6	855 ± 236	473.5 ± 163.5
5	29–32	8.17 ± 0.14	136.3 ± 28.8	289 ± 47	155.5 ± 31.8
6	60–63	7.90 ± 0.25	79.7 ± 30.9	727 ± 228	393.0 ± 137.2
7	60–63	8.19 ± 0.11	112.9 ± 12.2	369 ± 56	162.0 ± 20.8
8	60–63	8.15 ± 0.11	113.6 ± 11.1	364 ± 74	162.4 ± 25.5
9	61–64	8.21 ± 0.11	138.3 ± 17.1	302 ± 32	159.2 ± 19.7
10	60–63	8.26 ± 0.16	121.4 ± 15.8	285 ± 29	148.5 ± 15.2
11	57–61	8.21 ± 0.16	113.2 ± 11.7	346 ± 86	155.5 ± 35.8
12	59–62	8.15 ± 0.16	122.9 ± 17.2	294 ± 35	149.6 ± 19.9
13	60–63	8.30 ± 0.17	121.0 ± 9.5	359 ± 69	179.4 ± 27.3
14	59–62	8.25 ± 0.10	125.1 ± 9.0	374 ± 68	196.0 ± 33.3
15	60–63	8.19 ± 0.11	155.3 ± 34.3	366 ± 80	185.1 ± 43.8

[a]Site codes correspond to those shown in Fig. 3.1. The number of observations (N) used to compute the mean and standard deviation for the water quality parameters at each site is given as a range because the number differed slightly from parameter to parameter due to intermittent analytical errors that required the culling of some values. Data are for tests 1 through 9 (March through November 1986).

[b]In milligrams of $CaCO_3$ per liter.

[c]In microsiemens per centimeter, corrected to 25°C.

balance of water in Fifth Creek, relative to those in First Creek, were also evident in comparing downstream changes in the ratios of key water quality parameters for these two streams. In upper First Creek, for example, the ratio of hardness to alkalinity averaged 0.94. Farther downstream 1 km, this ratio declined to 0.87. Over a similar distance in Fifth Creek, the ratio of hardness to alkalinity dropped from 0.92 to 0.67. Over the entire five-site reach on WOC (a distance of about 4.8 km; Fig. 3.1), the hardness:alkalinity ratio declined from 0.94 to 0.70.

Although measurements of free chlorine were routinely made, logistics of sample collection seriously compromised the quality of data obtained for this toxicant. In some cases several hours elapsed between the time a sample was collected and the time it was analyzed. Levels of free chlorine can drop drastically in this length of time, and the rate of decline is affected by the temperature of the water. Hence, the concentration of free chlorine that was measured in the laboratory almost certainly underestimated to an unknown extent the concentration that was actually present at the time the sample was collected. Measurements of total residual chlorine are less subject to such errors. Therefore, concentrations of total residual chlorine across the suite of monitoring sites are presumed to reflect the probable distributions of free chlorine, which is highly toxic to fish and daphnids.

Average concentrations of total residual chlorine for all the monitoring sites are shown in Fig. 3.6. Total residual chlorine was never detected in the upstream reference sites in First Creek, WOC, or Melton Branch, but concentrations averaged 4.8 μg/L in the Fifth Creek reference site (Fig. 3.6). This average was biased, however, because the monitoring site was moved ~50 m upstream when chlorine inputs to this stream site from a leaking tap water main were first reported in April (Sect. 3.1.2). The highest concentrations of free and total residual chlorine noted for upper Fifth Creek before relocating the monitoring site were 40 and 90 μg/L, respectively. When the monitoring site in upper Fifth Creek was relocated farther upstream, free and total residual chlorine were no longer detected.

Figure 3.6 shows clearly that most of Fifth Creek and a three-site midreach section of WOC receive substantial inputs of chlorine. The three sites with the highest average values of total residual chlorine (sites 13 and 14 in Fifth Creek and site 11 in WOC) were also relatively toxic to *Ceriodaphnia* (Fig. 3.5).

For the five sites with the highest average concentrations of total residual chlorine, average total residual chlorine concentrations ranged from 27 to 87 μg/L. The coefficients of variation for total residual chlorine concentrations at the five sites most affected by chlorine were generally high (85.5% for site 8 and 83.6% for site 11 on WOC, 72.7 and 66.3% for the two midreach sites on Fifth Creek, and 178.6% for site 12 on WOC just upstream from Fifth Creek). The variance patterns for total residual chlorine (and presumably free chlorine), therefore, generally coincided with the variance patterns noted for the survival endpoints for the toxicity tests with fathead minnows and *Ceriodaphnia*. This correspondence suggests that episodic or periodic, rather than continuous, discharges of chlorine dominate ambient toxicity patterns and chlorine dynamics in midreach segments of Fifth Creek and WOC.

3.1.4 Discussion

Toxicity tests to monitor biological quality of ambient waters are being incorporated with increasing frequency into NPDES permits, but broad-based field studies to validate the utility of such approaches remain relatively rare (Roop and Hunsaker 1985). The data presented here (1) provide information regarding spatiotemporal distributions of ambient toxicity in streams within WOC watershed, and (2) highlight the useful and limiting aspects of the "minichronic" fathead minnow and *Ceriodaphnia* test systems used to quantify ambient toxicity.

In WOC watershed, the minichronic biotests of two species (*Pimephales promelas* and *Ceriodaphnia dubia/affinis*) implicate episodic releases, rather than continuous discharges, as controlling overall patterns of ambient toxicity. This view is substantiated by (1) the generally low toxicity of effluents from key wastewater treatment systems, (2) the general lack of concordance between patterns of survivorship and growth (for fathead minnow larvae) and survivorship and fecundity (for *Ceriodaphnia*) endpoints across sites and over time, and (3) the concordance in patterns of variation in survival between *Ceriodaphnia* and fathead minnow larvae. In the context of the third substantiating point, variance patterns in

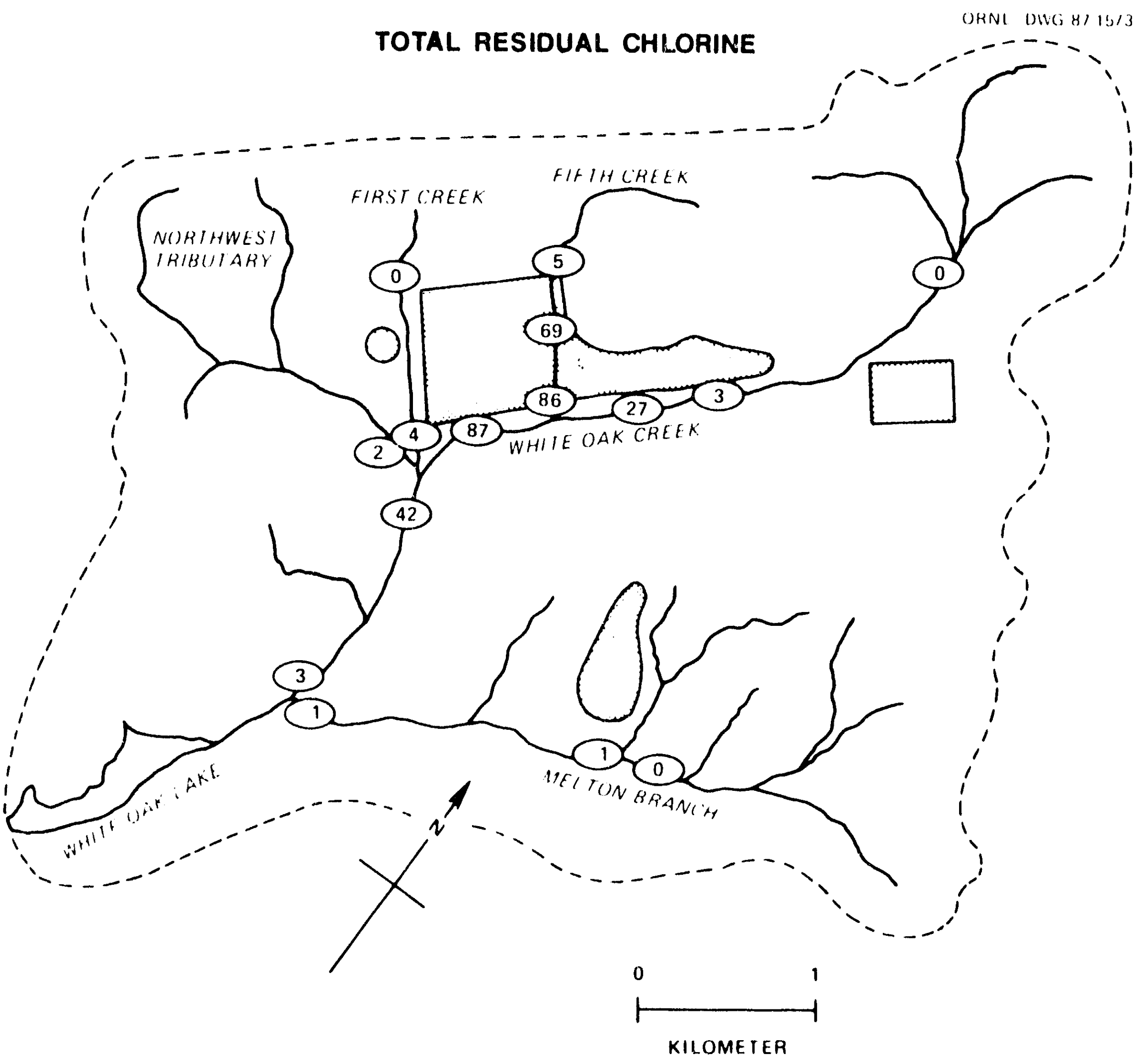

Fig. 3.6. Concentrations of total residual chlorine for the 15 monitoring sites (as for Fig. 3.1). Values are micrograms per liter (means for all tests).

the distribution of total residual chlorine and toxicity to both test species suggest that chlorine (free, total residual, or both) is, at least in Fifth Creek and in midreach sections of WOC, closely linked to the patterns of ambient toxicity. Ambient toxicity patterns in Melton Branch were clearly not controlled by chlorine, but variance patterns in the biotests for sites in downstream reaches of this stream still implicated the importance of episodic releases. Large episodic changes in concentrations of relatively innocuous chemical constituents (sulfates, calcium, magnesium, etc.) may regulate toxicity patterns in Melton Branch. With the recent shutdown of the HFIR (Fig. 2.7), it should be possible over the next several months to evaluate this hypothesis.

The reproductive endpoint of the 7-d *Ceriodaphnia* biotest is generally presumed to reflect physiological responses to sublethal concentrations of toxicants and consequently is considered to be a more sensitive indicator of water quality than survival. Compared with survival, the reproductive endpoint of the *Ceriodaphnia* test has statistical advantages as well: a healthy female typically produces 20 to 35 offspring over 3 brood cycles, whereas survival is computed only from the initial number of replicates that are used (usually 10). Marked reductions in fecundity of *Ceriodaphnia* precede reductions in survivorship for solutions containing various common cations and toxic metals (Taylor et al. 1987; J. M. Napier, ORNL, personal communication, 1986) and in dilutions of complex effluents (A. J. Stewart, unpublished data). These findings show that the fecundity endpoint of the *Ceriodaphnia* toxicity test is generically useful for evaluating many effluents and for determining the toxicity of specific chemicals or mixtures of chemicals. However, for those *Ceriodaphnia* tests in which mean fecundity was reasonably high (>12.0 offspring per female), reproduction variability was low both for the controls (C.V. = 10–15%) and between sites (C.V. = 20–40% for all sites combined). The relative similarity of monitoring sites in the fecundity endpoint of the test may be attributable to the nutritional requirements of the animals. Ambient stream water contains algae, bacteria, and detritus that are consumed by the filter-feeding *Ceriodaphnia*, and these food items can markedly augment *Ceriodaphnia* fecundity relative to that found in controls (L. F. Wicker and A. J. Stewart, unpublished data). Thus, site-to-site differences in suspended particulate matter could obscure site-to-site differences in fecundity that would otherwise be revealed in the absence of particulate matter serving as supplemental food.

Interpretations of *Ceriodaphnia* biotest results need to be made carefully on a test-by-test basis. A "budget" of animals in test 4, which had the lowest number of animals available for statistical comparisons between sites (Table 3.4), serves as a worst-case example of the considerations that may be required. Test 4 evaluated 14 sites (site 5 was dry) and a control. Ten replicates were initially set up for each of the 14 sites and for the control, for a total of 150 animals present at the start of the test. By day 7, only 107 animals remained alive: 41 had died, and 2 were missing and presumed dead. Twenty-eight of the 107 living animals were, by day 5, positively identified as males. By day 7, then, only 79 females (about 53% of the original number of animals) were available to contribute to the analysis. Male animals usually occur at much lower frequencies. [Their production, from normally parthenogenic females, indicates that the cultures used to produce neonates for a test were under stress (cf. Hutchinson 1967, p. 594).] If males are present in significant numbers, interpretation of the data can be confounded in two ways: (1) the presence of males reduces the numbers of animals potentially able to contribute to the fecundity endpoint, thus weakening the statistical power of the test, and (2) males are included in the analysis of survival but excluded from the analysis of fecundity. In most tests, large reductions in survival of the animals attributable to differences between sites occurred early in the test, before it was possible to reliably distinguish males from females. The distinction between sexes, however, may be important. Male *Daphnia magna*, for example, are less sensitive to toxicants than females (Breukelman 1932). Such biases in the survival endpoint of the *Ceriodaphnia* test may sometimes exist, but they have not yet been evaluated.

The high degree of test-to-test variability in growth of fathead minnow larvae is also a problem and is suspected to reflect variability in their food. In the fathead minnow test, the larvae are fed an excess of newly hatched brine shrimp three times daily (Horning and Weber 1985). The amount fed to the larvae is constant in terms of volume but variable in terms of dry weight. Additionally, differences in weight of the food supplied to the larvae may not correspond to the availability of the food items; unhatched brine shrimp cysts (which contribute to weight of the food) may not be eaten by the larvae. Intermittently poor hatches of brine shrimp may translate into lower rates of growth of the larval fish, and inadequate hatches of the shrimp would not be identified by measurements of the weights of food supplied to the fish. Studies of food-related sources of test-to-test variability in larval growth will be used to evaluate this possibility.

Survival of fathead minnow larvae in water from the uncontaminated reference sites on upstream reaches of WOC, First Creek, Fifth Creek, and Melton Branch was considerably more variable than survival of larvae in the controls (Table 3.3). Although the source of this variation is presently unknown, it must be identified to increase the ability of the fathead minnow test to detect toxicity in contaminated waters. A sample-handling aspect of the test may contribute to this variability. For example, all four reference sites are spring fed, and on average, relatively cool. (Mean temperatures for sites 1, 2, 3, and 5 were 15.5, 13.7, 13.7, and 11.5°C.) Immediately after collection, water from each site is warmed to 25°C, as recommended by Horning and Weber (1985). Rapid warming of the samples without subsequent aeration may result in supersaturating levels of dissolved nitrogen, carbon dioxide, and oxygen, which can adversely affect the fish (Horning and Weber 1985). The possibility of supersaturation will be evaluated in future tests.

3.1.5 Future Studies

The results of the ambient toxicity monitoring highlight two considerations that should be addressed. First, if episodic excursions in chlorine are instrumental in controlling patterns in toxicity over fairly large reaches of Fifth Creek and WOC, the frequency and magnitude of the excursion and the sources of the toxicant should be identified. The sampling procedures and the toxicity and chemical tests presently used must be modified to obtain these data. The data from the ambient tests suggest that 5 of the 15 sites must be examined from this perspective. One possible method is the use of relatively infrequent (e.g., three or four times per year) extended (e.g., 21-d) in situ tests at each of the five sites, with daily evaluations of survival of the test organism. For such tests, the locally common snail *Elimia* (= *Goniobasis*) *clavaeformis* Lea could be used with great effectiveness. The use of *Elimia* to determine ambient toxicity is also advantageous because the species is numerically abundant in many local, undisturbed streams but is absent from stressed streams within the Reservation (Sect. 6.1; J. M. Loar, unpublished data). This observation suggests that this species is relatively susceptible to adverse changes in water quality. Data from a study such as that described in this section would define the frequency of acutely toxic conditions in sites already defined as particularly problematic.

Continuous instream monitoring of chlorine concentrations by automated instrumentation, particularly in lower Fifth Creek and in WOC just upstream from its confluence with Fifth Creek, is also planned. In these areas, concentrations of chlorine frequently exceeded a "safe criterion" value of chlorine of 3-5 $\mu g/L$ (DeGraeve et al. 1979) by more than an order of magnitude.

A second consideration focuses on the observation that survival, not growth, is the more reliable and sensitive endpoint of the fathead minnow test. Investigations by others substantiate this finding (Mayer et al. 1986). Similarly, but probably for different reasons, the "preferred" chronic endpoint for the *Ceriodaphnia* test (fecundity) may be less useful than survival for tests of ambient waters unless filtration or centrifugation is used to eliminate confounding effects due to particulate matter that augments the food supply of *Ceriodaphnia*. To determine how particulate matter in water can affect the interpretation of patterns in ambient toxicity, the fecundity of *Ceriodaphnia* will be compared in chronic tests with filtered and nonfiltered water samples from the monitoring sites.

The frequency of monitoring for ambient toxicity will be reduced from monthly to bimonthly in the second year. Such a change will accommodate the need to focus more intensively on the sources of toxicity at those monitoring sites where ambient toxicity problems were identified during the first year. This modification in testing frequency will still provide sufficient information to document alterations in stream water quality resulting from planned changes in ORNL operations.

3.2 INSTREAM MONITORING OF THE PERIPHYTON AND MICROBIAL COMMUNITIES

Periphyton, a complex matrix of algae and heterotrophic microbes attached to submersed surfaces, serve as a major food source for many stream invertebrates (Minshall 1978) and for herbivorous fishes (Power et al. 1985). Due to their high biotic activity and organic content, periphyton can concentrate certain contaminants (e.g., mercury and cadmium) by several orders of magnitude relative to concentrations in the water (Huckabee and Blaylock 1973, Selby et al. 1985). These two facts implicate periphyton as critical components in the transfer of various contaminants to higher trophic levels in stream communities. From a toxicological perspective, monitoring of the periphyton is also advantageous because their high turnover rates (1-3 d under favorable conditions) permit detectable responses to short-term environmental changes (e.g., infrequent pulses of toxicants that might be "invisible" when considering organisms with longer life spans). Periphyton, therefore, can be used to (1) characterize the biological communities, (2) evaluate contaminant transport and food-chain accumulation, (3) identify sources and frequencies of toxicant entry into the system, and (4) predict and evaluate biotic responses to remedial action measures.

Monitoring of the periphyton/microbial communities was initiated in April 1986 and, to date, has focused largely on characterizing the biotic community (algal biomass and production). Determining contaminant transport and transfer and utilizing the periphyton as sensitive bioindicators are viewed as critical components of this effort in relation to the goals of BMAP (Loar et al. 1991) and will be evaluated beginning in 1987.

3.2.1 Methods

Samples were collected monthly to initially characterize the periphyton in streams in WOC watershed. These samples were collected from ten sites (Fig. 3.7) that generally coincided with the fish and benthic invertebrate sampling stations (Fig. 2.3). Five sites receive drainage or are downstream from effluent inputs from ORNL, including four on WOC (WCK 3.9, WCK 3.4, WCK 2.9, and WCK 2.3) and one on Melton Branch (MEK 0.6). The remaining five upstream sites—located on WOC (WCK 6.8), Melton Branch (MEK 1.8), Northwest Tributary (NTK 1.0), First Creek (FCK 1.0), and Fifth Creek (FFK 1.1)—were used as reference sites.

The use of uniform artificial substrata to monitor algal growth rates, standing crop, and production, as discussed in Loar et al. (1991), was abandoned in favor of natural substrata. In using artificial substrata (ceramic tiles), it was found that (1) these artificial surfaces required relatively long periods in situ (>6 weeks) before algal biomass (measured as chlorophyll *a*) was similar to that found on natural substrata (rocks) and (2) the algal community on artificial surfaces can differ from that present on natural surfaces. Methodological difficulties involved in the determination of periphyton production in situ were avoided by using short-term laboratory measurements for comparisons of periphyton primary production.

To measure algal biomass and production, four small relatively flat rocks (10-60 cm^2) were collected 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 in water from that site containing 10 μCi $NaH^{14}CO_3$. During the 2-h incubation, the water temperature was maintained within 2°C of the ambient stream temperature. Approximately 500 $\mu E \cdot m^{-2} \cdot s^{-1}$* of photosynthetically active radiation (400-700 nm) was provided by a 1000-W metal halide lamp (~25% full sun), and the water was circulated by a submersible pump to simulate natural conditions. After incubation, the rocks were rinsed twice in distilled water to remove residual inorganic ^{14}C, placed in 30 mL of dimethyl sulfoxide (DMSO), and kept in darkness for 24 h to extract soluble organic compounds and chlorophyll (Filbin and Hough 1984). The DMSO and rock were then heated to ~50°C for 45 min to complete the extraction. Five milliliters of extract was diluted 1:1 with 90% acetone, and the chlorophyll *a* content was determined spectrophotometrically by 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 content of the aliquot was determined by liquid scintillation spectrometry. The surface area of each rock was determined by covering the upper surface with aluminum foil, determining the weight of the foil, and converting to surface area based on a known weight per unit area of foil. Chlorophyll *a* and the rate of carbon incorporation were then expressed on a surface-area basis.

3.2.2 Results and Discussion

Periphyton chlorophyll *a* provided a measure of algal biomass at each of the ten monitoring sites from April through December 1986. Chlorophyll *a* per unit rock surface

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*The symbol *E* denotes einstein: 1 einstein = 1 mole of photons.

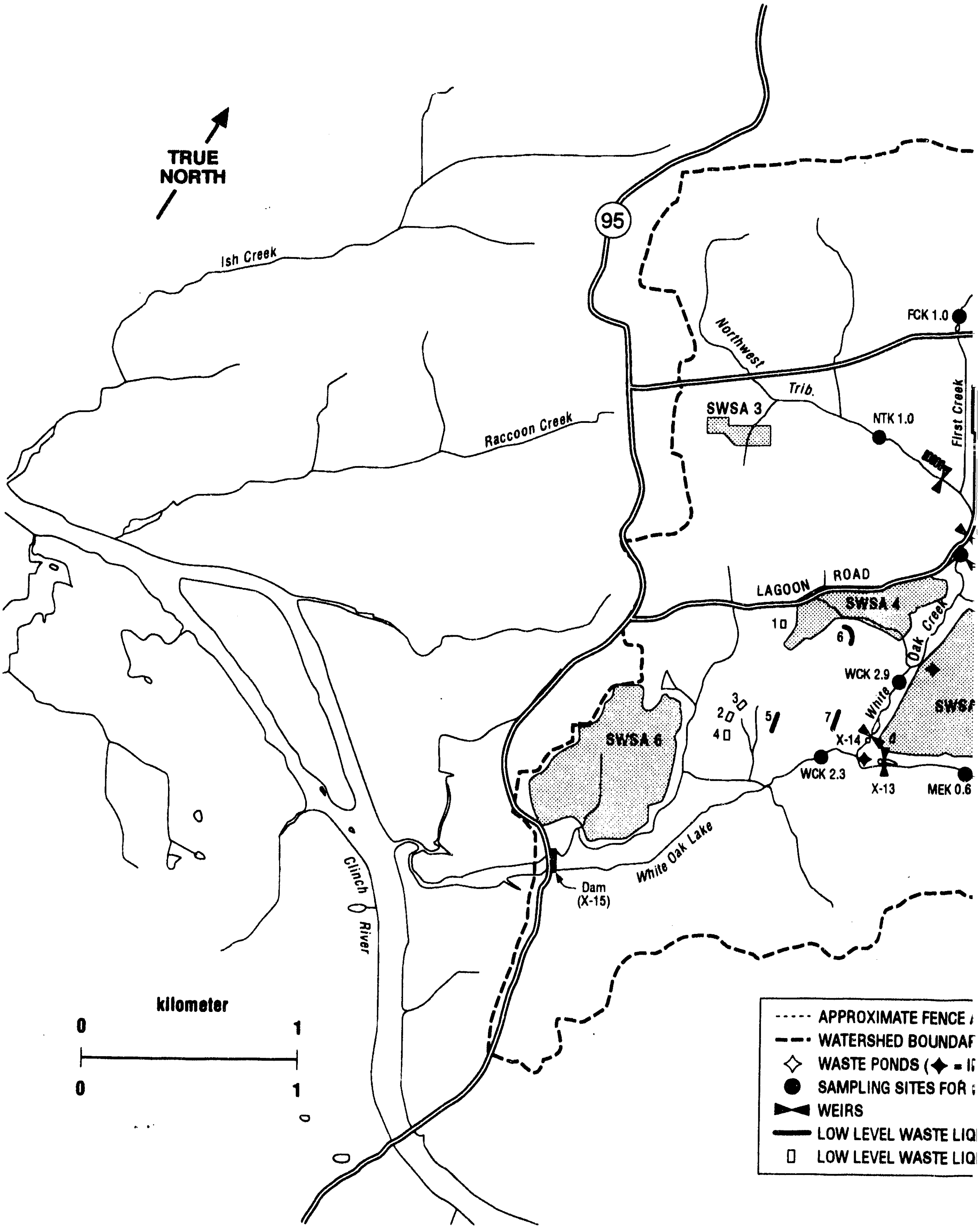

Fig. 3.7. Periphyton-microbial community study sites in the White Oak Creek drainage system. FCK = First Tributary kilometer; WCK = White Oak Creek kilometer; HFIR = High Flux Isotope Reactor; SWSA = solid r

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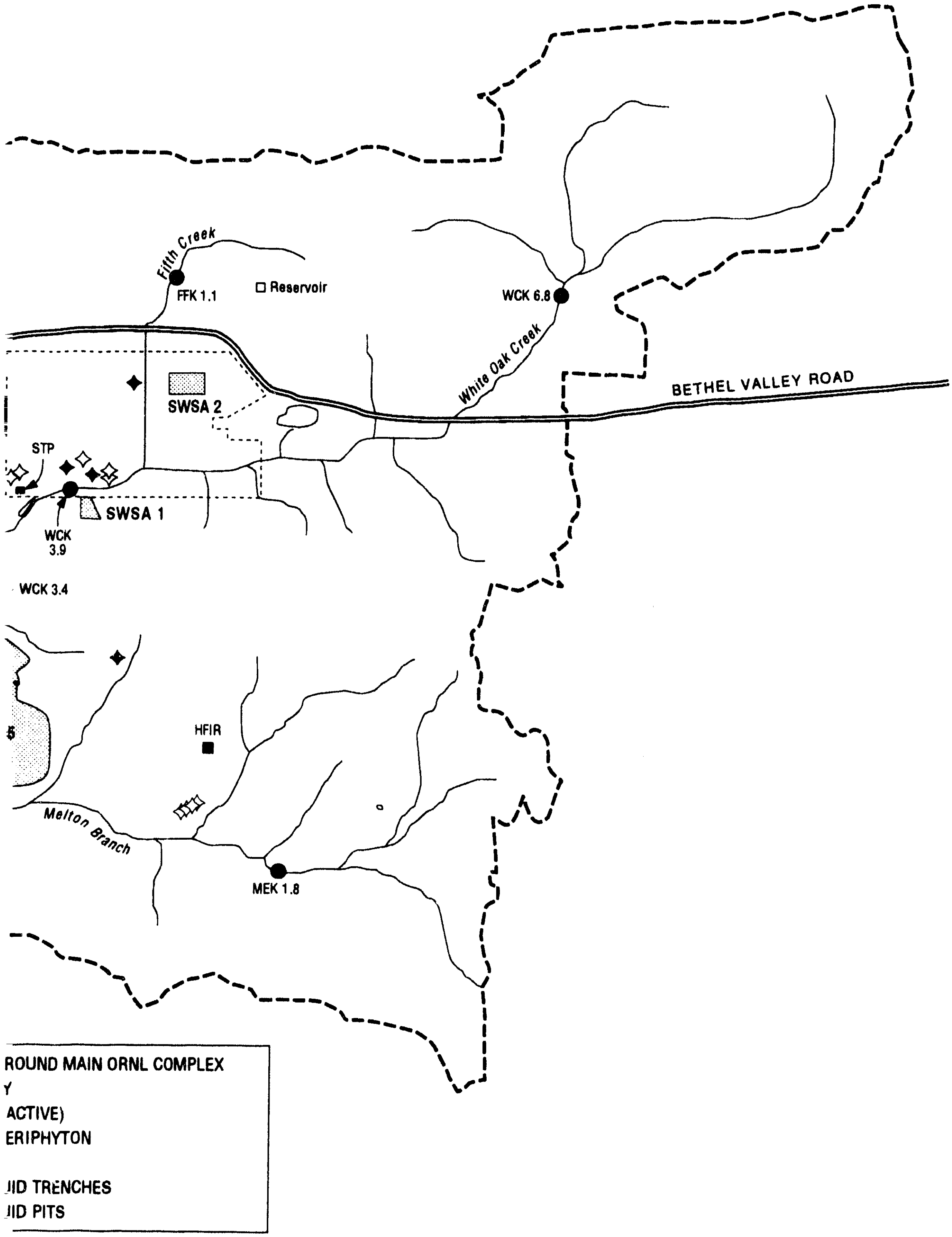

Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest
dioactive waste disposal/storage area.

area for each of the five downstream sites and a pooled value for the mean of the five upstream reference sites are shown in Fig. 3.8. The seasonal pattern for algal-periphyton biomass (estimated by the chlorophyll *a* content of the periphyton) was similar for all downstream sites: maximum biomass typically occurred in May and November, and minimum biomass occurred in July and December. The reference sites showed a similar pattern. Monthly data for periphyton primary production (data not shown) had a pattern similar to that of chlorophyll, although maxima typically occurred in September (rather than May) and November, and minima usually occurred in July.

The reference sites were located on small headwater streams with substantial riparian cover and generally had low concentrations of most nutrients; in contrast, several of the downstream sites had no riparian cover and all had high concentrations of soluble nutrients (Table 2.5). The generally high chlorophyll concentrations at the reference sites for November may have resulted from increased light and nutrients following leaf fall. Although no obvious explanation exists for the increased chlorophyll at the open, nutrient-rich downstream sites, a decreased discharge, or a dilution, of toxicants at these sites could have occurred [e.g., the shutdown of the HFIR may have influenced MEK 0.6 (Sect. 2.2)]. At the upstream sites, low levels of chlorophyll and production in July may have resulted from shading and nutrient limitation. No explanation is offered for the similar decline at the downstream sites, although the potential for toxicity during low flow cannot be eliminated from consideration. The influence of grazing on this seasonal pattern may be minimal because invertebrate biomass followed a seasonal pattern similar to that of periphyton chlorophyll *a*, and the seasonal pattern is similar for downstream sites WCK 3.9 and MEK 0.6, where grazers are few (Sects. 6.1.3 and 6.2.3.2).

Limited information on environmental parameters and the absence of direct evidence of toxicity to periphyton complicate the interpretation of the results for the downstream sites in the context of toxicity. At WCK 3.9 and MEK 0.6, however, substantial toxic effects are hypothesized. At WCK 3.9, for example, some rocks support very thick periphyton coatings, while nearby rocks are totally barren. This observation suggests episodic exposure to transient toxicants, such as chlorine (Sect. 3.1). Here, cells deep within a thick periphyton matrix are protected from acute exposure to toxicants and should recover rapidly, allowing the accumulation of additional periphyton. Conversely, cells on rocks with poorly developed mats of periphyton may be totally eliminated, resulting in the very patchy rock-to-rock spatial distribution of periphyton at this site. At MEK 0.6, the algal-periphyton biomass was consistently lower than expected based on the nutrient-rich conditions and the low apparent grazing pressure (see Sect. 6.1 and 6.2). Data on toxicity (Sect. 3.1) and on such indicators of stress as the composition and richness of the benthic invertebrate and fish communities at these sites (Sects. 6.1 and 6.2) corroborate these hypotheses.

Ranking the sites based on mean periphytic algal biomass (chlorophyll), as determined by monthly sampling (Table 3.8), revealed a more than tenfold range among the sites, with the reference sites generally being the lowest. Reference sites NTK 1.0 and MEK 1.8 were dry from July through October; values for these sites are means for May, April, November, and December. Values of 5 micrograms of chlorophyll *a* per square centimeter or less at several upstream sites were typical of other local reference streams (H. L. Boston and A. J. Stewart, unpublished data). Concentrations of chlorophyll *a* at the two WOC sites below the Sewage Treatment Plant (WCK 3.4 and WCK 2.9) were similar to the concentrations observed in other area streams with elevated nutrient levels (H. L. Boston and A. J. Stewart,

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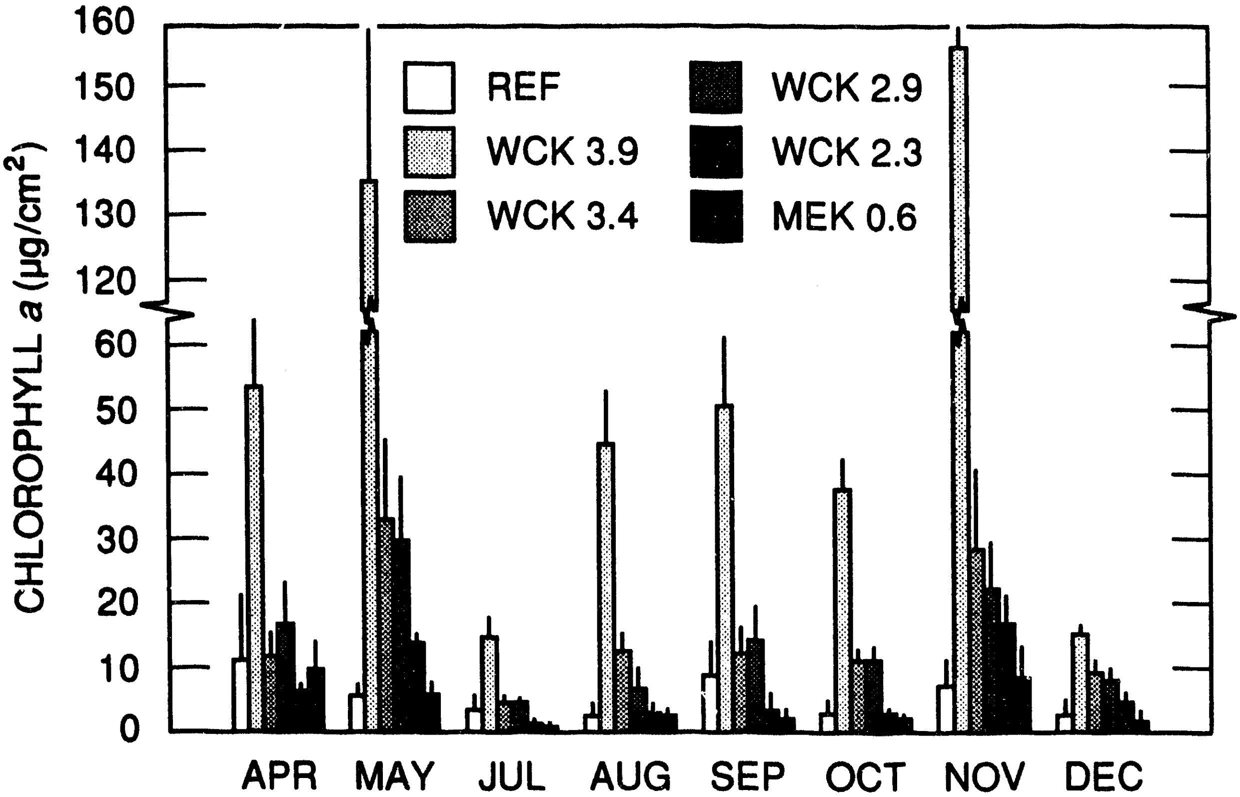

Fig. 3.8. Mean (± 1 s.d.) chlorophyll *a* concentrations in 1986 (as micrograms per square centimeter of riffle zone) in periphyton from five downstream sites and the mean and range of five reference sites (only three sites during July through October) in streams within White Oak Creek watershed. REF = reference site; WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer.

Table 3.8. Mean chlorophyll *a* and carbon incorporation rates of periphyton based on monthly sampling at ten sites from April through December 1986[a]

Site	Mean mass of chlorophyll *a* ($\mu g/cm^2$)	Site	Mean rate of production ($mg\ C \cdot m^{-2} \cdot h^{-1}$)
MEK 1.8[b]	2.9 (±1.8)	MEK 1.8[b]	7.5 (±5.4)
FCK 1.0[b]	3.1 (±2.1)	FCK 1.0[b]	7.5 (±2.0)
NTK 1.0[b]	4.0 (±1.9)	WCK 6.8[b]	11 (±6.2)
MEK 0.6	4.6 (±3.0)	MEK 0.6	12 (±3.2)
WCK 6.8[b]	5.0 (±3.0)	NTK 1.0[b]	14.5 (±5.1)
WCK 2.3	7.1 (±5.2)	WCK 2.3	19 (±6.8)
FFK 1.1[b]	10.8 (±6.7)	WCK 2.9	26 (±11)
WCK 2.9	15 (±8.3)	WCK 3.4	26 (±7.8)
WCK 3.4	16 (±9.3)	FFK 1.1[b]	35 (±15)
WCK 3.9	64 (±50)	WCK 3.9	69 (±41)

[a]Sites are arranged vertically as increasing average values. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer. Values shown represent the mean of eight monthly values ± 1 s.d., except for sites NTK 1.0 and MEK 1.8. These sites were dry from July through October, and values are for 4 months only.

[b]Reference site.

unpublished data). The mean chlorophyll content of periphyton at WCK 3.9 (monthly values up to 156 micrograms of chlorophyll *a* per square centimeter) was four times that of any other site in WOC watershed, whereas chlorophyll concentrations 1.6 km downstream (at WCK 2.3) were similar to those of the upstream reference sites. Interestingly, the mean periphyton chlorophyll concentration at FFK 1.1 was more than twice that observed at the other four reference sites.

The ten sites were also ranked on the basis of periphyton production rates (Table 3.8). The nearly tenfold range in mean production (photosynthetic carbon incorporation) among the sites was similar to that for chlorophyll. As for chlorophyll, primary production at the downstream sites reflected nutrient enrichment (Table 2.5) and high algal biomass.

Periphyton production at the downstream sites was high relative to natural (nonenriched) streams of similar size (e.g., Naiman and Sedell 1980, Hornick et al. 1981, Bott et al. 1985). Among the reference sites, Fifth Creek (FFK 1.1) periphyton had the highest concentration of chlorophyll *a* and the highest production rates. The average production rate at FFK 1.1 was also substantially higher than that of several of the more nutrient-rich downstream sites. Although Fifth Creek and First Creek seem to originate from the same groundwater source (see cation composition, conductivity, and hardness, Table 2.5), site FFK 1.1 had more available phosphorus and apparently more CO_2 (note lower pH but similar conductivity and alkalinity, Table 2.5), probably due to its proximity (within 20 m) to the springwater source. Favorable conditions for production (abundant inorganic carbon and mineral nutrients, little siltation, minor grazing pressure) likely facilitated the better than expected performance at this site. Site FCK 1.0 was located farther downstream of its spring source, and heavy grazing pressure (visual observations of an abundant snail population) probably contributed to the lower algal biomass and production at this site compared with that at FFK 1.1.

Of the downstream sites, WCK 2.3 was most similar to the upstream reference sites. A comparison of the monthly data for WCK 2.3 and WCK 6.8 suggests that, during the summer months, light availability (both are heavily canopied sites) and heavy grazing pressure (Sect. 6.1.3) may limit periphyton biomass (chlorophyll *a* concentrations) and production rates at both sites, despite higher nutrient levels farther downstream (Table 2.5). Data for benthic invertebrates and fishes (Sect. 6.1 and 6.2) also suggest that environmental quality at this downstream site had recovered, although there was no indication that full recovery of these communities had occurred.

To evaluate the month-to-month variability in chlorophyll *a* and production, the coefficient of variation was calculated for each site (Table 3.9). When the sites were ranked on the basis of increasing coefficient of variation for mean chlorophyll, moderate to substantial amounts of variation were observed at the reference and downstream sites. The ranking is difficult to interpret because values for reference and downstream sites, high- and low-nutrient sites, and woodland and open sites are interspersed throughout the table. To aid the evaluation, an analogous data set from similar studies in East Fork Poplar Creek (EFPC) and Brushy Fork was included in the analysis (H. L. Boston, unpublished data). When considered in the context of these additional data, the results of the periphyton analyses for WOC watershed suggested that variability in periphyton chlorophyll increases for sites that are subject to (1) disturbances, such as scouring, siltation, and possible toxic inputs, and (2) seasonal releases from one or more significant limitations (e.g., shaded sites where low light intensity may limit biomass regardless of nutrient regime).

It was hypothesized that the coefficient of variation for mean production would be positively correlated with the coefficient of variation for mean chlorophyll because production is chlorophyll dependent. With only ten data points, however, no clear relationship was evident. Production rates were less variable than chlorophyll *a* concentrations for all sites except WCK 3.9, where they were similar, and MEK 1.8, where the production rate was apparently more variable. Production values may be inherently less variable because, as chlorophyll per unit area increases, production per unit area first rises and then asymptotes (or declines) with increasing chlorophyll, due to self-limitation. While self-limitation clearly occurs at sites with very high chlorophyll (see Fig. 3.9), it cannot account for observed patterns in the data. Elucidation of the factors controlling the variability in production vs biomass can provide useful information concerning the influence of grazing, nutrient

Table 3.9. Coefficient of variation (C.V.) in chlorophyll *a* and production per unit surface area based on eight monthly samples from April through December 1986[a]

Site	C.V. in chlorophyll *a* (%)	Site	C.V. in production (%)
NTK 1.0[b]	47	FCK 1.0[b]	27
WCK 2.9	55	WCK 3.0	30
WCK 6.8[b]	58	NTK 1.0[b]	35
WCK 3.4	58	WCK 2.3	36
MEK 1.8[b]	61	WCK 2.9	43
FFK 1.1[b]	62	FFK 1.1[b]	43
MEK 0.6	66	MEK 0.6	46
FCK 1.0[b]	69	WCK 6.8[b]	56
WCK 2.3	74	WCK 3.9	60
WCK 3.9	78	MEK 1.8[b]	72

[a]Sites are arranged vertically as increasing values of coefficient of variation. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer. Values for NTK 1.0 and MEK 1.8 are for 4 months only.

[b]Reference site.

limitation, toxicants, or other factors on periphyton biomass and production. As more data become available, these relationships can be evaluated, thus increasing the capability to predict the responses of biota to remedial measures and to identify toxic vs natural stresses.

Chlorophyll-specific production values (carbon uptake per unit chlorophyll per unit time) can be used to assess the relative physiological state of the algal component of the periphyton. Chlorophyll-specific production normalizes production data for the amount of biomass present, thus providing additional insight into variations in conditions controlling photosynthesis and the physiological state of the periphytic algae. For this evaluation, monthly data for chlorophyll-specific production rates were plotted as a function of periphyton chlorophyll *a* concentrations for each site (Fig. 3.9). This analysis showed that chlorophyll-specific production declined as chlorophyll increased. Such a relationship results from increased self-shading and increased resistance to the inward diffusion of carbon and mineral nutrients and the outward diffusion of waste products for the lower layers of the algal periphyton as the thickness of the periphyton increases. As periphyton thickness increases, the average age of the algae may also increase, which, along with decreased light and rates of diffusion, should accentuate the observed relationship.

The data for chlorophyll and chlorophyll-specific production were transformed (natural log) plotted as shown in Fig. 3.10; the resulting relationship was described by a simple linear regression ($r^2 = 0.63$, $n = 68$). When data for a given site consistently lie above the line (i.e.,

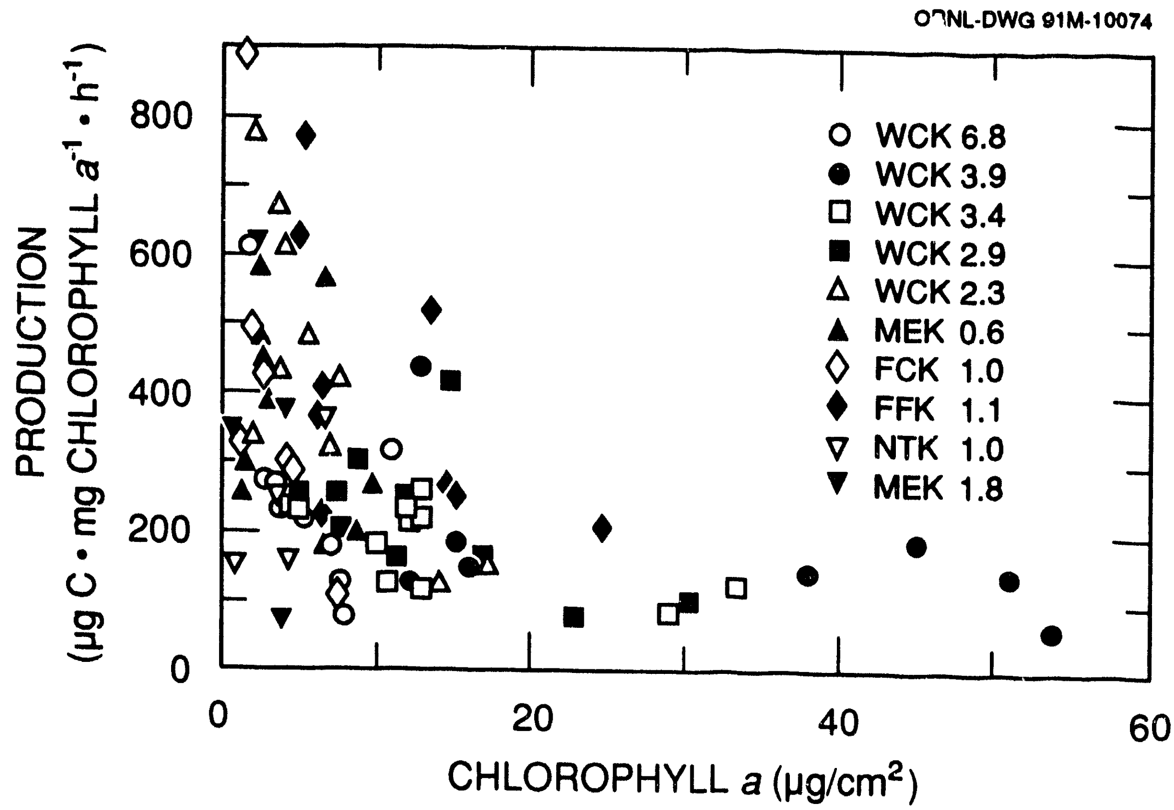

Fig. 3.9. Mean monthly values for chlorophyll-specific production plotted as a function of chlorophyll per unit area. Values represent the mean of four rocks. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

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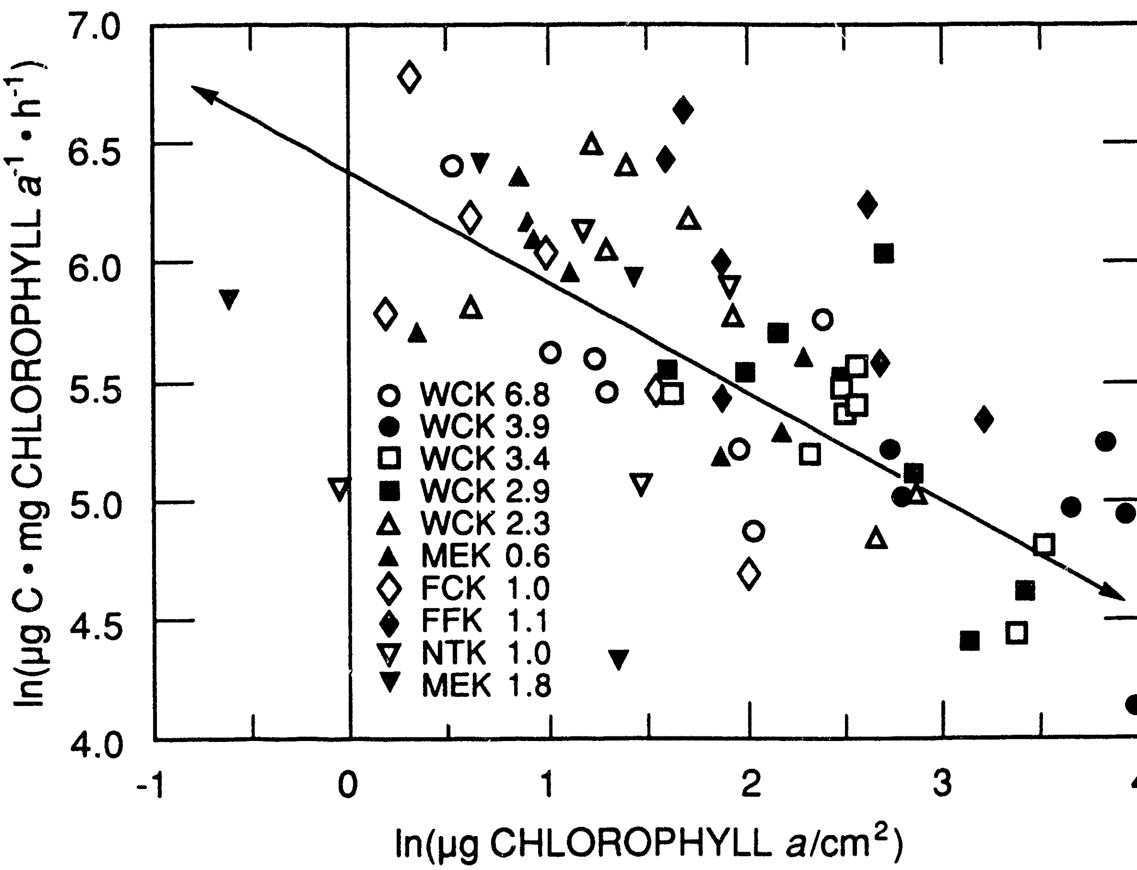

Fig. 3.10. Transformed (natural log) mean monthly values for chlorophyll-specific production plotted as a function of chlorophyll *a* for ten sites in streams within White Oak Creek watershed. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

higher than predicted chlorophyll-specific production), the periphyton are in better than average physiological condition. The converse is true for sites consistently falling below the line. Ideally, only data from the reference sites would be used to establish such a relationship, and data from downstream sites would be evaluated by plotting as shown in Fig. 3.10 (or more formally by ANOVA). However, the relatively few data for reference sites to date and the absence of data for chlorophyll-rich reference sites preclude the use of this more ideal procedure at this time.

Based on the information available, only three sites are clearly above or below the average (Fig. 3.10). With the highest nutrient levels among the reference sites (Table 2.5) and substantial rates of periphyton production (Table 3.8), the reference site FFK 1.1 consistently showed better than average performance. Two other reference sites—WCK 6.8 and FCK 1.0—had low nutrient levels and low chlorophyll *a* concentrations and production rates (Table 3.8) and were heavily grazed by snails. These two sites were consistently below the average line, indicating a poorer than average physiological state. At this time, neither sufficient data nor an adequate understanding of the influence of various factors on chlorophyll-specific production exists to explain the interaction of the positive influence of nutrient enrichment and the adverse influence of toxic stress that may determine the physiological condition of the periphyton at sites farther downstream.

3.2.3 Summary

Much of the initial work to date on the periphyton-microbial community has focused on the characterization of the periphytic algal community. Periphytic algal biomass was measured as chlorophyll *a* per unit rock area in riffle zones at ten sites from April through December 1986. Mean values during this period ranged from ~3 to >60 micrograms of chlorophyll *a* per square centimeter. Values of <10 $\mu g/cm^2$ at the upstream reference sites were typical of small woodland streams. Higher values reflected nutrient enrichment at the downstream sites, although grazing also appeared to be an important influence on periphyton biomass at several sites. Periphyton production was measured at these same sites and ranged from about 8 to almost 70 milligrams of carbon per square meter per hour. Again, lower values at the reference sites were typical of those found in similar natural systems. Low values for periphyton chlorophyll *a* and production at MEK 0.6 and visual observations of the periphytic algae at WCK 3.9 suggest that these sites experience occasional toxic stresses. The assessment of physiological status, using rating curves of chlorophyll-specific production vs chlorophyll, shows promise as a tool for evaluating environmental conditions, including stresses.

3.2.4 Future Studies

Initially, the periphyton component of BMAP focused on characterizing the algal component of the periphyton. Although these studies will be continued in 1987, methods for data analysis and interpretation will be expanded. For example, data can be plotted by month, rather than by site, to identify temporal changes in periphyton condition resulting from seasonal factors or toxic episodes. As more data become available, more-powerful statistical tools can be employed (e.g., two-way analysis of covariance) to evaluate the temporal and spatial variation in chlorophyll-specific production. In subsequent reports, such performance curves will be used to compare sites and seasonal changes in the physiological condition of periphyton community.

Chlorophyll-specific production provides a useful index of the physiological state of the algal periphyton. As described in Subtask 1c of BMAP (Loar et al. 1991), the levels of adenosine triphosphate (ATP), C, N, and P in periphyton will be assessed. These parameters will be used in a manner analogous to chlorophyll-specific production to describe the physiological state of the periphyton (Healey and Hendzel 1980, Bothwell 1985, Palumbo et al. 1987).

Future efforts will also focus on (1) use of the periphyton as indicators of occasional toxic stresses, (2) evaluation of the role of the periphyton in the transfer of contaminants to higher trophic levels (invertebrates and fishes), and (3) characterization of the microbial component of the periphyton.

3.2.4.1 Contaminant transfer and toxicity screening

Beginning in 1987, screening for contaminants will evaluate the role of the periphyton in contaminant dynamics. Preliminary data on the mercury content of periphyton indicated that several downstream sites in WOC were enriched by more than 50-fold relative to periphyton at reference sites. The use of periphyton responses (e.g., physiological parameters) to intermittent toxic stress will also be evaluated. Algal periphyton grown in the laboratory on artificial substrata (ceramic tiles) will be placed at sites in WOC known to be intermittently toxic to invertebrates and fish. Tiles will be removed at weekly intervals for 1 month and analyzed for species composition, total dry weight, organic carbon content, and chlorophyll-specific carbon uptake.

3.2.4.2 Microbial component of the periphyton

The heterotrophic microbial component of the periphyton also plays an important role in stream ecosystems. Because this component is regulated by a different set of environmental parameters, it may show different responses to perturbations than does the algal component of the periphyton. Characterization and monitoring of this component of the periphyton will be initiated in 1987.

4. BIOACCUMULATION STUDIES

4.1 IDENTIFICATION OF CONTAMINANTS THAT ACCUMULATE IN AQUATIC BIOTA

4.1.1 Introduction

As described in Task 2a of BMAP, fish were collected from seven sites in WOC watershed in December 1986-February 1987 for analysis of a wide range of metals and organic priority pollutants. Only results of mercury analyses at some of the sampling sites were available for this report. These preliminary data are included at this time because they may be of value in structuring and implementing the ORNL Mercury Monitoring Plan stipulated in the NPDES permit (EPA 1986, Part III F) and may impact plans for further studies as part of Subtask 2b: identifying contaminant sources of BMAP (Loar et al. 1991).

4.1.2 Methods

Fish were collected by electrofishing at three sites on WOC and at single sites on WOL, WOC embayment, Northwest Tributary, and Melton Branch. Sites on WOC, Melton Branch, and Northwest Tributary correspond closely to fish/benthos population survey sites (Fig. 2.3). At each site, fish were generally concentrated in one or two large pools; only at WCK 2.9 was it necessary to electrofish a larger reach of stream (~ 150 m).

Twelve fish were collected at each site to provide samples for analysis of metals, organics, and radionuclides and for archival storage. Samples were taken from eight fish for each purpose. Bluegill sunfish (*Lepomis macrochirus*) and redbreast sunfish (*Lepomis auritus*) were collected in equal numbers at a site where possible; however, redbreast sunfish were restricted to those portions of WOC watershed downstream of the weir at WCK 3.41. 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 and Brushy Fork (Anderson County, Tennessee) in order to estimate background levels of contaminants and 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 and measured, and scale samples were taken for age determination. The fish was fileted, and skin removed from the filet. A 1- to 2-g portion of the anterior dorsal portion of the axial muscle filet was excised for the determination of mercury, and the remainder of the filet was retained for analysis of other metals and radionuclides. The remaining filet was used for a duplicate sample, archived, or analyzed for organic contaminants. All samples 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.

Mercury determinations were carried out by ACD through the use of procedure EC 420 (Martin Marietta Energy Systems 1983). 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.

Statistical and quality assurance procedures are described in Sect. 4.2.2.

4.1.3 Results

Mercury in sunfish was found to be significantly ($p < 0.05$) above background levels at three sites in WOC and a site in Northwest Tributary (Table 4.1, Appendix B). Levels in fish from Melton Branch and WOL were lower and did not differ significantly from those in controls. The highest levels were observed in reaches of WOC downstream from SWSA 4 and SWSA 5 (Fig. 2.3). No fish collected in this study exceeded the 1-ppm U.S. Department of Agriculture Food and Drug Administration (FDA) action limit (FDA 1984a); the highest value measured was 0.73 ppm in a fish from site WCK 2.3 (Fig. 2.3).

Mercury levels in redbreast sunfish and bluegill sunfish did not differ significantly ($p > 0.05$) in WOC and Melton Branch at sites where both species occur (Appendix B), but the average level of mercury in bluegill (0.09 ppm) was significantly lower than that in redbreast sunfish (0.27 ppm) in WOL. The latter value significantly exceeded the level in bluegill controls (Hinds Creek), but did not differ significantly from redbreast sunfish controls (Brushy Fork). A possible explanation for higher levels of mercury in redbreast sunfish, a species typically found in stream environments, is that they may move back and forth between WOL and the lower reaches of WOC, where they accumulate mercury, while bluegill remain in the lake. An unlikely alternative explanation would be that the bluegill collected in WOL in winter were recent immigrants from the Clinch River/Watts Bar Reservoir and thus contain mercury levels typical of that system. Although White Oak Dam would provide an obstacle

Table 4.1. Total mercury (milligrams per kilogram) in sunfish (*Lepomis auritus* and *Lepomis macrochirus*) collected in White Oak Creek watershed, winter 1986–87
(n = 8 fish per site)

Site[a]	Mean ± s.d.	Range
NTK 0.2	0.26 ± 0.05	0.19–0.32
MEK 0.16	0.13 ± 0.04	0.08–0.20
WCK 3.5	0.25 ± 0.12	0.14–0.52
WCK 2.9	0.44 ± 0.09	0.30–0.58
WCK 2.3	0.49 ± 0.18	0.20–0.73
White Oak Lake	0.16 ± 0.10	0.14–0.52
Hinds Creek (control)	0.06 ± 0.02	0.05–0.09
Brushy Fork (control)	0.10 ± 0.06	0.04–0.24

[a]MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

to such migration, it might not be a barrier, especially during high flows. Thirdly, redbreast in WOL may not contain elevated levels of mercury, and larger sample sizes would result in convergence of the mean mercury concentration in these fish toward the control level.

4.1.4 Discussion

Elevated mercury levels were observed in sunfish collected in WOL in 1979 (Loar et al. 1981a) and in 1984 (TVA 1985a). In 1979, bluegill from WOL averaged 0.70 ppm mercury, with 20% (two of ten) of the fish containing more than the FDA action limit of 1 ppm (FDA 1984a). The 1984 study found much lower levels, averaging 0.23 ppm mercury in bluegill from WOL. The most recent data (Table 4.1, Appendix B) indicate continued improvement in mercury levels in WOL fish, with bluegill containing 0.09 $\pm$ 0.01 ppm (mean $\pm$ 1 s.d.) at this site and redbreast and bluegill together averaging 0.16 ppm.

While mercury levels in WOL fish give little evidence of active contamination, the levels of mercury in fish in WOC upstream from the lake indicate the probability of an active source or sources of mercury contamination in ORNL or subsurface waste disposal sites. The highest levels of mercury were observed in fish from site WCK 2.9, located immediately downstream from SWSA 4 and SWSA 5, and those from site WCK 2.3, located downstream from the tributary draining the chemical waste pits near the confluence of WOC and Melton Branch (WCK 2.49). However, elevated mercury levels were observed in fish upstream from those sites in WOC and Northwest Tributary. Because the accumulation of mercury by fish is an incompletely understood process involving the transformation of inorganic mercury to methylmercury species, it is possible that the ultimate source of mercury contamination is some distance upstream of sites exhibiting the highest levels of biotic contamination.

4.1.5 Conclusions

Levels of mercury in fish in WOL and Melton Branch are well below the FDA action limit and differ little from levels observed in fish from uncontaminated sites. An active source or sources of mercury contamination exist in WOC, but these data cannot isolate a specific source. Elevated levels in fish from the uppermost site sampled in WOC (WCK 3.5) indicate possible sources in the main ORNL complex. The discharge from the 3539/3540 ponds contained elevated levels of mercury in 1985, as did WOC downstream of this discharge (Martin Marietta Energy Systems 1986a). However, sources farther downstream on WOC may also be contributing to abnormally high mercury levels in fish. The slight elevation in mercury levels of fish from NTK 0.2 is probably due to the immigration of fish from WOC, although upstream sources of mercury (e.g., SWSA 3 or the 1505 complex) cannot be ruled out.

4.2 EVALUATION OF POLYCHLORINATED BIPHENYL CONTAMINATION

4.2.1 Introduction

A Tennessee Valley Authority (TVA) survey conducted in 1984 of contamination in fish in the vicinity of the DOE Oak Ridge facilities found significant levels of polychlorinated biphenyls (PCBs) in channel catfish collected in the WOC embayment of Watts Bar Reservoir

(TVA 1985a). Levels of PCBs in all nine channel catfish from this site exceeded the 2-ppm maximum permissible level established by FDA (FDA 1984b). These data raised concerns that such levels of contamination were representative of the levels of PCBs in channel catfish in the Clinch River in the vicinity of WOC. This portion of the river below Melton Hill Dam is intensively utilized by sport fishermen. Since virtually all liquid effluents from ORNL are released into the WOC drainage (and ultimately into the downstream embayment), it is possible that the source of PCBs in catfish in WOC embayment was past and/or present discharges from ORNL.

WOC immediately downstream from ORNL has been contaminated with PCBs in the past. Creek sediments were collected and analyzed for PCBs in 1974-75 (Energy Research and Development Administration, personal communication, 1986) and in 1979-80 (Boyle et al. 1982); in both cases levels of 1-3 ppm were typical. The 1984 survey by TVA found up to 2 ppm PCBs in sediments from WOC embayment (TVA 1985b, 1985c). The Clinch River-Watts Bar Reservoir system contains a number of other possible sources of PCB contamination. Both the Oak Ridge Y-12 Plant and the Oak Ridge Gaseous Diffusion Plant (ORGDP) have used quantities of PCBs in the past, and sediments from EFPC, Poplar Creek, and Bear Creek, which drain these plants, contained significant levels of PCBs in surveys conducted within the past decade (Loar et al. 1981b; TVA 1985b, 1985c; L. L. McCauley, Oak Ridge Y-12 Plant, personal communication, 1985). Two large coal-fired electric generating plants are located on the Clinch River/Watts Bar Reservoir: one on Melton Hill Reservoir, ~43 km upstream from the mouth of WOC embayment, and the other, 37 km downstream of the mouth on Watts Bar Reservoir. The system also receives treated sewage from the cities of Clinton, Oak Ridge, and Kingston.

The 1984 TVA survey of PCB contamination in catfish in the Clinch River/Watts Bar Reservoir depicted a pattern of ubiquitous contamination (0.5-1.0 ppm) throughout the system with high levels in WOC embayment (TVA 1985a). No fish were collected from the Clinch River at the mouth of WOC embayment in that study. Carp collected from WOL contained an average of only 0.4 ppm PCBs, while catfish in the downstream WOC embayment averaged 3.1 ppm. PCBs were monitored in carp, bass, bluegill, and gizzard shad in the Clinch River at the mouth of WOC and other sites downstream in 1984 by ORNL (Martin Marietta Energy Systems 1985) and monitored in carp only in 1985 (Martin Marietta Energy Systems 1986a). PCB levels in fish collected in the vicinity of WOC were not higher than in fish from downstream sites; the highest levels generally occurred near the mouth of Poplar Creek.

Thus, recent data from two sources suggest that PCB levels in channel catfish in WOC embayment may be high enough to warrant concern for public health, but that levels in other species of fish in the vicinity of the embayment are not of great concern. The primary goals of this task of BMAP were to (1) determine the likelihood that channel catfish caught by fishermen in the tailwaters of Melton Hill Dam and upper Clinch River/Watts Bar Reservoir might contain PCBs in excess of the FDA tolerance limit and (2) establish the extent to which ORNL, via WOC, WOL, and WOC embayment, is the source of PCB contamination in channel catfish in public waters downstream.

The task of determining the importance of ORNL and the WOC system as a source of PCBs in catfish is complicated by the presence of several other known and possible sources of PCBs on Clinch River/Watts Bar Reservoir. If channel catfish in this system have restricted home ranges, levels of PCBs in fish from a specific locale may be indicative of proximity to the primary source of contamination. However, if fish do not reside in any specific area for very long, a point source could produce a geographic distribution of PCBs in fish that is broad and relatively uniform and does not necessarily vary as a function of distance from the source. Studies have shown that the movements of channel catfish can be rather restricted (i.e., remaining within ~0.4 km of one site for several months) (Ziebell 1973) or cover distances of many kilometers (Hubley 1963).

A number of radionuclides are continuously discharged to WOC as a result of seepage from radioactive waste disposal sites and ORNL operations (Cerling and Spalding 1982, Martin Marietta Energy Systems 1986a). One of these radioisotopes, ^{90}Sr, is concentrated in fish bone, where it is retained with a very long biological half-life (Nelson 1967, Rosenthal 1963). PCBs are concentrated in fish lipids, where they are retained with a very long biological half-life (Niimi and Oliver 1983). The only significant source of ^{90}Sr to the Clinch River/Watts Bar Reservoir is ORNL. Thus, if the WOC system is the predominant source of PCB contamination in channel catfish in nearby reaches of the Clinch River/Watts Bar Reservoir, the geographical patterns of levels of PCBs in flesh and ^{90}Sr in bone of catfish collected in that system should be very similar, and concentrations of PCBs and ^{90}Sr should be highly correlated.

The specific objectives of this study (Subtask 2d of BMAP, as described in Loar et al. 1991) were

1. to determine the levels and geographical distribution of PCB contamination in channel catfish in the Clinch River/Watts Bar Reservoir near the mouth of WOC, and

2. to compare the pattern of PCB levels in channel catfish among the various sites with that observed for ^{90}Sr in catfish bone and then use that comparison to assess the importance of ORNL as a source of any observed PCB contamination.

4.2.2 Methods

Catfish were collected in July and August 1986 by trotline, gill net, and setline at sites in the WOC-Clinch River-Watts Bar Reservoir-Melton Hill Reservoir systems. Sampling sites are identified in Fig. 4.1. One site was located in WOL, a shallow 8-ha impoundment formed by a small dam on WOC. White Oak Dam acts as an obstacle, but not necessarily a barrier, to the movement of fish between the lake and the downstream embayment. Two sites (sites 2 and 3, Fig. 4.1) were located in WOC embayment. WOC embayment is a 1-km-long arm of Watts Bar Reservoir that impounds the old WOC bed below White Oak Dam. The embayment fluctuates in level in response to hydroelectric power generation at Melton Hill Dam several kilometers upstream. Such fluctuations result in tidelike flows of water into and out of the embayment several times a day, providing considerable flushing and dilution of the discharge of WOC. The embayment also fluctuates in level on a seasonal basis. Watts Bar Reservoir is drawn down by ~2 m each autumn to provide additional storage for flood control through the winter and early spring. During this time, the upper reaches of the embayment (site 2) are an exposed mud flat, and WOC is confined to its original channel. The lower site in WOC embayment is deeper and narrower. Winter pool levels do not expose mud flats, and numerous fallen trees provide cover for fish. A wire

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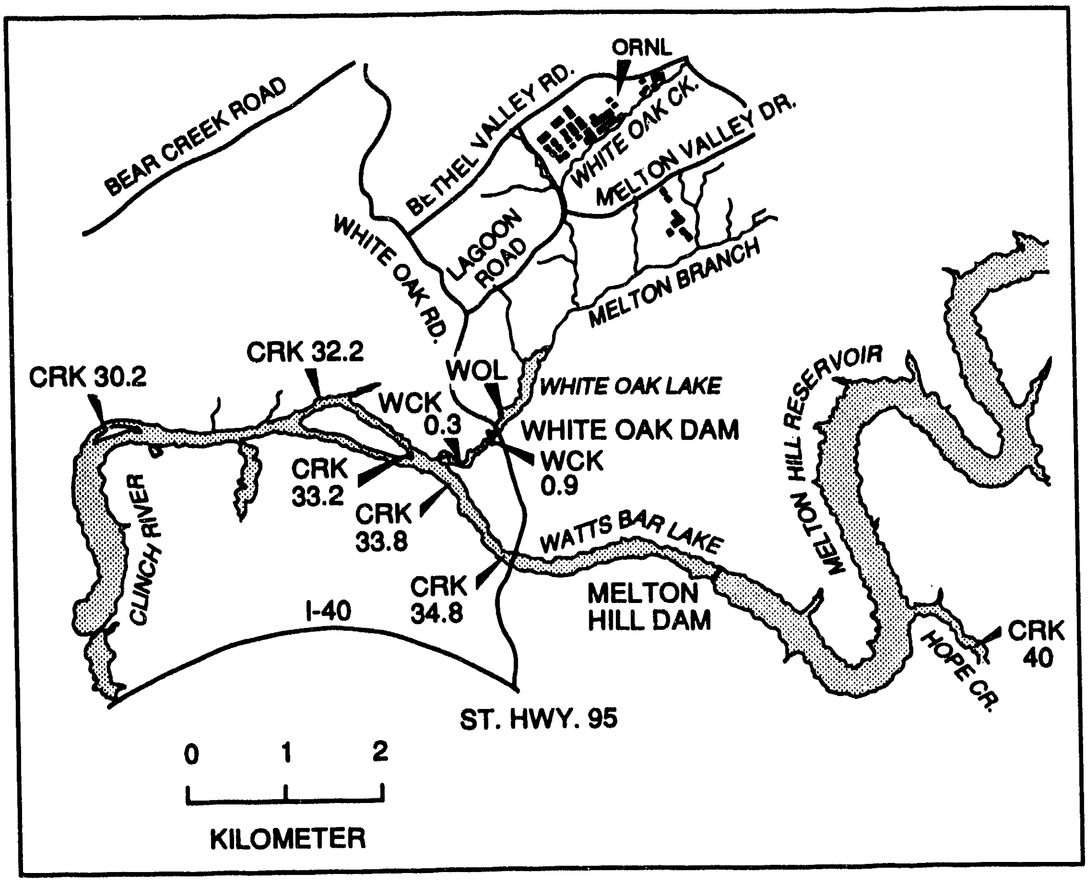

Fig. 4.1. Location of catfish collection sites in White Oak Creek embayment, White Oak Lake (WOL), and the Clinch River. CRK = Clinch River kilometer; WCK = White Oak Creek kilometer. CRK 40 is located in Melton Hill Reservoir.

fence with a sign warning of a radiation hazard blocks public access to WOC embayment at the point where it joins the Clinch River.

Five sites were sampled in the Clinch River/Watts Bar Reservoir. Two sites (sites 4 and 6) were close to the mouth of WOC embayment, 0.3 km upstream and downstream. Another pair of sites (sites 5 and 7) were located 1.0 km upstream and downstream from the pair closest to the embayment, and the most remote site (site 8) was located 3.3 km downstream from WOC. Flow in this reach of the Clinch River fluctuates daily in response to the regulated discharge at Melton Hill Dam. When discharging, flow is moderately fast and, in summer, cool (18-20°C) as a result of hypolimnetic discharges from the upstream reservoirs. The river channel is scoured in this reach and has a substrate of exposed bedrock (Loar et al. 1981a). Hope Creek embayment of Melton Hill Reservoir (site 9) was sampled as a measure of contamination at an upstream site relatively inaccessible to fish from WOC embayment.

The sampling program attempted to capture eight channel catfish (*Ictalurus punctatus*) weighing 440 g or more at each site. This objective was not met in WOL, where the only catfish collected were yellow bullhead (*Ictalurus natalis*) and black bullhead (*Ictalurus melas*). They were judged to be the best surrogate species for channel catfish in WOL and were therefore included in the study. Blue catfish (*Ictalurus furcatus*) and flathead catfish (*Pylodictus olivaris*) were occasionally collected and kept in case adequate numbers of channel catfish could not be obtained at a site. While eight channel catfish were not collected at all sites, the numbers of these other species were neither adequate nor appropriately distributed to include in the study.

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 and measured, and the dorsal fin was removed for possible age determination. The fish was then fileted and the skin removed. After rinsing in running water, one filet was wrapped in heavy-duty aluminum foil and stored in a locked freezer at -20°C for archival purposes, the other was frozen and then ground three times in a hand-operated meat grinder. A 10- to 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 ACD for analysis.

A 4- to 5-cm portion of the vertebral column was removed from the tail of each fish, air dried, wrapped, labeled, and submitted to ACD for analysis for ^{90}Sr. PCBs in fish were also analyzed by ACD through the use of procedure EPA 600/4-81-055 (EPA 1980b). This procedure utilizes extraction with methylene chloride followed by adsorption column cleanup, solvent exchange, and evaporative concentration prior to analysis by packed-column gas chromatography using electron capture detection. Strontium-90 was determined by beta counting with the use of 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).

Statistical evaluations of the data were made by using SAS procedures and software (SAS 1985a, 1986b) for ANOVA; Duncan's multiple range test; linear regression analysis; and the calculation of means, standard deviations, standard errors, and coefficients of variation.

Quality assurance was maintained by 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 fish reference standards and uncontaminated fish spiked with PCBs. Details and results are summarized in Appendix C.

4.2.3 Results

PCBs were detected in all catfish collected in WOC embayment and the Clinch River/Watts Bar Reservoir (Table 4.2 and Appendix D). The PCB mixtures recovered from fish contained predominantly tetrachlorobiphenyl and pentachlorobiphenyl congeners characteristic of Arochlor 1254 and 1260 commercial mixtures, in roughly equal amounts. Total PCBs (PCB-1254 + PCB-1260) ranged from a high of 2.6 ppm in a fish collected in WOC embayment to a low of 0.1 ppm in a fish collected at CRK 33.9. The largest number of fish containing PCBs in excess of the FDA tolerance limit of 2 ppm was found in WOC embayment, where 25% of the fish (3 of 12) contained more than 2 ppm. Only 5% of the fish (2 of 39) captured in the Clinch River/Watts Bar Reservoir exceeded the statutory limit, one from 0.3 km downstream from the mouth of WOC embayment (CRK 33.2) and the other from a site 3 km farther downstream.

Contaminants having long biological half-lives tend to accumulate to ever-higher levels throughout much of a fish's lifetime. Thus, larger, older fish may contain higher contaminant levels than smaller, younger fish. The effects of such a bias can be significant if fish collected from some sites are predominantly small while those at other sites are predominantly large. Such effects can often be avoided by collecting adult fish of similar size. Therefore, channel catfish weighing less than 440 g were not included in this study if adequate numbers of larger fish were collected at a site. Mean weights of collections were similar at all sites (640–780 g), except at CRK 33.8, where the mean weight was 1400 g. Results of regressions of ^{90}Sr, total PCB, PCB-1254, and PCB-1260 vs fish weight at each site showed no significant relationship (slope not different from zero, $p > 0.05$) in 31 of 32 possible comparisons. Although a significant relationship was noted for PCB-1260 at the site with the highest catfish mean weight, it was well within the probability expected using $p = 0.05$. Therefore, comparisons of mean contaminant levels among sites could be made without normalizing for variations in fish weight.

The highest mean concentration of PCBs in catfish (1.4 ppm) was found in fish collected at WCK 0.3, a site in WOC embayment 0.3 km upstream from the confluence with the Clinch River. However, statistical analysis using ANOVA and Duncan's multiple range test indicated no significant differences in mean PCB levels among sampling sites, except at the site 8 km upstream on Melton Hill Reservoir, which was significantly lower than WCK 0.3 ($p < 0.05$). The levels of PCB-1254 in catfish from WOL, WOC embayment, and the Clinch River 1.3 km upstream from the mouth of WOC (i.e., CRK 34.8) were not significantly different, but of these sites only the embayment site nearest the river (WCK 0.3) had a level of PCBs significantly higher than the remaining sites (CRK 33.8, CRK 33.2, CRK 32.2, CRK 30.2, and Melton Hill Reservoir). Levels of PCB-1260 did not differ significantly among any of the sampling sites.

Table 4.2. Levels of polychlorinated biphenyls (PCBs) in milligrams per kilogram wet weight and ^{90}Sr in becquerels per kilogram of bone in White Oak Creek/Clinch River catfish[a]

Site[b]	PCB-total	PCB-1254	PCB-1260	^{90}Sr
WOL	0.97±0.47 0.59–1.76 (5)	0.52±0.27 0.29–0.96 (5)	0.46±0.21 0.20–0.80 (5)	4420±672 3700–5500 (5)
WCK 0.9	0.80±0.16 0.68–0.91 (2)	0.49±0.15 0.38–0.59 (2)	0.31±0.01 0.30–0.32 (2)	1035±516 670–1400 (2)
WCK 0.3	1.4± 0.73 0.32–2.62 (10)	0.87±0.54 0.09–1.70 (10)	0.53±0.23 0.23–0.92 (10)	676±478 120–1500 (10)
CRK 33.8	0.62±0.44 0.10–1.35 (8)	0.19±0.17 0.02–0.46 (8)	0.43±0.34 0.07–0.89 (8)	706±299 350–1200 (7)
CRK 34.8	1.07±0.57 0.27–1.84 (8)	0.56±0.32 0.19–1.10 (8)	0.52±0.31 0.08–1.00 (8)	295±374 30–1100 (8)
CRK 33.2	0.85±0.73 0.22–2.40 (7)	0.30±0.14 0.10–0.49 (7)	0.55±0.79 0.07–2.30 (7)	1283±1009 78–2500 (7)
CRK 32.2	1.01±0.48 0.47–1.88 (8)	0.42±0.49 0.10–1.60 (8)	0.60±0.32 0.28–1.30 (8)	57±43 15–120 (8)
CRK 30.2	0.88±0.64 0.32–2.36 (8)	0.29±0.16 0.10–0.56 (8)	0.59±0.52 0.16–1.80 (8)	22±8 14–37 (8)
CRK 40	0.46±0.22 0.20–0.80 (6)	0.14±0.10 0.01–0.29 (6)	0.32±0.24 0.10–0.79 (6)	14±17 1–44 (6)

[a]For each PCB and for ^{90}Sr, the first value listed is mean ± 1 s.d.; the second value is range; and the third value, indicated parenthetically, is number of samples.

[b]CRK = Clinch River kilometer; WCK = White Oak Creek kilometer; WOL = White Oak Lake. WCK 0.9 and WCK 0.3 are locations in White Oak Creek embayment. CRK 40 is located in Melton Hill Reservoir. Refer to Fig. 4.1 for site locations.

Levels of contaminants accumulated by different organisms under identical exposure conditions are highly variable. Often, a lognormal distribution more appropriately describes this variability than does a normal distribution. Because the appropriateness of either underlying distribution was not obviously superior to the other in this data set, the data were also analyzed after logarithmic transformation to evaluate possible differences among sites. Such a transformation has the effect of making variances among sites more homogeneous and of reducing the influence of very high and very low values in the comparisons of means. When comparisons of PCBs among sites were made using log-transformed values, the comparisons were unchanged for total PCBs and PCB-1260. However, results of the comparison of PCB-1254 levels among sites indicated that fish from WOL, WOC, and CRK 34.8 had significantly higher levels than fish at other sites.

The levels of ^{90}Sr in catfish vertebrae exhibited much greater differences among sampling sites than did PCBs (Table 4.2). The highest mean concentration of ^{90}Sr (4420 Bq/kg) was found in bullheads from WOL, while the lowest value (14 Bq/kg) was in fish collected from Melton Hill Reservoir. Mean levels in fish collected in WOC embayment and the two Clinch River sites nearest the mouth of the embayment (at CRK 33.2 and CRK 33.8) were about 20% of those observed in fish from WOL. This may reflect both dilution of aqueous ^{90}Sr in the embayment, which receives tidelike flushings several times daily as a result of water level fluctuations in the Clinch River in response to hydroelectric discharges at Melton Hill Dam, and movement of fish in and out of the embayment. The dilution factor for the flow of WOC in the embayment was estimated from the differences in annual mean ^{3}H and ^{90}Sr concentrations measured in WOL and WOC embayment (Martin Marietta Energy Systems 1986a). Tritium is elevated in WOC due to seepage from ORNL waste disposal sites (Sect. 7.1). The dilution factor was $\sim$0.2, suggesting that ^{90}Sr differences between fish from the two sites are due primarily to dilution. Surprisingly, the mean level of ^{90}Sr in fish collected in the Clinch River at a site 1.3 km upstream from the mouth of WOC (CRK 34.8) was five times higher than that in fish collected 1.3 km downstream from the mouth (CRK 32.2) (Table 4.2). However, this difference was not statistically significant ($p > 0.05$).

Statistical evaluation of differences in ^{90}Sr levels as a function of sampling site required that the data be log transformed, since the assumption of homogeneous variances among sites was clearly not met. Results of Duncan's multiple range test indicated that ^{90}Sr levels in fish from WOL were significantly ($p < 0.05$) higher than in fish from all other sites. Mean levels of ^{90}Sr did not differ among fish collected in WOC embayment and sites in the Clinch River 0.3 km upstream and downstream of the mouth of WOC. However, fish from these four sites did contain significantly more ^{90}Sr than fish collected from sites farther from the mouth of WOC. Strontium-90 in catfish from the most remote site sampled in the Clinch River (3.3 km below the mouth of WOC) was significantly lower than in fish from a site 1.3 km above the mouth (CRK 34.8), which did not differ significantly from the site 1.3 km downstream from WOC, despite the fivefold difference noted previously. Fish from Melton Hill Reservoir, the most remote site, contained significantly less ^{90}Sr than fish from all other sites.

A comparison of the patterns of PCB-1254 and ^{90}Sr among sampling sites is shown in Figs. 4.2 and 4.3. The pattern for ^{90}Sr is that expected for downstream dilution of a point source. Strontium-90 is highest in WOL fish and decreases as it is diluted in WOC embayment and again in the Clinch River. The pattern exhibited by PCB-1254 is not obviously similar, having a maximum concentration at a site within WOC embayment and

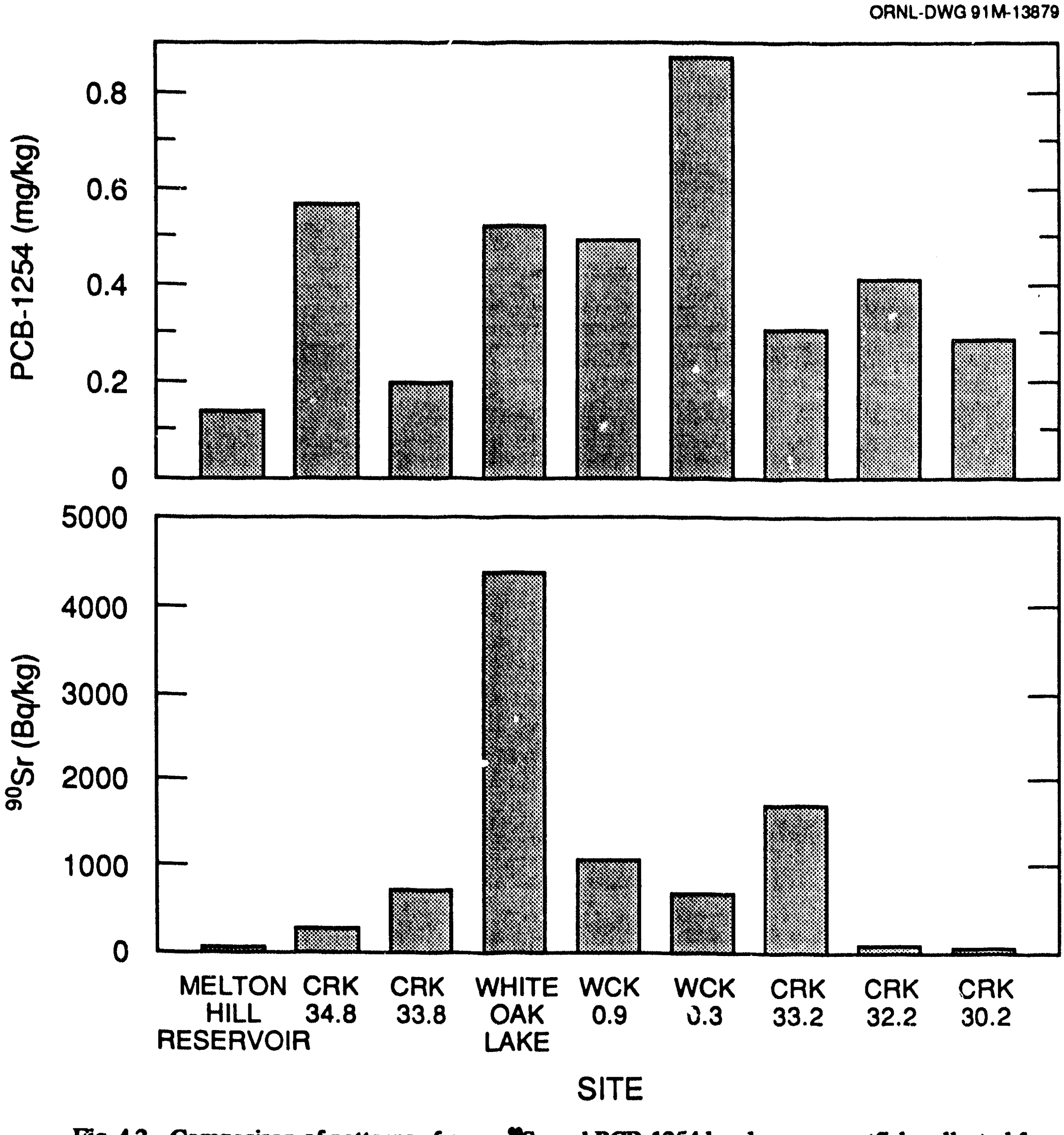

Fig. 4.2. Comparison of patterns of mean ^{90}Sr and PCB-1254 levels among catfish collected from nine sites (shown in Fig. 4.1), including White Oak Lake. CRK = Clinch River kilometer; WCK = White Oak Creek kilometer. WCK 0.9 and WCK 0.3 are locations in White Oak Creek embayment.

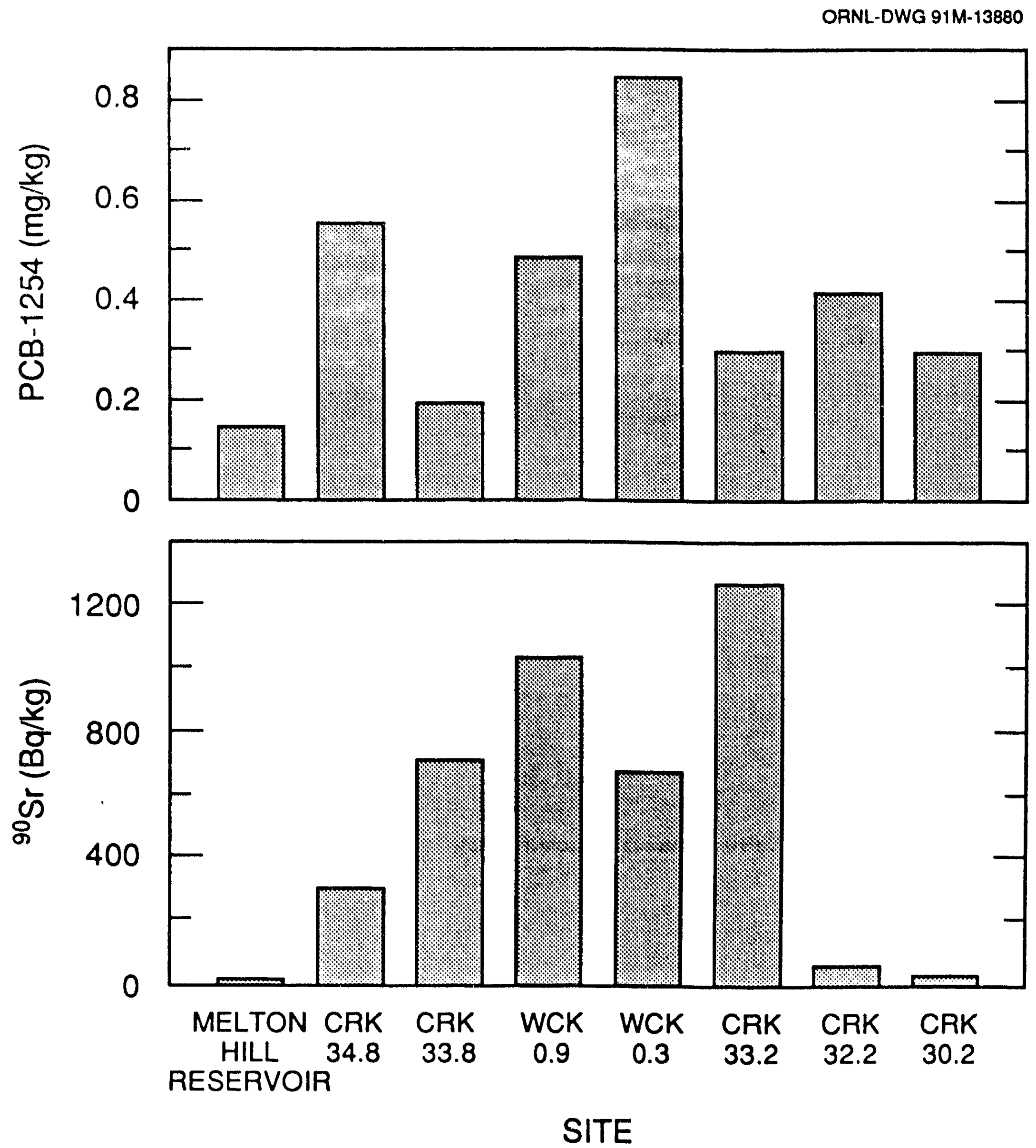

Fig. 4.3. Comparison of patterns of mean ^{90}Sr and PCB-1254 levels among catfish collected from nine sites (shown in Fig. 4.1), excluding White Oak Lake. CRK = Clinch River kilometer; WCK = White Oak Creek kilometer. WCK 0.9 and WCK 0.3 are locations in White Oak Creek embayment.

relatively similar levels at all other sites in the Clinch River and WOL. The slopes of linear regressions of [PCB-1254] vs [^{90}Sr] and [PCB-1254] vs ln[^{90}Sr] were not significantly different from zero ($p > 0.05$), while the regression of ln[PCB-1254] vs ln[^{90}Sr] was significant. Similar treatments of PCB-1260 and total PCBs were not statistically significant.

If WOC embayment was the source of PCB contamination in Clinch River catfish rather than WOL or WOC, a stronger relationship between PCBs and ^{90}Sr should have been observed when data from WOL were excluded from the analysis. When linear regressions were performed excluding that data set, the results were essentially unchanged. Once again, the only significant relationship was between ln[PCB-1254] and ln[^{90}Sr]. While this relationship was statistically significant in both cases, it explained relatively little of the variation in PCB-1254 levels that was observed, having R^2 values of 0.11 and 0.09 for the two regressions. The two significant expressions (the second excluding WOL) were virtually identical:

$$\ln[\text{PCB-1254}] = 0.154 \times \ln[^{90}\text{Sr}] - 2.02, \text{ and} \tag{1}$$

$$\ln[\text{PCB-1254}] = 0.157 \times \ln[^{90}\text{Sr}] - 2.03 \ . \tag{2}$$

If ^{90}Sr values typical of WOC embayment (1000 Bq/kg) and Melton Hill Reservoir (10 Bq/kg) are inserted into Eq. 1 or Eq. 2 to calculate PCB-1254 levels at those sites, values of 0.4 and 0.2 ppm are obtained respectively. The difference, 0.2 ppm, indicates the maximum amount of PCB accounted for by variations in ^{90}Sr.

4.2.4 Discussion

4.2.4.1 PCBs in Clinch River Catfish

In 1984, all catfish collected in WOC embayment contained PCBs in excess of the FDA limit (TVA 1985a). Two years later, 30% of the catfish collected from the site of the 1984 study (WCK 0.3) exceeded the 2-ppm limit. The mean level of PCBs in catfish from this site was 3.1 ppm in 1984, while PCBs averaged 1.4 ppm in 1986. The more highly chlorinated congeners, PCB-1260, predominated in the 1984 collection, while PCB-1254 was relatively more abundant in fish captured in 1986. Levels of PCB-1254 changed little between 1984 and 1986, averaging 0.76 ± 0.49 and 0.87 ± 0.54 (mean $\pm$ 1 s.d.), respectively. It appears as though a considerable decrease has occurred in the aqueous release of PCBs at this site in the past 2 years. A substantial fraction of channel catfish from the embayment contain excessive levels of PCBs, however. Whether or not this situation will continue to improve cannot be foreseen from the present data.

PCB contamination is clearly evident in catfish collected in the Clinch River/Watts Bar Reservoir in the vicinity of WOC. Catfish collected at the five Clinch River sites averaged 0.89 ± 0.57 ppm. These levels are not atypical of those reported in 1984 for catfish collected at sites throughout Watts Bar Reservoir (TVA 1985a). The TVA study reported a mean value of 0.9 ppm PCBs at a site 16 km downstream from WOC (the site closest to WOC in that study) and an average of 0.7 ppm for all sites in Watts Bar Reservoir. Less than 2% (1 of 64) of the catfish collected in Watts Bar Reservoir in 1984 exceeded 2 ppm PCBs. A higher proportion of fish containing more than 2 ppm PCBs was noted in 1986 (5% or 2 of 39 fish exceeded the FDA limit). The small numbers involved make statistical comparisons of these proportions tenuous; however, it seems likely that PCB levels in channel catfish in the several kilometers of the Clinch River below Melton Hill Dam are similar to concentrations observed 2 years previously farther downstream in this system.

The levels of ^{90}Sr in catfish collected in the Clinch River indicate that fish do move between the embayment and the Clinch River. Three of eight fish caught 1.3 km upstream from the mouth of WOC contained $\geq$400 Bq/kg ^{90}Sr, indicating that they had significant exposure to the discharge of WOC. Strontium-90 levels in fish collected in the Clinch River at sites 0.3 km upstream and downstream of WOC were essentially the same as those in fish caught in WOC embayment, indicating substantial localized movement in and out of the embayment. The two sites farther downstream did not produce fish with ^{90}Sr levels typical of WOC embayment. Two fish from the site 1.3 km downstream contained 110–120 Bq/kg. Such levels may indicate previous residence in WOC embayment or may be due to the WOC discharge plume remaining near the north bank of the Clinch River for much of the 1.3-km reach. At the site 3.3 km downstream, ^{90}Sr ranged from 11 to 37 Bq/kg. If the ^{90}Sr content of vertebrae from catfish captured in WOL (4420 Bq/kg) is multiplied by the dilution factor of the flow of WOC into the Clinch River (0.0025), a ^{90}Sr content of 11 Bq/kg is calculated for fish in reaches of the Clinch River below a mixing zone close to the creek mouth. This corresponds relatively well to the mean value of 22 Bq/kg observed in fish 3.3 km downstream.

It appears as though most fish caught in the study reach of the Clinch River/Watts Bar Reservoir in summer contain levels of long-lived contaminants representative of exposure regimes in the environment within 1 or 2 km of the site of capture. Catfish that reside for periods of time in WOC embayment move into the Clinch River, where they are accessible to sport fishermen. Such fish appear to be more likely to be found upstream of the mouth of WOC than downstream. Because fishing pressure is higher in the upstream tailwater, such movements could increase the likelihood of those fish being caught by anglers. The extent or duration of such movements is unknown.

4.2.4.2 WOC as Source of PCBs

If WOC is the only source of PCBs (or, more appropriately, PCB-1254, which was significantly higher in fish from the embayment) to fish in WOL, WOC embayment, and nearby reaches of the Clinch River, ^{90}Sr content of fish bone should be a marker for the degree and duration to which a fish is exposed to the WOC discharge. Patterns of ^{90}Sr and PCB-1254 content of catfish among sites should be similar, and ^{90}Sr and PCB-1254 levels in fish should be highly correlated. Examination of Fig. 4.2 shows little similarity between the patterns of ^{90}Sr and PCB-1254 among sampling sites. If contaminated sediments in the embayment are the major source of PCB contamination, ^{90}Sr would still be a marker for PCB exposure at all sites except WOL. However, the patterns of ^{90}Sr and PCB-1254 among sites (with WOL excluded) were also not similar (Fig. 4.3), and the only significant relationship was a weak correlation between ln[^{90}Sr] and ln[PCB-1254]. The presence of significant levels of PCBs in fish collected in Melton Hill Reservoir and apparently ubiquitous levels approaching 1 ppm in fish from the 4.6-km reach of the Clinch River sampled in this study suggest that other sources of PCB contamination are important in this system.

Nevertheless, the data do indicate that the WOC system is a source of PCB contamination to catfish. PCBs in WOL bullheads could come only from upstream sources or residues in lake sediments. Significantly higher levels of PCB-1254 were observed in channel catfish collected in WOC embayment than in fish from Clinch River sites, and PCB-1254 was weakly correlated with ^{90}Sr levels. The levels of ^{90}Sr in WOL catfish were more than five times higher than they were in WOC embayment catfish; however PCBs, and PCB-1254 in particular, did not decrease in similar fashion, but rather showed increased levels in embayment fish (Table 4.2). While differences in accumulation of PCBs and ^{90}Sr between bullheads and channel catfish may obscure somewhat the ability to make such distinctions, the large magnitude of this difference indicates that the aqueous phase discharge of WOL is not the major source of PCBs in catfish in WOC embayment.

The elevated levels of PCBs in WOC embayment thus appear to originate from a source within the embayment, almost certainly PCB-contaminated sediments transported downstream from WOL. White Oak Dam was extensively renovated in 1981–83, a procedure that involved constructing a new spillway on the west side of the dam. During the construction period, the water level in WOL was lowered, exposing contaminated sediments along the shoreline and making the remaining submerged lake bed more susceptible to resuspension due to wind and wave action. Deep sediments near the dam face undoubtedly were transported downstream during excavation of the new spillway. These activities could have introduced a pulse of PCB-laden sediments to the embayment, resulting in the high levels of PCBs in catfish that were observed there in 1984. Since that time the pulse of contamination has probably been transported out of the embayment or been buried by more-recent sediments, accounting for the decrease in PCB levels in embayment catfish between 1984 and 1986. It is not possible to ascertain if the levels of PCBs presently observed in WOC embayment catfish represent a steady state produced by ambient exposure or if they were caused by much higher exposures several years previously. The preponderance of PCB-1254 over PCB-1260 in fish collected in 1986, in contrast to the opposite pattern in 1984, suggests that 1986 levels represent a steady state with present exposure conditions.

The downstream variation in ^{90}Sr in catfish can be used to infer the importance of WOC embayment as a source of PCB contamination of fish in the Clinch River arm of Watts Bar Reservoir. The Clinch River in the study reach is fast flowing with a highly scoured bed; fine sediments are not retained. Studies of the longitudinal distribution of PCBs in sunfish in a small stream downstream from a point source showed that PCBs declined approximately in proportion to the aqueous dilution ratio (G. R. Southworth, unpublished data). Therefore, it appears appropriate to use a dilution ratio of ^{90}Sr in fish bone to assess the importance of this source. Assuming that all PCBs in WOC embayment catfish come from the lower portion of the embayment, the predicted concentrations of PCBs in downstream fish resulting from this source were calculated by multiplying the ratio of ^{90}Sr in downstream fish to ^{90}Sr in embayment fish by the mean level of PCBs in embayment fish. The predicted PCB concentrations calculated by this method were 0.12 and 0.05 ppm at sites 1.3 and 3.3 km below the mouth of WOC respectively. In both cases, this represents a small fraction of the total PCBs observed in fish from those sites, indicating that the source of PCBs in WOL embayment appears to have little impact on total PCB levels in fish outside the immediate vicinity of the embayment.

4.2.5 Conclusions

Some channel catfish caught by fishermen in the 7-km reach of the Clinch River/Watts Bar Reservoir downstream from Melton Hill Dam are likely to contain PCBs in excess of the

2-ppm FDA limit. However, the proportion of fish containing such levels is low, approximately 5%. While a significant portion of the PCB burden of catfish captured in WOC embayment and nearby locations in the Clinch River may result from recent or ongoing releases from sources within the embayment or upstream, little of the total PCB content of fish from other sites in the Clinch River can be attributed to these sources.

Channel catfish residing in WOC embayment, which is closed to public access, move in and out of the embayment enough to be vulnerable to capture in nearby waters open to public fishing. Contaminant levels in these fish should be evaluated by using the assumption that such fish are available for consumption by sport fishermen.

4.3 FUTURE STUDIES

The presence of elevated levels of mercury in fish from WOC indicates a need to proceed to Subtask 2b of BMAP: identification of contaminant sources (Loar et al. 1991). Further studies will be designed and implemented in this subtask after discussion with the DEM staff responsible for implementing the NPDES Mercury Monitoring Plan (Sect. 10).

Annual monitoring of PCB contamination in channel catfish in WOC embayment will be continued in order to detect any increase in PCB levels at this site as remedial actions are carried out in WOC watershed. In conjunction with this activity, organisms used as food by catfish will be analyzed for PCBs, and the role of the food-chain pathway in determining PCB levels in Clinch River catfish will be evaluated.

The annual screening of fish in WOC for metals and organic contaminants outlined in BMAP will be continued; results of the initial screening for metals (other than mercury) and organics in fish collected from seven sites in WOC watershed in winter 1986-87 will be presented in the second annual BMAP report for ORNL. Efforts will be initiated on Subtask 2c: integration of water quality and bioaccumulation data (Loar et al. 1991). This subtask will evaluate the effectiveness of bioconcentration data in detecting exposure to contaminants measured in routine water quality monitoring programs (Sects. 2.2.3 and 2.2.4) or inferred from bioindicators (Sect. 5), such as liver detoxification enzyme levels.

5. BIOLOGICAL INDICATORS OF CONTAMINANT-RELATED STRESS

5.1 INTRODUCTION

This task involves the development, screening, and application of biological indicators to evaluate the responses of fish populations in WOC watershed to point- and area-sources of contamination (Task 3 of BMAP; Loar et al. 1991). Biological indicators have an advantage over other biomonitoring approaches in that they (1) can provide early warning signals of potential ecological effects because of their sensitivity to water quality degradation; (2) can be used to establish the relationship between water quality degradation and important biological responses, such as growth; (3) can be used to evaluate the effects of remedial actions on water quality; and (4) are, in general, easy to measure and cost-effective for long-term biomonitoring.

During the past year two major subtasks were addressed: selection/screening of bioindicators (Subtask 3a) and application of bioindicators for long-term field monitoring (Subtask 3b). The strategy utilized in the design and application of these studies was to measure a suite of indicators representing a series of biological responses along a gradient of relatively short-term to long-term responses, to analyze and evaluate these indicators for their ability to provide the information related to advantages (1) through (3) in the preceding paragraph, and to select a subset of the most relevant indicators for long-term monitoring. The basic approach involved a comparison of both individual and integrated biological responses of fish from various areas in WOC watershed with the responses of fish from noncontaminated reference areas.

5.2 METHODS

Sampling was conducted during late November and early December 1986 at sites immediately below the weir at WCK 3.41, above the weir at WCK 2.65, above the weir at MEK 0.16, in WOL, and in Brushy Fork (BFK 7.6), a reference stream (Sect. 2.3). From 10 to 14 adult redbreast sunfish were electroshocked at each site, and blood samples were obtained from each fish with the use of unheparinized syringes within 1-2 min following capture. Each fish was identified with a numbered tag for future reference and transported alive to the laboratory for analysis of various tissues and organs. Because no centrarchids (sunfishes) were found above the weir at MEK 0.16, this site could not be included in the data set as initially planned.

5.2.1 Analytical Procedures

Total lengths and weights of each fish were recorded before the liver, gonads, gills, and kidney were removed for analysis. Liver and ovaries were weighed, and sections of each organ were preserved for histopathological analyses. The remainder of the liver was used for enzyme assays and deoxyribonucleic acid (DNA)-ribonucleic acid (RNA) determinations. Blood samples were centrifuged and the resultant serum frozen for subsequent biochemical analysis. Blood analysis was performed on a centrifugal fast analyzer in the Health Department at ORNL by using the procedures described by Adams et al. (1985).

Livers removed from fish for enzyme assays were homogenized in chilled 0.1 *M* phosphate buffer (pH 7.5) and 0.15 *M* KCl. The homogenates were centrifuged at 3000 × *g* for 10 min and at 10,000 × *g* for 20 min.* The resulting supernatants were centrifuged at 105,000 × *g* for 60 min and resuspended in 0.1 *M* Tris buffer (pH 7.4), 1 m*M* ethylenediaminetetraacetic acid (EDTA), and 10% glycerol by sonication. Proteins were determined by the Bio-Rad reagent method using bovine serum albumin as a standard. The activity of 7-ethoxyresorufin O-deethylase (EROD) was measured fluorometrically at 30°C with a centrifugal fast analyzer coupled to an argon-ion laser and computer. The assay was performed in HEPES buffer (pH 7.8) with 5 m*M* magnesium acetate, 1 m*M* 7-ethoxyresorufin, 0.25 m*M* nicotinamide-adenine dinucleotide phosphate (NADPH), and 1 m*M* EDTA (Egan et al. 1983). The electron transfer enzyme NADPH-cytochrome *c* reductase was assayed spectrophotometrically by a modification of Phillips and Langdon (1962) with cytochrome *c* as the electron donor. Reduced cytochrome *c* was determined by using an extinction coefficient of $21.1 \cdot cm^{-1} \cdot mM^{-1}$. The reaction mixture contained 50 m*M* Tris (pH 7.4), 20% glycerol, 1 m*M* dithiothreitol (DTT), 1 m*M* EDTA, 1.1 mg/mL horse heart cytochrome *c*, 0.175 m*M* NADPH, and 2–10 µg of microsomal protein. Cytochrome P-450 and b_5 were each assayed by their characteristic oxidized and reduced spectra (Omura and Sato 1964) with several modifications (Johannesen and DePierre 1978). For cytochrome P-450, samples were oxidized with CO and reduced with sodium diothionite; cytochrome b_5 was reduced with nicotinamide-adenine dinucleotide (NADH).

Sections of liver tissue were extracted for nucleic acid (RNA and DNA) content by using a modification of the Schmidt-Thannhauser method (Munro and Fleck 1966). The RNA and DNA in the resulting supernatants were quantified spectrophotometrically at maximum UV absorbance. Liver proteins were analyzed by a modification of the Lowry method (Hartree 1972).

5.2.2 Statistical Procedures

To test for differences in each bioindicator between sites, and between male and female fish within each site, ANOVA using SAS procedures was applied. Due to unequal sample sizes and some unpaired data sets, tests appropriate to assumptions of unequal and unpaired data were applied. Statistical significance of interactions attributed to sex and sampling sites was also considered in the models of main effects.

The significance of integrated biological responses between fish from various sites was determined by canonical discriminant analysis (Seal 1968). This method provides a graphical representation of the positions and orientations of the various site responses relative to each other. In addition, the discriminant analysis variable selection technique of McCabe (1975) was used to determine if particular subsets of the full set of variables could discriminate as well among the four sites as could the full set of variables. 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.

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

5.3 RESULTS AND DISCUSSION

The objectives of the initial studies were to measure a suite of bioindicators over a range of short- to long-term biological responses (Subtask 3a) and, based on the analyses presented in this report, to select the most ecologically relevant and cost-effective indicators (Subtask 3b). Blood biochemistry, liver enzymes, indices of overall body condition, and indicators of growth (RNA:DNA ratio) were quantified in fish collected during the late fall of 1986 at three sites in WOC watershed and a reference site. Data are presented first for individual bioindicators and then for integrated bioindicator responses.

5.3.1 Individual Indicator Responses

The individual indicator responses measured in this study can be conveniently placed into one of five groups: (1) protein metabolism [serum bilirubin, serum glutamic oxaloacetic transaminase (SGOT), and total protein]; (2) carbohydrate-lipid metabolism (serum glucose, triglycerides, and cholesterol); (3) condition indices, including the liver-somatic index, visceral-somatic index, and condition factor; (4) liver (detoxification) enzymes; and (5) growth. These groups are discussed in the following paragraphs.

5.3.1.1 Indicators of protein metabolism

The transaminase enzyme (SGOT), total serum proteins, and bilirubin were used as indicators of protein metabolism. Both SGOT and bilirubin are involved in protein catabolism and, therefore, are indicative of tissue damage in organs such as the liver. There were no significant differences in the levels of SGOT or bilirubin between the reference (control) site and the three WOC sites even though there were differences among the WOC sites themselves (Fig. 5.1 and Table 5.1). In contrast, total serum protein in Brushy Fork fish was significantly higher than in fish from WOL (Fig. 5.1 and Table 5.1). Total serum protein may indicate differences in overall protein metabolism between fish experiencing varying levels of environmental stress. Preliminary studies of reproductive success in another local stream indicate that vitellogenin, a high-density lipoprotein, is highly correlated with total serum protein (S. M. Adams, unpublished data).

5.3.1.2 Indicators of carbohydrate and lipid metabolism

Serum glucose was measured as an indicator of carbohydrate metabolism. There were no significant differences in this parameter between fish sampled from the various sites (Table 5.1) even though the reference site appeared to have the lowest concentrations (Fig. 5.2). Hyperglycemia is a generalized stress response in fish to a broad spectrum of environmental perturbations (Silbergeld 1974) and, therefore, may not be a useful indicator to single out the effects of toxicant stress in WOC fish.

Serum triglycerides and cholesterol both serve as measures of lipid metabolism and nutritional status. For triglycerides, there were significant differences (1) between the reference site and all the WOC sites and (2) between WOC sites (Table 5.1). Fish from WCK 3.4, WCK 2.7, and WOL had significantly lower levels of triglycerides than did fish from the reference site (Table 5.1). However, cholesterol levels showed no clear pattern of site differences (Table 5.1) even though cholesterol levels for WOC fish were lower (Fig. 5.2).

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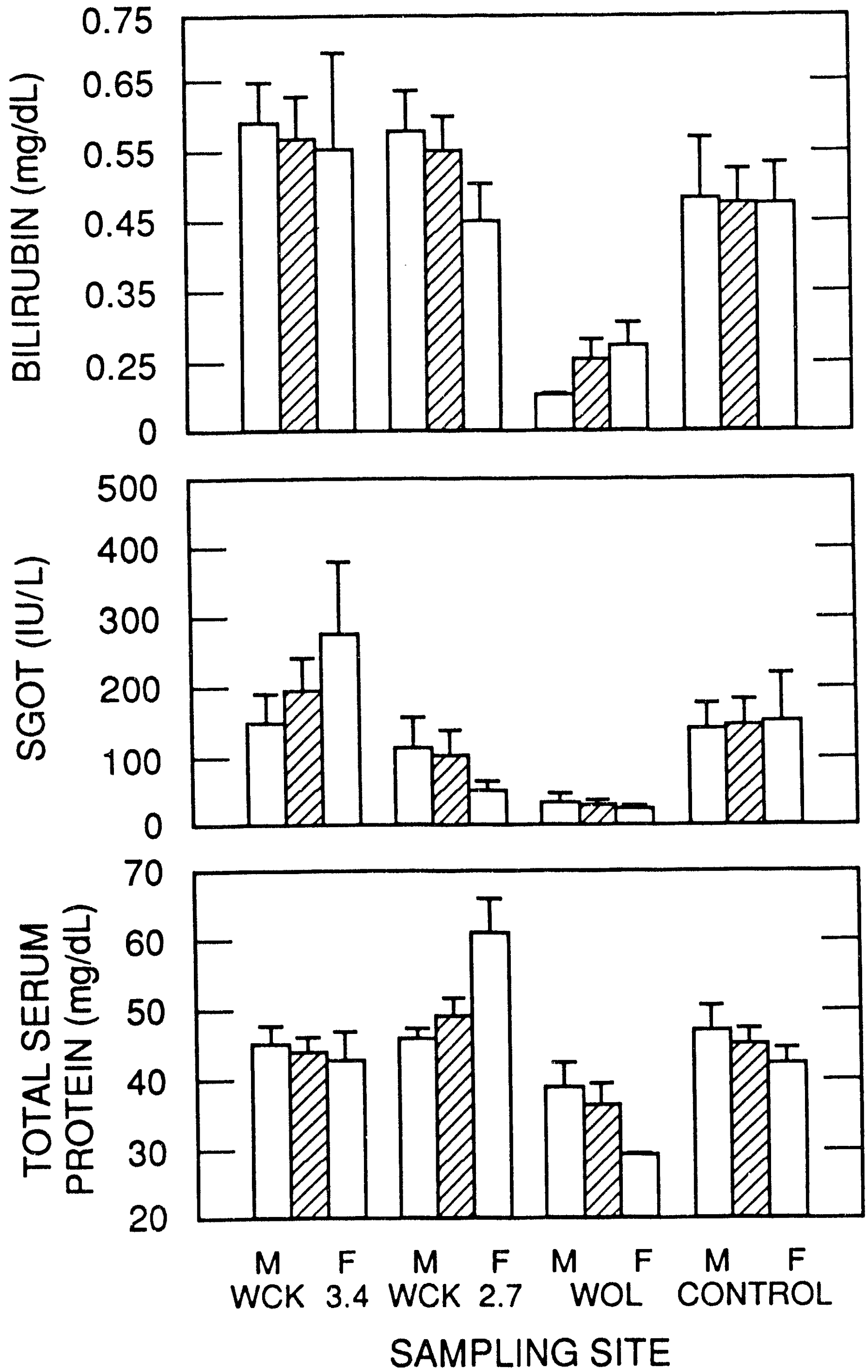

Fig. 5.1. Response of protein metabolism indicators [bilirubin, serum glutamic oxaloacetic transaminase (SGOT), and total serum protein] in redbreast sunfish from sampling sites in White Oak Creek [White Oak Creek kilometer (WCK) 3.4 and WCK 2.7], White Oak Lake (WOL), and Brushy Fork (reference or control stream) in fall 1986. For each site, the values for males (M), females (F), and the average of the two sexes (hatched bars) are given. Lines extending above the bars indicate 1 s.e. of the mean. IU = international units.

Table 5.1. Statistical comparisons between sites for individual bioindicators

Bioindicator[a]	Site comparisons[b]				
	Reference vs WCK 3.4	Reference vs WCK 2.7	Reference vs WOL	Reference vs all WOC and WOL	Among WOC sites
Serum bilirubin					*
Total serum protein		–	+		**
SGOT					*
Serum glucose					
Serum triglycerides	+	++	++	**	*
Serum cholesterol					
Cytochrome P-450	– –	–		**	*
Cytochrome b_5	– –	– –		**	**
NADPH	– –	– –		**	**
Condition factor		–		*	
Visceral-somatic index			+		
Liver-somatic index	++	++	++	**	
RNA:DNA ratio	++	++	++	**	
RNA	++	+	+	**	

[a]SGOT = serum glutamic oxaloacetic transaminase; NADPH = nicotinamide-adenine dinucleotide phosphate; RNA = ribonucleic acid; DNA = deoxyribonucleic acid.

[b]The symbols ++ and + denote values at Brushy Fork (reference stream) significantly higher at the 99 and 95% confidence levels, respectively, than at the comparison site; the symbols – – and – denote values at Brushy Fork significantly lower at the 99 and 95% confidence levels, respectively, than at the comparison site. The symbols ** and * indicate statistical differences at the 99 and 95% confidence levels respectively. WCK = White Oak Creek kilometer; WOC = White Oak Creek; WOL = White Oak Lake.

Blood triglyceride levels can indicate both the magnitude of feeding (nutrition) and the mobilization of energy (fat) reserves during stress, if adequate fat reserves are available. Lipids can be mobilized by fish to partially mediate the effects of stress (Lee et al. 1983).

Serum triglyceride concentrations differed among sites and may serve as indicators of nutritional status and/or chronic stress. Which, if either, of these two general response categories is most applicable in interpreting site differences will be investigated further. This evaluation can be accomplished, in part, through integration of other indicators of nutrition, such as feeding habits, to partition nutritional responses from metabolic stress responses. Cholesterol does not appear to be as good an indicator as triglycerides and will probably be excluded from future analyses.

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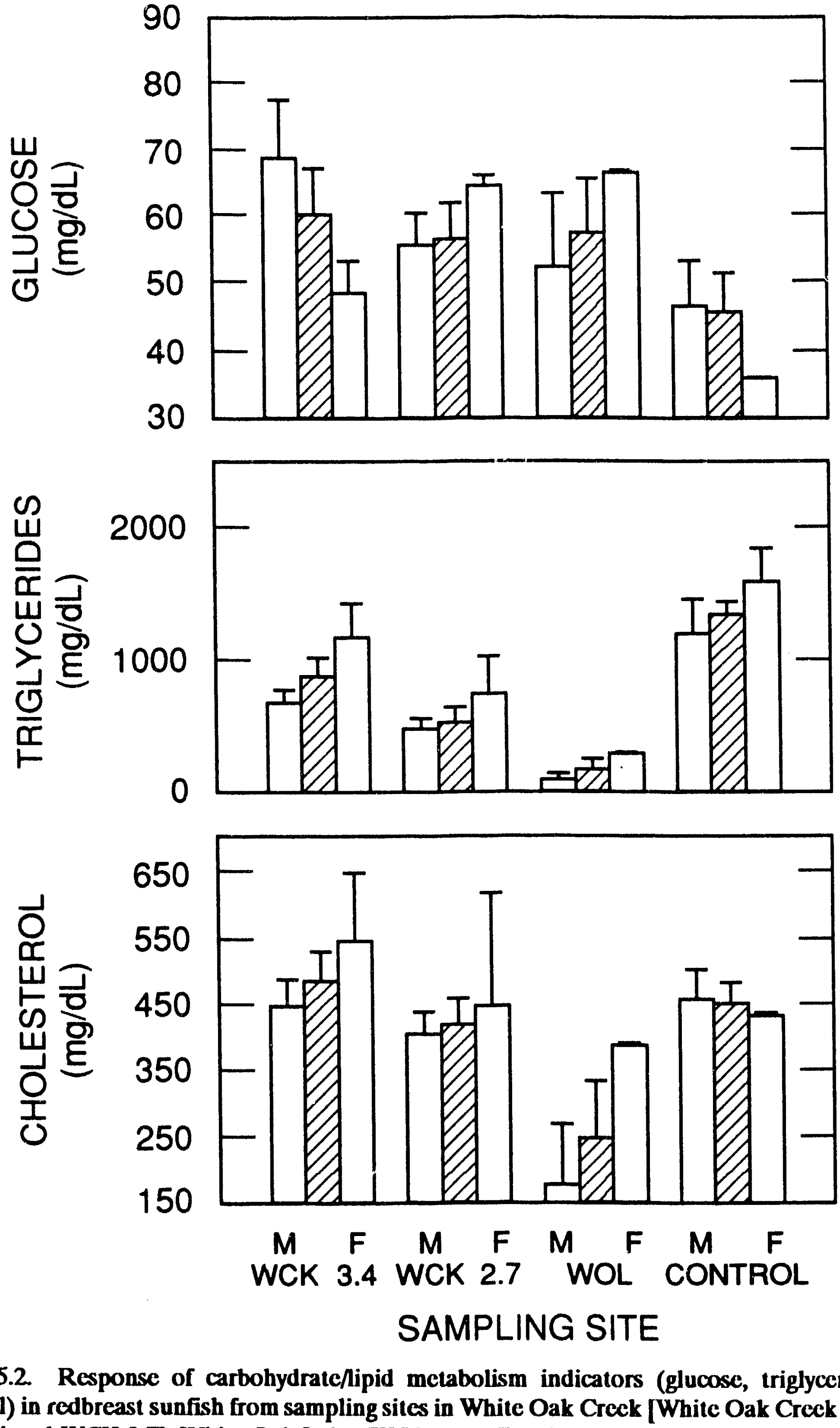

Fig. 5.2. Response of carbohydrate/lipid metabolism indicators (glucose, triglycerides, and cholesterol) in redbreast sunfish from sampling sites in White Oak Creek [White Oak Creek kilometer (WCK) 3.4 and WCK 2.7], White Oak Lake (WOL), and Brushy Fork (reference or control stream) in fall 1986. For each site, the values for males (M), females (F), and the average of the two sexes (hatched bars) are given. Lines extending above the bars indicate 1 s.e. of the mean.

5.3.1.3 Condition indices

The condition factor [(total body weight)/(total body length)^3] is a generalized indicator of overall body fitness or "plumpness" of a fish. Because this factor is a generalized indicator, it reflects both the consequences of feeding and the degree of metabolic stress. The mean condition of fish at the reference site was significantly lower than that of fish at WCK 2.7 (Table 5.1, Fig. 5.3), although the condition of fish at all sites appeared similar (Fig. 5.3).

The visceral-somatic index, or VSI [(total visceral weight – stomach contents)/(total body weight)], is used as a general indicator of lipid reserves in fish. Most sunfishes store lipids in the mesenteries of the viscera, making VSI a cost-effective indicator of energy reserves. Fish from WOL had a significantly lower average VSI than that in reference fish (Table 5.1 and Fig. 5.3). There was no significant difference in the average VSI among fish in WOC and WOL even though the WOL fish appeared to have a lower average VSI (Fig. 5.3).

Of all the indicators measured in this study, the liver-somatic index, or LSI [(total liver weight – gall bladder)/(total body weight)], appears to be one of the most useful and informative. Not only did LSI vary with sex of fish, but large differences were observed in LSI between sites (Fig. 5.3, Table 5.1). LSI reflects both short-term nutritional status and metabolic energy demands (Heidinger and Crawford 1977, Adams and McLean 1985). In addition, LSI is a sensitive indicator of toxicant stress; liver enlargement (hyperplasia) has been reported in fish exposed to various types of pollutants (Chambers 1979, Poels et al. 1980, Sloof et al. 1983). Even though no significant differences were observed between sites in the WOC system (Table 5.1), LSI was significantly higher in fish from the reference site than in fish collected from both WOC sites and WOL (Table 5.1). The higher LSI in fish from Brushy Fork probably indicates that these fish were experiencing less metabolic stress and higher nutritional levels than the fish in the WOC system.

5.3.1.4 Liver detoxification enzymes

Liver detoxification enzymes, such as cytochrome P-450-dependent monooxygenases, play a major role in the biotransformation or metabolism of xenobiotics, such as pesticides, hydrocarbons, chlorinated hydrocarbons, and numerous biological molecules, including steroid hormones (Payne and Penrose 1975, Lidman et al. 1976, Stegeman 1981). This detoxification system is composed of membrane-bound hemoproteins, which coordinate the substrate and molecular oxygen at its active site, and has been referred to as the mixed function oxydase (MFO) system. Other enzymes associated with this detoxification system are the electron transfer enzymes, which transfer electrons to cytochrome P-450, and thus are important in the oxidation of xenobiotics. The electron transfer enzymes measured in this study include cytochrome b_5 and NADPH cytochrome P-450 reductase.

Fish sampled from the reference site had significantly lower concentrations of all three enzymes (NADPH, P-450, and cytochrome b_5) than did fish collected from the two WOC sites (Table 5.1). However, levels of these three enzymes in fish from Brushy Fork and WOL were similar. Although there were no significant differences in the concentrations of P-450 between males and females at any site, females at both WOC sites had significantly higher levels of NADPH and cytochrome b_5 (WCK 3.4 only) (Fig. 5.4).

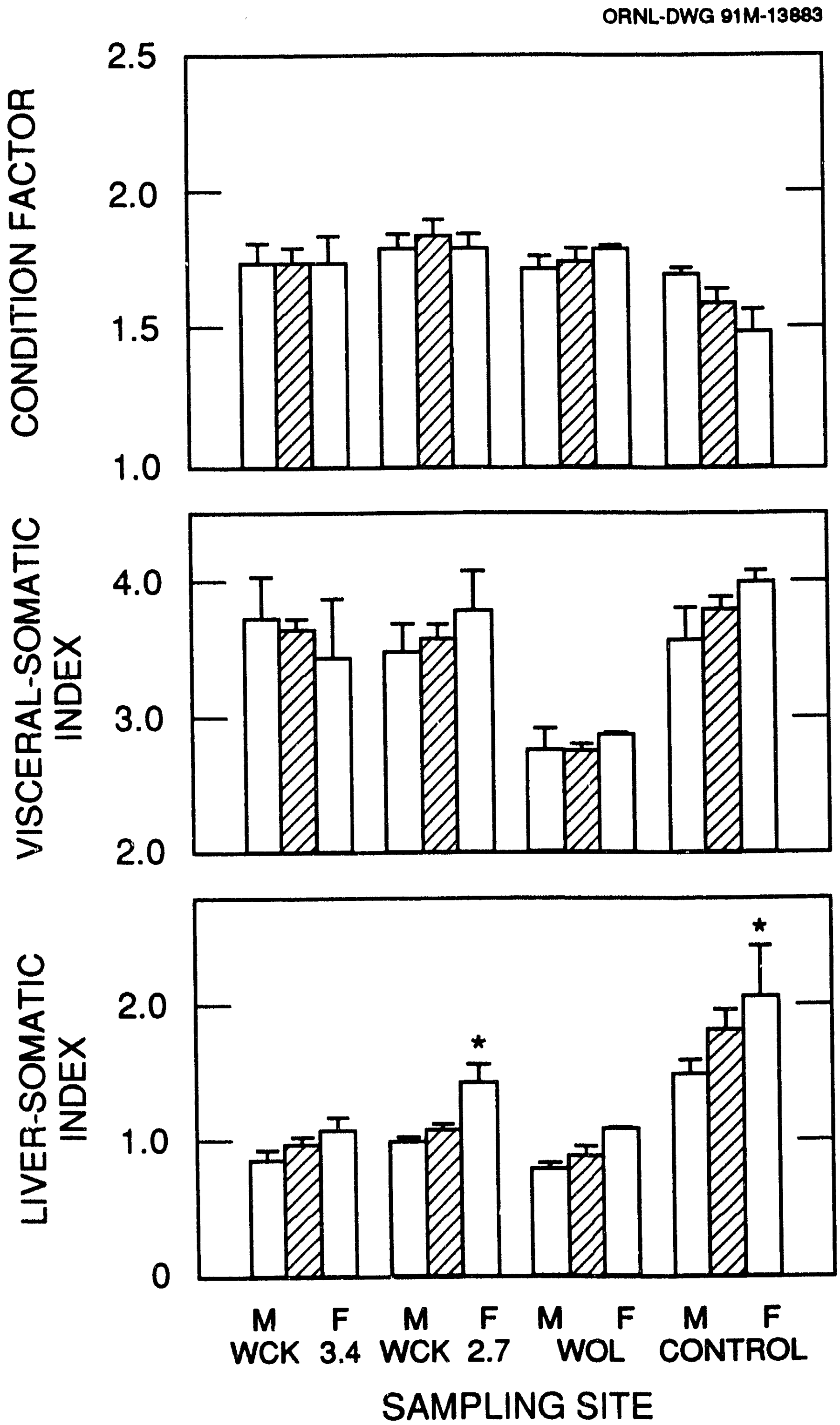

Fig. 5.3. Response of the condition indices (condition factor, visceral-somatic index, and liver-somatic index) in redbreast sunfish from sampling sites in White Oak Creek [White Oak Creek kilometer (WCK) 3.4 and WCK 2.7], White Oak Lake (WOL), and Brushy Fork (reference or control stream) in fall 1986. For each site, the values for males (M), females (F), and the average of the two sexes (hatched bars) are given. Stars above bars indicate that males are significantly different from females within a site ($p < 0.05$). Lines extending above the bars indicate 1 s.e. of the mean.

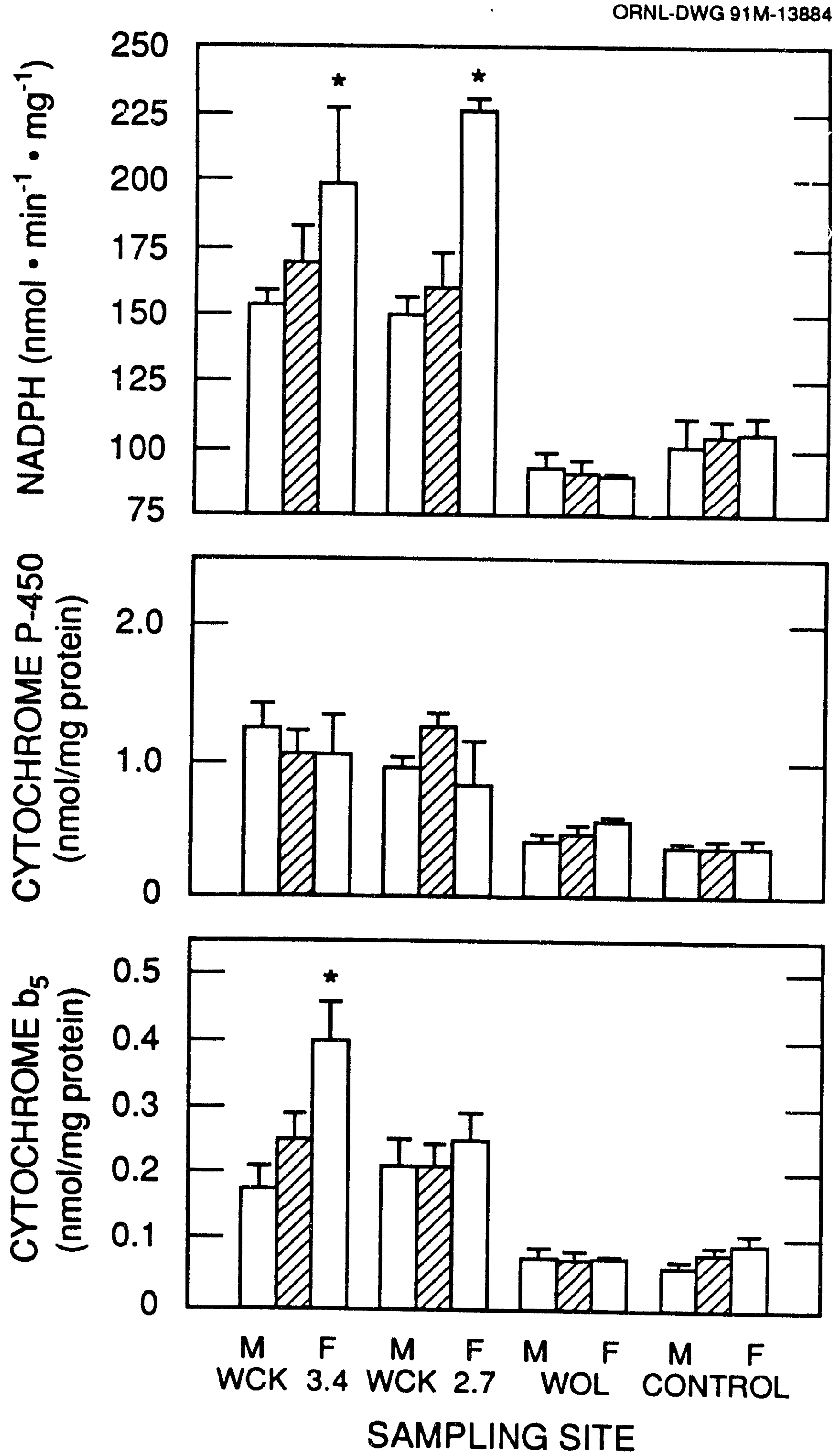

Fig. 5.4. Response of the liver detoxification enzymes [nicotinamide-adenine dinucleotide phosphate (NADPH), cytochrome P-450, and cytochrome b_5] in redbreast sunfish from sampling sites in White Oak Creek [White Oak Creek kilometer (WCK) 3.4 and WCK 2.7], White Oak Lake (WOL), and Brushy Fork (reference or control stream) in fall 1986. For each site, the values for males (M), females (F), and the average of the two sexes (hatched bars) are given. Stars above bars indicate that males are significantly different from females within a site ($p < 0.05$). Lines extending above the bars indicate 1 s.e. of the mean.

Enzyme levels in fish from the two WOC sites were significantly higher than those in fish from the reference site on Brushy Fork, but fish in WOL had levels similar to the reference fish. One explanation for this similarity is that the contaminants responsible for induction of enzymes in WOC fish are significantly diluted by WOL. In addition, sedimentation processes in the lake probably remove many contaminants from the water column. If, in fact, most contaminants are sequestered in the sediments, direct exposure of fish would decrease, but indirect exposure at a reduced level through the food chain might increase. Studies during the next year will include both diet and histopathological analyses to evaluate the causal mechanisms responsible for these observed similarities between fish in WOL and Brushy Fork.

5.3.1.5 Indicators of growth

The ratio of RNA to DNA concentrations in body tissues has been used as an indicator of immediate or short-term fish growth in natural systems (Bulow 1970, Haines 1973), as well as an indicator of exposure to sublethal concentrations of toxicants (Barron and Adelman 1984). This ratio is generally considered a more accurate index of growth than RNA concentration alone, since the ratio is not affected by cell number or size in the tissue sampled (Hotchkiss 1955).

Short-term growth of fish in Brushy Fork was significantly higher than that in fish from the two WOC sites or from WOL, as indicated by the RNA:DNA ratio (Fig. 5.5 and Table 5.1). All sites in the WOC system had similar levels of growth. Because the amount of RNA in tissue is dependent on the number and size of cells, which may not be consistent between tissues of different fish, the differences between sites in RNA levels alone were not as significant as those site differences reflected by the RNA:DNA ratio (Fig. 5.5). Based on the results of this initial study, the RNA:DNA ratio appears to be a good indicator of fish response to stress because it reflects short-term growth in situations in which growth is an integrator of many environmental variables acting on an organism. Additional studies are planned to evaluate the relative importance of water quality and other environmental factors, such as food availability, on growth responses.

5.3.2 Integrated Bioindicator Responses

To examine the integrated response of fish to water quality, a canonical discriminant analysis procedure was used to evaluate blood biochemical parameters, liver detoxification enzymes, nucleic acids, and condition indices within a multivariate context. With canonical discriminant analysis, the mean values of the first two canonical variables (the two varibles that account for most of the ability to separate integrated fish responses at each site) were plotted for fish from each site, along with the statistical confidence region (90%). Sites were considered to be significantly different if the 90% confidence radii of the site means did not overlap. This integrated response analysis consist of two components: (1) an evaluation of differences in sites based on the total set of response variables measured and (2) variable selection/screening to determine the best subset of variables for evaluating the response of fish to environmental stress.

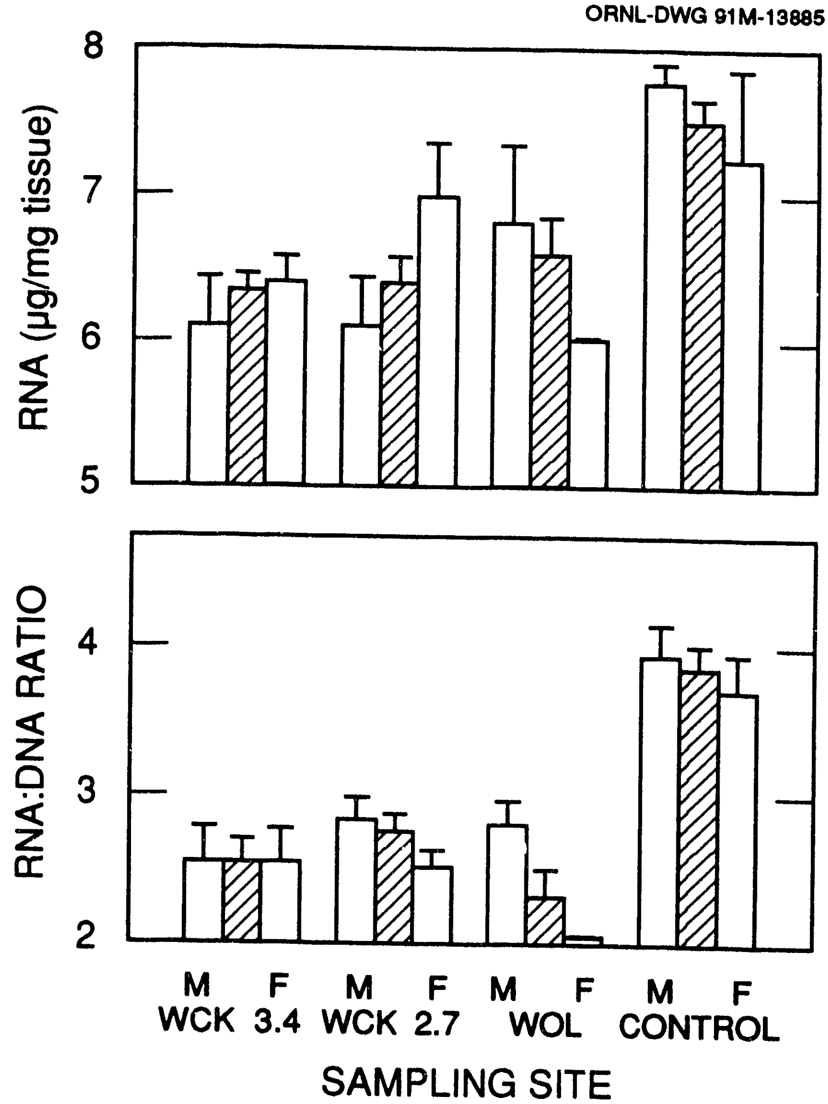

Fig. 5.5. Response of the growth indices [ribonucleic acid (RNA), RNA: deoxyribonucleic acid (DNA) ratio] in redbreast sunfish from sampling sites in White Oak Creek [White Oak Creek kilometer (WCK) 3.4 and WCK 2.7], White Oak Lake (WOL), and Brushy Fork (reference or control stream) in fall 1986. For each site, the values for males (M), females (F), and the average of the two sexes (hatched bars) are given. Lines extending above the bars indicate 1 s.e. of the mean.

5.3.2.1 Integrated site evaluation

The multivariate analysis using the full set of 13 variables indicated that fish sampled from each of the 4 sites were segregated and distinct in their integrated response (Fig. 5.6). The most dramatic difference occurred between the reference site (Brushy Fork) and the three WOC sites. (The reference fish were separated by a relatively large distance from the other sites.) Even though fish from the three WOC sites (two stream sites and WOL) were segregated from each other, they tended to cluster near each other (Fig. 5.6). Thus, the integrated responses of fish from the various sites in WOC watershed were more similar to each other than they were to the responses of fish from Brushy Fork.

5.3.2.2 Variable selection/screening

Statistical procedures were used to identify the subsets or combinations of variables that could discriminate as well between the 4 sites as could the full set of 13 variables. The U-ratio (McCabe 1975) is a statistical measure of the ability of variable subsets to discriminate between sites (i.e., in this case the integrated fish responses between sites). The higher the ratio, the less discriminatory power a particular variable subset has. A plot of the U-ratio for subsets of variables ranging from 1 to 13 (Fig. 5.7) indicated that a subset of ~7–9 variables is needed (approximate breakpoint in line) to provide results similar to those demonstrated by the full 13-variable set. This relationship was demonstrated in another manner when the integrated site responses were plotted for the full set and for subsets of 7 and 8 variables (Fig. 5.8). Segregation of sites by using the 7-variable subset did not result in the same discriminant pattern as with the full set because the 90% confidence radii of sites 2 and 3 overlapped (Fig. 5.8). However, by using an 8-variable subset, all sites were segregated in a manner similar to the pattern observed for the full set of variables. Through the use of this screening procedure and the field data obtained to date, the variables that appeared to provide the most useful information and, consequently, have been selected for future studies include (listed in order of importance) (1) the RNA:DNA ratio, (2) SGOT, (3) LSI, (4) serum protein, (5) P-450 enzyme, (6) cytochrome b_5 electron-transport enzyme, (7) glucose, and (8) bilirubin.

5.3.3 Major Response Categories of Bioindicators

The bioindicators measured in this study can be classified into one of three major groups, depending on the general types of causal agents ultimately responsible for each type of response (Table 5.2): (1) indicators of water quality or those that reflect direct (metabolic) effects of contaminant stress, such as the liver detoxification enzymes, bilirubin, serum protein, SGOT, and LSI, if liver enzymes are also high; (2) indicators of the nutritional status of fish or those that reflect indirect effects of environmental conditions mediated through the food chain (see Fig. 7 in Loar et al. 1991), including triglycerides, cholesterol, and LSI, if liver enzymes are low; and (3) indicators that are integrative in nature and reflect both direct metabolic effects and indirect nutritional effects of toxicant stress, such as the RNA:DNA ratio and other indicators of fish growth.

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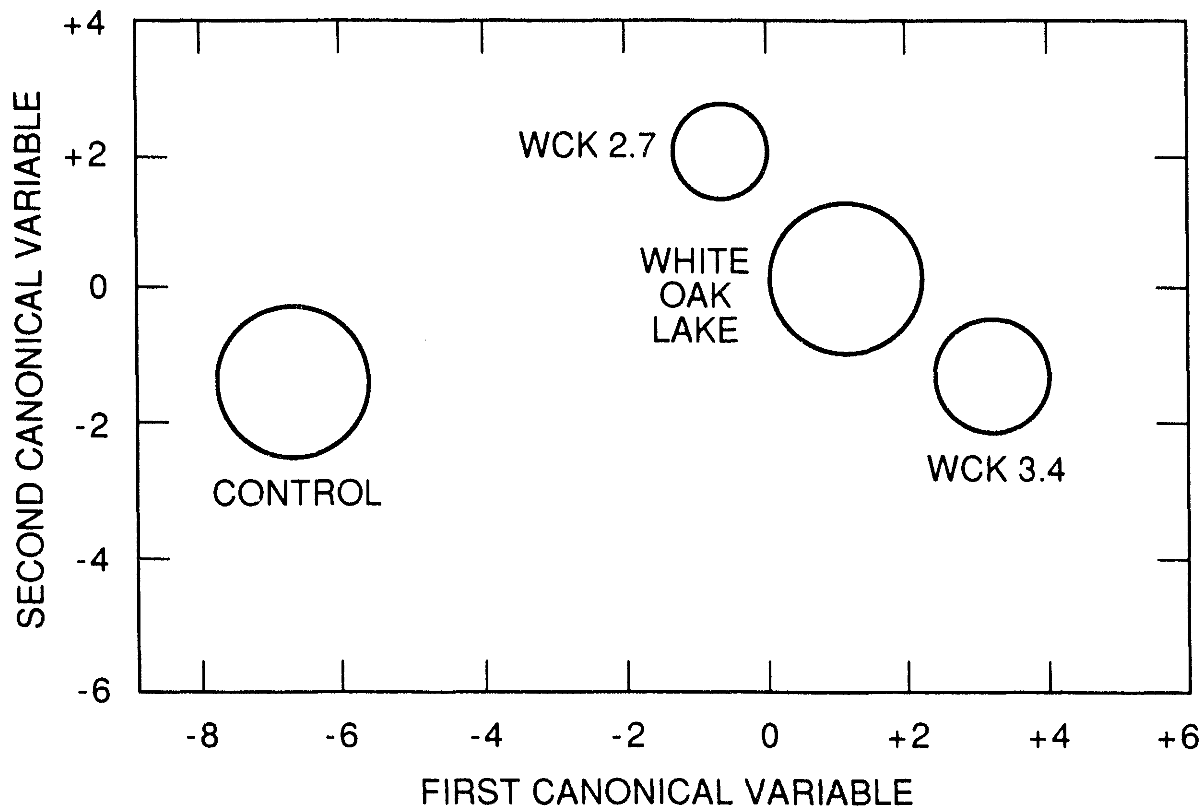

Fig. 5.6. Segregation of integrated bioindicator responses (blood chemistry, liver enzymes, condition indices, and growth indices) for redbreast sunfish from two White Oak Creek sites, White Oak Lake, and Brushy Fork (reference or control stream) in fall 1986. Circles represent site means of the integrated responses and the 90% confidence radii of the site means. WCK = White Oak Creek kilometer.

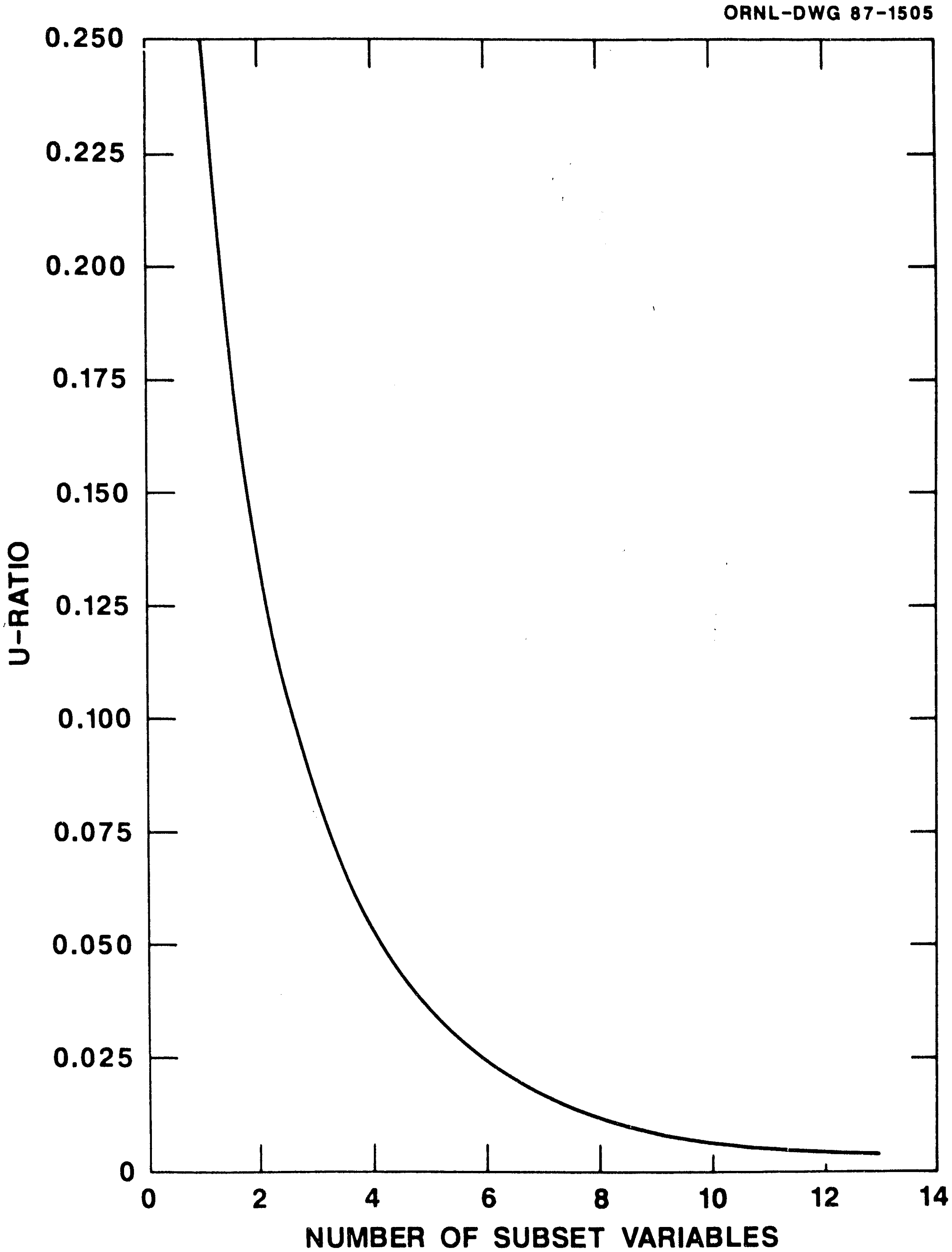

Fig. 5.7. Response of the U-ratio (a statistical measure of the ability to discriminate between sites) to the number of variables in a subset.

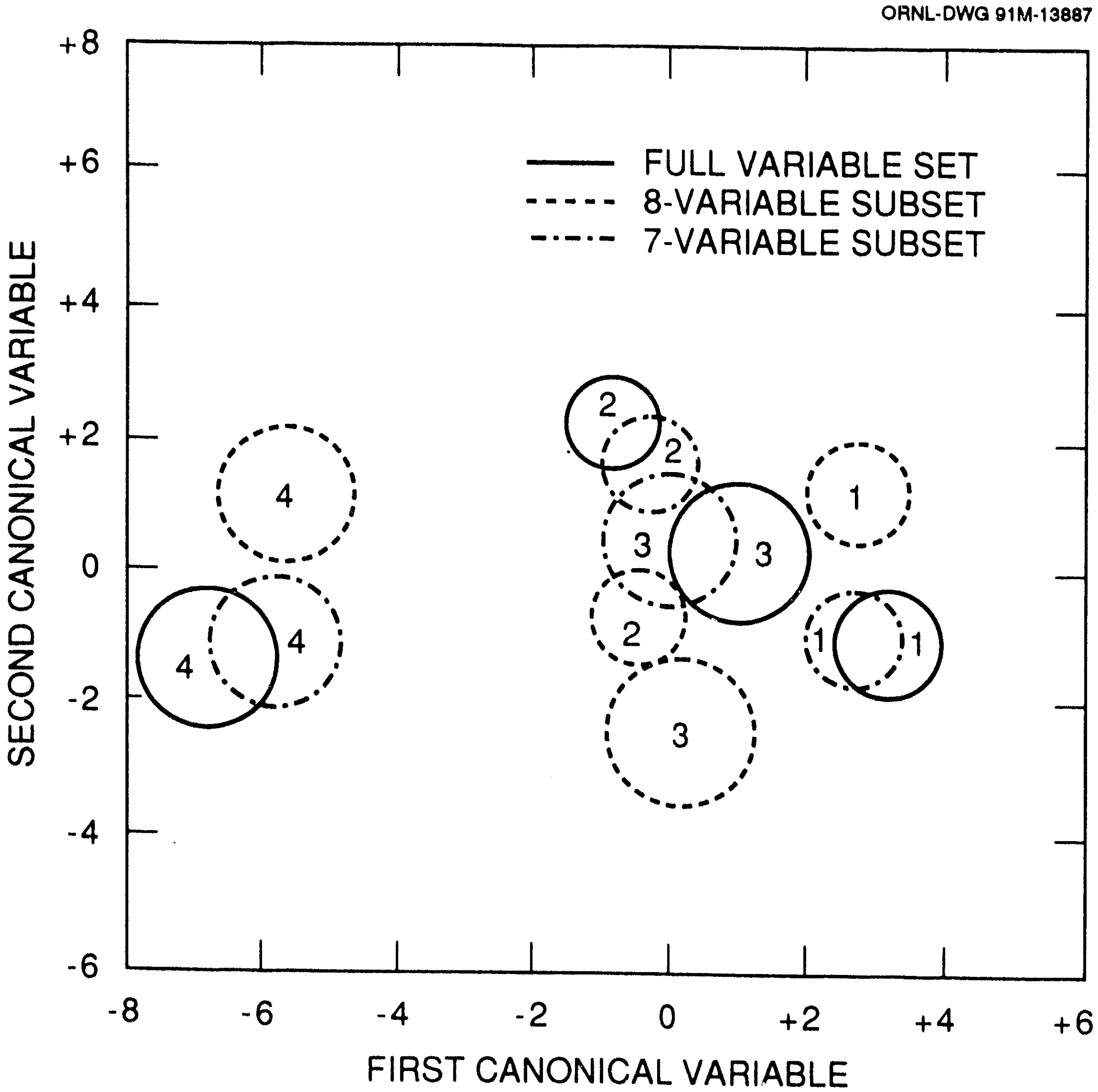

Fig. 5.8. Segregation of integrated bioindicator responses for redbreast sunfish from two White Oak Creek sites (sites 1 and 2), White Oak Lake (site 3), and Brushy Fork, the reference or control site (site 4), in fall 1986. The results are based on the full set of 13 variables (solid circles), a subset of 8 variables (broken circles), and a subset of 7 variables (dash-dot circles).

Table 5.2. Major response categories of bioindicators, grouped according to the general types of causal agents ultimately responsible for each response

Category	Specific indicators[a]
Indicators of nutrition or indirect effects via the food chain	Liver-somatic index (if liver enzymes low) Triglycerides Cholesterol Feeding and nutrition Body lipids
Indicators of water quality or direct effect of contaminants	Liver-somatic index (if lever enzymes high) Liver (detoxification) enzymes Histopathology Bilirubin Serum protein SGOT Phospholipid ratios
Indicators that integrate direct and indirect effects	RNA:DNA ratio Growth Reproductive success

[a]SGOT = serum glutamic oxaloacetic transaminase; RNA = ribonucleic acid; DNA = deoxyribonucleic acid.

5.4 CONCLUSIONS

Major accomplishments during the first sampling season included field measurements of a suite of bioindicators and identification of a preliminary subset of the most ecologically relevant and cost-effective indicators for further monitoring. The bioindicators used in future studies are also those that are sensitive to water quality change and can be used to evaluate the effects of remedial actions on water quality.

In addition to screening and selecting a set of bioindicators for additional study, several conclusions were drawn from the initial study. First, the integrated responses of fish at the reference site on Brushy Fork were significantly different from those of fish in WOC and WOL, and the integrated responses of fish from the two WOC sites were significantly different from those of fish in WOL. Second, many bioindicators are interrelated and affect each other in various ways. Therefore, investigations of the responses of fish to environmental stress are more effectively evaluated by using integrated response variables instead of individual variables. Finally, processes within biological communities exhibit strong seasonality. Consequently, the effects of contaminant stress on organisms cannot be adequately evaluated without the integration of data from several seasons.

5.5 FUTURE STUDIES

This initial study identified a preliminary subset of bioindicators that should provide ecologically relevant and cost-effective information for evaluating the effects of remedial actions on the biological integrity of WOC. The relevant indicators include the RNA:DNA ratio (indicator of growth), LSI, indicators of protein metabolism (SGOT and serum protein), and the liver detoxification enzymes (P-450 and cytochrome b_5). Other variables included in the initial screening, such as cholesterol, glucose, triglycerides, and NADPH, will probably be excluded from future studies. Indicators of heavy metal stress, the metallothionein enzymes, will be analyzed in fish from WOC to (1) determine if concentrations of heavy metals in WOC are high enough to elicit responses in fish and (2) identify/evaluate sources of metals to WOC.

Results of the initial study also provided insights into the direction of other investigations in 1987. Such studies fall into three major areas:

1. Identification and refinement of indicators that reflect the response of organisms to specific types of environmental factors. Bioindicators can be grouped into three major categories based on responses that reflect (a) the direct effects of water quality; (b) the indirect effects of water quality, as mediated through the food chain; and (c) both direct and indirect effects. Specific studies that relate to this effort will include manipulative field experiments, food habits and nutritional value of the diet, quantitative histopathology of the liver (e.g., Hinton et al. 1973), and in situ measurements of fish metabolism and respiration.

2. Integration of information from other related tasks. Pertinent data from related tasks, such as toxicity monitoring (Sect. 3), bioaccumulation monitoring (Sect. 4), and instream ecological monitoring (Sect. 6), will be integrated with the bioindicator task to evaluate potential chronic effects of water quality and the effects of remedial actions on the biotic integrity of WOC.

3. Evaluation of Brushy Fork as an appropriate control site. To firmly establish the credibility of Brushy Fork as an appropriate reference stream, several similar-sized streams that also do not receive industrial or municipal effluents will be sampled in a one-time, intensive study in late spring 1987. The same bioindicators will be measured from each stream, and the data will be compared statistically to determine if Brushy Fork is, in fact, an appropriate and representative reference site for WOC.

Routine monitoring of the selected suite of bioindicators will be conducted twice annually. Sampling conducted as part of the manipulative field experiments and the in situ metabolism/respiration studies will be conducted at various times throughout the year, depending upon the availability of fish. Sampling sites will be located near large weirs, as discussed in BMAP (Loar et al. 1991). Because no sunfishes could be collected at the lower Melton Branch weir in fall 1986, another site (e.g., at the weir on lower Northwest Tributary) may be substituted if no fish are found again in spring of 1987.

6. INSTREAM ECOLOGICAL MONITORING

The objectives of the instream ecological monitoring task (Task 4 of BMAP) are (1) to characterize spatial and temporal patterns in the distribution and abundance of the benthos and fish populations in WOC watershed; (2) to identify contaminant sources that adversely affect stream biota, including differentiation between point sources and nonpoint (or area) sources, wherever possible; and (3) to monitor these populations and evaluate the 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 respectively.

6.1 BENTHIC MACROINVERTEBRATES

6.1.1 Introduction

Benthic macroinvertebrates are those organisms that are large enough to be seen without the aid of magnification and that live on or in the substrate of flowing and nonflowing bodies of water. Their limited mobility and relatively long life spans (a few weeks to more than 1 year) make them ideal for use in evaluating the ecological effects of effluents to streams (Platts et al. 1983). Thus, the composition and structure of the benthic community reflect the relatively recent past and can be considerably more informative than methods that rely solely on water quality analyses but ignore the potential synergistic effects often associated with complex effluents. The objectives of the initial phase of this study were to spatially and temporally characterize the benthic invertebrates of WOC watershed. This information will be used as a baseline from which change can be followed during the monitoring phase of the program. The data will also be used to provide direction in future studies.

6.1.2 Materials and Methods

Benthic macroinvertebrates were sampled at approximately monthly intervals from May through October 1986, from 16 sites in WOC watershed. The sampling sites included six sites on WOC, three each on First Creek* and Melton Branch, and two each on Northwest Tributary and Fifth Creek (Fig. 2.3; Table 6.1); the uppermost site of each stream served as

*At the start of the sampling program, the upstream reference site on First Creek was located north of Bethel Valley Road but was later moved downstream (south of the road) where fish were also being sampled; thus, two sets of samples were collected from FCK 1.0, and five sets of samples were collected from FCK 0.8 (Table 6.1). A preliminary analysis of some samples collected from these two sites indicated that, although species composition differed somewhat, both sites had healthy benthic communities, as indicated by the presence of several stonefly, mayfly, and caddisfly taxa. It was also concluded that the site just south of Bethel Valley Road was a more suitable reference site because, like the lowermost site on this stream, there was little canopy cover. Data obtained from the uppermost site (FCK 1.0) are presented in tables and figures, but are not discussed in the text of the report.

a reference site. Samples were collected from WOL at approximately bimonthly intervals (Fig. 2.3).

At all stream sites, benthic invertebrates were collected with a Surber bottom sampler ($0.09\ m^2$, or $1\ ft^2$) fitted with a 363-µm-mesh collection net. Three randomly selected samples were collected from designated riffle areas at each site. In WOL, benthic invertebrates were collected with a hand-operated Ponar dredge having dimensions of 15 × 15 × 15 cm. Five samples were collected at ~10-m intervals across a transect located ~100 m upstream of the dam. All samples were placed in prelabeled plastic or glass jars and preserved in 80% ethanol; the ethanol was replaced with fresh 80% ethanol within 1 week.

Various supplemental information was also recorded at the time of sampling. At each stream site, water temperature and specific conductance were measured with a Col-Parmer Model R-1491-20 LCD temperature/conductivity meter. Recorded for each sample were water depth; location within the riffle area (distance from permanent headstakes on the stream bank); relative stream velocity (visually determined as very slow, slow, moderate, or fast); and substrate type, which was determined with the use of a modified Wentworth particle size scale (Loar et al. 1985). At WOL, a single measurement of water temperature and specific conductance was obtained at the surface, and water depth and substrate type were recorded for each sample.

All samples were washed in the laboratory by using a standard no. 50 mesh (297-µm-mesh) sieve and placed in a large white tray. Organisms were removed from the debris with forceps and placed in labeled vials containing 80% ethanol. Organisms were identified to the lowest practical taxonomic level with the use of a stereoscopic dissecting microscope. A blotted wet weight (biomass) of all individuals in each taxon was determined to the nearest 0.01 mg on a Mettler analytical balance.

Chironomid larvae were identified from permanent slide mounts. Larvae were initially sorted into groups based on morphological similarities (e.g., body size, head capsule shape and coloration, and abdominal setae), and then the head capsule and body of one or two larvae in each group were mounted on a microscope slide by using a small drop of CMC-10 mounting medium. The head capsules of larger larvae were first cleared in hot 10% potassium hydroxide solution for 10 min. After drying overnight, the mounted larvae were identified with the use of a compound binocular microscope. Slides of mounted larvae were stored in slide boxes and retained for reference.

Individual taxa from a given site and sampling date were preserved in separate vials in 80% ethanol. A reference collection, for which the identification of each taxon has been verified, will be maintained at ORNL.

SAS procedures (SAS 1985a, 1985b) were used to perform all calculations and statistical analyses on transformed data [$\log_{10}(x + 1)$] (Elliott 1977). The Shannon-Wiener index (H') was used to calculate the taxonomic diversity of benthic macroinvertebrates at each site (Pielou 1977):

$$H' = -\Sigma p_i \log_2 p_i \ ,$$

Table 6.1. Number of samples collected at each benthic macroinvertebrate sampling site in White Oak Creek watershed, May through October 1986

Site[a]	Month						Total
	May	June	July	Aug.	Sept.	Oct.	
FCK 0.1	3	3	3	3	3	3	18
FCK 0.8	3	NS[b]	3	3	3	3	15
FCK 1.0	3	3	NS	NS	NS	NS	6
FFK 0.2	3	3	3	3	3	3	18
FFK 1.0	3	3	3	3	3	3	18
MEK 0.6	3	3	3	3	3	3	18
MEK 1.2	3	3	3	3	3	3	18
MEK 2.1	3	3	Dry[c]	Dry	Dry	Dry	6
NTK 0.2	3	3	3	3	3	3	18
NTK 1.0	3	3	3	Dry	3	Dry	12
WCK 2.3	3	3	3	3	3	3	18
WCK 2.9	3	3	3	3	3	3	18
WCK 3.4	3	3	3	3	3	3	18
WCK 3.9	3	3	3	3	3	3	18
WCK 5.1	3	3	3	3	3	3	18
WCK 6.8	3	3	3	3	3	3	18
WCK 1.1[d]	5	NS	5	NS	5	NS	15

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

[b]NS = site not sampled

[c]Dry = dry site that was therefore not sampled.

[d]Site located in White Oak Lake.

where p_j is the proportion of the benthic invertebrate community made up by species *j*. A blocked (site and date) ANOVA and Duncan's new multiple range test were used to compare mean values for density, biomass, taxonomic richness (number of taxa per sample), and diversity. Because differences occurred in the sampling frequency for some sites (Table 6.1), statistical analyses were limited to only those months in which samples were collected from both the reference and downstream sites of each stream. Thus, the analyses were performed as follows: First Creek (FCK 0.1 and FCK 0.8), May through October (excluding June); Fifth Creek (FFK 0.2 and FFK 1.0), May through October; Melton Branch (MEK 0.6, MEK 1.2, and MEK 2.1), May and June only; Northwest Tributary (NTK 0.2 and NTK 1.0), May through July only (comparisons did not include September since flow in NTK 1.0 was the result of a rainstorm that had occurred only a few days before); and WOC (WCK 2.3, WCK 2.9, WCK 3.4, WCK 3.9, WCK 5.1, and WCK 6.8), May through October. Additionally, the two sites on lower Melton Branch (MEK 0.6 and MEK 1.2) were compared statistically for the entire 6-month period. Unless stated otherwise, statements of significant differences are based on accepting $p < 0.05$.

6.1.3 Results

6.1.3.1 White Oak Creek and tributaries

Taxonomic Composition. In the quantitative samples taken from WOC and its tributaries from May through October 1986, 161 benthic macroinvertebrate taxa were represented (Appendix F, Table F.1); at least 141 of these taxa were insects. Also collected in the streams were Planariidae (planarians); Nematoda (roundworms); Oligochaeta (aquatic earthworms); crustaceans, including Isopoda (aquatic sow bugs), Amphipoda (sideswimmers), and Decapoda (crayfish); Gastropoda (snails); and Bivalvia (clams and mussels). There were ten orders of insects represented, including Collembola (springtails); Ephemeroptera (mayflies); Odonata (dragonflies and damselflies); Plecoptera (stoneflies); Hemiptera (true bugs); Megaloptera (alderflies, dobsonflies, hellgrammites); Neuroptera (spongillaflies); Trichoptera (caddisflies); Coleoptera (beetles); and Diptera (true flies). By far the most common order of insects was the dipterans, which were represented by 74 taxa. Of the dipterans, 57 taxa were of the family Chironomidae (nonbiting midges). Of the remaining orders of insects, the greatest number of taxa were collected in the orders Trichoptera, Ephemeroptera, Odonata, and Coleoptera, which contained 19, 13, 10, and 7 taxa respectively.

Insects were the most commonly collected taxa at each site and, except for MEK 1.2, chironomids were represented by the most taxa; at MEK 1.2 both the chironomids and the odonates were represented by five taxa. The reference sites of each stream were characterized by a greater variety of taxa than the downstream sites. Each reference site had two or more stonefly and mayfly taxa and four or more caddisfly taxa. Mayflies were totally absent from all but the reference sites, while only one to three caddisfly taxa (*Cheumatopsyche*, *Hydropsyche*, and/or *Chimarra*) were found at each of the downstream sites.

Density and Biomass. Considerable monthly variation occurred within and between all sites in both density and biomass (excluding Decapoda and Mollusca)* of the benthic invertebrates in WOC and its tributaries (Figs. 6.1 and 6.2). Peaks in both density and biomass generally occurred in May or June, while minima were generally observed in July or August. The most dramatic changes typically occurred at those sites numerically dominated by only one to three taxa (e.g., FCK 0.1, MEK 0.6, and WCK 2.3). Density and biomass were consistently greater in the upstream reference sites, except for density and biomass in Melton Branch and Northwest Tributary and biomass in WOC, where relatively large numbers of caddisflies were sometimes collected. Notable differences were observed between the taxa contributing the most to the biomass and density of the upstream (reference) and downstream sites. In general, several taxa contributed substantially to the total density and biomass of the

*The weights presented in Fig. 6.2 exclude Mollusca (snails and clams) and Decapoda (crayfish) in order to make the weight of benthic organisms at each station more comparable without the biasing effect of a few heavy-bodied organisms. However, these taxa are included in the estimates of mean monthly density shown in Fig. 6.1.

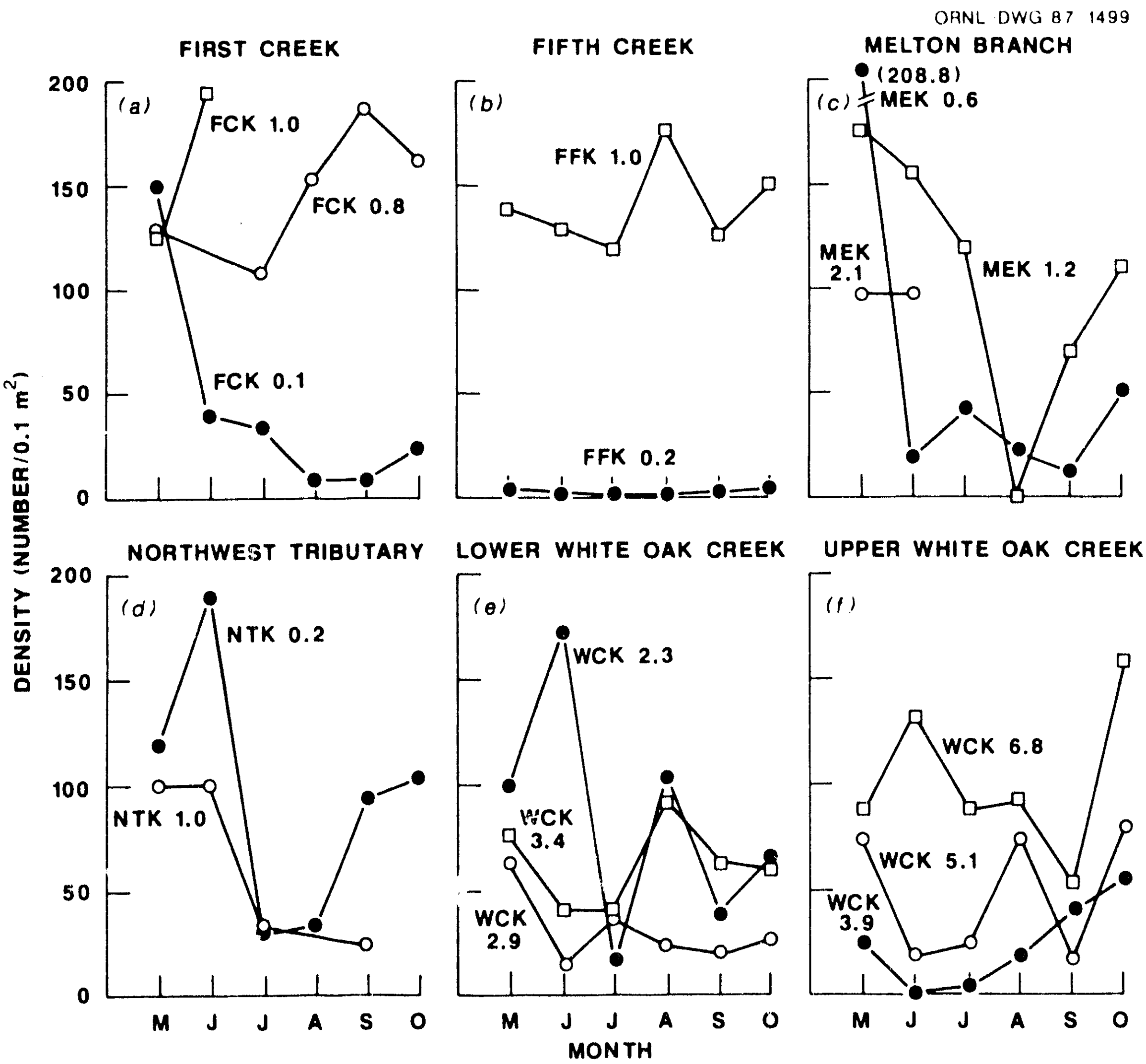

Fig. 6.1. Monthly mean density (number per 0.1 m^2) of benthic macroinvertebrates in White Oak Creek watershed, May through October 1986. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

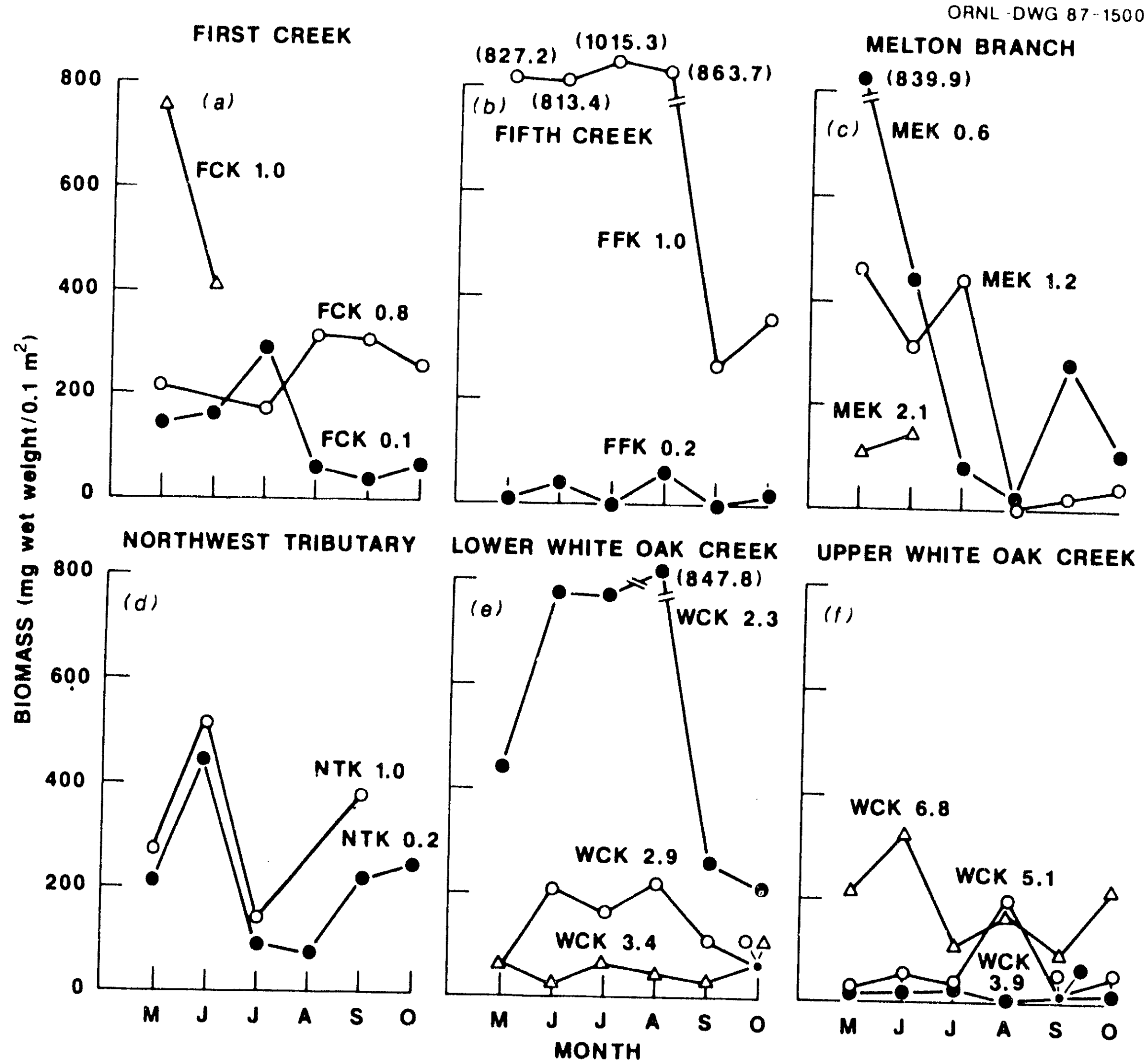

Fig. 6.2. Monthly mean biomass (excluding Decapoda and Mollusca) (milligrams per 0.1 m^2) of benthic macroinvertebrates in White Oak Creek watershed, May through October 1986. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

reference sites, whereas at the downstream sites a considerable portion of the total density and biomass was largely that of one to three taxa.

In First Creek, mean monthly density and biomass (excluding decapods and mollusks) at FCK 0.8 ranged from 109.1 to 188.7 individuals per 0.1 m^2 and 177.9 to 320.0 mg wet wt/0.1 m^2, respectively; at FCK 0.1 density and biomass ranged from 11.1 to 151.4 individuals per 0.1 m^2 and 43.9 to 297.3 mg wet wt/0.1 m^2 respectively (Figs. 6.1 and 6.2). Both density and biomass were significantly greater at FCK 0.8 than at FCK 0.1. FCK 0.8 was numerically dominated by the amphipod *Gammarus* (39.3 to 75.3%) and, to a lesser extent, by the snail *Elimia* (5.7 to 31.2%). Stoneflies were sometimes abundant, making up as much as 7.2% of the mean monthly density. Much of the total biomass at this site was contributed by *Gammarus* (38.3 to 64.4%). FCK 0.1 was numerically dominated by both *Cheumatopsyche* (Trichoptera) and *Cricotopus/Orthocladius* (Diptera: Chironomidae); *Cheumatopsyche* accounted for 10.7 to 39.0%, and *Cricotopus/Orthocladius* accounted for 6.1 to 50.7% of the density. A majority of the biomass at this site was generally contributed by a few large oligochaetes (13.7 to 81.3% of the mean monthly biomass).

Density and biomass were both significantly higher in upper (FFK 1.0) than in lower Fifth Creek (FFK 0.2) (Figs. 6.1 and 6.2). Mean monthly density and biomass (excluding decapods and mollusks) at FFK 1.0 ranged from 118.4 to 177.2 individuals per 0.1 m^2 and from 272.1 to 1015.3 mg wet wt/0.1 m^2 respectively. *Lirceus* (Isopoda), chironomids, caddisflies (primarily *Diplectrona* and *Cheumatopsyche*), and stoneflies (primarily *Leuctra*) were typically the most abundant taxa at this site, combining for 49.5 to 89% of the total density. Major contributors to the biomass at this site were caddisflies and dipterans, particularly the heavy-bodied cranefly, *Tipula*; these two groups accounted for 20.1 to 79.8% of the total biomass. The number of benthic invertebrates at FFK 0.2 was very low, with densities ranging from 0.4 to 4.3 individuals per 0.1 m^2 and biomass ranging from 0.1 to 68.4 mg wet wt/0.1 m^2. The highest density of any taxon at this site never exceeded 2.2 individuals per 0.1 m^2.

The upper site on Melton Branch (MEK 2.1) was dry from July through October. Both mean density and biomass remained relatively stable at this site in May and July, exhibiting maxima of 96.8 individuals per 0.1 m^2 and 146.6 mg wet weight/0.1 m^2 (excluding decapods and mollusks), respectively (Figs. 6.1 and 6.2). The most abundant taxonomic group at MEK 2.1 was the chironomids (>64% of the total density), although other taxa were also relatively abundant, including beetles (2.6 to 11.6%), mayflies (4.4 to 6.3%), and isopods (2.2 to 6.7%). No significant differences were found in density or biomass between this site and the two downstream sites during this period.

Mean density and biomass (May through October) at MEK 0.6 ranged from 12.6 to 208.8 individuals per 0.1 m^2 and from 24.1 to 839.9 mg wet wt/0.1 m^2 (excluding decapods) respectively. At MEK 1.2, density and biomass ranged from 1.8 to 175.4 individuals per 0.1 m^2 and from 3.2 to 463.6 mg wet wt/0.1 m^2 respectively; mollusks were not collected at either site, and crayfish were not collected at MEK 1.2. From May through October neither the density nor the biomass of MEK 0.6 differed significantly from those values determined for density and biomass at MEK 1.2. Chironomids and caddisflies were numerically dominant at both sites. At MEK 1.2, caddisflies (primarily *Cheumatopsyche*) and chironomids (primarily *Polypedilum*) made up 0 to 92.6% and 3.0 to 99.0%, respectively, of the total community density. At MEK 0.6, caddisflies (primarily *Cheumatopsyche* and *Chimarra*) and chironomids made up 0 to 62.7% and 19.6 to 78.0%, respectively, of the total community density.

Caddisflies, chironomids, and dragonflies/damselflies were the primary contributors to the biomass at MEK 1.2, with relative contributions ranging from 0 to 93.7%, 0.3 to 71.8%, and 0 to 36.4% respectively. The greatest contributions of the chironomids occurred only in those months when they were also numerically dominant (i.e., September and October). Much of the biomass at MEK 0.6 was that of the caddisflies, nonchironomid dipterans (primarily *Tipula*), and dragonflies/damselflies, which made up 0 to 74.9%, 0 to 46.9%, and 0 to 95.3%, respectively, of the mean monthly biomass.

Likc the reference site on Melton Branch, streamflow at the reference site on Northwest Tributary (NTK 1.0) was intermittent. This site first became dry in August, had flow again in September, and then was dry in October. Neither the density nor biomass of NTK 1.0 differed significantly from values observed at NTK 0.2. Mean monthly densities at NTK 0.2 ranged from 34.4 to 190.2 individuals per 0.1 m^2; at NTK 1.0 the range was 25.5 to 101.2 individuals per 0.1 m^2 (Fig. 6.1). Mean monthly biomass (excluding mollusks and decapods) at NTK 0.2 ranged from 79.6 to 450.8 mg wet wt/0.1 m^2, and at NTK 1.0 the range was 146.1 to 528.4 mg wet wt/0.1 m^2 (Fig. 6.2). Considerable differences in the dominant taxa were found between sites. The most abundant taxa at NTK 0.2 were the caddisflies (almost exclusively *Cheumatopsyche*) and chironomids (primarily *Thienemannimyia*), with these two groups accounting for 48.9 to 64.3% of the mean monthly density. Relatively large numbers of beetles (mostly *Stenelmis*) and clams (*Corbicula*) were also sometimes collected, accounting for up to 29.4 and 35.9%, respectively, of the density in at least 1 month. Chironomids (mostly *Thienemannimyia*) and isopods (*Lirceus*) were usually the most abundant taxa at NTK 1.0, accounting for 18.8 to 63.2% of the mean monthly density. Other abundant taxa at this site were beetles (*Stenelmis* and *Optioservus*), craneflies, and stoneflies (primarily *Amphinemura* and *Perlesta*), with relative abundances ranging from 3.2 to 39.5%, 3.6 to 20.9%, and 0 to 16.1% respectively.

Caddisflies made up most of the biomass (excluding mollusks and crayfish) at NTK 0.2, with a relative biomass of 22.5 to 58.2%. Nonchironomid dipterans, dragonflies/damselflies, and oligochaetes each made up more than 20% of the biomass in at least 1 or 2 months. With few exceptions, the biomass (excluding mollusks and crayfish) in May and June at NTK 1.0 was distributed relatively equally among several taxa, including chironomids, nonchironomid dipterans, mayflies, isopods, oligochaetes, stoneflies, and caddisflies. In July and September, a few large oligochaetes dominated the biomass, making up 54.8 and 94.0% in each respective month.

Highest densities of benthic invertebrates in WOC were generally found in the upstream reference site (WCK 6.8) (Fig. 6.1). Density then declined sharply at WCK 5.1 and reached minimum levels at WCK 3.9. Downstream of WCK 3.9, densities tended to increase to levels comparable to those at WCK 6.8. With the exception of WCK 2.3, the downstream trend in biomass (excluding mollusks and decapods) was similar to that of density (Fig. 6.2). Biomass at WCK 2.3 was considerably greater than at any other site on WOC with the exception of WCK 6.8, primarily because of the large number of caddisflies. Statistical analysis of both density and biomass confirmed this general trend (Table 6.2). Mean densities in WOC ranged from a low of 1.1 individuals per 0.1 m^2 at WCK 3.9 to a high of 172.9 at WCK 2.3. Mean biomass (excluding mollusks and decapods) varied from a low of 9.5 mg wet wt/0.1 m^2 at WCK 3.9 to a high of 847.8 mg wet wt/0.1 m^2 at WCK 2.3.

Table 6.2. Statistical comparisons of mean benthic macroinvertebrate density, biomass, taxonomic richness (number of taxa per sample), and taxonomic diversity in White Oak Creek

	Site[a]					
Density	WCK 6.8	WCK 3.4	WCK 2.3	WCK 5.1	WCK 2.9	WCK 3.9
Biomass	WCK 2.3	WCK 6.8	WCK 2.9	WCK 3.4	WCK 5.1	WCK 3.9
Taxonomic richness	WCK 6.8	WCK 5.1	WCK 2.3	WCK 3.4	WCK 2.9	WCK 3.9
Taxonomic diversity	WCK 6.8	WCK 5.1	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9

[a]Sites not joined by lines are significantly different ($p < 0.05$). Sites are arranged in order of decreasing values. WCK = White Oak Creek kilometer.

A marked difference in the taxa contributing the most to both density and biomass existed between WCK 6.8 and all other sites on this stream (Table 6.3). Considerable monthly variability occurred in the taxa numerically dominating WCK 6.8, but, in general, several taxa were relatively abundant, including chironomids, beetles, mayflies, snails, stoneflies, oligochaetes, and caddisflies. Particularly notable was the relatively large number of stoneflies, which made up greater than 27% of the mean density in all but 1 month. With the exception of snails (primarily *Elima*), several genera within each of these groups contributed to the total density. A similarly even distribution among several taxa was also observed for biomass, although chironomids contributed very little to the total biomass.

Much of the total density and biomass at the remaining five sites on WOC was typically that of one to three groups: chironomids, caddisflies, and/or oligochaetes. Where caddisflies dominated numerically, they usually contributed the most to the biomass also. The most abundant group at WCK 5.1 was the chironomids (19.2 to 91.5%), but the caddisfly *Cheumatopsyche* (0 to 44.2%) and oligochaetes (0.9 to 11.5%) were also sometimes relatively abundant. Chironomids, caddisflies, and oligochaetes also contributed the most to total biomass at WCK 5.1, contributing 3.7 to 84.5%, 0 to 75.0%, and 0.6 to 51.8% respectively. Chironomids, particularly the *Cricotopus/Orthocladius* group, were the most abundant (33.3 to 99.3%) and generally contributed the most to total biomass at WCK 3.9.

The community at WCK 3.4 was made up primarily of chironomids, and their small size resulted in the sustained low biomass of this site. Caddisflies (*Hydropsyche* and *Cheumatopsyche*), however, were collected regularly. The chironomids (primarily the *Cricotopus/Orthocladius* group) made up 84.8 to 100.0% of the mean monthly density; their relative biomass ranged from 46.7 to 100%.

Table 6.3. Relative abundance (percentage of monthly mean density) and biomass (percentage of monthly mean biomass) of the numerically dominant benthic macroinvertebrate taxa in White Oak Creek, May through October 1986

Site[a]	Dominant taxa	Range of relative abundance (%)	Range of relative biomass (%)
WCK 2.3	Caddisflies	48.0–92.4	4.7–97.7
	Chironomids	3.1–39.6	0.1–3.3
	Dragonflies	0.0–13.3	0–94.6
WCK 2.9	Caddisflies	6.7–81.2	14.3–45.6
	Chironomids	11.9–89.4	0.8–33.7
	Oligochaetes	2.8–56.9	49.2–83.2
WCK 3.4	Caddisflies	0–9.1	0–47.1
	Chironomids	84.8–100	46.7–100
WCK 3.9	Chironomids	33.3–99.3	0–99.1
	Oligochaetes	0–66.7	0–98.0
WCK 5.1	Caddisflies	0–44.2	0–75.0
	Chironomids	19.2–91.5	3.7–84.5
	Oligochaetes	0.9–11.5	0.9–11.5
WCK 6.8	Beetles	4.9–14.4	3.2–39.5
	Caddisflies	0.8–8.2	1.3–22.2
	Chironomids	5.2–15.1	0.9–3.1
	Mayflies	3.6–21.8	10.7–38.3
	Oligochaetes	5.7–33.9	3.4–22.2
	Snails[b]	6.8–36.7	–
	Stoneflies	4.3–51.6	4.6–18.4

[a]WCK = White Oak Creek kilometer.

[b]Biomass of snails relative to that of other taxa at WCK 6.8 was not determined; mean monthly biomass of this group ranged from 515.2 to 4263.59 mg wet wt/0.1 m^2.

At WCK 2.9, the numerically dominant taxa were chironomids (primarily the *Cricotopus/Orthocladius* group), oligochaetes, and caddisflies (*Cheumatopsyche* and *Hydropsyche*), with relative abundances ranging from 11.9 to 89.4%, 2.8 to 56.9%, and 6.7 to 81.2% respectively. These taxa also contributed extensively to the biomass, contributing 0.8 to 33.7%, 49.2 to 83.2%, and 14.3 to 45.6% respectively.

At WCK 2.3, the numerically dominant taxa were caddisflies (primarily *Cheumatopsyche* and *Hydropsyche*) and chironomids (primarily *Cricotopus/Orthocladius*); the relative abundance of these taxa ranged from 48.0 to 92.4% and 3.1 to 39.6% respectively. The biomass of the caddisflies was also considerable (4.7 to 97.7%), although dragonflies (1.0 to 94.6%) also contributed substantially.

Community Structure. A useful parameter for assessing the status of the benthic macroinvertebrate community is taxonomic richness in terms of both total and mean number of taxa per sample (Platts et al. 1983). Under environmental stress, the number of taxa would be expected to decrease. Total richness at each site is presented in Fig. 6.3. Total richness ranged from a low of 12 at WCK 3.9 to a high of 66 at FFK 1.0, the upstream reference site on Fifth Creek. With the exception of Melton Branch, the greatest number of taxa was collected at the reference site of each stream; in the case of the reference sites on First Creek, Melton Branch, and Northwest Tributary, the total richness is based on fewer samples than collected for their corresponding downstream sites (see Table 6.1). In WOC, there was a sharp reduction in total richness from the reference site (WCK 6.8) to the next site downstream (WCK 5.1), which is located just below the 7000 area. A considerable decrease occurred again at the next downstream site (WCK 3.9). This decline was followed by a general increase in total richness with increasing distance downstream from the main ORNL complex.

Mean richness (Fig. 6.4) showed the same general pattern as total richness; that is, on average, the greatest number of taxa was generally found at the upstream reference sites. Mean richness consistently exceeded 22 taxa per sample in the upper reference sites, while, with few exceptions, mean richness at the respective downstream sites rarely exceeded 20 taxa per sample and, in many cases, did not exceed 15 taxa per sample. These differences between the reference sites and their corresponding downstream sites were significant in all streams except Northwest Tributary, where the mean number of taxa at the lower site (NTK 0.2) was never less than 17 and was greater than 22 taxa per sample in 3 months. In both Melton Branch and WOC, there was a trend towards maximum depression in mean richness in the midreaches, followed by an increase at the lowermost site(s). In Melton Branch, mean richness at the lowermost site (MEK 0.6) was found to be significantly greater than at the midreach site (MEK 1.2). Statistical comparison of the WOC sites confirmed that maximum depression of mean richness occurred in the midreaches of the stream, and, at WCK 3.9, mean richness was significantly lower than at any other site (Table 6.2). The number of taxa per sample at WCK 2.3 was similar to that observed at WCK 5.1 (Table 6.2 and Fig. 6.4).

Because taxonomic diversity indices combine information on both taxonomic richness and evenness of a community into a single dimensionless value, they can provide a useful tool for interpreting changes in relative abundance of organisms even where the total number of taxa changes very little (Weber 1973, Platts et al. 1983). However, reliance on diversity indices

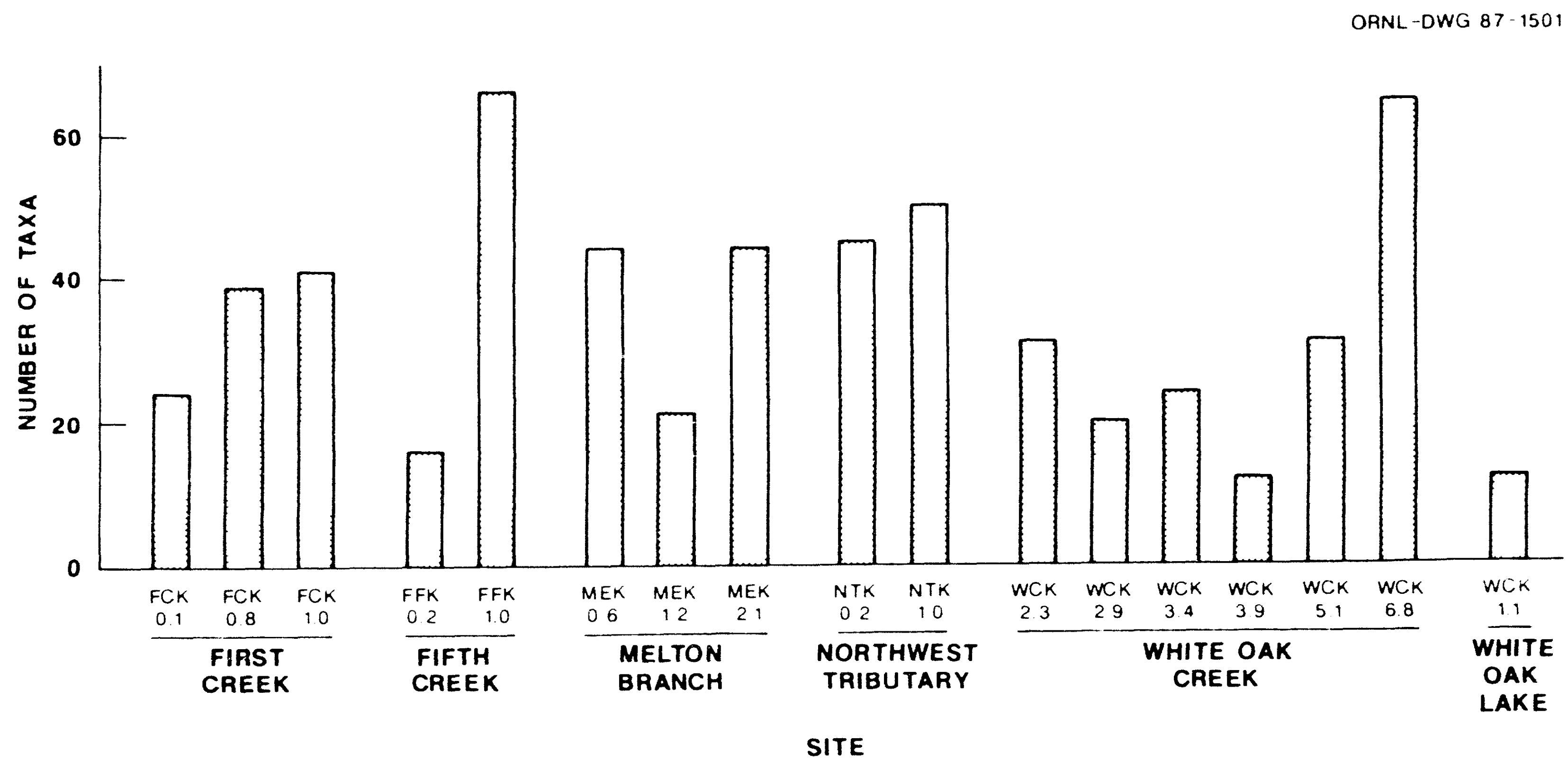

Fig. 6.3. Total richness (number of taxa collected) per site from White Oak Creek watershed. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

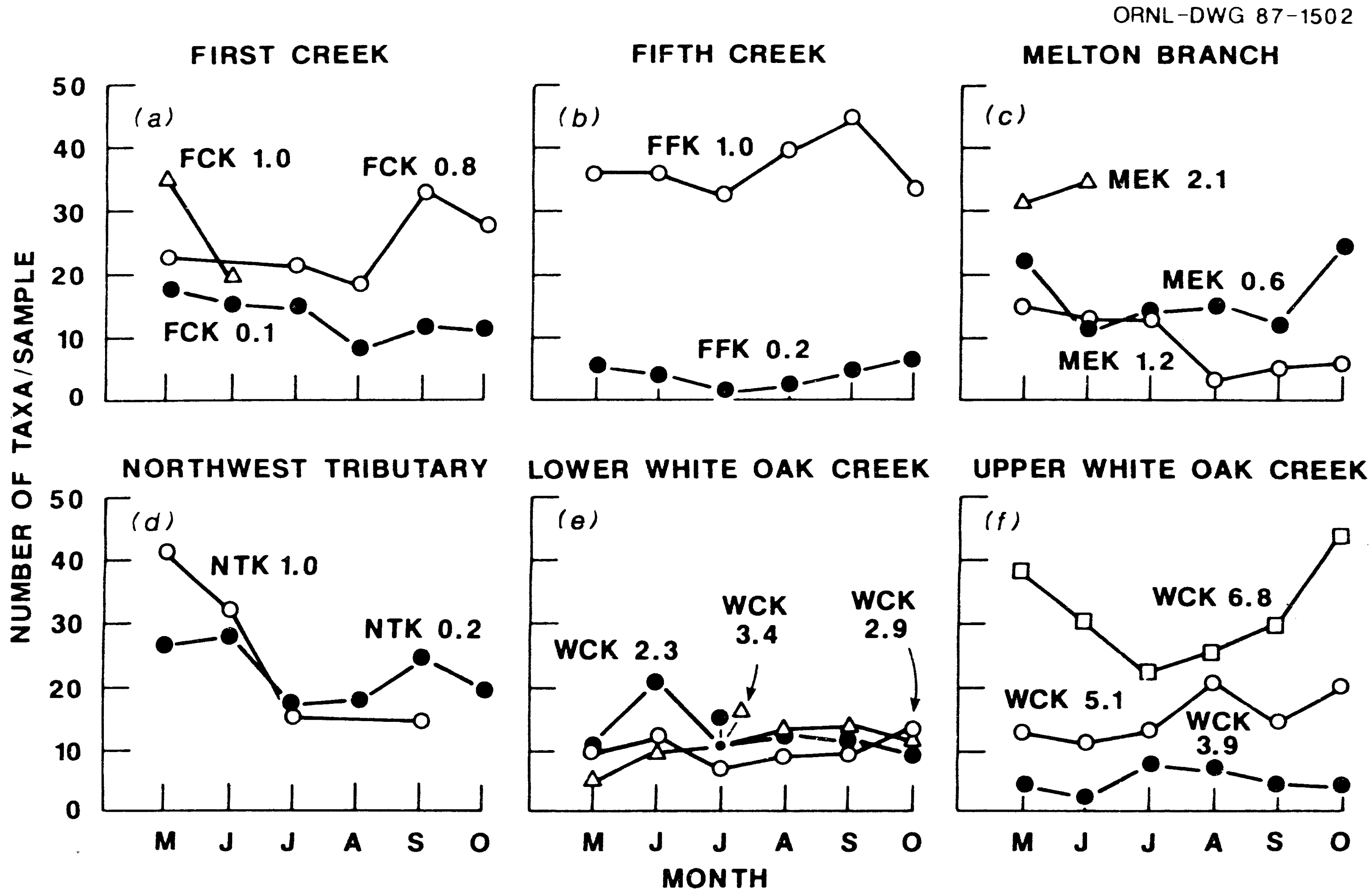

Fig. 6.4. Mean richness (number of benthic macroinvertebrate taxa per sample) in White Oak Creek watershed, May through October 1986. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

alone to interpret data is risky; under some circumstances, the abundance may be evenly distributed among the taxa although the number of taxa is very low, resulting in a misleadingly high estimate of diversity (Rosenberg 1976). Values of H′ equal to or greater than 3 are typical of good water quality, while values of 1 to less than 3 occur in areas of moderate pollution; values of less than 1 characterize areas of heavy pollution (Platts et al. 1983).

In general, diversity in WOC watershed followed the same trends shown by density, biomass, and taxonomic richness, with maximum diversity usually occurring in the upstream reference site of each stream (Fig. 6.5). Except for First Creek, diversity was consistently higher in the reference site, although in Northwest Tributary and the two upper WOC sites (WCK 5.1 and WCK 6.8), this difference was not significant; the First Creek sites (FCK 0.8 and FCK 0.1) also did not differ significantly.

The lowest diversity values in Melton Branch and WOC occurred in the midreaches of each stream; in both streams, there was a general increase in diversity at the lower sites (Fig. 6.5). A significant reduction in diversity was observed in the two downstream sites of Melton Branch, but diversity at the lowest site (MEK 0.6) was significantly greater than at the middle site (MEK 1.2). In WOC, diversity was significantly higher at WCK 6.8 than at all other sites except WCK 5.1, with maximum depression occurring at WCK 3.9 (Table 6.3).

6.1.3.2 White Oak Lake

Twelve distinct taxa were represented in quantitative samples taken from WOL (WCK 1.1) during May through September 1986 (Table F.1). Of these taxa, nine were insects, two were oligochaetes, and one was a snail. Eight of the insect taxa were dipterans (true flies), of which six were chironomids; the only nondipteran insect was a caddisfly.

Density of benthic macroinvertebrates in WOL ranged from a low of 91.5 individuals per 0.1 m^2 in May to a high of 229.7 individuals per 0.1 m^2 in July (Table 6.4). Biomass (excluding snails) ranged from a low of 65.2 mg/0.1 m^2 in September to a high of 278.3 mg/0.1 m^2 in July (Table 6.4). The biomass of snails ranged from 0 in May to 31.4 mg/0.1 m^2 in July. Chironomids, primarily *Procladius* and *Tanypus*, were the numerically dominant taxa and made up 69.6% or more of the mean density in each month. *Chaoborus*, a nonchironomid dipteran, was relatively abundant in July (25.9%) but contributed 8.5% or less in May and October. Chironomids were also the most substantial contributors to the biomass each month, contributing 61% or greater.

The benthic fauna of WOL was not very rich: the mean number of taxa per sample did not exceed 6.2 in any month (Table 6.4). This was also reflected in taxonomic diversity, which ranged from 1.6 to 1.8 (Table 6.4).

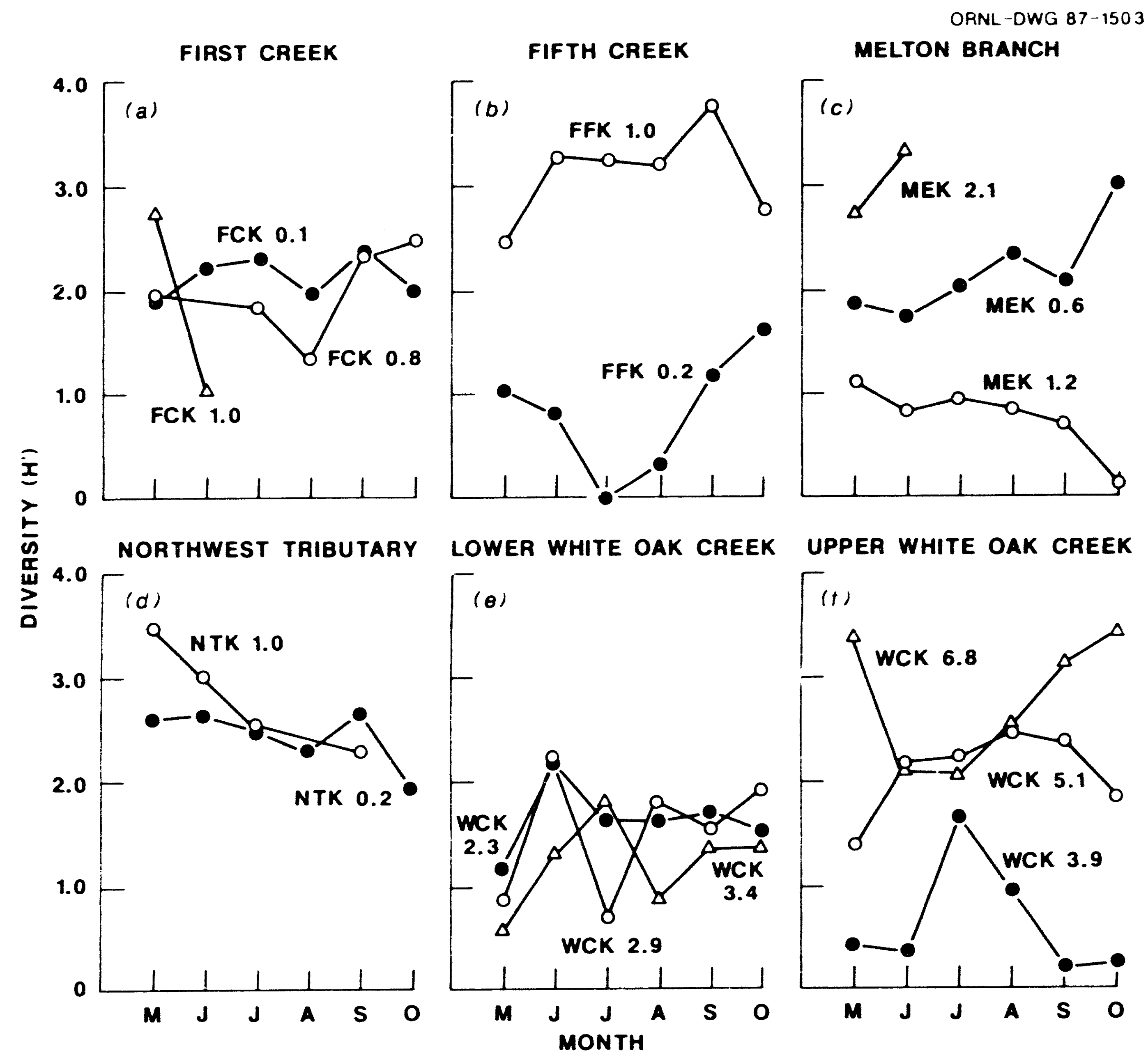

Fig. 6.5. Taxonomic diversity (H') of benthic macroinvertebrates in White Oak Creek watershed, May through October 1986. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

Table 6.4. Mean monthly density (number per 0.1 m^2), biomass (milligrams wet weight per 0.1 m^2), number of taxa per sample, and species diversity (H′) of benthic macroinvertebrates in White Oak Lake, May through October 1986

Month	Density	Biomass[a]	Number of taxa	Species diversity
May	91.5	166.8	4.2	1.6
July	229.7	278.3	6.2	1.7
September	92.4	65.2	4.8	1.8

[a]Excluding snails.

6.1.4 Discussion

6.1.4.1 White Oak Creek and tributaries

Current Study. Several natural factors control both the community structure and species composition of the benthos in streams, including chemical, physical, and biological factors (e.g., Hynes 1970). Selection of permanent monitoring sites that minimize differences in these natural factors reduces the possibility of making erroneous conclusions about impact-induced differences in the structure and composition of benthos between sites. Although it was not always possible, efforts were made to select permanent sampling sites in WOC watershed that were as similar to one another as possible, particularly with respect to water velocity and substrate type.

Another important consideration when monitoring benthic macroinvertebrates is the integrity of a reference site to which a supposed stressed site is being compared. Considerable differences in the structure of the benthic community can be found naturally both within a stream (e.g., Minshall et al. 1983) and between streams (e.g., Bunn 1986, Bunn et al. 1986). However, unimpacted streams are usually characterized by a wide assortment of benthic invertebrates such as mayflies, stoneflies, caddisflies, true flies, mollusks, and others (e.g., Hynes 1960). In addition to differences in taxonomic composition, the reference sites generally exhibited higher values for density, biomass, taxonomic richness, and diversity than their downstream sites. The reference sites were also characterized by relatively "healthy" communities, as evidenced by the occurrence of several pollution-intolerant taxa, such as mayflies and stoneflies (Hubbard and Peters 1978, Surdick and Gaufin 1978). Thus, preliminary analyses indicate that the reference sites selected are suitable for characterizing unimpacted benthic communities in this watershed.

Preliminary results of the characterization phase of the benthic macroinvertebrate monitoring program in First Creek, Fifth Creek, Melton Branch, Northwest Tributary, and WOC indicate that varying degrees of impact have occurred on the benthic communities in each of these streams. The most severely impacted site in the watershed appeared to be lower Fifth Creek (FFK 0.2). Density, biomass, and taxonomic richness of this site were significantly lower than at upper Fifth Creek (FFK 1.0) and typically were also much lower than at any other site in the watershed. Mean monthly density at this site never exceeded 4.3 individuals per 0.1 m^2, which is considerably lower than densities normally found in relatively undisturbed streams on the Oak Ridge Reservation (30 to 500 individuals per 0.1 m^2; Loar et al. 1981a; Sherwood and Loar 1987; G. F. Cada, unpublished data).

The low density and species richness at FFK 0.2 are indicative of a toxic stress (Hynes 1960, Wiederholm 1984). These data are also consistent with results obtained from ambient toxicity tests for this site, which showed significant reductions in survival of *Ceriodaphnia* and fathead minnows (Fig. 3.5). Although chlorine has been implicated as a possible toxicant at this site (Sect. 3.1.3.4), other factors, including elevated temperatures and/or artificially high and stable flow regimes, could also be causing stress to the benthic macroinvertebrates.

Major differences were noted in the benthic community between the upper and lower sites on First Creek. Density, biomass, and taxonomic richness were significantly higher at the reference site than at FCK 0.1. The benthic community of this stream shifted from one of relative complexity (snails, amphipods, beetles, worms, stoneflies, mayflies, and caddisflies) at FCK 0.8 to a much simpler community at FCK 0.1 (chironomids, caddisflies, oligochaetes, and beetles); at FCK 0.1, mayflies were totally absent, and only one stonefly was collected. These data strongly suggest that lower First Creek has been impacted, but the source of the impact is not known. Although an average level of 4 ug/L total residual chlorine observed in lower First Creek is not toxic (Fig. 3.6 and Sect. 3.1.5), higher concentrations may have occurred but were not detected due to the episodic nature of the inputs. There also appeared to be evidence of an increase in the amount of silt at FCK 0.1 relative to the reference site. Silt can affect invertebrates both directly (e.g., obstruction of food collection and respiration) and indirectly (e.g., habitat modification and change in food quality) (Minshall 1984, Wiederholm 1984). The source of the silt is most likely past construction activities in areas upstream of this site (e.g., construction of a weir).

Evidence of stress was less obvious in lower Northwest Tributary, where no significant differences were found in density, biomass, richness, or diversity. NTK 0.2 was consistently dominated numerically by chironomids and caddisflies (primarily *Cheumatopsyche*), while considerable monthly variability occurred in the taxa numerically dominating the community at NTK 1.0. Upper Northwest Tributary was typically dominated numerically by chironomids and isopods, but relatively large numbers of beetles, craneflies, and stoneflies were sometimes found. The most striking difference between upper and lower Northwest Tributary was the total absence of mayflies and stoneflies from NTK 0.2. This observation, along with the decreasing complexity of the community at the lower site, is suggestive of stress. Alteration of the benthic community at this site is most likely the result of discharges from the 1500 area, which may contain low concentrations of chlorine that could cause chronic stress to the benthic community.

The discharges into Northwest Tributary may also alter the natural flow and thermal regimes. Because some discharges are routed into ponds prior to discharge into the Northwest Tributary, retention of the water in these ponds increases its temperature due to longer and direct exposures to sunlight. Depending upon the extent of alterations in flow rate and temperature, density and biomass can either increase or decrease, while the number of taxa can either decrease or remain unchanged (Hynes 1970, Ward and Stanford 1979, Sweeney 1984, Wiederholm 1984).

Another possible impact of the ponds could be through alteration of the food of the benthic invertebrates. Retention of water in the ponds permits increases in plankton, which then serves as food for filter-feeding organisms. This hypothesis could possibly explain why filter feeders such as *Cheumatopsyche* and *Corbicula* do well at NTK 0.2.

Density and biomass of benthic invertebrates in the lower two sites of Melton Branch (MEK 0.6 and MEK 1.2) were considerably higher than at the reference site (MEK 2.1) in both May and June, although these differences were not significant. In contrast, the mean number of taxa per sample and diversity were significantly greater at MEK 2.1 during this same 2-month period. The most striking difference between MEK 2.1 and the lower two sites was the total number of taxa collected. At MEK 2.1, the total number of taxa collected during May and June equaled the total number collected from May through October at MEK 0.6 and exceeded the total for May through October at MEK 1.2 by 23 taxa. These data indicate that the benthic communities at MEK 0.6 and MEK 1.2 have been impacted. Maximum adverse effects on the benthic community in Melton Branch were found at the middle site (MEK 1.2), where, from May through October, the mean richness and taxonomic diversity were significantly lower and total richness was substantially lower than at MEK 0.6. The most likely sources of stress are the discharges from the HFIR/TRU ponds and cooling tower blowdown, both of which enter Melton Branch via a small tributary located ~200 m upstream of MEK 1.2. The tributary draining the Molten Salt Reactor Experiment facilities (located ~50 to 100 m upstream of MEK 0.6) may also be a source of stress to the benthos at MEK 0.6. However, the occurrence of a "healthier" community at MEK 0.6 relative to that at MEK 1.2 implies that the major source of stress to the system is the tributary draining the HFIR/TRU area.

The combination of relatively high densities of invertebrates and low species richness at the lower two sites on Melton Branch is indicative of nutrient enrichment (Wiederholm 1984) and is consistent with the finding of high levels of phosphorus in the lower reaches of the stream (Tables 2.3 and 2.5). Elevated temperatures could also stress the benthos in lower Melton Branch. Depending upon the extent of the increase, elevated temperatures can either reduce or increase the density, biomass, and richness of the benthic community (Wiederholm 1984). Temperature measurements taken within 30 min of each other at MEK 2.1 and MEK 1.2 in May and June showed that temperatures were at least 8°C higher at MEK 1.2. Elevated temperatures were less pronounced at MEK 0.6; the temperature was higher at MEK 0.6 than at MEK 2.1 in May, but the reverse was true in June. Flow augmentation via the HFIR tributary could have a profound effect on the benthos of Melton Branch. During dry periods in summer, little or no flow occurs immediately above the HFIR tributary (Sect. 2.1), which results in little or no dilution of the wastewater entering Melton Branch.

Varying degrees of impact were observed at all sites on WOC downstream of the reference site (WCK 6.8). Increasing degradation of the benthic community was observed from WCK 5.1 to WCK 3.9, where the maximum impacts were noted. At WCK 3.9, biomass, mean richness, and diversity were all significantly lower than at any other site on WOC. Densities were also significantly lower at this site compared with those at all others except WCK 2.9. Below WCK 3.9 there was a general trend of increasing improvement in the benthic community downstream to WCK 2.3. However, the density at WCK 2.9 was significantly lower than at WCK 3.4 and WCK 2.3, suggesting that there may be additional stress at this site. Although there was a trend of increasing improvement below the main ORNL complex, improvement can be considered only partial because species richness, relative to that at WCK 6.8, remained significantly depressed at the downstream sites and pollution-sensitive taxa, such as mayflies and stoneflies, were totally absent.

A number of factors could be contributing to the degradation of the benthic invertebrate communities in WOC. Very high concentrations of SRP were observed in WOC south of Bethel Valley Road (Table 2.5). Enrichment of streams with nutrients, such as phosphorus, can reduce species richness while providing ideal conditions for more-tolerant taxa (Hynes 1960, Wiederholm 1984).

Streamflow in WOC is augmented by several sources (Table 2.1). This factor not only alters the natural flow regime of WOC but also could alter the natural thermal regime. As discussed previously, modifications of natural thermal and flow regimes can affect the composition of benthic communities.

The low density, biomass, richness, and diversity at WCK 3.9 suggest that this site is impacted by one or more toxicants. This hypothesis is consistent with the results obtained from the ambient toxicity tests with *Ceriodaphnia* in which both fecundity and survival were reduced (Fig. 3.4 and Table 3.5). Chlorine is hypothesized to be a source of toxicity at this site (Sect. 3.1.3.4).

A slight depression in the density and species richness at WCK 2.9, relative to WCK 3.4 and WCK 2.3, may be indicative of additional perturbations at this site. Although a small tributary that drains a portion of SWSA 4 enters WOC ~100 m upstream of site WCK 2.9 (Fig. 2.3), water quality samples collected at WCK 2.9 in September 1986 (Table 2.5 and Appendix A) did not confirm the hypothesis that the observed ecological effects were related to contaminant releases from SWSA 4.

In addition to the previously noted site-specific perturbations, past land-use practices (e.g., clear-cutting, channelization, dredging) in WOC watershed have doubtlessly affected the benthic communities in most of the streams. Clear-cutting could affect benthic invertebrates by altering (1) the food base of a stream (e.g., greater autochthonous production due to increased light penetration and lower allochthonous production due to reduced terrestrial inputs, such as leaves) and (2) the natural thermal regime of the stream due to greater exposure to sunlight. Where substantial areas have been cleared of vegetation, both siltation and flow rates may increase. Thus, in the worst cases, clear-cutting could reduce the density, biomass, and species richness of the benthic community, although recent studies suggest that density, biomass, and production of invertebrates can increase when the canopy is removed (Hawkins et al. 1982, Behmer and Hawkins 1986). The most immediate effect of stream channel disturbance (e.g., dredging or channelization) is the temporary elimination of all invertebrates from the disturbed area. Invertebrates downstream of the sites may also be affected, primarily by siltation. Frequently, however, recovery of the benthic communities at such sites is rapid due to recolonization from unimpacted upstream areas. In the absence of historical data on invertebrate populations prior to and following such disturbances, the extent of their impact on the existing populations cannot be determined. Data collected to date, however, indicate that the benthic communities inhabiting certain streams within the watershed (e.g., lower Fifth Creek and the middle reaches of WOC and Melton Branch) are most likely a reflection of current ORNL operations.

Previous Studies. In a recent synoptic survey of the benthic macroinvertebrates of WOC and its tributaries (Sherwood and Loar 1987 but also reported in Loar et al. 1991), several conclusions were made from comparisons of the results of that survey with past studies (Krumholz 1954a; Blaylock, unpublished data, as reported in Loar et al. 1981a; Loar et al. 1981a). Those that are applicable to the results of the current study include the following:

(1) the reference site on WOC north of Bethel Valley Road appears "healthy" and unimpacted; (2) the benthic fauna of WOC south of Bethel Valley Road is still impacted but shows some evidence of improvement in lower WOC; (3) the benthos in Melton Branch below the tributary draining the HFIR area are still adversely impacted; and (4) the benthic community in lower Northwest Tributary remains moderately impacted, and little change has occurred since 1974.

Two of the sites included in the survey conducted in August–September 1985 (Sherwood and Loar 1987) were relocated at the beginning of the present study, including MEK 1.4, which was moved downstream ~200 m, and NTK 0.3, which was moved downstream ~100 m. One of the major differences observed between the new and old sites on each stream was the absence of mayflies. Based on the limited 1985 survey, it was hypothesized that many of the invertebrates collected at MEK 1.4 were not inhabitants but rather had drifted into the area from above the HFIR tributary. The results of the present study indicate that this was most likely the case; not only were no mayflies collected at MEK 1.2, but the total number of taxa collected in the earlier survey (at MEK 1.4), which included only a onetime collection of three samples, exceeded the total number of taxa collected in the present study (at MEK 1.2) by 10. The lower site on Northwest Tributary was moved downstream for the present study in order to provide a characterization of the benthos below all discharges into the stream. Although the absence of mayflies in this study suggests that discharges from the 1505 area may be impacting this stream, the benthic community at NTK 0.2 in August 1986 was remarkably similar to the benthic community at NTK 0.3 in August 1985, with both *Cheumatopsyche* (47.9% in 1986 and 37% in 1985) and elmid beetles (8.3% in 1986 and 36% in 1985) dominating (Sherwood and Loar 1987).

One of the most notable changes in WOC, observed both in this study and in the recent survey of Sherwood and Loar (1987), was the occasional appearance of large numbers of filter-feeding caddisflies from WCK 3.4 down to WCK 2.3. This group of insects was virtually absent during the 1979–80 study conducted by Loar et al. (1981a). This suggests some recovery of the benthic community in lower WOC. Although some improvement in water quality has probably occurred since the 1979–80 study, the upgrading of an existing weir at WCK 2.65 and the construction of a new weir at WCK 3.41 have probably had a significant influence on the appearance of this group of insects. Construction of weirs can be beneficial to benthic invertebrates in some cases (Hynes 1960). The retention of polluted water behind weirs provides more time for self-purification through biological processes and settling of suspended solids. Another effect of the retention of water is increased production of planktonic organisms. The combination of plankton production and retention of solids results in a highly concentrated source of high-quality food for filter-feeding organisms below the weirs. Construction of even very small dams can result in considerable increases in filter-feeding caddisflies (Mackay and Waters 1986). A negative aspect of weirs is that they can serve as barriers to recolonization by fish (see Sects. 6.2.3.1 and 6.3.3) and by invertebrates that lack aerial stages (Hynes 1960).

6.1.4.2 White Oak Lake

Since its impoundment in 1943 as a final settling basin for effluents discharged into WOC, WOL has undergone a number of significant changes, such as alterations in the water level and complete drainage (Loar et al. 1981a). Because WOL serves as a settling basin and has undergone these changes, an appropriate control was not available; therefore, it is best to assess the status of the benthos in WOL only in the context of the results of past studies.

Previous studies of the benthos in WOL were conducted by Krumholz (1954b) in the early 1950s, by Blaylock (unpublished data) in late 1974 and early 1975, and by Loar et al. (1981a) in 1979–80. Krumholz sampled the littoral zone extensively over a 3-year period and collected a total of 64 taxa (Krumholz 1954b, Tables 6 and 7). Quantitative sampling by Krumholz (1954b) and by Blaylock (unpublished data), however, was limited to late fall through early spring, and a total of eight and seven taxa, respectively, were collected. Loar et al. (1981a), on the other hand, collected quantitative samples from midspring through midfall and collected a total of 14 taxa, approximately the number collected in the present study (12 taxa). Insects, particularly dipterans, were found to be numerically dominant in all four studies. However, in some cases, considerable differences were found in the species composition of WOL. All studies reported similar densities of chironomids, and in all but the Krumholz study this group was usually dominant. In the early 1950s, the phantom midge, *Chaoborus*, was found to be the most abundant. This dipteran was rare during the studies by Blaylock (unpublished data) and Loar et al. (1981a), while in the present study, *Chaoborus* was collected in each of the three sampling periods. If, as Loar et al. (1981a) suggested, a real decline had occurred in the *Chaoborus* population 20 to 25 years after Krumholz's initial study, the results of the current study suggest that this taxon may have returned as a major component of the benthic community in WOL. Studies have shown that seasonal changes in the density of *Chaoborus* can be considerable (Czarnezki 1977, Hare and Carter 1986) and that the pattern of seasonal change can be highly species specific (Hare and Carter 1986). Furthermore, densities can be highly dependent on depth, with greater depths usually preferred (Czarnezki 1977, Hare and Carter 1986). Therefore, a number of factors could account for the differences between the studies on WOL, including (1) collection of samples during different seasons, resulting in the occurrence or absence of *Chaoborus* in the samples due to natural seasonal variations, and/or annual variations in the life history of the species due to annual variability in temperature and food availability; (2) differences in sampling technique; (3) possible differences in sample location; and (4) a real decline, possibly due to factors such as a change or changes in suitable habitat or water quality. Collection of benthos from WOL over a full annual cycle may provide insight into the reasons for the differences between these studies.

Some other notable differences were found between this study and that of Loar et al. (1981a). In the earlier survey, relatively high numbers of ceratopogonids (*Palpomyia*), mayflies (*Caenis* and *Callibaetis*), and snails (*Physa* = *Physella* in the present study) were collected in some months. In the present study, mayflies were never collected, and snails and ceratopogonids were very minor components of the benthic community of WOL. Subtle differences in the types of habitats sampled during the two surveys may account for the absence or low numbers of these taxa in the present study. For example, both *Caenis* and *Callibaetis* prefer the shallower regions of standing water, typically around rooted vegetation (Edmunds et al. 1976). In the present study, no samples were collected within 10 m of the shoreline, which may explain why these taxa were not collected. Timing of sample collection may also be important. For example, Loar et al. (1981a) found the greatest number of ceratopogonids in April, after which their numbers remained relatively low; samples were not collected in April in the present study. The absence or low numbers of these taxa could also be the result of an actual decline in abundance or extirpation of the species from WOL. A more extensive collection of the benthos in WOL is needed to help ascertain these differences.

6.1.5 Summary and Conclusions

Quantitative benthic macroinvertebrate samples were collected from May through October 1986, at approximately monthly intervals from 16 stream sites within WOC watershed, including First Creek, Fifth Creek, Melton Branch, Northwest Tributary, and WOC. Additionally, samples were collected at ~60-d intervals from WOL. A total of 161 taxa were collected from the stream sites, while 12 taxa were collected from WOL. A majority of the organisms collected at all sites were insects. Additional taxa collected included flatworms, roundworms, aquatic earthworms, aquatic sow bugs, sideswimmers, crayfish, snails, and mussels.

The upstream reference site on each stream was characterized by highly diverse and complex communities typically dominated numerically by several taxa and by relatively large numbers of pollution-intolerant taxa, such as mayflies and stoneflies. Varying degrees of impact were found at all stream sites below the respective reference sites. These downstream sites were typically characterized by a reduction in the complexity of the community, by a smaller total number of taxa collected, and, at all but NTK 0.2, by significant reductions in mean density, biomass, and taxonomic richness. Low to moderately impacted sites typically had only two or three numerically dominant taxa, including chironomids, hydropsychid caddisflies, and/or aquatic earthworms. The least-impacted downstream site was NTK 0.2; however, some impact at this site was implied by the absence of mayflies and stoneflies. Those downstream sites exhibiting moderate to moderately high impact included FCK 0.1, MEK 0.6, MEK 1.2, WCK 2.3, WCK 2.9, WCK 3.4, and WCK 5.1. The most severely impacted sites were lower Fifth Creek (FFK 0.2) and the middle reaches of WOC (WCK 3.9). Densities of invertebrates never exceeded 4.3 individuals per 0.1 m^2 at FFK 0.2, and collection of all taxa was sporadic. At WCK 3.9, a single group of chironomids (*Cricotopus orthocladius*) was numerically dominant.

In Melton Branch and WOC, which had three and six sampling sites, respectively, maximum impacts were noted in the middle reaches. Both streams showed trends of some improvement at their lower sites, although there were no indications that full recovery had occurred in either stream.

Several potential sources and causes of impact were identified within the watershed. In First Creek, the most likely sources and causes of impact were upstream discharges of chlorine and past weir construction that resulted in siltation and upstream discharges of chlorine from some unknown source or sources. The maximum total residual chlorine concentration, observed ~10 m above the mouth of First Creek (site 9 in Fig. 3.1), was 0.05 mg/L on March 20, 1986 (A. J. Stewart, ORNL, personal communication, 1986). Intermittent discharges of chlorine may also stress the benthos in lower Fifth Creek (Fig. 3.6), although elevated temperatures and increased flow rates cannot be ignored as potential sources of impact. The source of perturbation to the benthos in Melton Branch was hypothesized to be the discharge of effluents from the HFIR/TRU complex. In Melton Branch downstream of the tributary receiving these effluents, temperatures and phosphorus concentrations are elevated, and the natural flow regime has been modified. The cause of impact to lower Northwest Tributary was not clear; it is hypothesized that, in addition to containing low levels of chlorine, discharges from the 1500 area alter the natural flow regime. Additionally, since much of the water at NTK 0.2 is diverted into small ponds before it is discharged to the stream, the natural thermal regime has also been altered. In WOC, all sites are most likely impacted by nutrient enrichment, while WCK 3.9 is also impacted by chlorine.

The benthic community of WOL was found to be basically similar to that reported in earlier studies and to consist primarily of insects, especially chironomids. Although differences were found between the present study and earlier surveys (e.g., mayflies were absent in the present study), it is not clear whether these differences were the result of actual changes in the benthos or of differences in the sampling program.

6.1.6 Future Studies

The preliminary characterization of the benthic macroinvertebrates in WOC watershed indicates the need for a continued intensive sampling program over a complete annual cycle. Therefore, the current schedule of monthly sample collection on WOC, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, as well as the bimonthly sample collection on WOL, will be maintained through April 1988. This will provide a good data base from which the benthos of the watershed can be thoroughly characterized and from which future changes in the benthos can be monitored. Such a thorough characterization will also reduce the need for a continued intensive sampling program; therefore, beginning in May of 1988, the monitoring phase of the program will be initiated, and sampling frequency in the entire watershed will be reduced from monthly to quarterly.

To supplement the data obtained from the quantitative sampling program, qualitative samples will be collected from both reference and impacted sites in the spring of 1987 and 1988. After 1988, qualitative samples will be taken each spring from impacted sites only. In addition to the currently used method for the collection of quantitative samples in WOL, a program will be initiated to quantitatively collect benthic invertebrates in the littoral zones. Sampling will be conducted by using artificial substrates, and the frequency of collection will coincide with collections taken by the Ponar grab. In addition, Ponar grab samples will be collected from the shallow northeastern end of the lake. These data will be used to provide (1) a more complete characterization of WOL and (2) information required in Subtask 6b of BMAP (Sect. 8.2 and Loar et al. 1991).

Data analysis in future reports will incorporate information obtained from reference sites (e.g., Brushy Fork) used in other biological monitoring programs on the DOE Oak Ridge Reservation. This information documents the natural annual variability that might be expected for benthic invertebrate populations in unimpacted streams in the Oak Ridge area. Efforts will also be initiated to estimate secondary production of the benthic invertebrates in WOC watershed.

6.2 FISHES

6.2.1 Introduction

Fish population and community studies can be used to assess the ecological effects of water quality and/or habitat degradation. These studies offer several advantages over other indicators of environmental quality (see review by Karr et al. 1986) and are especially relevant to assessment of the biotic integrity of WOC. Fish communities, for example, comprise several trophic levels, and species that compose the potential sport fishery in WOC (e.g., bluegill, redbreast sunfish, largemouth bass) are at or near the end of food chains. Consequently, fish populations integrate the direct effects of water quality and habitat degradation on primary producers (periphyton) and consumers (benthic invertebrates) that are utilized 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. 1991) 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 (Sect. 2.2.2).

6.2.2 Methods

6.2.2.1 Population surveys

Quantitative sampling of the fish populations at 17 sites in WOC watershed (Fig. 2.3) was conducted by electrofishing in August–September 1985, April–May 1986, and August–September 1986 to estimate population size (densities in numbers and biomass per unit area). Sampling reaches ranged from 27 to 121 m in mean length at the sites on tributaries and from 44 to 160 m at the WOC sites (Table 6.5). Fish sampling sites either overlapped or were within 100 m of the sites included in the instream benthic invertebrate monitoring task (Sect. 6.1), except for FFK 0.4 and MEK 1.5, where benthos were not sampled. Lengths of the sampling reaches were determined by (1) the presence of at least one riffle-pool sequence, if possible; (2) location of suitable places for anchoring upstream and downstream block nets (seines); (3) stream size or order; and (4) the density of fish, as determined by the initial survey in August–September 1985. Results of this survey were also used to determine subsequent collection strategy and the need for additional sampling sites. For example, upstream reference sites on Northwest Tributary and Fifth Creek were added to the sampling program in 1986.

Field Sampling Procedures. All stream sampling was conducted with the use of 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 to 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) so that the electrofisher operator could also collect stunned fish.

After a 0.64-cm-mesh seine was stretched across the upper and lower boundaries of the reach to restrict fish movement, a two- to five-person sampling team made three consecutive passes through the study reach in an upstream direction. 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 passes could not always be made consecutively. 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 segregated by pass in wire mesh cages (0.64 cm diam) or buckets with small holes for storage during further sampling.

Table 6.5. Mean length, mean width, mean surface area, mean depth, and stream order of fish sampling sites in White Oak Creek watershed, August 1985–September 1986

Site[a]	No. of sampling periods	Mean length (m)	Mean width (m)	Mean area (m^2)	Mean depth (cm)	Stream order[b]
FCK 0.1	3	61.0	1.18	72.2	7.4	1
FCK 0.8	3	29.7	1.68	49.8	12.8	1
FFK 0.2	3	64.3	1.12	72.4	10.4	2
FFK 0.4	3	27.7	1.52	41.3	12.2	2
FFK 1.0	2[c]	27.0	1.09	29.3	5.4	2
MEK 0.6	3	51.3	3.02	155.1	15.1	3
MEK 1.4	3	51.0	2.61[d]	135.7[d]	10.8[d]	3
MEK 1.5	2[e]	121.0	1.61	194.4	5.1	2
MEK 2.1	2[f]	33.5	1.37	47.1	5.1	2
NTK 0.3	3	72.7	2.11	154.0	5.3	2
NTK 1.0	2[c]	38.5	1.85	71.0	5.6	2
WCK 2.3	3	91.0	5.12	466.7	27.3	4
WCK 2.9	3	104.7	4.89	510.2	26.3	3
WCK 3.4	3	131.0	3.40	455.6	29.2	3
WCK 3.9	3	160.3	2.74	438.6	14.3	3
WCK 5.1	3	44.7	2.02	90.3	12.1	2
WCK 6.8	3	54.3	2.36	128.2	6.9	2

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

[b]Based on the method of R. E. Horton, "Erosional Development of Streams and Their Drainage Basins," *Geol. Soc. Am. Bull.* **56**, 275–370 (1945).

[c]Sampling at this site was initiated in April 1986.

[d]Width and depth measured only once.

[e]Sampling at this site was terminated after May 1986.

[f]Site was dry in August and September 1986.

After collecting, fish were anesthetized with MS-222 (tricaine methanesulfonate), identified, measured to the nearest 0.1 cm (total length), and weighed by using Pesola spring scales to the nearest 0.1 g (for fish less than 100 g) or to the nearest gram (for fish greater than 100 g). At sites with high fish densities, individuals were recorded by 1-cm size classes and species. If 25 individuals of a species-size class were measured and weighed, additional members of that size class were only measured. Length-weight regressions (SAS 1985b) were later used to estimate missing weight data. Other data recorded included sex (if possible to determine), reproductive state, disposition (i.e., dead or kept for laboratory identification and reference collection), and presence of any abnormalities (e.g., external parasites and skeletal deformities.).

After processing fish from all passes, the fish were allowed to fully recover from the anesthetic 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 the individual fish, conductivity, water temperature, turbidity, dissolved oxygen, cloud cover, and shocking time—as well as length, width, and depth of sample reach—were recorded at each sampling site.

Data Analysis. After reviewing the information on the field data sheets for completeness and accuracy, the data were keypunched, stored on IBM 3033 computers, and analyzed with the use of SAS procedures (SAS 1985a, 1985b).

Species population estimates were calculated by 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 (in square meters) of the study reach. For each sampling date, surface area was estimated by multiplying the length of the reach by the mean width based on measurements taken at 5-m intervals.

Condition factors (K) were calculated for individual fish by site and species by using the formula

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

with weight in grams and total length in centimeters (Hile 1936). Fish without measured weights were not used in calculations of condition factors. The PROC GLM procedure (SAS 1985b) on untransformed data was used to compare condition factors between sites and between sampling periods 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.

6.2.2.2 Age determination

Scales were taken for age determination from target species (redbreast sunfish, bluegill, rock bass, and warmouth) collected during routine population surveys. Because of the low densities of these species at most sites in WOC (Sect. 6.2.3), only data on redbreast sunfish and bluegill collected during the fall of 1986 are included in this report. Scale data for bluegill and redbreast sunfish collected in October 1986 from a reference stream (Brushy Fork; see Sect. 2.3) were used for comparison.

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 by using a Wildco scale press and acetate slides; those that produced poor impressions (due to attached mucous or skin) were mounted between two glass microscope slides that were taped together. Because attempts to improve the impressions by treating scales with potassium hydroxide were not successful, scales were used without cleaning. Enlarged images of the scales were projected on a screen by using an Eberbach 2700 slide projector with a 16-mm lens. Where possible, at least ten scales from each fish were mounted and compared. For actual measurements of annuli, the best representative scale was used and identified on the slide. 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 because all scales were unsuitable. In this preliminary analysis, ages were determined by only one person; in future analyses, age determinations will be independently checked.

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 various annuli. 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.

6.2.3 Results and Discussion

6.2.3.1 Species composition and richness

A total of 16 species were collected from WOC watershed during the three sampling periods between August 1985 and September 1986. Table 6.6 shows the species composition at each site. The lowermost site on WOC (WCK 2.3) had the greatest diversity, 12 species, of which 6 were centrarchids. This site has transient species from WOL, such as the gizzard shad and carp, which were taken only at this locale. The remaining sites were restricted to simple fish communities of two to six species. These communities included blacknose dace/sculpin, blacknose dace/creek chub, and various combinations of these with bluegill and redbreast sunfish, stoneroller, fathead minnow, mosquitofish, and juvenile largemouth bass. Because most of the sites sampled were 1° or 2° streams,* blacknose dace and creek chubs should be, and were, the most common species. Fathead minnows also occurred at many sites, probably from use as laboratory animals in buildings and ponds adjacent to WOC. The fact that the bluegill was the most widespread centrarchid in the WOC system indicates the influence of several settling and fish culture ponds, weir impoundments, and WOL. Two sites, MEK 1.4 and FFK 0.2, consistently had no fish.

As expected, there was a general trend of increasing species richness with increasing stream size (Table 6.6). Although this trend was not well defined in the tributaries, WOC proper had a much richer fish community at the lowest site. A confounding factor in this analysis is the presence of three weirs on the WOC system (Fig. 2.3). These effectively prevent upstream migration of fish and perhaps are the dominant factor producing the difference between the fish community at WCK 2.3 and the communities at WCK 2.9 and WCK 3.4. The weirs also prevent recolonization of Melton Branch following toxic episodes (see Sect. 6.2.3.2).

In comparison with other systems, the fish fauna in WOC watershed seems somewhat depauperate. The majority of the fauna are considered to be tolerant species; rock bass, spotted bass, and banded sculpin were the most intolerant species collected. In this

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*As in Sect. 2.3, *1°* denotes a first-order stream, *2°* denotes a second-order stream, etc.

Table 6.6. Fish species composition in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, August 1985–September 1986

Species	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	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 *Lepomis macrochirus*	3		NF				NF			3	1[b]	3	2	3	2		
Largemouth bass *Micropterus salmoides*										1		2					
Redbreast sunfish *Lepomis auritus*						3						3	2	3			
Rock bass *Ambloplites rupestris*												3					
Spotted bass *Micropterus punctulatus*												1					
Warmouth *Lepomis gulosus*												3					
Clupeidae																	
Gizzard shad *Dorosoma cepedianum*												1					
Cottidae		2		2	2[b]												3
Banded sculpin *Cottus carolinae*																	
Cyprinidae																	
Carp *Cyprinus carpio*												1					
Blacknose dace *Rhinichthys atratulus*	3	3		2	2[b]	3		2[b]	2[b]	3	2[b]		2	2	1	3	3
Central stoneroller *Campostoma anomalum*												1				3	3
Creek chub *Semotilus atromaculatus*	2	3				3		2[b]	2[b]	1	2[b]		2	1	1	3	3

Table 6.6 (continued)

Species	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	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
Fathead minnow *Pimephales promelas*	3									3	2[b]		3	3	3	2	
Ictaluridae																	
Black bullhead *Ictalurus melas*												1					
Yellow bullhead *Ictalurus natalis*												2					
Poeciliidae																	
Mosquitofish *Gambusia affinis*	3									3	1[b]	3	3	3	1		
Number of species	5	3	NF	2	2	3	NF	2	2	6	5	12	6	6	5	4	4

[a]Values represent the number of sampling periods ($n = 3$) that a given species was collected at the site. NF = no fish collected. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

[b]Sampled only twice during this period.

discussion, species tolerance relates to ability to adapt to human-related stresses (e.g., chemical contamination and habitat alteration) as discussed by Karr et al. (1986). The basses occurred only at WCK 2.3, while the sculpin occurred at upper or headwater sites (FCK 0.8, FFK 0.4, FFK 1.0, and WCK 6.8). Also, two families of fishes (Percidae and Catostomidae) and several genera (*Notropis*, *Noturus*, and *Phoxinus*) commonly found in this area and indicative of high water quality (D. A. Etnier, The University of Tennessee, Knoxville, personal communication, 1986) were absent from WOC. The fish community in Brushy Fork, a tributary of Poplar Creek north of Oak Ridge, had many intolerant species, including several species of Percidae, Catostomidae, and *Notropis* (J. M. Loar, unpublished data). Earlier surveys of WOC included some members of these taxa, as well as *Pomoxis* and *Nocomis* (L. A. Krumholz, unpublished data, 1954a). Again, the disappearance may be a result of not only poor water quality but also effective isolation from the remainder of the Clinch River basin by White Oak Dam. Krumholz (1954a) observed a decline in certain genera (*Moxostoma* and *Pomoxis*) in his surveys of WOL, and, after White Oak Dam was completely closed in 1960 to Watts Bar backwater, these species were not identified in surveys by Kolehmainen and Nelson (1969).

6.2.3.2 Density and biomass

Total densities and biomass at each site for each sampling period are given in Table 6.7. Densities and biomass for individual species at each site and sampling period are given in Tables E.1 through E.6 (Appendix E).

A general pattern in the fish densities of WOC and its tributaries was noted. Densities were highest in the upper reaches of the streams, usually at the uppermost site. The values at upper sites were on the order of 2 to 40 times greater than those at the lower sites. The highest densities in the entire WOC watershed were found at WCK 5.1 and FFK 0.4, both of which were dominated by cyprinid species, the central stoneroller and blacknose dace respectively. Densities decreased downstream but not always linearly. The lowest densities in the watershed occurred at WCK 3.9, followed by WCK 2.9 and MEK 0.6. The low densities at WCK 3.9 and the absence of fish at FFK 0.2 can probably be attributed to toxic effects associated with chlorination of cooling water (see Sect. 3.1.4). Karr et al. (1985) documented similar effects of residual chlorine on the biotic integrity of fish communities over a short distance comparable to this reach of WOC. Just below the confluence of the HFIR tributary with Melton Branch (MEK 1.4) and at MEK 0.6, the absence or low numbers of fish may result, in part, from the elevated temperature of the HFIR discharge (G. F. Cada, ORNL, personal communication, 1986) and perhaps from toxic inputs from a tributary just east of SWSA 5 (Fig. 2.3). Cada (ORNL, personal communication, 1986) observed a few creek chubs at MEK 1.4 in December 1985 when stream temperatures dropped below lethal limits for that species. The causes of low densities at WCK 2.9 are more uncertain; they may be related to inputs from seeps and a tributary that drain SWSA 4, although water quality data from site WCK 2.9 did not confirm this theory (Table 2.5 and Appendix A).

These patterns were consistent over all sampling periods. Overall, densities at all sites were frequently higher in late summer, with very high densities in 1986. In comparison with Bear Creek and EFPC, area streams with similar species, densities were up to four times higher in WOC at WCK 5.1 (J. M. Loar, unpublished data). The large increase in density at FFK 0.4 from 0 in late summer of 1985 to 6.29 in spring of 1986 was due to a slight shift

Table 6.7. Total density (number of individuals per square meter) and total biomass (grams per square meter) for three sampling periods in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Sampling period	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	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
August–September 1985																	
Total density	2.48	4.14	NF	NF	NS	0.43	NF	1.17	5.40	0.68	NS	0.28	0.03	0.20	<0.01	10.12	3.26
Total biomass	6.13	4.45	NF	NF	NS	3.40	NF	0.48	0.90	1.00	NS	7.24	0.36	4.50	0.05	20.19	6.82
April–May 1986																	
Total density	1.31	2.14	NF	6.29	4.08	0.23	NF	0.65	2.52	0.88	1.49	0.24	0.29	0.18	<0.01	8.12	1.20
Total biomass	3.92	3.37	NF	21.58	14.94	2.40	NF	0.75	2.40	1.09	2.93	4.90	1.24	8.80	<0.01	33.56	2.99
August–September 1986																	
Total density	2.89	5.50	NF	10.83	2.79	0.07	NF	NS	DS	3.84	0.44	1.00	1.36	1.77	0.02	15.71	2.66
Total biomass	2.72	6.25	NF	18.02	6.35	0.57	NF	NS	DS	1.73	1.18	8.08	2.64	16.37	0.34	25.16	5.37

[a]NF = no fish taken in sample; NS = site not sampled on this date; DS = site dry at time of sampling. FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

in the sampling reach. This change resulted in the inclusion of a large pool and avoided an area that is influenced by cooling tower blowdown.

A similar shift in the densities at MEK 0.6 may have been related to discharges from the Silver Recovery Facility in the HFIR area. During May 1986, releases from this facility lowered dissolved oxygen levels, resulting in two fish kills involving "small minnows and bluegill"; totals of 7 and 50 fish were collected at the weir at MEK 0.16 (Shoemaker et al. 1986). Because Melton Branch is isolated from the remainder of the WOC by this weir and normal fish densities are low, such kills may have accounted for the continued decline of densities from 0.43 fish per square meter on August 23, 1985; to 0.23 fish per square meter on April 30, 1986; to 0.07 fish per square meter on September 24, 1986. The first part of the decline followed a pattern common throughout WOC and also observed in Melton Branch by Cada (ORNL, personal communication, 1986); that is, spring densities were lower than the previous fall densities. However, unlike other sites, the density at MEK 0.6 continued to decline. Thus, the total impact of Silver Recovery Facility discharges may have been greater than that indicated by the size of the fish kill.

The blacknose dace was usually the species with the highest densities wherever it occurred in WOC watershed. (The mean of all sites was 1.96 fish per square meter.) Creek chubs, although occurring widely in WOC, did not have high densities. (The mean of all sites was 0.27 fish per square meter.) Densities of other species, such as mosquitofish and bluegill, usually remained constant at all sites. Bluegill densities were comparable to densities in EFPC and Brushy Fork (J. M. Loar, unpublished data). Mosquitofish densities increased abruptly in late summer 1986 from previous levels. (The mean of all sites increased from 0.12 fish per square meter to 0.90 fish per square meter.) It is not clear if this was a biological change or reflected more-intensive sampling for this species in 1986 than in 1985.

An interesting pattern was observed with the densities of fathead minnows in the WOC system (Tables E.1 through E.3). In fall 1985, the mean density of all WOC watershed sites (five) was only 0.01–0.03 fish per square meter. In the spring and late summer of 1986, however, the values increased to 0.15 fish per square meter and 0.10 fish per square meter, respectively, at seven sites. The fathead minnow was not reported in WOC in surveys prior to 1985 (L. A. Krumholz, unpublished data, 1954b; Kolehmainen and Nelson 1969; Loar et al. 1981a) or even in surveys conducted in the early 1940s of this area of the Clinch River basin (D. A. Etnier, The University of Tennessee, Knoxville, personal communication, 1978). The presence of fathead minnow in WOC can be traced to their use as a laboratory species; their recent population growth may reflect increased use, and associated escape, in laboratory situations and/or may indicate establishment of a reproducing population in WOC.

A general pattern of decreasing mean biomass with increasing stream size was observed at the sites on WOC. Again, WCK 5.1 had the highest biomass due to an apparently robust population of all age groups of three cyprinids. Sites WCK 2.3 and WCK 3.4 also exhibited high biomass due to sunfish populations. These peaks were interrupted by dramatically lower biomass levels at WCK 2.9 and WCK 3.9. The lowest mean total biomass in WOC was observed at WCK 3.9. Biomass at most sites in WOC was comparable to that in EFPC (2–10 g/m^2) but was lower than the biomass in Brushy Fork (17–19 g/m^2) (J. M. Loar, unpublished data). Mean total biomass in the tributaries was extremely variable; no consistent pattern occurred between the upstream and downstream sites. Sites on Fifth and First creeks generally had higher values among the tributary sites, while FFK 0.4 had the second highest biomass in WOC watershed. The lowest mean total biomass of the tributaries occurred at

NTK 0.3, where the community consisted of immature centrarchids, cyprinids, and mosquitofish.

Unlike density, no seasonal pattern was apparent in the biomass data. Values for the late summer samples were not consistently higher or lower than those for the spring sample.

Biomass measurements of individual species indicated that the blacknose dace was consistently the greatest contributor to biomass at the upper WOC and tributary sites. (The mean total biomass of all sites was 2.60 g/m^2.) The creek chub (mean total biomass = 0.79 g/m^2) and bluegill (mean total biomass = 1.47 g/m^2) also contributed substantially to the biomass throughout the WOC system. The central stoneroller had a high mean total biomass (7.79 g/m^2) due to the contribution of only one (WCK 5.1) of the three sites where it was found.

6.2.3.3 Growth and condition

Growth and condition of the WOC watershed fish were analyzed by calculating condition factors for all species and by estimating a population growth rate for redbreast and bluegill sunfish. The latter analysis required determining the age classes of the sunfish by interpretation of scale annuli.

The population growth rate, or apparent growth rate, is a comparison of mean size of surviving fish at successive ages (Ricker 1975). It is not always an accurate estimate of the true growth rate because size-selective mortality within an age class can affect (usually by lowering) the resulting growth curve. For comparisons of growth rates between sites, however, similar biases could be assumed, and the resulting curves would indicate site-specific growth differences. The population growth rate for redbreast sunfish at sites WCK 2.3, WCK 3.4, and a reference stream (Brushy Fork) foı the period of late September to early October 1986 is shown in Fig. 6.6. The 0+ age class represents young-of-the-year fish that had not formed an annulus; succeeding age classes demonstrated the same number of annuli, as indicated by their age-class designation. The curve for Brushy Fork is smooth, with rapid initial growth followed by progressively slower growth. This pattern would be expected in a stream with a typical temperate-region temperature regime, like Brushy Fork (J. M. Loar, unpublished data), and few atypical growth disruptions (e.g., thermal pollution). The curves for WCK 3.4 and WCK 2.3 generally indicate a rate similar to that of Brushy Fork. The frequently higher mean weights for older age classes in WOC might be a result of increased food availability, a hypothesis that will be examined further in 1987 (Sect. 5.5).

A similar pattern of growth was observed in bluegill sunfish at the same sites (Fig. 6.7). The greater variability in growth that occurred at WCK 2.3 (at age 2+) may be a reflection of sample size. In both species, growth at site WCK 3.4 appeared to be consistently higher than at WCK 2.3 or Brushy Fork. Population growth rates, as shown in Figs. 6.6 and 6.7, are a long-term, integrative measure of growth and may differ from estimates of short-term growth rates (Sect. 5.3.1.3) that reflect only conditions in the immediate past.

Condition factors (K) for the more common species in WOC, calculated based on the methods of Hile (1936) provide an evaluation of the relative well-being of the fish, because fish with more weight per length have a higher condition factor (Everhart et al. 1975). Caution must be exercised in the interpretation of the condition factor because it appears to be relatively insensitive to environmental conditions or nutritional status (Loar et al. 1985).

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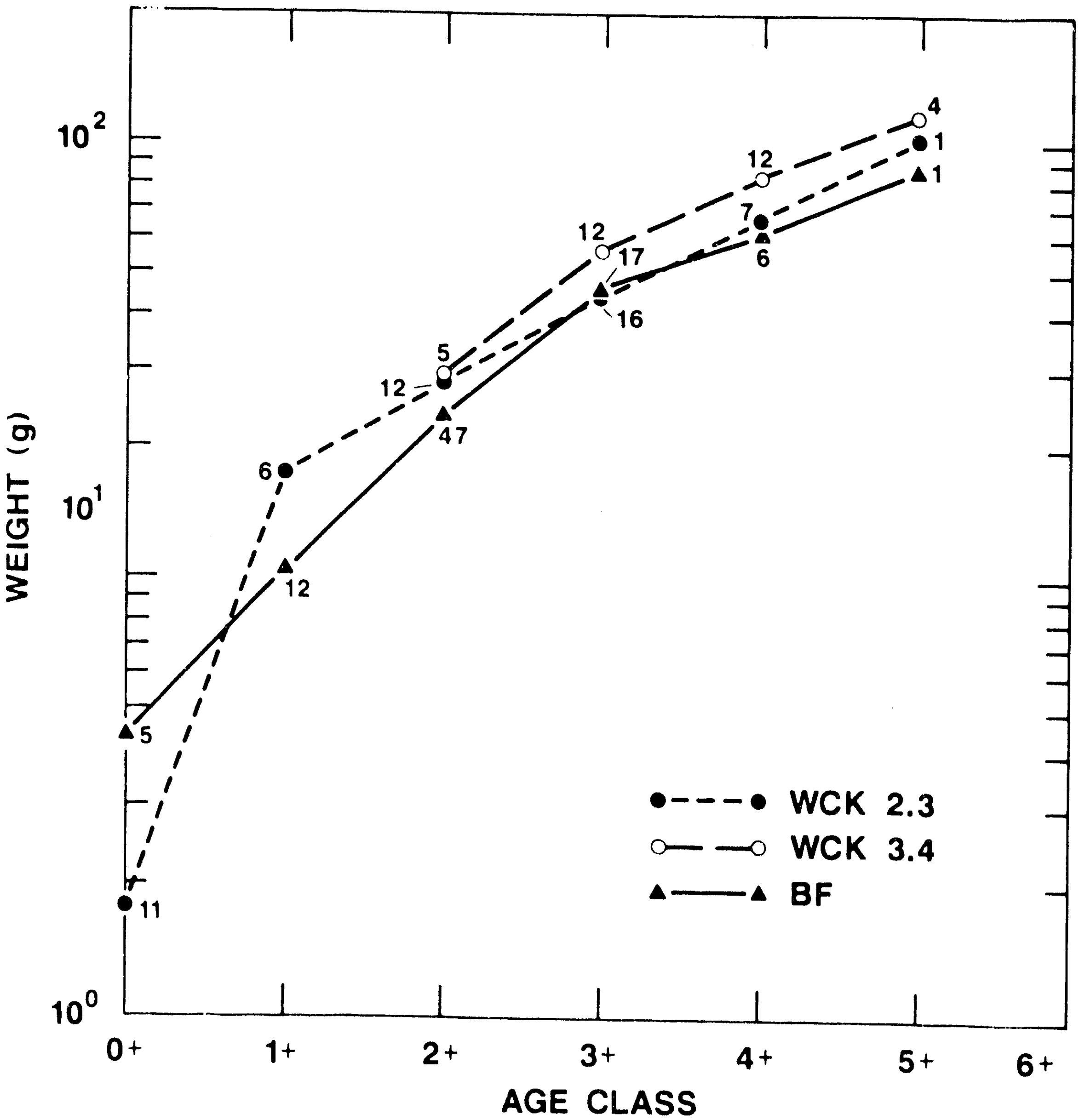

Fig. 6.6. Mean weight at age of redbreast sunfish at White Oak Creek kilometer (WCK) 2.3, WCK 3.4, and Brushy Fork (BF), the reference site, for September–October 1986. Numbers on curves represent sample size.

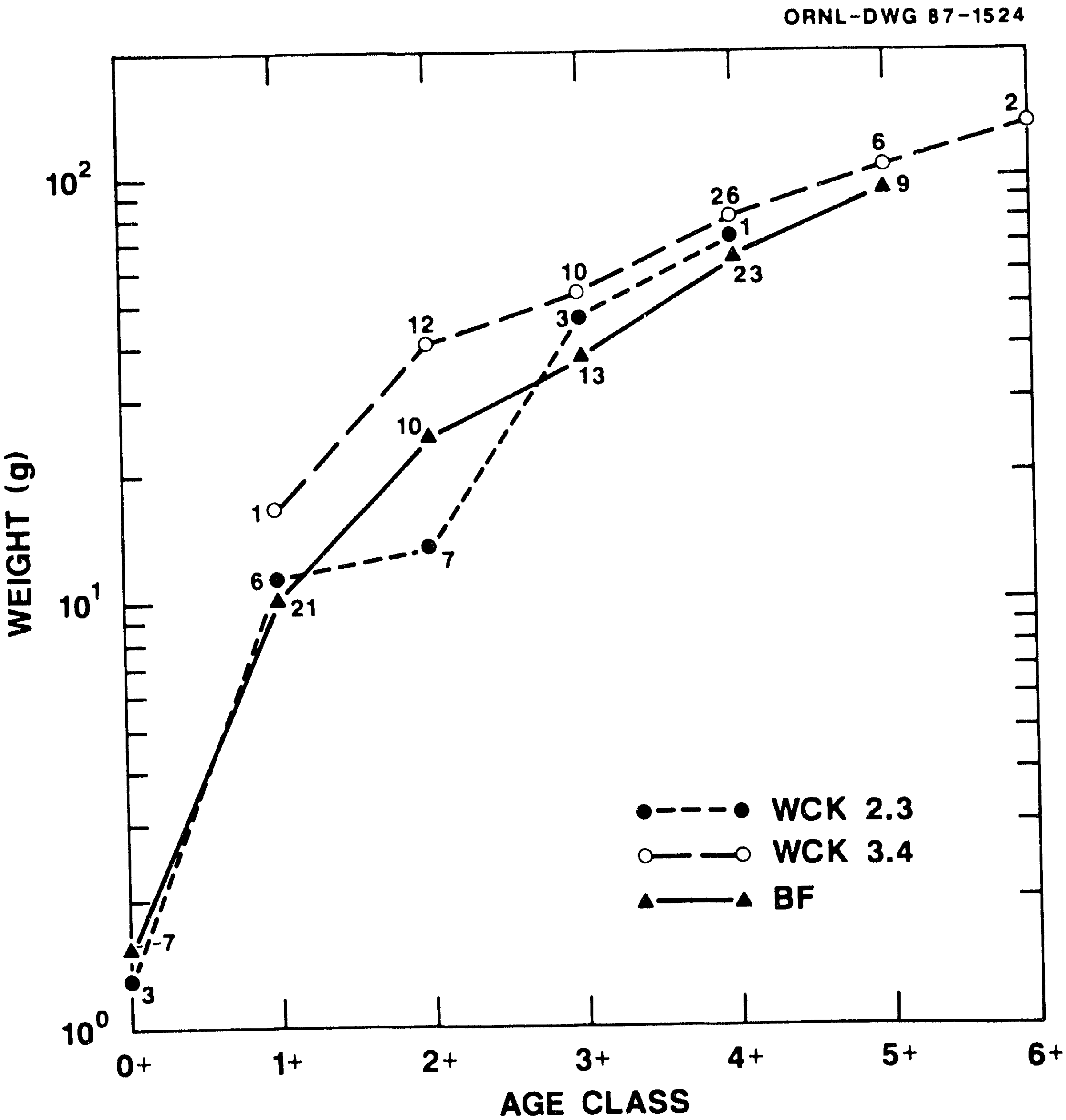

Fig. 6.7. Mean weight at age of bluegill sunfish at White Oak Creek kilometer (WCK) 2.3, WCK 3.4, and Brushy Fork (BF), the reference site, for September–October 1986. Numbers on curves represent sample size.

Condition factors for species that occurred at more than one site were compared statistically within sampling periods (Appendix E, Tables E.7 through E.9). Few statistically significant differences ($p < 0.05$) between sites were found for most species during any sampling period. Similar results were found for redbreast sunfish at three WOC sites that were sampled in the bioindicator studies (Fig. 5.3). No site consistently produced higher mean K values for a majority of species. Also, there was not a tendency for one species to have consistently higher K's at a particular site. The species with the most significant differences between sites was the creek chub; these differences were probably due to a few large individuals (with high K's) inhabiting sites not used as rearing areas by the creek chub. In comparison with the population growth rates of bluegill and redbreast sunfish (Figs. 6.6 and 6.7), higher mean condition factors were not found at site WCK 3.4 than at WCK 2.3 in late summer 1986, and the differences in mean K between the two sites were not statistically significant (Table E.9). Condition factors have been found to vary inversely with density (Wiener and Hanneman 1982), and a similar relationship was found for bluegill and redbreast sunfish at WCK 2.3 compared with WCK 3.4 during most sampling periods. In these comparisons, however, the condition factors were not significantly different, and the trend did not hold for other sites.

Condition factors were also compared between sampling periods at each site (Appendix E, Tables E.10 through E.12). In these comparisons, a trend toward higher mean K in the spring and more statistically significant differences from other sampling periods was apparent. These higher condition factors demonstrated (1) the expected increase in K associated with enlargement of gonads for spawning and (2) winter mortality of smaller, young-of-the-year fish. A trend toward decreasing K over the three sampling periods at MEK 0.6 was not apparent (Tables E.10 and E.11). Such results might indicate that discharges from the HFIR area that were responsible for fish kills did not affect the long-term well-being of survivors.

6.2.4 Summary

Data obtained in Subtask 4b of BMAP included fish population estimates and growth parameters at 17 sites in WOC watershed over the period between August 1985–September 1986. Data analyses included species composition and richness, density, biomass, growth of target species, and condition factor for two sampling periods in late summer and one in the spring. The purpose of these analyses was to characterize the structural properties of fish populations and communities in WOC watershed and to determine impacts of the ORNL facilities on these populations.

A total of 16 species were collected from all sites in WOC watershed during the three sampling periods. Site WCK 2.3 had the greatest diversity (12 species); the remaining sites had simple communities consisting of 2 to 6 species. Although blacknose dace and creek chubs were the most common species, fathead minnows, mosquitofish, and bluegills also were abundant at several sites. No fish were collected in Melton Branch just below the confluence of the HFIR tributary (MEK 1.4) and in lower Fifth Creek (FFK 0.2).

There was a general trend of increasing species richness with increasing stream size. The presence of three weirs on the WOC system effectively prevents upstream migration of fish and may be a dominant factor controlling fish community structure between WCK 2.3 and upper WOC. The weirs would also prevent recolonization of Melton Branch following toxic discharges. The majority of the community is composed of tolerant species; rock bass, spotted bass, and sculpins are the most intolerant. Low species richness appears to be a result not only of poor water quality at several locations but also of effective isolation from the remainder of the Clinch basin.

Densities were highest in the upper reaches of the streams, usually at the uppermost (or reference) site. These values were 2 to 40 times greater than at lower sites. The highest densities were found at WCK 05.1 and FFK 0.4; the lowest densities occurred at WCK 3.9, WCK 2.9, and MEK 0.6. The low densities or absence of fish at WCK 3.9, WCK 2.9, FFK 0.2, MEK 0.6, and MEK 1.4 occurred downstream of known discharge locations. Densities at all sites were higher in the late summer sampling periods, especially in 1986. Blacknose dace was usually the species with the highest densities.

A pattern of decreasing mean fish biomass per unit area with increasing stream size was apparent in WOC. Site WCK 5.1 had the highest biomass in WOC due to robust cyprinid populations, and sites WCK 2.3 and WCK 3.4 had high biomass values due to centrarchid populations. The lowest mean total biomass was at WCK 3.9. Sites on Fifth and First creeks generally had the highest values among tributary sites, with FFK 0.4 having the second highest biomass of all WOC watershed sites. Seasonal biomass patterns were not apparent for all sites. Blacknose dace biomass was generally the highest of all species.

Growth of bluegill and redbreast sunfish was evaluated by using the population growth rate, a comparison of mean size of surviving fish at successive ages. Fish at the reference site (Brushy Fork) demonstrated a smooth pattern of growth, with a high initial growth rate. The growth rates for sunfish at WCK 2.3 and WCK 3.4 had the same general pattern, but weights at age were greater than similarly aged fish in Brushy Fork. The higher absolute weights may be due to elevated temperatures or increased food availability.

Few statistically significant differences in K values between sites were indicated for most species. In comparison with the population growth rates observed for bluegill and redbreast sunfish, higher mean condition factors were not found at site WCK 3.4 than at WCK 2.3 in late summer 1986. Comparison of the condition factors between sampling periods indicated a tendency for the spring 1986 sample to have higher mean K's and more statistically significant differences than those from other sampling periods. The higher K in spring reflected (1) the expected increase in condition factor associated with enlargement of gonads for spawning and (2) winter mortality of smaller, young-of-the-year fish.

6.2.5 Conclusions

At this stage of the biological monitoring and characterization of WOC watershed, only general indications and some trends can be identified. All conclusions of this report are presented as hypotheses requiring further investigation in 1987.

Areas of WOC watershed represented by the headwater sites FFK 1.0, WCK 6.8, and NTK 1.0 appear to have healthy fish communities relative to other WOC sites. The species composition, densities, and biomass at these sites do not indicate any significant problems related to ORNL facilities. Sites FFK 0.4 and WCK 5.1 represent areas with levels well above those expected, perhaps because of phosphorus enrichment (see Sect. 3.1.2). Species density and biomass indicated significant increases above levels that are typical of 2° streams. Significant adverse impacts of ORNL operations were demonstrated at sites FFK 0.2, WCK 3.9, MEK 1.4, MEK 0.6, and possibly at WCK 2.9. These areas all showed depressed species diversity, densities, and biomass relative to other sites in the watershed. Potential agents responsible for these depressed conditions include elevated, near-lethal temperatures; nutrient enrichment; episodic releases of various contaminants; and chlorine. Two WOC sites, WCK 3.4 and WCK 2.3, had intermediate species richness, density, or biomass conditions relative to the other WOC watershed sites.

The trend over the August 1985 to September 1986 sampling period indicated a general improvement in the WOC system. Species richness, density, and biomass were all elevated compared with earlier surveys (e.g., Loar et al. 1981a). This trend is typical of a recovering system (Turnpenny and Williams 1981) with more-tolerant species (e.g., mosquitofish) demonstrating earlier and more-extensive recovery than more-sensitive species (e.g., sculpins). A major factor in interpreting the extent of current recovery and the potential for future recovery is the isolation of WOC by several weirs and White Oak Dam from the remainder of the Clinch system. Without access, intolerant species can never recolonize the WOC area regardless of the degree of water quality improvement (Turnpenny and Williams 1981). Within the WOC system, Melton Branch demonstrates the effect of isolation to a greater degree. For example, density and biomass showed a steady decrease following various impacts, and recovery and recolonization from WOC have been prevented.

6.2.6 Future Studies

As described in BMAP (Loar et al. 1991), sampling in WOC will continue to focus on characterizing the fish fauna of the watershed in 1987. Such sampling will include quantitative population estimates at the sites described in this report; MEK 1.5, however, has been dropped because of similarities with MEK 2.1. The population estimate schedule will be stabilized to a spring/fall sampling cycle. This schedule will still provide sufficient data to estimate fish production and to evaluate effluent impacts. Elimination of a late-summer sample will also reduce inadvertent mortality during sampling by avoiding periods of high water temperatures when fish are more vulnerable to additional levels of stress. To provide an estimate of total biomass in WOL, as required in Subtask 6b of BMAP, a quantitative estimate of the size of the fish population will be obtained by using a mark-recapture technique in late spring (Sect. 8.2).

In addition to quantitative sampling, qualitative sampling will be conducted throughout WOC watershed during the summer. The sampling sites will include the various experimental ponds (near the 1500 complex), settling basins, and weir impoundments associated with WOC. The goal of the qualitative sampling is to provide a more comprehensive species list and additional data on areas of WOC and WOL potentially affected by effluents.

Data for age and growth evaluations of the target sunfish species (bluegill and redbreast) will be collected during the fall sampling period. These data will include scales taken during the quantitative population estimates at WCK 2.3 and WCK 3.4 as well as additional sampling done above and below the sites to provide a sufficient sample size. Scales will also be collected from target species in WOL and in the various ponds/impoundments to allow comparison of growth rates and to evaluate any growth difference in fish from the lentic/lotic environments in WOC watershed.

6.3 INTERPRETATION OF BIOTIC CHANGES

6.3.1 Introduction

Previous studies of the fish and benthic invertebrate communities of WOC watershed discussed in Loar et al. (1981a) and the results of present studies (Sects. 6.1 and 6.2) indicate that, although conditions remained stable or improved somewhat between 1980 and 1986, significant adverse impacts on the biota of some stream reaches are evident. These impacts are shown most convincingly by both low biomass and low numbers of species, as at sites FFK 0.2 and WCK 3.9.

Low biomass or low species diversity can be caused by a wide variety of natural conditions. However, under natural conditions there is usually an inverse correlation between these two measures of community structure: where biomass is high, species diversity is often low, and where species diversity is high, biomass is often low (Huston 1979). The combination of both low biomass and low diversity at several sites suggests that nonnatural factors are influencing the stream communities.

Interpretation of both the differences in community structure between different sections of WOC watershed and the changes in community structure that occur through time requires an understanding of the responses of community biomass and species diversity to conditions that would be naturally found in streams of the size and condition of the various streams in WOC watershed. "Improvement" of conditions in a stream could result in either an increase or decrease in diversity or biomass, depending on the initial conditions of the stream community and the processes that are affecting it. For this reason, a much more broadly based control is needed than can be provided by a few reaches within the target watershed and nearby areas. In a situation where conditions are expected to change, such a control must include a wide variety of natural conditions and, most importantly, an understanding of how natural streams respond to changing conditions.

The objective of this study (Subtask 4c of BMAP, as discussed in Loar et al. 1991) is to develop a methodology, based on ecological theory and data from a large number of streams within the region, that will allow evaluation of the ecological condition of the various streams in WOC watershed and of the significance of changes in those conditions that occur following implementation of remedial actions.

6.3.2 Methods

The first stage of this study is to assemble a data base that will help place WOC watershed in a regional context and allow each site to be compared with streams of similar size and structure that have been less severely affected by human activity. No single stream can serve as an adequate control, because every stream differs in some way from every other. However, a regional data base from a large number of streams can help detect general patterns of community structure that would not be obvious from a study of only a few streams.

The major progress to date has been the initiation of a data base of fish and benthic invertebrate community structure, in collaboration with Dr. David Etnier, The University of Tennessee. The principal sources of data are a series of TVA studies of streams and rivers of the Tennessee Valley region (7 river systems with over 100 stream sites studied between

1968 and 1972) and The University of Tennessee Baseline Stream Survey of 20 stream sites conducted in 1981 (Etnier et al. 1983). In addition to the numbers and relative abundances of fish and invertebrates, the data base includes measure of stream size, structure, and water quality. Compilation of the core data is nearly complete. This data set will be supplemented with additional comparable data from Drs. D. A. Etnier and G. A. Vaughan, The University of Tennessee Department of Zoology. A major task in 1987 is the analysis of this data base and the evaluation of WOC in this regional context.

Community structure can be characterized not only by the number or identity of species that are present but also by the functional types of organisms, the relative proportion of each functional group, and the number of species composing that group. This information allows a much more sensitive evaluation of the ecological condition of a stream, as affected by both natural processes and human disturbances. This approach, in addition to more-conventional measures, will be used in the analysis of the data base and will permit a more accurate prediction of how the stream community should respond to changing conditions, such as those resulting from remedial actions.

The second stage of this task is to develop a theoretical model of stream community structure that will allow prediction of the response of the biotic community to changes resulting from remedial actions designed to "improve" conditions in the stream. Such a model must also incorporate the effects of natural processes, such as droughts or increased rainfall, that will inevitably be superimposed on the intentional changes. The model to be developed will be based on the dynamic equilibrium theory of community structure (Huston 1979, 1985), which incorporates processes that affect growth, reproduction, and productivity, as well as factors that cause mortality.

The long history of research on aquatic ecosystem models and food-web structure in the Environmental Sciences Division (ESD) at ORNL will greatly facilitate the development of this model. Data on growth rates and mortality of test organisms (Sect. 3.1) and native species (Sect. 6.2) for sites in WOC watershed will allow the model to be fitted to specific conditions in WOC.

6.3.3 Discussion

At this point in BMAP, it is too early to draw any conclusions about the way in which the biotic community of WOC is responding to changing conditions. Some interesting phenomena have been identified, such as the apparent invasion and spread of the fathead minnow (*Pimephales promelas*), that give some insight into the condition of the biotic community.

While certain taxa are known to be indicators of specific environmental conditions, either favorable or unfavorable, many species are essentially interchangeable in the functions they perform in a community. For this reason, the degree of overlap in the species lists of two streams indicates relatively little about the similarity or difference between their communities. An assessment of trophic structure based on the functional roles of the species present gives a much better indication of community structure and will be used in the analysis of the regional data base.

The community structure of a stream is an indication of the stream's ecological conditions only if all functional types of organisms that could naturally occur there actually have the opportunity to occur there. For example, a species of fish that could occur in a stream may not be present because it was eliminated by temporarily unfavorable conditions and then was unable to recolonize the stream due to obstacles, such as waterfalls, dams, or thermal barriers, even though conditions above the obstacle are suitable for its survival. This phenomenon may explain the low fish species diversity in parts of WOC watershed and may prevent an increase in fish species diversity even if conditions improve sufficiently for more species to live there (see Sect. 6.3.2.1). This sort of biogeographical constraint on immigration and dispersal probably influences fish communities throughout the Oak Ridge Reservation and the entire Tennessee Valley. Once a species has been eliminated for whatever reason from a reach or stream, its return may be more limited by barriers to dispersal than by conditions in the site where it previously occurred.

Benthic invertebrate communities are much less likely to be influenced by such biogeographical constraints because most stream insects are able to fly during the adult stage of their life cycle. For this reason, the insect community is likely to show a much faster response to improving conditions than is the fish community.

The apparent invasion and spread of the fathead minnow (discussed in Sect. 6.2.3.2) may be a consequence of such physical constraints on the natural fish community. If native fish species that are functionally similar to the fathead minnow were eliminated from WOC during the past when water quality was much poorer, they may still be absent even though water quality has improved enough for them to survive. Thus an "empty niche" may exist in the fish community because the invasion of native species, which could fill that niche, has been prevented by some sort of barrier. In this situation, a nonnative species, such as the fathead minnow, could rapidly populate the stream following its accidental introduction because of the lack of competition from native species with the same role in the community.

An understanding of the structure of the fish and benthic invertebrate communities of WOC watershed and of the principal natural factors that influence them is essential for interpreting the changes in community structure that are anticipated over the course of BMAP, due to both intentional alteration of stream conditions and unintentional changes resulting from natural processes.

7. ASSESSMENT OF CONTAMINANTS IN THE TERRESTRIAL ENVIRONMENT

Development of an effective biological monitoring program for the terrestrial environment presents special difficulties not encountered in aquatic environs. Fewer data exist on contaminant types, amounts, and sources for the terrestrial environment near ORNL, and much of the available data for radionuclides were obtained many years ago. Thus, these data are not reliable indicators of current levels of contamination. The slower movement of contaminants in soils compared with that in air and water, as well as the greater potential for spatial variation of contaminant concentrations in soils, complicates the task of surveying large land areas. Sampling must be conducted more intensively within a given area, and sources or hot spots cannot be readily tracked from flow patterns as in aquatic systems. Screening for contaminants is further complicated by the fact that chemicals present in soils may not reach either surface waters or groundwaters in sufficient concentrations to be detected during routine water quality surveys. Because organic contaminants with high sorption coefficients (K_p) in soil may not be found in surrounding waters yet may still be available to terrestrial biota to produce toxic effects, a separate terrestrial monitoring program is needed to supplement the aquatic monitoring activities.

Terrestrial biota can be used to overcome some of the problems of identifying contaminants in terrestrial systems because they are temporal and spatial integrators of toxicant exposures and can reveal the biological availability of many chemical toxicants; however, terrestrial animals are more difficult to sample than aquatic species. Whereas most aquatic animals can be obtained by electroshocking, seining, and netting, most terrestrial species must be trapped. The handling and transport of wild animals to the laboratory is more difficult and poses greater hazards of disease and bodily injury to workers.

While methods for biological monitoring in aquatic environments have been intensively studied and extensively developed over the past decades, resulting in standard procedures and approaches for conducting many aspects of a biological monitoring program (Roop and Hunsaker 1985, Bergman et al. 1986), the same is not true for terrestrial monitoring. The merit of using wild species as indicators of chemical contamination has been recognized in wildlife toxicology, yet the methods have not been developed to correlate biological endpoints with contaminant exposure and dose. Attempts to apply the techniques of wildlife and population biology to detect toxicological effects of contaminants on natural populations of animals have proved ineffective because they are time-consuming, expensive, and typically yield equivocal results, a problem resulting in part from the high variability inherent in natural systems.

The objectives of Task 5, as stated in BMAP (Sect. 3.5), are (1) to document what radioactive and organic contaminants are present in elevated amounts in the terrestrial environment at ORNL, (2) to examine the potential for mobility and availability of these contaminants to terrestrial biota, and (3) to select the appropriate organisms and monitoring approaches for more-detailed biological monitoring, as needed (Loar et al. 1991).

The approach of the terrestrial monitoring program is to focus on analysis of plants and animals that can provide specific data on the mobility and biological availability of contaminants. Analyses of soil and water are undertaken only as appropriate to document indications of contamination. New techniques for biochemical detection of animal exposures are being investigated for their appropriateness for a more detailed biological monitoring program that will be based on initial scoping data.

The specific objectives of Task 5 of BMAP are (1) to document what radioactive and organic contaminants are present in elevated amounts in the terrestrial environment at ORNL, (2) to examine the potential for mobility and availability of these contaminants to terrestrial biota, and (3) to select the appropriate organisms and monitoring approaches for more-detailed biological monitoring, as needed (Loar et al. 1991).

Studies undertaken in 1986 were directed at surveying the terrestrial environment in the vicinity of SWSA 4, SWSA 5, the 3524 Equalization Basin, the ORNL Steam Plant, WOL, WOC, and Bearden Creek (reference site). Samples were also collected from the vicinity of EFPC, Brushy Fork, and Bull Run Steam Plant for use in comparative analyses. Contaminants surveyed in these scoping activities included ^{90}Sr, ^{137}Cs, ^{60}Co, ^{75}Se, benzo[a]pyrene (BaP), and mercury. In addition, an intensive field survey was conducted of tritium (^{3}H) in the area adjacent to the ORNL SWSAs. Organisms sampled included several species of mammals and turtles. Grasses, tree parts, water, and soil samples were collected and analyzed as needed.

7.1 FIELD SURVEY OF ENVIRONMENTAL TRITIUM IN AREAS ADJACENT TO ORNL SOLID RADIOACTIVE WASTE DISPOSAL/STORAGE AREAS

7.1.1 Introduction

Studies were initiated in early 1986 on ^{3}H in terrestrial environments adjacent to the ORNL SWSAs where ^{3}H contamination is known to occur. These studies were undertaken to meet several objectives: (1) to survey the spatial pattern of ^{3}H in surface waters and shallow well waters of WOC watershed as an indicator of ^{3}H sources; (2) to determine the levels of ^{3}H in atmospheric moisture and the relationship of such levels to ^{3}H levels in soil; (3) to determine the human health significance, if any, of atmospheric ^{3}H concentrations in the watershed; and (4) to determine the history of ^{3}H releases from SWSAs prior to 1964 from tree cores and predict future trends in ^{3}H releases. A summary of this investigation follows; for a more comprehensive discussion of the results of this study, see Amano et al. (1987).

7.1.2 Methods

7.1.2.1 Survey of ^{3}H in surface waters and shallow groundwaters

In February and March 1986, water samples were taken from major tributaries and shallow groundwater wells throughout WOC watershed. Groundwater wells were identified from maps and the metal tags present on each well casing. Water samples were collected in plastic 1-L bottles and distilled to separate ^{3}H from other radionuclides. The condensate was analyzed for ^{3}H by liquid scintillation counting in Aquasol. The detection limit was ~0.5 nCi/L.

7.1.2.2 Survey of ^{3}H in surface soils and air moisture

Based on the survey of ^{3}H in surface waters and well waters, seven sites were selected throughout the watershed for sampling of ^{3}H in soil water and atmospheric moisture in March 1986. These sites were two locations south of SWSA 4, two locations south of SWSA 5, the east seep between pits 2–4 and Trench 5, north WOL bed, and near Building 1505 (Figs. 2.3 and 2.5).

Soil samples were taken with a 2-cm-diam soil probe. Samples were divided into two sections: top (surface to 10 cm deep) and bottom (10 to 20 cm deep). Each section was placed in a glass test tube and stoppered with a rubber stopper to prevent evaporation. Soil water was removed by vacuum distillation (~760 mm negative pressure at room temperature), trapped in liquid nitrogen, and sampled for liquid scintillation counting. Air moisture was collected at each site by suspending two clean glass flasks filled with dry ice at a distance of 25 and 80 cm above the soil. After ~1 h, the air moisture, which had crystallized on the exterior portion of the flask, was removed, melted, and sampled for liquid scintillation counting.

7.1.2.3 Pine survey south of SWSA 5

Loblolly pines (*Pinus taeda*) were selected along the south perimeter of SWSA 5 and sampled for ^{3}H. A core was removed from each tree with an increment corer and immediately placed in sealed glass tubes to prevent evaporation of water. In the laboratory, each entire core was soaked for 1 week in a known amount of distilled water that then was sampled to determine the concentration of free ^{3}H in pine core water. Extraction experiments showed that ^{3}H in the soak water reached steady state in less than 1 week. Cores were then dried at 60°C and cut into yearly increments based on the visible tree rings. Each increment was burned in a Packard Tri-Carb Sample Oxidizer, and the vaporized ^{3}H was trapped and dispensed into a mixture of water and scintillator.

7.1.3 Results

7.1.3.1 Concentrations of ^{3}H in surface waters and shallow groundwaters

Tritium concentrations were below the detection limit in Northwest Tributary and in WOC within the ORNL complex. The highest ^{3}H concentration (17 μCi/L) was found in a small stream (middle drainage tributary) draining the south side of SWSA 5. Tritium concentrations in WOC were less than those found in the tributary immediately south of SWSA 4 because of dilution. Concentrations in Melton Branch at the confluence with WOC (3.2 μCi/L) were two orders of magnitude greater than concentrations in WOC. This survey showed that SWSA 4 and SWSA 5 (south) were major sources of ^{3}H contamination to WOC and Melton Branch in the spring of 1986.

Tritium concentrations in shallow wells were generally less than concentrations observed in surface waters, except for wells immediately south of SWSA 5. The highest ^{3}H concentration in well water (370 μCi/L) was found in a shallow (8-m-deep) unidentified well located on the Melton Branch floodplain south of SWSA 5. As was the case for the survey of surface waters, well water samples indicated that SWSA 4 and SWSA 5 (south) were major sources of ^{3}H contamination to the watershed in the spring of 1986.

7.1.3.2 Concentrations of ^{3}H in surface soils and air moisture

Tritium concentrations in soil water were usually less in topsoil (top 10 cm) than in subsoils (10–20 cm). The highest ^{3}H concentrations in soil water (>10 μCi/L) were found along the Melton Branch floodplain south of SWSA 5. Soils in this area were also very wet during the spring of 1986. Concentrations of ^{3}H in soil water at sites south of SWSA 5 were ~100 times greater than ^{3}H concentrations in soil water at any of the other six sampling locations.

Tritium concentrations in air moisture collected immediately south of SWSA 5 along Melton Branch floodplain were ~1000 times greater than concentrations measured in air moisture collected near Building 1505 near the main ORNL complex (Fig. 7.1). Tritium concentrations in air moisture immediately south of SWSA 4 and on the northern portion of WOL bed were also elevated above concentrations measured near Building 1505. Building 1505 is not a "control" site, because of atmospheric releases of ^{3}H from the main ORNL complex.

As expected, ^{3}H concentrations in air moisture close to the ground (25-cm height) were one to three times higher than concentrations in air moisture collected 80 cm above the ground. There was a strong positive correlation between ^{3}H concentrations in surface air moisture and ^{3}H concentrations in surface soil water across the seven sites surveyed in March 1986 (Fig. 7.2). The relationship between surface air (25- and 80-cm height) and surface soil was described by the equation:

$$Y = 0.61X^{0.88},\ r^2 = 0.82,\ n = 14\ ,$$

where

X = ^{3}H concentration in surface soil water,
Y = ^{3}H concentration in surface air.

Relative humidity at the time of the March air sampling was 30 to 36% at the seven sites sampled. (Dry bulb temperature was 25°C.) Relative humidity during the December 1986 air sampling below SWSA 5 was 60%. (Dry bulb temperature was 8.5°C.) Based on the air temperature and relative humidity data, ^{3}H concentrations in air moisture were converted to microcuries of tritium per cubic centimeter of air for comparison with levels that may give an annual dose of 5 rem/year to workers. Based on International Commission on Radiological Protection Publication 30 (ICRP 30), a ^{3}H concentration of 22 pCi/cm^3 air over a 40-h workweek and a 50-week year is sufficient to deliver a dose of 5 rem/year to soft tissue (ICRP 1980). Air concentrations at two locations south of SWSA 5 were more than an order of magnitude below the 22 pCi/cm^3 air value.

7.1.3.3 Pine survey south of SWSA 5

Tritium concentrations in the tree cores along the southern perimeter of SWSA 5 increased in a westerly direction as one approached the middle drainage tributary of SWSA 5. The pattern of ^{3}H concentrations in tree cores indicated that ^{3}H seepage from SWSA 5 was

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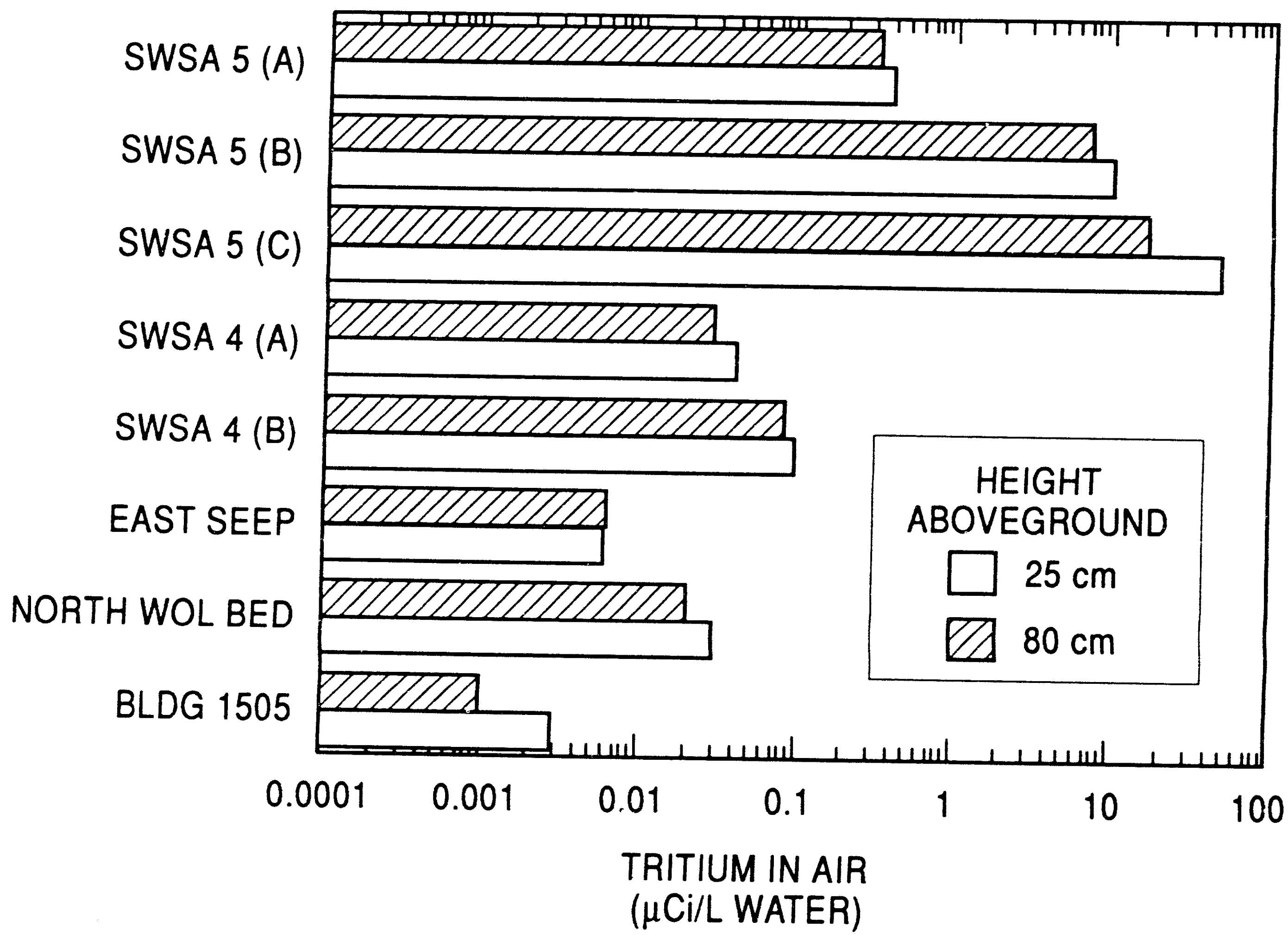

Fig. 7.1. Tritium concentrations in atmospheric moisture at 25 and 80 cm aboveground at eight sites in White Oak Creek watershed (spring 1986). SWSA = solid radioactive waste disposal/storage area; WOL = White Oak Lake.

ORNL-DWG 91Z-14887

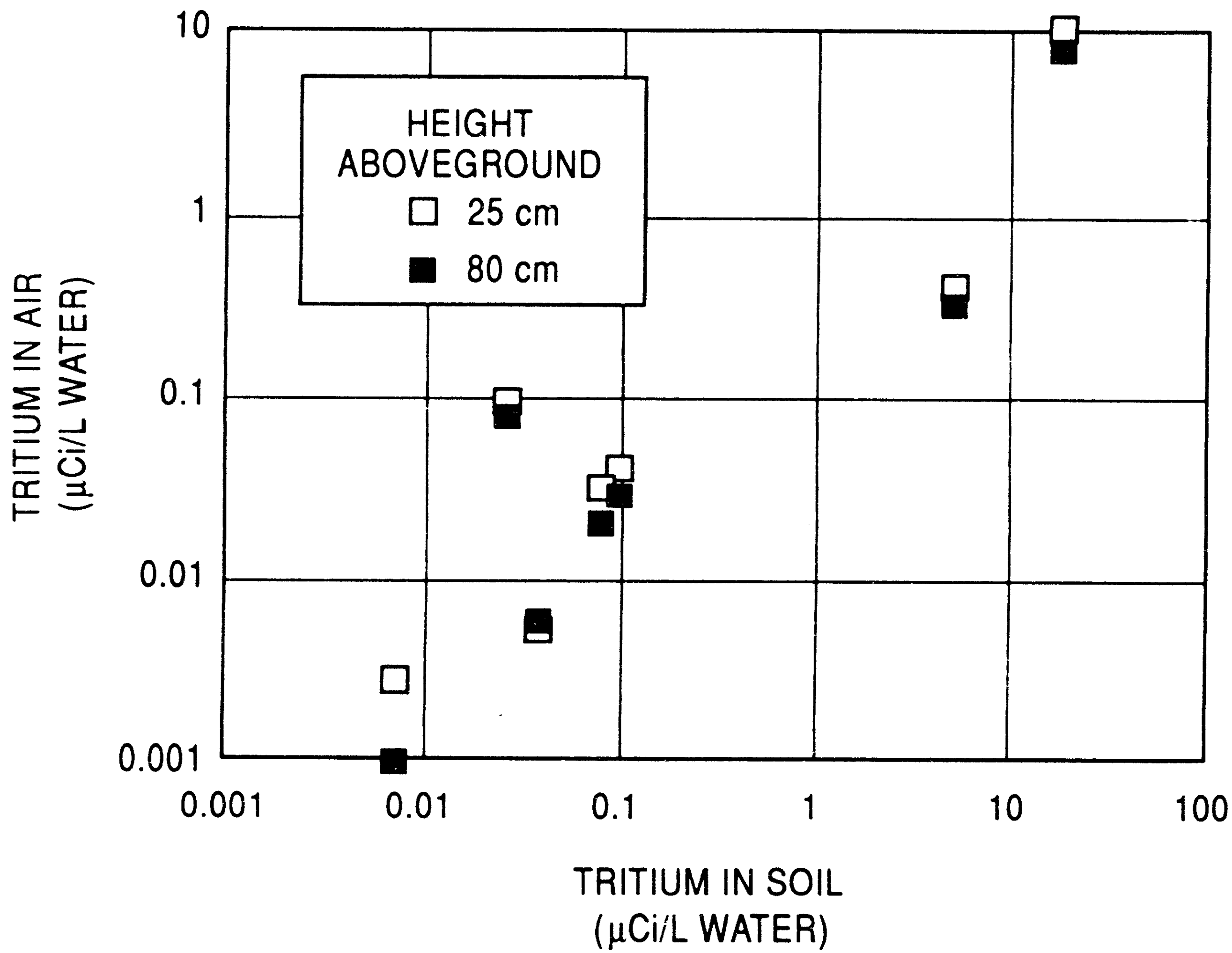

Fig. 7.2. Relationship between tritium in atmospheric moisture and tritium in soil water (0–10 cm) from sites within White Oak Creek watershed (1986).

greatest in the vicinity of the middle drainage, but there was some indication of increased seepage from the southeast corner of the burial ground.

Cores from trees 11 and 12 were selected for interpretation of time trends because these two trees were the most highly contaminated of those surveyed. Concentrations of ^{3}H were averaged over 5-year increments to reduce the influence of year-to-year fluctuations. Time history data for these trees indicate that ^{3}H releases have been increasing from 1965 to 1985 in the vicinity of the middle drainage tributary of SWSA 5. Both trees show similar time history trends after 1965. Prior to 1965, tree 11 showed a peak in ^{3}H concentration in the period 1951 to 1955, which was not evident in tree 12 (Fig. 7.3). Prior analysis of tree core data from pines located south of SWSA 5 indicated that ^{3}H releases from the burial ground were increasing from 1960 to 1975 (Auerbach et al. 1976). The present data are consistent with that prior interpretation and indicate that, during the last decade, ^{3}H migration from SWSA 5 has perhaps continued to increase beyond 1975 levels (Fig. 7.3). However, it is uncertain to what extent the patterns observed from these trees are representative of discharges from all of SWSA 5.

7.1.4 Conclusions

A survey of ^{3}H concentrations in surface waters and shallow well waters during 1986 showed that SWSA 4 and SWSA 5 are major contributors of ^{3}H to WOC watershed. Tritium concentrations in Melton Branch were higher than those in WOC at the confluence of these two streams.

Tritium concentrations in soil water and atmospheric moisture were greater in areas immediately south of SWSA 5 than in areas immediately south of SWSA 4. Tritium concentrations in air moisture increased in a westerly direction along the southern portion of SWSA 5 as one approached the middle tributary drainage of the burial ground.

Tritium concentrations in air showed both spatial and seasonal variation. Concentrations in air moisture at different heights above the ground were more uniform during summer than during winter. This difference can be attributed to the presence of tritiated water vapor transpired by tree foliage and the drying of the surface soil during summer. During winter the water table is nearer to the soil surface; consequently, ^{3}H concentrations in air moisture collected near the ground were elevated. There was a strong positive correlation between ^{3}H concentrations in air moisture and ^{3}H concentrations in surface (0- to 10-cm) soils.

During this study, the highest concentrations of ^{3}H in air did not exceed ICRP 30 recommended limits for occupational exposure (40-h workweek). Nevertheless, ^{3}H concentrations in air were high enough to merit periodic monitoring, particularly in seepage areas south of SWSA 4 and SWSA 5 and in the immediate vicinity of the middle drainage tributary on SWSA 5, where remedial action engineering measures might be undertaken by laboratory or contractor personnel.

Patterns of ^{3}H in air moisture, surface waters, and pine cores indicate that there is a major area of ^{3}H seepage from SWSA 5 near the middle drainage tributary. Tritium concentrations in tree cores from pines south of SWSA 5 indicate that ^{3}H migration from SWSA 5 in the vicinity of the middle drainage tributary has increased during the last decade to a level above that which occurred prior to 1975.

ORNL-DWG 91Z-14888

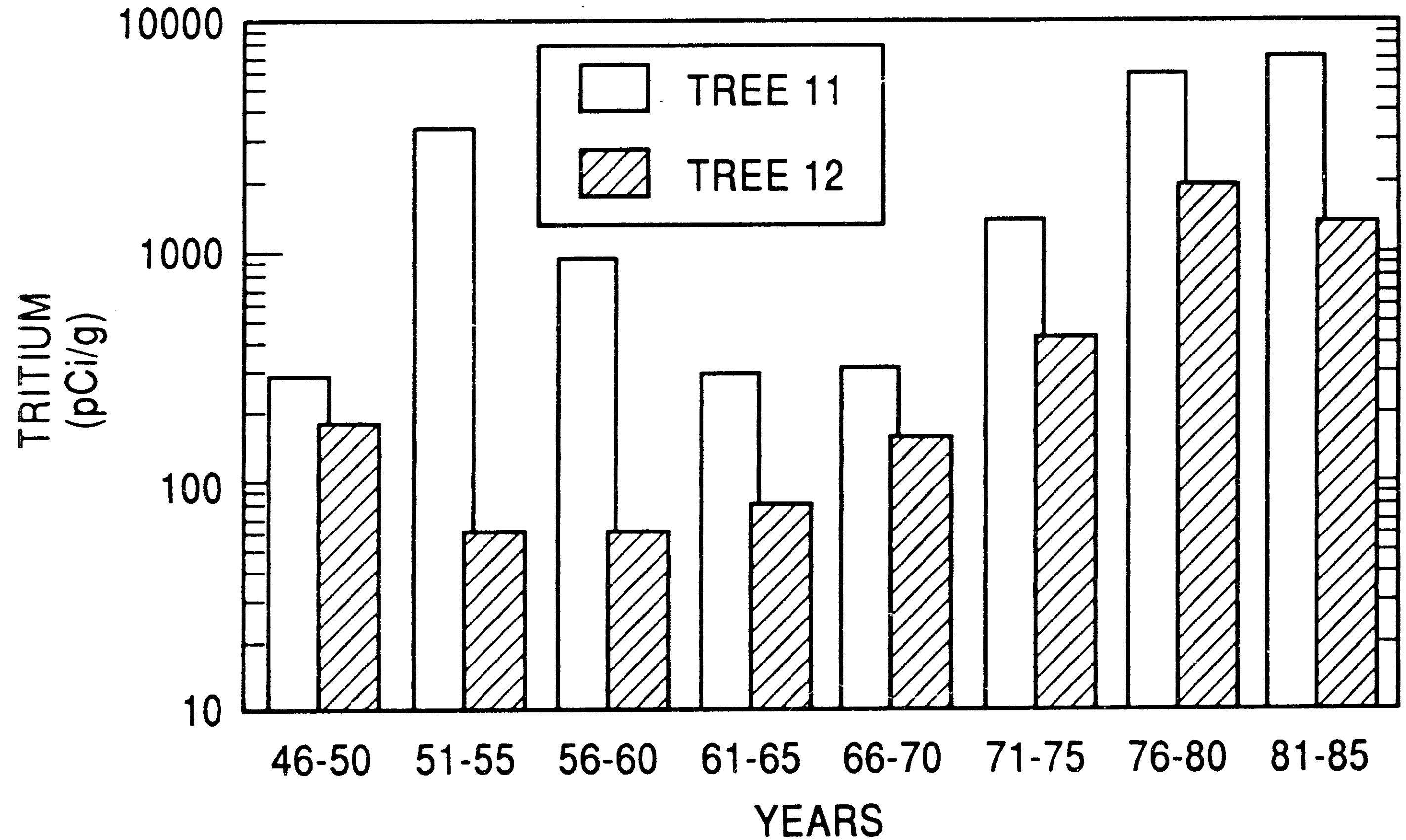

Fig. 7.3. Mean concentrations of tritium in pine cores collected on the south perimeter of solid radioactive waste disposal/storage area 5 for 5-year periods from 1946 to 1985.

7.2 ANALYSES OF VERTEBRATES FOR EXPOSURE TO RADIONUCLIDES, HAZARDOUS ORGANICS, AND MERCURY

7.2.1 Introduction

WOL has been contaminated with low-level radioactive waste since 1943 (Oakes et al. 1982). Numerous studies have been conducted in the area since 1955 to assess the levels of radionuclides in resident small-mammal populations and to evaluate the effects of environmental radiation (Auerbach 1958, 1959, 1961; Dunaway and Kaye 1961; Kaye and Dunaway 1963); however, very little monitoring of radionuclide levels in WOL mammals has been done since the early 1960s. Because additional releases of contaminants may have occurred, a reexamination of animals in these areas is being conducted as part of Subtasks 5a and 5b of BMAP on radionuclides and organics, respectively, in the terrestrial environment (Loar et al. 1991). The purpose of these studies is to (1) provide useful information on exposures of animals inhabiting contaminated sites, using new and sensitive techniques developed since the studies of 20 to 30 years ago; (2) determine contaminant body burdens; and (3) document the quantities of contaminants in mammal populations.

7.2.2 Description of Study Sites

WOL acts as a settling basin for WOC, which receives chemicals and radionuclides from ORNL operations. SWSA 6 is located to the north of the lake. Leachates from other SWSAs within the watershed may enter the lake via WOC and adjacent uplands. Strontium-90, ^{137}Cs, and ^{60}Co are the predominant radionuclides in the lake (Oakes et al. 1982). The specific site used in this study was located at the northwest end of WOL and southwest of SWSA 6. This area was selected for its dense shrub-grassland vegetation, which is flanked on both sides by a forest stand, thus providing ideal habitat for many species of small mammals. Preliminary trapping confirmed selection of the site on this basis.

A section of EFPC floodplain located in the City of Oak Ridge was selected as a primary reference site positive for BaP and mercury and negative for radionuclides based on previous sediment analyses (Hoffman et al. 1984). The creek originates at the Oak Ridge Y-12 Plant site, flows through New Hope Pond and the City of Oak Ridge, and enters Poplar Creek east of ORGDP. Near EFPC kilometer 18.2, the floodplain is low and the creek periodically overflows, depositing sediment. The floodplain at this point contains abundant vegetation including sneezeweed, jewelweed, and grass. A box elder grove and an old field border the area.

Another selected field site was the 3524 Equalization Basin, which is used as a process waste disposal basin at ORNL. The unlined pond, which is located in the 3500 area of ORNL, is 85 by 24 m and has a volume of ~3800 m^3 (one million gallons). Of the 5.6 TBq (150 Ci) of nuclides estimated to be in the basin sediments, ^{137}Cs makes up 68%, and ^{90}Sr contributes 19% (Braunstein et al. 1984). Groundhogs were common in the area at the time trapping was initiated.

Additional mammal trapping sites included the vicinity of the ORNL Steam Plant, along the banks of WOC, a grass/shrub area near the Bull Run Steam Plant, and along the banks of Brushy Fork, a tributary of Poplar Creek located north of Oak Ridge (Fig. 2.1). The latter two areas were included as additional reference sites.

7.2.3 Methods

Several types of traps and baits were used to capture mammals. Small animals were trapped by using 22.5 × 7.5 × 7.5 cm Sherman live traps, typically placed in grassy areas and along logs and baited with sunflower seeds. Pitfall traps baited with anchovies were used at some sites to capture shrews. Larger mammals were trapped with 65 × 22.5 × 22.6 cm Tomahawk live traps placed along stream and lake banks. Apples served as bait for groundhogs and muskrats.

Trapping was carried out in the summer and early fall of 1986. Animals were transported to the laboratory each day for weighing, species identification, and blood collection. Animals were anesthetized with Metafane® (2,2-dichloro-1,1-difluoroethyl methyl ether) for blood collection and either sacrificed in the laboratory by exsanguination or returned to the field for release after recovery from the anesthesia. Blood samples were taken by heart puncture for small mammals and a snake and by puncture of the caudal artery for larger mammals, such as muskrats. Corpses were frozen and kept for gamma radiation counting and organ analyses.

Because of the small size of the animals examined in this study, kidney samples (0.5–1.0 g) from several individuals of the same species at each site (EFPC, WOL, and Bull Run Steam Plant) were combined for mercury analysis. Analyses were done by ACD at ORNL, through the use of atomic absorption spectroscopy.

Body burdens of gamma radiation were determined by using a gamma ray spectrometer (Nuclear Data, Inc., 6620 microprocessor) coupled to an intrinsic germanium detector having a relative efficiency of 25% and a resolution of 1.8 keV for the ^{60}Co 1332 photon. Dead small animals and tissue samples of larger animals were fitted into vials of predetermined geometry and counted for ~22 h each.

Blood samples were analyzed for BaP metabolites by the method of Shugart (1985). Diol-epoxide metabolites of BaP covalently bound to hemoglobin were released by acid hydrolysis. Resulting tetrols were separated by high-performance liquid chromatography (Perkin-Elmer Series IV Liquid Chromatograph) and detected by fluorescence spectrometry (Perkin-Elmer LS-4). Separations were achieved by a reversed phase technique with the use of a Vydac column and methanol:water (45:55) as the eluent. Two metabolites of BaP—benzo(a)pyrene,r-7t-8,9,c-10-tetrahydrotetrol and benzo(a)pyrene,r-7,t-8,c-9,10-tetrahydrotetrol, referred to as tetrols I-1 and II-2, respectively—served as indicators of BaP exposure in the animals.

7.2.4 Results

A variety of mammal species were collected at the six sampling sites (Table 7.1). The white-footed mouse (*Peromyscus leucopus*) was the most abundant species at both EFPC and WOL. The shorttail shrew (*Blarina brevicauda*) was the second most common small mammal. Muskrats were present along steep, sunny banks where pools were present in the streambed; no muskrats were captured at WOL. Only groundhogs (*Marmota monax*) were captured at the 3524 Equalization Basin; no mammals were captured at WOC. Several nonmammalian species were also captured at EFPC, including the black rat snake (*Elaphne obsoleta obsoleta*) and the snapping turtle (*Chelydra serpentina*).

Table 7.1. Animal species collected near Oak Ridge National Laboratory and reference sites, 1985–86

Site/species	No. collected
East Fork Poplar Creek	
White-footed mouse (*Peromyscus leucopus*)	27
House mouse (*Mus musculus*)	1
Shorttail shrew (*Blarina brevicauda*)	2
Muskrat (*Ondatra zibethica*)	9
Cotton rat (*Sigmodon hispidus*)	1
Norway rat (*Rattus norvegicus*)	1
Black rat snake (*Elaphne obsoleta obsoleta*)	1
Snapping turtle (*Chelydra serpentina*)	1
White Oak Lake	
White-footed mouse (*Peromyscus leucopus*)	14
House mouse (*Mus musculus*)	1
Cotton rat (*Sigmodon hispidus*)	3
Shorttail shrew (*Blarina brevicauda*)	7
Prairie vole (*Microtus ochrogaster*)	1
ORNL Steam Plant	
Eastern cottontail rabbit (*Sylvilagus floridanus*)	1
Bull Run Steam Plant	
Pine vole (*Pitymys pinetorum*)	4
Shorttail shrew (*Blarina brevicauda*)	3
Cotton rat (*Sigmodon hispidus*)	1
Brushy Fork	
Groundhog (*Marmota monax*)	1
Opossum (*Didephis marsupialis*)	1
3524 Equalization Basin	
Groundhog (*Marmota monax*)	4

7.2.4.1 Mercury

Analyses of pooled kidney samples from captured animals (Table 7.2) showed that mercury concentrations in the shorttail shrew at EFPC were almost two orders of magnitude higher than concentrations in shrews at WOL or Bull Run Steam Plant, 21 µg/g wet wt compared with 0.55 and 0.08 µg/g wet wt respectively. Cotton rats and white-footed mice collected at EFPC also had elevated concentrations of mercury compared with those of species collected at the other two sites.

Mercury levels in animals from WOL were intermediate between the concentrations found at EFPC, which is known to contain elevated mercury concentrations (e.g., TVA 1985b, 1985c) and those observed at Bull Run Steam Plant, for which data on mercury residues in soils are not available. Additional residue analyses will be conducted to determine the significance of mercury contamination in the WOL area.

Table 7.2. Concentrations of mercury in pooled kidney tissue of small mammals collected at East Fork Poplar Creek, White Oak Lake, and Bull Run Steam Plant, 1985–86

Site/species	Concentration of mercury (µg/g wet wt)
East Fork Poplar Creek	
White-footed mouse	0.78
Shorttail shrew	21
Cotton rat	1.9
White Oak Lake	
White-footed mouse	0.18
Shorttail shrew	0.55
Cotton rat	0.04
Bull Run Steam Plant	
Shorttail shrew	0.08
Cotton rat	0.05
Pine vole	0.07

7.2.4.2 Gamma radiation

Gamma radiation counting of small mammals and groundhogs is incomplete at this time. However, preliminary results indicate that, although ^{60}Co and ^{106}Ru have been reported in mammals from previous studies on WOL (Dunaway and Kaye 1961, Kaye and Dunaway 1963), these radionuclides were not detected in the present study. These differences may be due, in part, to their relatively short half-lives (^{60}Co = 5 years, ^{106}Ru = 382 d) and to the fact that animals counted in previous WOL studies were collected from the drained lake bed itself and may have been exposed to higher concentrations of radionuclides than animals from the adjacent upland. Cesium-137 (half-life = 30 years) was detectable in some, but not all, of the white-footed mice counted.

Both ^{137}Cs and ^{60}Co were detected in groundhogs collected from the 3524 Equalization Basin. Cesium-137 ranged from 3.7 to 4.2×10^3 Bq/kg dry wt in muscle and from 1.5 to 1.6×10^3 Bq/kg dry wt in bone. Cobalt-60 ranged from 15.5 to 26.2 Bq/kg dry wt in muscle and from <0.37 to 17.4 Bq/kg dry wt in bone. No ^{60}Co and only trace quantities of ^{137}Cs were detected in control groundhogs.

7.2.4.3 Blood analyses for benzo[a]pyrene

Results of blood analyses showed site and species differences in exposure to BaP. Only mammals collected at EFPC had detectable amounts of the BaP tetrol, indicating BaP exposure (Table 7.3). Although several species from EFPC had detectable concentrations of BaP tetrols, none of the white-footed mice were positive (Table 7.3). These data suggest that white-footed mice, which are granivores and live aboveground, may not be suitable for determining the bioavailability of polycyclic aromatic hydrocarbons (PAHs) in soil. The fact that BaP tetrols were found in shrews, muskrats, and a snapping turtle, species that have a great deal of soil and/or sediment contact, indicates that these species (or others with similar habits) are preferred for biological monitoring of PAHs.

7.2.5 Conclusions

Several common mammalian species with different food preferences and habitats occur near ORNL and can be used for contaminant monitoring of the terrestrial environment. Preliminary evidence indicates that white-footed mice, which are more abundant than shrews, are likely to be the most useful species for monitoring those ORNL sites where radionuclides are available to terrestrial biota; however, this species is not well suited as a biological monitor for PAHs. Groundhogs at the 3524 Equalization Basin probably represent the highest radiation exposure of wild mammals at ORNL because of the high radionuclide concentrations at the site. They are excellent candidates for evaluating the use of biochemical indicators of genotoxic exposure for the routine terrestrial monitoring program. Small mammals collected near seeps in SWSA 4 and SWSA 5 will also be analyzed.

Analyses of kidney tissues from small mammals showed that mercury concentrations are elevated in mammals from WOL (relative to those from Bull Run Steam Plant) but are much lower than those in animals from EFPC floodplain. Additional analyses are being conducted to determine the relative importance of mercury as a contaminant to terrestrial species in the WOL area. Preliminary results showing an absence of BaP metabolites in blood of small

Table 7.3. Concentrations of benzo[a]pyrene (BaP) tetrol in the hemoglobin (Hb) of animals from the Oak Ridge National Laboratory (ORNL) environs, East Fork Poplar Creek floodplain, and reference sites

Site/species	Sample size	BaP tetrol (pg/mg Hb)[a]
East Fork Poplar Creek		
Muskrat A	1	1.28
B	1	0.85
C	1	0.34
D-I	6	ND
Shorttail shrew A	1	0.42
B	1	0.27
Snapping turtle	1	0.16
Norway rat	1	0.15
Cotton rat	1	ND
Black rat snake	1	ND
House mouse	1	ND
White-footed mouse	27	ND
White Oak Lake		
Shorttail shrew	7	ND
Cotton rat	3	ND
Prairie vole	1	ND
House mouse	1	ND
White-footed mouse	14	ND
3524 Equalization Basin (ORNL)		
Groundhog	4	ND
ORNL Steam Plant		
Eastern cottontail rabbit	1	ND
Bull Run Steam Plant		
Shorttail shrew	3	ND
Cotton rat	1	ND
Pine vole	4	ND
Brushy Fork		
Groundhog	1	ND
Opossum	1	ND

[a]ND = not detectable; limit of detection is 10 pg. Reported values were normalized to 1 mg of Hb.

mammals indicate that PAHs are not likely to be of concern as contaminants of the terrestrial environment at ORNL.

7.3 RADIATION MONITORING AND BIOCHEMICAL ANALYSES OF FRESHWATER TURTLES

7.3.1 Introduction

Of the vertebrates that inhabit both aquatic and terrestrial habitats, freshwater turtles have a close contact with sediments and are well suited as monitors of the biological availability of radionuclides from a contaminated water body. Freshwater turtles have proved to be good sentinel species for monitoring contaminants, such as PCBs (Helwig and Hora 1983, Olafsson et al. 1983), organochlorine pesticides (Pearson et al. 1973, Owen and Wells 1976) and heavy metals, such as mercury and cadmium (Helwig and Hora 1983).

Many of the attributes of turtles that made them valuable bioindicators of contamination in these studies apply to sites containing radionuclides as well. Because most turtles are omnivorous, radionuclides in both plants and animals can contribute to the body burden. Furthermore, because most turtle species are relatively long-lived, a lengthy exposure to contaminants may occur. In addition, adult turtles are of interest in critical pathway analyses for risk assessments because they are consumed by humans. Similarly, turtle eggs and young are often ingested in the wild by predators, such as skunks and raccoons, and can be evaluated as a potential source of contamination to terrestrial wildlife. Turtles are being used as sentinel species for the detection of contaminants at the DOE Savannah River Ecology Laboratory (SREL) (J. W. Gibbons, Savannah River Ecology Laboratory, personal communication, 1986). The present study was undertaken to (1) determine whether turtles can be useful in monitoring the bioavailability of radionuclides in aquatic sediments and (2) evaluate the utility and appropriateness of measuring DNA damage to determine exposure of organisms to genotoxic agents (such as radionuclides). Turtles were trapped from WOL (see Sect. 7.2.2) and from Bearden Creek embayment on Melton Hill Reservoir, a noncontaminated reference site east of ORNL (Fig. 2.1).

7.3.2 Methods

Although some preliminary trapping was conducted in 1985, most of the turtles were collected from June to September 1986. Animals were captured by using hoop nets, 0.625-cm wire mesh funnel traps, 2.5-cm wire mesh box traps (Tomahawk Company, Tomahawk, Wisconsin), and trotlines. Traps baited with fish were placed near the lakeshore and checked daily during trapping periods.

Turtles were measured, weighed, and aged. They were subsequently either (1) marked and returned to the site or (2) sacrificed for dissection, liver biopsy, and gut content analysis and then frozen for later analyses of radionuclides. Dissected animals were separated into bone, muscle, plastron, carapace, gastrointestinal tract, and various internal organs. Whole small turtles and the tissues from sacrificed turtles were counted for gamma radiation as described in Sect. 7.2.3. All soft samples were fitted into containers of predetermined geometry and counted until an error term of 10% or less was obtained. To measure ^{90}Sr in turtles, samples were ashed and acid digested for Cerenkov counting (Lauchli 1969, Larsen 1981) in a liquid scintillation counter (Packard Tri-Carb 4640).

To estimate abundance, turtles were marked prior to release by using a procedure developed at SREL, in which the marginal scutes on the carapace are designated with a letter and marked on individual animals by drilling holes (J. W. Gibbons, Savannah River Ecology Laboratory, personal communication, 1986).

7.3.3 Results

A total of 53 turtles of 6 species were trapped, indicating that WOL has a highly diverse turtle community. The yellow-bellied slider (*Pseudemys scripta*) was the most abundant species, although stinkpots (*Sternotherus odoratus*), painted turtles (*Chrysemys picta*), common snapping turtles (*Chelydra serpentina*), Eastern spiny soft-shell turtles (*Trionyx spiniferus spiniferus*), and map turtles (*Graptemys geographica*) were also present (Table 7.4). Turtles collected from Bearden Creek embayment (reference site) included the yellow-bellied slider, which was the most abundant species, the stinkpot, and the painted turtle.

Table 7.4. Capture frequency of White Oak Lake turtles, 1985–86

Species	Disposition				
	Sacrificed	Marked	Recaptured	Released (not marked)	Total
Snapping turtle (*Chelydra serpentina*)	3	—	—	2	5
Soft-shelled turtle (*Trionyx spiniferus spiniferus*)	4	—	—	1	5
Stinkpot (*Sternotherus odoratus*)	3	—	—	1	4
Yellow-bellied slider (*Pseudemys scripta*)	10	15	4	—	29
Painted turtle (*Chrysemys picta*)	1	8	—	—	9
Map turtle (*Graptemys geographica*)	1	—	—	—	1

7.3.3.1 Radionuclide analyses

Analyses of dissected organs and tissues for gamma radiation are in progress. Preliminary data from these analyses show ^{137}Cs throughout the body, with the majority found in muscle (230 to 240 Bq/kg wet wt). Cobalt-60 accumulated primarily in the liver (37 to 41 Bq/kg wet wt) and kidneys (36 to 56 Bq/kg wet wt). Selenium-75 was found at the highest concentrations in the unlaid eggs (premature eggs with yolks but no albumin or shells), ranging in value from 30 to 44 Bq/kg.

Because the majority of ^{90}Sr in turtles is likely to be found in shells, the carapaces and plastrons of four yellow-bellied sliders collected from WOL during July 1985 were acid digested, ashed, and counted to determine radioactivity. Preliminary analyses show concentrations of ^{90}Sr in the carapace ranging from 4.4×10^3 to 10.1×10^3 Bq/kg wet wt; concentrations in the plastron ranged from 6.8×10^3 to 12.6×10^3 Bq/kg wet wt. These data are not yet corrected for possible interference by ^{137}Cs and ^{60}Co (Larsen 1981) and thus must be regarded as preliminary.

7.3.3.2 Gut content analyses

The gastrointestinal contents of six turtles from WOL were examined to assess the influence of food preference on radionuclide concentrations found in various species. Eastern spiny soft-shell turtles ingested primarily fish and crustaceans (88 to 98% of diet), map turtles appear to feed exclusively on snails, whereas snapping turtles appear to be omnivorous, based on the mixture of detritus, sediment, and plant and animal matter that was recovered from the gastrointestinal tracts. The differences in food habits can be useful in explaining species differences in body burdens of radionuclides from turtles in WOL.

7.3.4 Conclusions

Those contaminants that predominate in WOL sediment and water (Cerling and Spalding 1982) were also found in turtles (^{90}Sr, ^{137}Cs, ^{60}Co). In addition, ^{75}Se was present. Considerable species differences were found in the body burdens of specific radionuclides in turtles. Preliminary data indicate that variability may be a function of differences in food preferences and sediment contact among species. Additional data are being collected to establish such relationships. Data on radionuclide concentrations in turtles to be obtained in this study will also be used in the screening analysis described in Sect. 8.1 to determine potential exposure pathways to humans who ingest these animals.

7.4 FUTURE STUDIES

Results obtained in this first phase of monitoring in the terrestrial environment will be useful for delineating the scope and focus of future monitoring and remediation activities. Although many species are available as biological indicators of contaminant exposure, results to date show that species selection is an important consideration in designing a monitoring program. Appropriate selections depend not only on the food preferences and habitats of the animals but also on the type of contaminant under study.

Preliminary studies indicate that PAHs in terrestrial species at ORNL are unlikely to exceed background levels, while radionuclides are elevated in several wildlife species consumed by humans. Further analyses are planned to confirm these preliminary findings. Analyses of radionuclides in small mammals will focus on identifying areas potentially contributing to the uptake of radionuclides by deer, which emerged as a problem during 1986. Additional sampling will be conducted to evaluate mercury as a contaminant of terrestrial biota at WOL.

Fat analysis of wild animals for PCBs was not conducted as planned (Subtask 5b of BMAP), due to the lack of sufficient fatty tissue in the animals. Unlike their counterparts in the laboratory, wild mice and rats had negligible amounts of fat. Other tissues (e.g., brain) are under consideration for PCB analysis, but these studies will not be initiated until the radionuclide analyses have been completed.

The groundwork has been laid during this past year for achieving the goal of determining the utility of new biochemical techniques for detecting exposure of wild animals to environmental contaminants. Studies during the next year will determine whether these state-of-the-art assays can replace many presently used, time-consuming, and inconclusive approaches to wildlife toxicology.

Because SWSA 4 and SWSA 5 were found to be major contributors of ^3H to WOC watershed and because ^3H concentrations in tree cores in the vicinity were found to increase during the last decade, additional studies of this radionuclide are planned. Strong seasonal changes in the ^3H concentration in soil water were identified in floodplain soils below SWSA 5. Future studies will seek to quantify ^3H fluxes in this area.

8. RADIOECOLOGY OF WHITE OAK LAKE

8.1 PRELIMINARY SCREENING OF RADIONUCLIDES IN WHITE OAK LAKE

8.1.1 Introduction

The purpose of conducting a preliminary screening analysis on a contaminated environment is to identify contaminants and areas of concern from the standpoint of protecting human health. Equally important is the identification of contaminants that are present at such low levels that they should not be of concern. Such analyses are important because the results can be used in decision making and to identify areas where additional research and data collection are needed as well as those areas of less concern.

The approach taken in this screening analysis involves the use of conservatively biased calculational procedures (i.e., procedures that are not likely to underestimate the actual exposure of humans to a given radionuclide). If calculated exposures do not approach or exceed specified limits or established health standards, individual radionuclides and food-chain pathways can be screened out and thus given a low priority for future consideration. Individual radionuclides that are calculated to approach or exceed established limits are designated as potential pollutants that warrant further investigation.

It must be emphasized that the values estimated by screening techniques are strictly for comparison with environmental standards (limiting values) and are not intended to represent actual doses to humans. Because of the nature of the assumptions and methods incorporated in these techniques, the actual doses, in most situations, will be significantly less than the values calculated for screening.

In this study, a screening analysis was conducted for radionuclides in WOL as part of Subtask 6a of BMAP (Loar et al. 1991). A screening analysis for metals and organic pollutants in WOL will be reported in the next annual report. The current screening study was conducted for exposure pathways leading to humans for three possible scenarios that could result from remedial actions that are being considered for WOL.

8.1.2 Methods

WOL has received radioactive effluents from ORNL operations since 1943. Metals and organic pollutants have also been released into the lake, although monitoring programs and environmental data collections have emphasized radioactive pollutants. Some information has been obtained recently on concentrations of heavy metals and organic pollutants in the lake (e.g., TVA 1985a, 1985b, 1985c). Data available from monitoring programs and environmental studies are reviewed and summarized in Sherwood and Loar (1987). The most recent data on radionuclides, if applicable, were used in the present analysis.

8.1.2.1 Scenarios

The potential human exposure pathways from proposed remedial actions for WOL were considered for the following three scenarios.

Scenario 1. The lake is maintained at its current status, and the public is allowed free access to it and the surrounding environs. The potential for human exposure includes the aquatic and terrestrial food-chain pathways and external exposure from the floodplain, sediment, and water. The various exposure pathways are shown in Fig. 8.1, and the components considered in these exposure pathways are listed in Table 8.1.

Scenario 2. The lake is completely drained, and the public is allowed access. Potential exposure to humans in this scenario is the terrestrial food-chain pathway and external exposure from the floodplain and drained lake bed. It is assumed that WOC is routed around or through the lake bed but that the stream does not support adequate biota for human consumption.

Scenario 3. The lake is drained and capped or excavated, with a canal constructed around it to divert the flow of WOC into the Clinch River. The radioactivity in the WOL environment is isolated from the public by removing or entombing the contaminated material. The public is allowed free access but is no longer exposed to any critical pathways. External exposure from residual material would depend upon the level to which the radioactive material is either cleaned up or isolated. Although radiation exposure to humans would occur during the process of isolating the contaminated material, evaluation of the exposure during this period is beyond the scope of this screening evaluation.

8.1.2.2 Description of exposure pathways

Aquatic. The aquatic food-chain pathways include consumption of water, fish, turtles, clams, crayfish, and aquatic plants. The species of fish consumed are bluegill (*Lepomis macrochirus*), carp (*Cyprinus carpio*), and largemouth bass (*Micropterus salmoides*). Aquatic plants, namely *Elodea* and *Potomageton* spp. as well as other macrophytes, are considered potential exposure pathways; however, the use of aquatic plants from WOL would probably be very limited. The annual consumption rates for aquatic and terrestrial foods used in this analysis are listed in Table 8.2. As conditions exist today, most of the aquatic food-chain pathways, with the exception of fish and turtles, would be of little significance because of the limited quantity of biota available for consumption by humans. However, for screening purposes the aquatic foods listed in Table 8.1 were used in the analysis.

Terrestrial. The terrestrial food-chain pathways considered for screening purposes are illustrated in Fig. 8.1. They include the consumption of agricultural crops grown on either the drained lake bed or the floodplain, inadvertent consumption of bottom sediments or floodplain soil, meat from large and small game mammals (e.g., deer and rabbits), milk from either cows or goats grazing on the drained lake bed, beef, poultry, eggs, honey, and waterfowl.

External Exposure. External exposure pathways include swimming, boating, sunbathing, handling of fishing gear, and working over the contaminated sediments on the floodplain and lakeshores in tasks such as gardening. For this preliminary screening exercise, default values for occupancy rates were taken from the International Atomic Energy Agency (IAEA) Technical Report 57 (IAEA 1982) and are listed in Table 8.3.

Table 8.1. Food-chain and external exposure pathways corresponding to each of the three remedial action scenarios

Pathway	Pathway component		
	Scenario 1	Scenario 2	Scenario 3
Aquatic	Fish Turtles Invertebrates Crayfish Aquatic plants Waterfowl Water	No aquatic components	No aquatic components
Terrestrial	Rabbits Deer Cows Chickens Vegetables Honey Sediments	Rabbits Deer Cows Chickens Vegetables Honey Sediments (soil)	No terrestrial components
External	Working over sediments Boating Swimming Sunbathing over sediments Handling fishing gear	Working over sediments (soil) Sunbathing over sediments (soil)	No external pathways

8.1.2.3 Methods for dose calculations

Data Base. Data used for the screening analysis are based on the most recent WOL data available; if no data were available on WOL, then literature values and mathematical models were used to estimate the radionuclide concentrations in food-chain components. Concentrations for the aquatic food-chain pathway are listed in Table 8.4. In several cases, the measured concentration in biota was based on samples collected in the early or mid-1970s. These values were used for screening instead of using calculated concentrations because the quantity of radionuclides, except for tritium (Ohnesorge 1986), released into the lake has not changed substantially during the last 15 years and because pr edicted values tend to significantly overestimate measured values.

The average measured value for a radionuclide concentration in a food-chain component was used when a data set contained an adequate number of samples. For example, the highest average concentrations of ^{137}Cs and ^{60}Co in muscle tissue of the three fish species listed in Table 8.4 were used to calculate doses to humans. The average concentration of ^{90}Sr in eviscerated carcasses of bluegill, instead of the concentration in muscle tissue, was used to calculate doses to humans because these values would conservatively represent a food-chain

Table 8.2. Annual consumption rates of aquatic and terrestrial foods

Pathway component	Consumption rate (kg/year)
Aquatic plants	20[a]
Fish	7.3[b]
Turtles	7.3
Clams	1.8[a]
Crayfish	1.8[a]
Beef	100[c]
Venison	75[a]
Small game mammals	25[a]
Honey	1.8[a]
Water	730[c]
Milk (cow and goat)	300[c]
Vegetables (fresh)	200[c]
Sediment	0.365[b]
Poultry	9.5[a]
Waterfowl	9.5[a]
Eggs	15.0[a]

[a]E. M. Rupp, F. L. Miller, and C. F. Baes III, "Some Results of Recent Surveys of Fish and Shellfish Consumption by Age and Region of U.S. Residents," *Health Phys.* **39**, 165–175 (1980).

[b]Martin Marietta Energy Systems, Inc., *Environmental Monitoring Report, U.S. Department of Energy Oak Ridge Facilities, Calendar Year 1984*, ORNL-6209, Oak Ridge National Laboratory, Oak Ridge, Tenn., 1985.

[c]*Screening Techniques for Determining Compliance with Environmental Standards: Releases of Radionuclides to the Atmosphere*, NCRP Commentary No. 3, National Council on Radiation Protection and Measurements, Bethesda, Md., 1986.

Table 8.3. Default values for occupancy rates (U_p)

Pathway	Occupancy rate (h/year)
Working over contaminated sediments	2000
Sunbathing	1000
Handling fishing gear	2000
Swimming	300
Boating	2000

Source: *Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases*, Safety Series No. 57, International Atomic Energy Agency, Vienna, 1982.

Table 8.4. Concentrations of radionuclides (in becquerels per kilogram wet weight, unless otherwise noted) in components of the aquatic food-chain pathway in White Oak Lake[a]

Component	^{137}Cs	^{60}Co	^{90}Sr	^{3}H
Fish				
Carp	260	6.3	19	12,000
Largemouth bass	590	8.5	8.5	12,000
Bluegill	360	9.6	890[b]	12,000
Turtles	630	480	560	12,000
Crayfish	330	52	1,100	12,000
Clams	160	470	1,100	11,000
Aquatic plants				
Elodea	2,200	13,000	5,600	11,000
Potomageton	960	7,400	5,600	11,000
Macrophytes	590	560	5,600	11,000
Water, Bq/L	1.6	2.3	11	13,000
White Oak Dam			2,500	93,000
Melton Branch				

[a]Underlining indicates model-derived values.
[b]Measured concentration in eviscerated carcass.

pathway that included the consumption of fish bones as well as muscle tissue. When the number of samples composing the data set was limited, the maximum value was used.

When WOL data were not available, literature values and mathematical models (IAEA 1982, Ng et al. 1982, NCRP 1986) were used to calculate radionuclide concentrations in food-chain components. For Scenario 2, in which measurements were not available for the terrestrial food-chain pathway, the concentration in the following food-chain components was calculated: vegetables grown on the floodplain or drained lake bed, milk from cows and goats consuming vegetation grown on the lake bed and drinking water from Melton Branch, beef, poultry, and eggs (Table 8.5). These calculations were based on conservative default values for concentration ratios (curies per kilogram of dry or fresh weight of vegetation:microcuries per kilogram of dry weight of soil) and transfer coefficients (i.e., the fraction of radionuclide ingested daily by an herbivore that can be measured per unit mass of animal product or that is secreted in milk when at steady-state conditions). Although the calculated values may deviate from measured values, they are expected to result in an overestimation of the actual concentration. The concentration of tritium (^3H) in Melton Branch, a tributary of WOC, is over an order of magnitude higher than in WOL (Martin Marietta Energy Systems 1986a). If WOL is drained, less dilution of ^3H would occur; therefore, for conservative purposes, the concentration of ^3H in Melton Branch was used to calculate doses to man from drinking water and from ingestion of milk, beef, and other foods. Deer were considered to be free ranging and would not spend all of their time on the WOL bed. Table 8.6 provides references for the data for specific pathway components and indicates the components for which values were derived by mathematical model calculations.

Table 8.5. Concentrations of radionuclides in components of the terrestrial food-chain pathway (in becquerels per kilogram wet weight, unless otherwise noted)[a]

Component	^{137}Cs	^{60}Co	^{90}Sr	^3H
Rabbits	1,900	1,500	--	--
Deer (venison)[b]	<19	<3.0	4.8	<u>46,000</u>[c]
Beef	<u>5,600</u>	<u>1,700</u>	<u>210</u>	<u>46,000</u>[c]
Milk[d]	<u>2,000</u>	<u>152</u>	<u>280</u>	<u>46,000</u>[c]
Poultry	<u>8,300</u>	<u>925</u>	<u>44</u>	--
Eggs	<u>1,500</u>	<u>200</u>	<u>590</u>	--
Vegetables	<u>2,200</u>	<u>190</u>	<u>1,300</u>	--
Honey	4.1	0.7	--	--
Sediment, dry wt	110,000	11,000	4300	<u>600</u>
Waterfowl[e]	<u>8,900</u>	<u>300</u>	--	--

[a]Underlining indicates model-derived values.

[b]*Environmental Surveillance of the U.S. Department of Energy Oak Ridge Reservation and Surrounding Environs During 1986*, ES/ESH-1/V1 and V2, Martin Marietta Energy Systems, Inc., Oak Ridge, Tenn., 1987.

[c]Value is based on model using the concentration of ^3H in water from Melton Branch.

[d]Includes goat milk.

[e]D. K. Halford, J. B. Millard, and O. D. Markham, "Radionuclide Concentration in Waterfowl Using a Liquid Radioactive Waste Disposal Area and Potential Radiation Dose to Man," *Health Phys.* **40**, 173–181 (1981).

Table 8.6. Sources or methods used to obtain radionuclide concentrations in food-chain components

Pathway component	Data sources: reference[a] or model[b]
Fish	Sherwood and Loar (1987)
Aquatic plants	B. G. Blaylock (unpublished data, 1976)
Turtles	Model
Crayfish	B. G. Blaylock (unpublished data, 1976)
Clams	Model
Rabbits	Tsakeres (1982)
Waterfowl[c]	Halford et al. (1981)
Honey	P. S. Rohwer (ORNL, personal communication, 1985)
Deer	Martin Marietta Energy Systems (1985, 1987); model[d]
Water	Martin Marietta Energy Systems (1986a)
Sediments	Sherwood and Loar (1987)
Vegetables	Model
Milk	Model
Poultry	Model
Eggs	Model
Beef	Model

[a]Complete citations to published references are included in Sect. 11 of this report.

[b]*Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases,* Safety Series No. 57, International Atomic Energy Agency, Vienna, 1982.

[c]Data from radioactive leaching ponds at the Idaho test reactor area.

[d]Model was used to calculate the concentration in flesh of deer drinking water in Melton Branch.

Screening Limits. Two screening limits were used in this analysis. The Uranium Fuel Cycle Standard promulgated by EPA is 25 mrem/year (0.25 mSv/year) (EPA 1977), and the current standard for ORNL is 100 mrem/year (1.0 mSv/year).* The calculated doses for the different radionuclides and the various food-chain pathways were screened against the EPA 0.25-mSv/year and the DOE 1.0-mSv/year standards. The ratio of the calculated dose to the screening limits for each radionuclide and for each food-chain pathway indicates the contribution of that particular pathway and radionuclide to the total dose commitment. If the ratio is much less than 1, then the radionuclide via that particular pathway can be given a low priority for further consideration. When the ratio approaches or exceeds 1, that radionuclide pathway is designated as one that warrants further investigation.

Dose Calculation for Exposure to Food-Chain Pathways. The 50-year dose equivalent to man was calculated for aquatic and terrestrial food-chain pathways based on average concentrations of four radionuclides (^{137}Cs, ^{60}Co, ^{3}H, and ^{90}Sr) that are found at relatively high concentrations in WOL and its environs. Dose to man from ingesting a pathway component (food, etc.) contaminated with a specific radionuclide or radionuclides is calculated as the product of three terms (IAEA 1982):

$$D_{ic} = C_{ic} \times AI_c \times CF_i$$

where

D_{ic} = the 50-year dose equivalent to man of nuclide i in component C (millisieverts per year),
C_{ic} = the concentration of nuclide i in the edible portion of component C (becquerels per kilogram per year or becquerels per liter per year fresh weight),
AI_c = the average annual rate of consumption of component C (kilograms per year or liters per year),
CF_i = the 50-year dose conversion factor for nuclide i (millisieverts per becquerel).

Dose Calculation for External Exposure to Sediment or Water. For screening purposes, estimates of external exposure were calculated by using a simple model that embodies realistic assumptions but that leads to higher than average estimates of exposure (IAEA 1982). The dose rate due to external radiation from either sediment or water was calculated by the following equation:

$$H_i = D_{ip} \times U_p$$

where

D_{ip} = the whole-body dose rate for gamma emitters or the skin dose rate for beta emitters,
U_p = the occupancy rate for the external exposure pathway.

*The symbol *mSv* denotes millisievert: 1 sievert = 100 rems.

8.1.3 Results and Discussion

The estimated 50-year committed doses for the different food-chain pathways for the ingestion of contaminated food via the aquatic and terrestrial food-chain pathways are given in Table 8.7. Total doses for individual pathways as well as dose contribution from individual radionuclides are listed in Table 8.7. For most food-chain pathways, ^{137}Cs is the major dose contributor (57%), followed by ^{90}Sr (38%), ^{60}Co (3.3%), and ^{3}H (1.6%). Food-chain pathways for which ^{90}Sr is the major dose-contributing radionuclide include aquatic (fish, turtles, and aquatic plants) and terrestrial (eggs and vegetables) components. Except for fish, concentrations of ^{90}Sr in components of these food chains were derived from models and, therefore, are probably overestimations of the actual values (see Tables 8.4 and 8.5 for derived values).

Table 8.7. Fifty-year committed dose to man [in millisieverts per year (1 sievert = 100 rems)] and the contributing dose from individual radionuclides for components of the aquatic and terrestrial food-chain pathways

	Dose	^{137}Cs	^{60}Co	^{90}Sr	^{3}H
Fish	0.28	0.054	0.0002	0.22	0.0021
Turtles	0.15	0.057	0.009	0.15	0.002
Clams	0.076	0.0035	0.0022	0.07	0.0005
Crayfish	0.08	0.0075	0.0024	0.07	0.0005
Aquatic plants	5.1	0.53	0.66	3.90	0.0052
Waterfowl[a]	1.00	1.0	0.01	--	--
Water[b]	0.53	0.014	0.0044	0.29	0.23
Rabbits	0.84	0.58	0.096	0.16	0.0063
Deer	0.11	0.017	0.0006	0.013	0.08
Beef	0.83	7.0	0.44	0.73	0.11
Milk[c]	11.0	7.4	0.12	2.90	0.33
Poultry	1.0	0.98	0.023	0.015	--
Eggs	0.6	0.28	0.0076	0.31	
Vegetables	18.0	8.28	0.17	9.0	--
Honey	0.0001	0.0001		--	--
Sediment	0.57	0.50	0.01	0.055	--

[a]D. K. Halford, J. B. Millard, and O. D. Markham, "Radionuclide Concentration in Waterfowl Using a Liquid Radioactive Waste Disposal Area and Potential Radiation Dose to Man," *Health Phys.* **40**, 173-181 (1981).

[b]Dose from Melton Branch water.

[c]Includes cow and goat milk.

The food-chain pathways having the highest total estimated doses were the vegetable, milk, and beef pathways respectively. Concentrations of radionuclides in the components in these pathways were estimated by using mathematical models. If measured concentrations of radionuclides had been available, estimated doses would probably be reduced; however, these food-chain pathways would, more than likely, remain the major food-chain pathways in regard to dose.

The ratios of the estimated dose for each food-chain pathway to both the 0.25- and 1.0-mSv/year screening limits are shown in Table 8.8. Pathways having a ratio of much less than 1 would have a low priority in regard to their contribution to total food-chain dose to humans and thus a low priority for further study. Food-chain pathways having ratios approaching or greater than 1 would warrant consideration for further study. At a screening limit of 0.25 mSv/year, five ingestion pathways (clams, crayfish, turtles, deer, and honey) have ratios of less than 1 (Table 8.8). At a screening limit of 1.0 mSv/year, food-chain pathways, except waterfowl, aquatic plants, beef, milk, poultry, and vegetables, have ratios of less than 1 (Table 8.9). Concentrations of radionuclides for components of the food-chain pathways that exceeded the 1.0-mSv/year screening limit were derived values, except for ^{137}Cs and ^{60}Co in aquatic plants. As discussed previously, using mathematical models to derive radionuclide concentrations in food products induces additional conservatism and results in a greater overestimation of the dose.

Doses for external radiation exposure to individuals engaged in various activities in the WOL environment are listed in Table 8.10. These are estimated doses assuming that the exposure is from the concentration of radionuclides in sediment (dry weight) and that the occupancy rate and activities are those listed in Table 8.3. The major dose-contributing radionuclides are ^{137}Cs and ^{60}Co. Individuals engaged in activities in which they would be exposed to radioactivity in the drained lake bed would receive the highest estimated doses of all exposure pathways. The estimated doses and percentages of the total dose for the aquatic and terrestrial food-chain pathways and the external exposure pathways are shown in Table 8.11. External exposure to humans from gamma radiation exceeds the food-chain pathway doses by about a factor of 2.

Draining the lake, as proposed in Scenario 2 (Sect. 8.1.2.1), would increase the external irradiation exposure of individuals working in the WOL area because the dry sediments would become a potential source of radiation exposure. Under current conditions, lake water acts as a partial shield by attenuating the gamma radiation from the radioactive sediment; as a result, the external exposure is reduced.

Total estimated doses given in Table 8.11 for the food-chain and external exposure pathways exceed the 0.25- and 1.0-mSv/year screening limits many times; therefore, for the scenarios included in this analysis, neither pathway can be excluded from consideration. In Table 8.8 the ratio for various food-chain pathways that contribute to the total food-chain dose provides a method for estimating the importance of the various food-chain pathways.

Table 8.8. Ratio of radiation dose for different food-chain pathways to U.S. Environmental Protection Agency (EPA) screening limit (0.25 mSv/year) and U.S. Department of Energy (DOE) screening limit (1.0 mSv/year)[a]

Pathway	Dose:DOE limit	Dose:EPA limit
Fish	1.1	0.28
Turtles	0.6	0.15
Clams	0.30	0.076
Crayfish	0.32	0.08
Aquatic plants	20.4	5.1
Waterfowl	4.0	1.00
Water	2.1	0.53
Rabbits	3.4	0.84
Deer	0.46	0.11
Beef	33.2	8.3
Milk	42.0	10.0
Poultry	4.1	1.0
Eggs	2.4	0.6
Vegetables	70.0	17.5
Honey	0.0004	0.0001
Sediment	2.3	0.57

[a]The symbol *mSv* denotes millisievert: 1 sievert = 100 rems.

8.1.4 Conclusions and Recommendations

Providing the scenarios considered for screening are viable options for remedial action and based on the results of the screening analysis and on the screening limit of 1.0 mSv/year under which ORNL currently operates, actual measurements of radionuclide concentrations should be obtained for those components of the food-chain pathways that exceeded the screening limit. Doses should then be calculated based on these measured values for food-chain components instead of derived values. Radionuclide concentrations should be determined for small game animals, waterfowl, and aquatic macrophytes, which have not been sampled for many years, to obtain a more realistic dose for these food-chain components.

The concentration of radionuclides in sediment was the basis for many of the derived values in the food-chain components and for the external exposure pathways. Since the radionuclide concentrations for sediment used in this analysis were based on three core samples collected in 1986 and nine surface sediment samples collected in 1985, additional sediment sampling is necessary because of the heterogeneity in the distribution of the radionuclides in WOL sediments.

Table 8.9. Food-chain and external exposure pathways exceeding 0.25- and 1.0-mSv/year dose limits[a]

Pathway	Pathway components	Components exceeding 0.25 mSv/year	Components exceeding 1.0 mSv/year
Aquatic	Fish Turtles Clams Crayfish Aquatic plants Waterfowl Water	Fish Aquatic plants Waterfowl Water	Aquatic plants Waterfowl
Terrestrial	Rabbits Deer Milk Beef Poultry Eggs Vegetables Sediment	Rabbits Milk Beef Poultry Eggs Vegetables Sediment	Milk Beef Poultry Vegetables
External	Working over sediments Handling fishing gear Sunbathing over sediments Boating	Working over sediments Handling fishing gear Sunbathing over sediments	Working over sediments Handling fishing gear Sunbathing over sediments

[a]The symbol *mSv* denotes millisievert: 1 sievert = 100 rems.

Draining the lake (Scenario 2) would eliminate the aquatic food-chain pathways but would result in an exposed lake bed with a potential for higher radiation doses for external exposures and would create a larger area for the terrestrial food-chain components to accumulate radionuclides.

This screening analysis is based on mathematically derived estimates, available data, and scientific judgment, all of which are conservatively biased toward higher dose estimates. As more data become available and assumptions change, results of the screening analysis are expected to change. This screening analysis was performed for the remedial action scenarios previously described for WOL only and does not include the radioactive waste ponds or upper WOC.

8.2 FUTURE STUDIES

Future studies in WOL were identified as discrete subtasks of Task 6 (radioecology of WOL) in Loar et al. (1991) and are briefly described in Sects. 8.2.1 through 8.2.3.

Table 8.10. Dose to humans for beta and gamma irradiation from external exposure pathways
[All doses in millisieverts per year (1 sievert = 100 rems)]

		Total dose	^{137}Cs	^{60}Co	^{90}Sr	^{3}H
Working over sediments	Beta[a]	19.30[b]	16.00	0.56	2.70	0.0018
	Gamma[c]	52.70	37.70	15.00	--	--
Handling fishing gear over sediments	Beta	19.30	16.00	0 56	2.70	
	Gamma	5.27	3.77	1.50	--	--
Sunbathing on sediments	Beta	9.63	8.00	0.27	1.36	0.0009
	Gamma	26.60	18.90	7.70	--	--
Swimming in lake	Beta	0.0076	0.00003	0.00002	0.0011	0.0064
	Gamma	0.0011	0.00015	0.00099	--	--
Boating	Beta	0.0003	0.0002	0.0001		
	Gamma	0.0035	0.0005	0.003	--	--

[a]Skin dose rate.
[b]Conversion factor of 0.3 [International Commission on Radiological Protection (ICRP) 26] used to convert to effective whole-body dose rate.
[c]Effective whole-body dose rate.

Table 8.11. Estimated dose contributions and percentage of total dose for food-chain and external exposure pathways for the White Oak Lake environment

Pathway	Dose (mSv/year)[a]	Percentage of total dose
Aquatic food chain	7.0	4.8
Terrestrial food chain	40.00	27.4
External exposure		
Beta	14.50[b]	9.9
Gamma	84.50	57.9
Total dose	146.00	

[a]The symbol *mSv* denotes millisievert: 1 sievert = 100 rems.
[b]Dose conversion factor of 0.3 [International Commission on Radiological Protection (ICRP) 26] was used to convert skin dose from beta irradiation to effective whole-body dose.

8.2.1 Radionuclide Screening

With the completion of the initial radionuclide screening for WOL and its environs, the emphasis of Subtask 6a will shift to a similar screening of toxic metals and organic contaminants in the lake. This screening will be completed in 1987, and the results will be presented in the next annual report.

The initial screening identified the need for additional data on radionuclide concentrations in waterfowl and small game animals (e.g., rabbits). Waterfowl that frequent WOL may be killed by hunters off-site. Limited data on waterfowl from WOL and information from the literature indicate that the amount of radioactivity in these birds would be very low. This conclusion is based on the low levels of radioactivity in WOL that the waterfowl have access to and the short time that transient birds spend on the lake. Nevertheless, to document that transient waterfowl contain relatively low levels of radioactivity, a study will be initiated in 1987 to census waterfowl populations of WOL. This study will establish the existence of a resident population and provide an estimate of the number of transient waterfowl. During the course of the study, samples of several species of waterfowl will be collected and analyzed for radioactivity. Data on radionuclide concentrations in rabbits from the vicinity of WOL will also be obtained.

8.2.2 Current Radioecological Status of White Oak Lake

A report on the current radioecological status of WOL is in preparation (Sherwood and Loar 1987). This report will compare the current status of the lake with conditions in 1950, 1960, and 1970. Studies are planned to determine the concentration and quantity of radionuclides in the invertebrate, fish, and aquatic macrophyte populations in WOL (Sects. 6.1.6, 6.2.6, and 8.2.3 respectively).

Based on the screening analysis, additional information is needed on the concentration of radionuclides in WOL sediment. Data on radionuclide concentrations in sediments and on the concentrations of radionuclides in the biota will be required to establish an inventory of the radionuclides in various components of WOL as part of Subtask 6b of BMAP. These baseline data will be used to establish the current radioecological status of WOL and its environs in order to assess the effectiveness of future remedial actions.

8.2.3 Remobilization of Radionuclides from Sediments

The concentrations of radionuclides in WOL sediment are several orders of magnitude higher than in the aqueous phase. The contribution of remobilized radionuclides to aquatic food chains in WOL is unknown. Also, the contribution of radionuclides from sediments, either in the form of suspended matter or in the dissolved form, to the quantity of radioactivity passing over the dam is unknown. If the lake is maintained, it will be necessary, for remedial purposes, to know (1) the potential for radionuclides in the sediments to accumulate in aquatic food chains and (2) the potential for release of radioactivity over the dam from sediment remobilization. This information will be obtained in Subtask 6c of BMAP.

As part of this subtask, a study was designed to determine the role of aquatic macrophytes in the remobilization of contaminants from the sediments of WOL. This study also includes determination of the seasonal and spatial distribution of above-surface biomass, by species, and the annual production and biomass turnover of the macrophytes. In addition to examining the question of remobilization, this study will contribute information to Subtask 6b.

Macrophyte photosynthesis, biomass accumulation, and eventual decay can have profound effects on energy flow and biogeochemical cycles in the lake ecosystem. Unlike other aquatic plants, rooted macrophytes have access to material buried in the sediments and may act as a "living conduit" for the remobilization of these materials to the water column. Macrophytes may have a significant effect on the remobilization of radionuclides from the sediments in WOL because the greatest percentage of the lake biomass is composed of macrophytes. Other studies will be implemented, as required, to determine the biological, chemical, and physical factors that affect remobilization of radionuclides from the sediments of WOL.

9. CONTAMINANT TRANSPORT, DISTRIBUTION, AND FATE IN THE WHITE OAK CREEK EMBAYMENT-CLINCH RIVER-WATTS BAR RESERVOIR SYSTEM

9.1 INTRODUCTION

Contaminants from ORNL operations are released from WOL and transported downstream through WOC embayment, into the Clinch River (at CRK 33.5, 3.5 km below Melton Hill Dam), and ultimately into Watts Bar Reservoir. In Task 7 of BMAP (Loar et al. 1991), coordinated ecological and biogeochemical studies are being conducted to determine the transport, distribution, and ecological fate of contaminants introduced to the embayment-river-reservoir system as a result of waste disposal activities at ORNL. These studies are closely coordinated with and complementary to other biological monitoring tasks included in BMAP.

Task 7 focuses on (1) preliminary efforts to assess the extent of off-site contamination of the aquatic environment downstream of WOL and (2) prediction of the effectiveness of remedial action measures at ORNL and other Oak Ridge DOE facilities to reduce contaminant releases to the Clinch River. The primary objectives of Task 7 are

1. to integrate previous information and data on the ecological fate of radionuclides and other contaminants discharged from WOL to the WOC embayment-Clinch River-Watts Bar Reservoir system;

2. to determine the extent and spatial distribution of contaminant accumulation in the sediments and in selected biota of the embayment-river-reservoir system;

3. to identify and quantify the hydrodynamic, biological, and geochemical factors that determine the transport, distribution, and ecological fate of DOE-derived contaminants in the Clinch River and Watts Bar Reservoir system; and

4. to predict the effect of ORNL remedial actions by using simulation models that incorporate the hydrodynamic, biological, and geochemical factors to be investigated in pursuing Objective 3.

To accomplish these objectives, four subtasks (Fig. 9.1) that involve an interdisciplinary combination of simulation modeling, field sampling and measurements, and laboratory experimentation and analysis are being conducted.

9.2 LITERATURE REVIEW AND SYNTHESIS

The available information on the fate of radionuclides and other contaminants discharged from WOL to the embayment-river-reservoir system is being reviewed, integrated, and synthesized (Subtask 7a of BMAP). To date, we have concentrated on the identification

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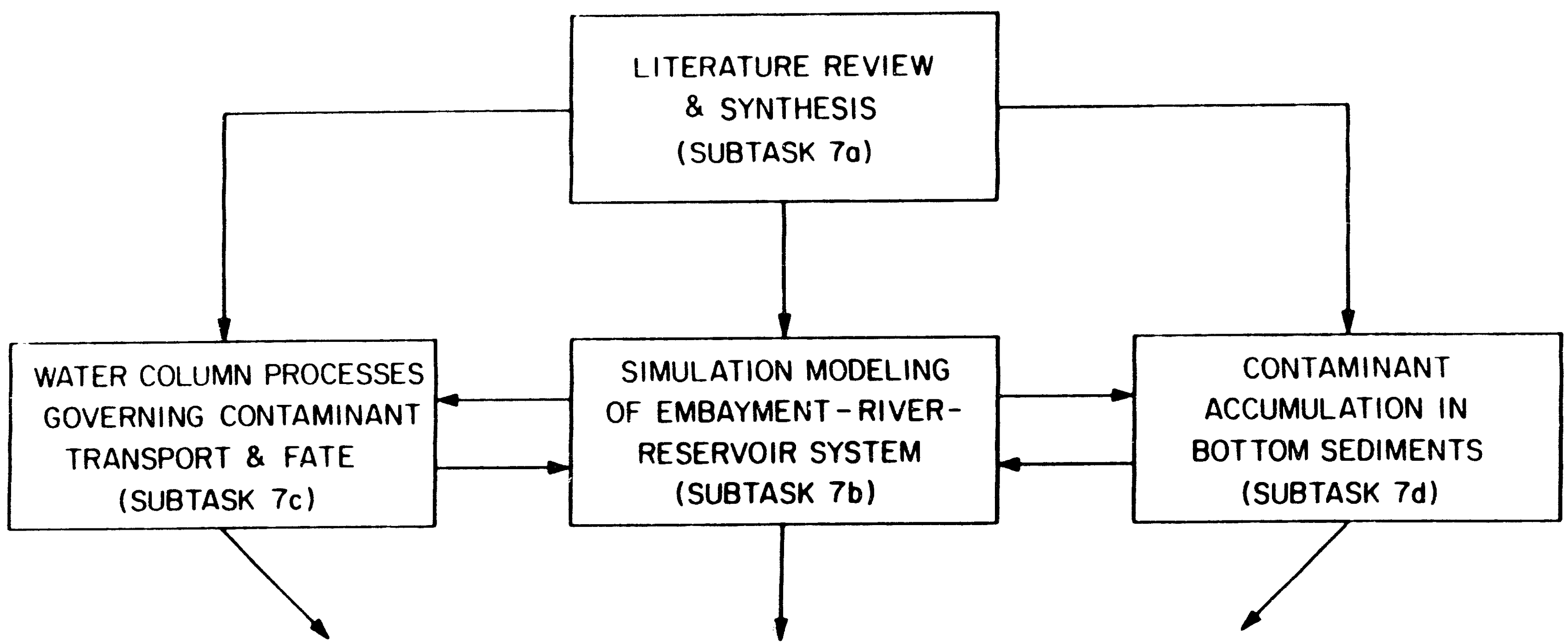

RESEARCH PRODUCTS

1. PREDICTION AND EVALUATION OF REMEDIAL ACTION ALTERNATIVES
2. ESTABLISHMENT OF QUANTITATIVE BASELINE REQUIRED FOR EVALUATING LONG-TERM EFFECTIVENESS OF THE ORNL REMEDIAL ACTION PROGRAM

Fig. 9.1. Interrelationships between the proposed subtasks and the research products.

of long-term water quality data sets that can be used to test and calibrate the embayment-river-reservoir model (see description of Subtask 7b in Sect. 9.3).

9.3 SIMULATION MODELING OF THE EMBAYMENT-RIVER-RESERVOIR SYSTEM

The transport, distribution, bioavailability, and ecological fate of contaminants discharged from WOL (or from a discharge canal, if WOL is bypassed) are strongly influenced by the hydrodynamic regime and by suspended-particle dynamics. The embayment-river-reservoir system is highly dynamic, as hydropower and flood-control operations at the Melton Hill, Fort Loudon, and Watts Bar dams result in both daily and seasonal variations in flow velocity and direction, water column mixing, suspended-particle transport, water residence times, and water levels. Similarly, the transport, bioavailability, and fate of contaminants that become associated with suspended particles are affected by biological particle production, complexation of contaminants with dissolved organic matter in the water column, contaminant sorption kinetics and particle sedimentation, resuspension, and accumulation.

As part of Subtask 7b of BMAP, a branched two-dimensional (longitudinal-vertical) model (Fig. 9.2) of the embayment-river-reservoir system is being developed by Dr. R. T. Brown at the Water Research Center, Tennessee Technological University. The general scheme for the model follows that of previous successful two-dimensional models applied to Fort Loudon, Cherokee, Pickwick, Boone, Normandy, Old Hickory, and Douglas reservoirs in Tennessee. The Watts Bar version will be the most involved application to date because of the complex branching flow patterns in the Clinch, Emory, and Tennessee River arms of the reservoir.

The reservoir has been segmented into 52 columns with 10 vertical layers. The Tennessee and Emory rivers are divided into 3- to 5-mile segments, while the Clinch River is divided into 1-mile segments to allow more-detailed simulation of the flow and mixing patterns between WOC and ORGDP down to the mouth of the Clinch River. The model includes the effects of the wedge storage created on the Clinch River during peaking-power releases from Melton Hill Dam and the flushing of WOC embayment as river elevation rises and falls during these peaking-release cycles. The model simulates seasonal patterns of physical and chemical variables in the entire reservoir through the use of daily inputs and model time steps. The hourly flows and releases from Melton Hill Reservoir and from WOL are simulated by using hourly inputs and model time steps for the Clinch and Emory River arms.

Inflows and inflow concentrations for the modeled variables are specified for ten locations: Fort Loudon releases, Little Tennessee River inflow (from Tellico Reservoir), Clinch River inflow (from Melton Hill Reservoir), WOC inflow, Poplar Creek inflow, Emory River inflow, White's Creek inflow, Piney Creek inflow, and inflow from local drainage into Watts Bar Reservoir. The initial model calibration will use data collected during the Clinch River Study in 1961–62. TVA conducted a seasonal water quality survey of Watts Bar

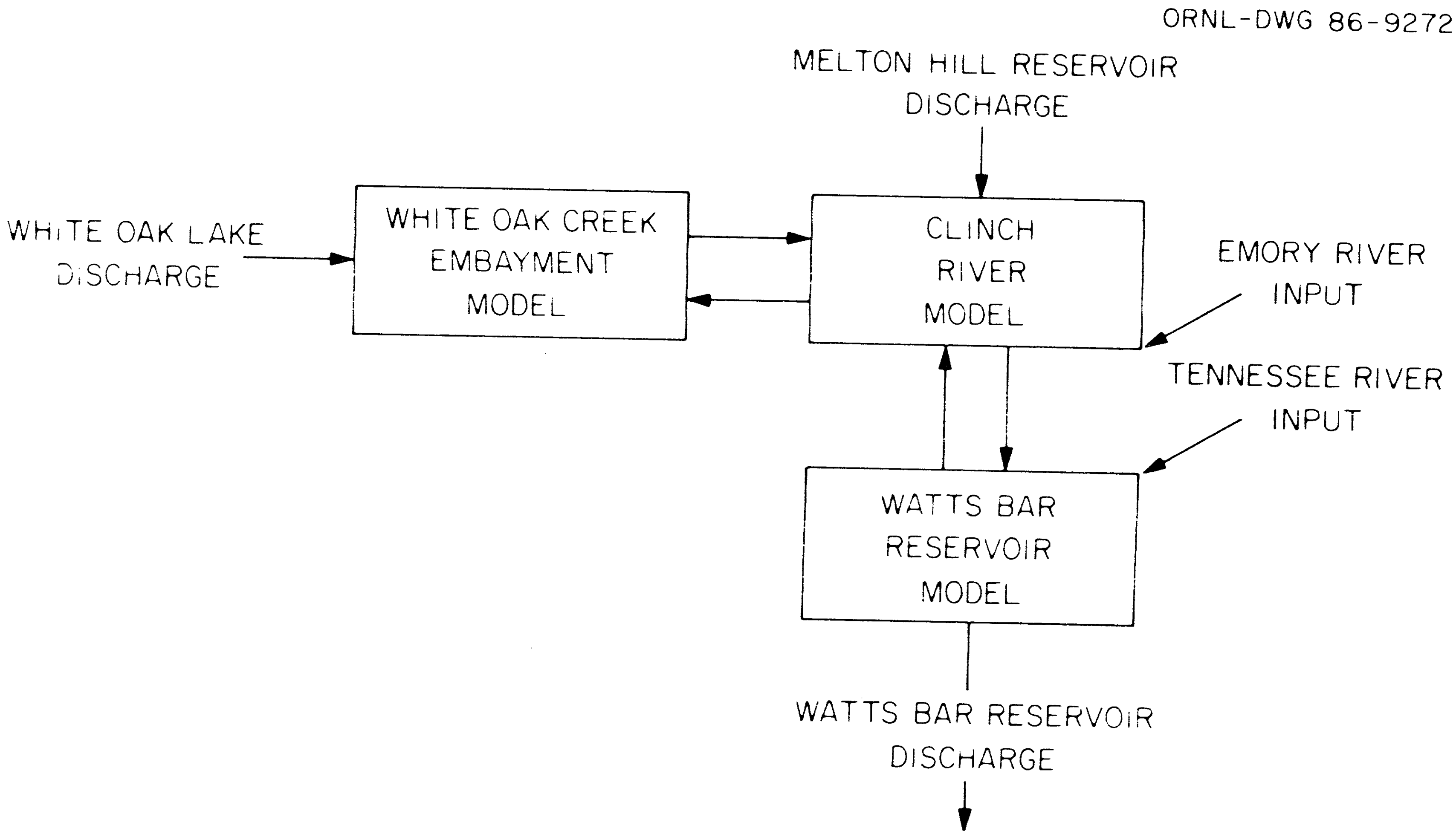

Fig. 9.2. Linked simulation models of the embayment, river, and reservoir. Initially, simple two-dimensional flow models will be constructed, based on the morphology of embayment, river, and reservoir portions of the system and using White Oak Lake and Melton Hill Dam discharge data as the primary input variables. Information derived from field measurements will be incorporated into expanded versions of these flow model to permit integration of data on physical, chemical, and biological processes controlling contaminant transport and fate in the river-reservoir system. Both initial and expanded versions will be employed throughout the project to (1) guide sampling and data collection efforts and (2) predict and evaluate the relative effect of remedial action alternatives.

Reservoir with monthly profiles of temperature, dissolved oxygen, turbidity, and nutrients during 1962. These data will also be used to test model predictions. Data collected during the December 1985 strontium release can also be used to verify the hourly flow and mixing patterns in the Clinch River.

The model is being modified to include particle production by phytoplankton growth, the sorption of radionuclides onto suspended particles, and the deposition and resuspension of particles during high-flow episodes. Initially, ^{90}Sr and ^{137}Cs are being used as representative soluble and particle-reactive radionuclides respectively. The biological components of the model will also be expanded to allow the estimation of biological uptake of the radionuclides and other contaminants and of food-web bioaccumulation patterns.

A working model has been formulated and is now being debugged. The inflow data necessary for model calibration with the 1961–62 Clinch River have been prepared, and the morphological data for the embayment, river, and reservoir are ready to be incorporated into the model. An initial run of the model is expected by late March 1987. The emphasis will then shift to preparing historical data sets that will serve to calibrate and guide modifications of the model to support the range of applications required by ORNL-ESD projects concerned with the fate of materials discharged and transported into the Clinch–Watts Bar system.

9.4 WATER COLUMN PROCESSES GOVERNING CONTAMINANT TRANSPORT AND FATE

Some contaminants and radionuclides, such as ^{3}H, ^{90}Sr, and ^{131}I, are relatively soluble in freshwater systems; consequently, their transport and biogeochemical fate are mediated by movements of water masses and by biological uptake from the water phase. In Subtask 7c of BMAP, ^{90}Sr is used as a representative of a contaminant that remains in solution when discharged into the embayment-river-reservoir system, and analytical techniques are being developed for accurately measuring the low concentrations of ^{90}Sr that occur in the Clinch and Tennessee River system. However, most contaminants and radionuclides (such as PCBs, Hg, ^{60}Co, ^{137}Cs, and $^{239,240}Pu$) are chemically and biologically reactive and rapidly become associated with particles in freshwater systems. Consequently, the transport and biogeochemical fate of these contaminants are governed to a great extent by sorption kinetics and fine-particle dynamics, as well as by water movement.

The hydrodynamics and limnology of WOC embayment are strongly influenced by hydropower operations at the Melton Hill Dam (Loar et al. 1981a), located just 3.7 km upstream. Because the Norris and Melton Hill reservoirs (located on the upper Clinch River) function as efficient traps for suspended particles, the water discharged from the Melton Hill Dam usually contains low concentrations of suspended particles. Therefore, in the Clinch River and Watts Bar Reservoir, the photosynthetic production of organic matter by planktonic algae is an important source of suspended particles available for contaminant sorption and of dissolved organic compounds capable of complexing radionuclides and other contaminants. Because of the factors affecting phytoplankton particle production and the rapid downstream transport of water through the river segment between Melton Hill Reservoir and upper Watts Bar Reservoir, it is hypothesized that most contaminant sorption, biological uptake of contaminants by planktonic biota, and removal of particle-associated contaminants from the water column by settling will occur primarily in three areas: (1) WOC embayment, (2) other downstream marginal embayments along the Clinch River, and (3) Watts Bar Reservoir.

An initial set of field measurements was made in July 1986 to quantify the flow regime, thermal stratification and relative water column stability, suspended-particle concentrations, and phytoplankton biomass and distribution at stations in the Clinch River and in Watts Bar Reservoir. Physical-chemical variables and chlorophyll concentrations (indicative of phytoplankton biomass levels) were measured along a transect from the mouth of WOC embayment (CRK 33.5) to the uplake portion of Watts Bar Reservoir (TRK 910.1). These field data, collected during a relatively slack water period (i.e., during a period of low releases from Melton Hill Dam) in late July 1986, indicated that phytoplankton biomass, and thus suspended-particle availability for contaminant sorption, in near-surface water layers were relatively low (~8 mg chlorophyll *a* per cubic meter) in the river, but increased significantly (to ~21 mg chlorophyll *a* per cubic meter) in the upper reservoir. Additional field measurements will be conducted on a seasonal basis and during a variety of flow conditions (e.g., during low-flow vs peaking-power releases from the Melton Hill Dam) to document a range of flow velocities, mixing regimes, levels of biological activity, particle production rates, and suspended-particle concentrations.

In conjunction with the hydrologic, biomass, and suspended-particle data, measurements were initiated on the partitioning of particle-reactive contaminants between aqueous and particulate phases. The partitioning data are presented in Table 9.1. Because of the relatively low concentrations of radionuclides in the water column, large-volume water samples (830 L) were processed and the suspended matter removed by continuous-flow centrifugation.

Three large-volume water and suspended-matter samples were collected in Watts Bar Reservoir (near the Kingston city water intake) on December 1, 5, and 17, 1986. These samples were collected (1) to determine whether the abnormally high concentrations of ^{60}Co measured in WOC on November 25-26, 1986, could be traced into Watts Bar Reservoir, and, if they could, then (2) to obtain field data on the transport rates for particle-reactive contaminants. Also included in Table 9.1 are concentration and particle-to-water distribution data for mercury and for several other radionuclides, including plutonium isotopes and naturally occurring ^{7}Be. Beryllium-7 (53.3-d half-life) is introduced into aquatic systems as a dissolved constituent of rainwater, and its distribution is being used to quantify the sorption rates and removal behavior of particle-associated contaminants in aquatic ecosystems. The average particle:water distribution ratio (K_d) for ^{7}Be (9×10^4) is typical of other particle-reactive contaminants in the river-reservoir system (Table 9.1).

The particle:water sorption ratios (R_s) listed in Table 9.1 are numerically equal to K_d, but are determined in the field; they are site-specific empirical values that do not imply chemical equilibrium or reversibility. Values of R_s are important for quantifying radionuclide sorption in models used to assess radionuclide transport and fate. Although the sorption ratios for radiocesium, radiocobalt, plutonium, and mercury are high (about 10^5), it should

Table 9.1. Contaminant distributions between aqueous and particulate phases

Date	Nuclide	Suspended load (mg/L)	Dissolved concentration (mBq/L)	Particulate concentration (mBq/g)	Distribution[a] (R_s)
			Kingston City Park—Clinch River		
12/1/86	^{60}Co	14	0.9	41	5×10^4
	^{137}Cs		1.3	250	2×10^5
	^{7}Be		3.4	200	6×10^4
	$^{239,240}Pu$		0.014	1.5	1×10^5
	^{238}Pu		0.004	0.19	5×10^4
12/5/86	^{60}Co	11	9.3	160	2×10^4
	^{137}Cs		1.8	540	3×10^5
	^{7}Be		2.4	195	8×10^4
	Hg (ppb)		0.005	2360	5×10^5
12/17/86	^{60}Co	7	8.0	270	3×10^4
	^{137}Cs		3.8	980	3×10^5
	^{7}Be		2.8	295	1×10^5
			Thief Neck—Watts Bar Reservoir		
12/22/86	^{60}Co	7	0.44	34	8×10^4
	^{137}Cs		0.63	190	3×10^5
	^{7}Be		2.9	322	1×10^5

[a]Particle:water distribution (R_s) = concentration per kilogram of particles:concentration per liter of water.

be remembered that at suspended-matter concentrations of 10 mg/L (which is typical during normal flow conditions), there is 10^5 times more water per liter than suspended matter. Therefore, about 50% of the total radioactivity in a liter of water will be in the dissolved phase, and the remaining 50% will be associated with the suspended matter. However, during rainstorms, floods, and other periods of high discharge when suspended-matter concentrations increase to about 100 mg/L, more than 90% of the total radioactivity in the water column will be transported with particles. Since transport in river systems is dominated by periods of high discharge, the transport, distribution, and fate of these radionuclides will be governed to a great extent by fine-particle dynamics.

On December 1, 1986, the dissolved concentration of ^{60}Co was 0.9 mBq/L near Kingston City Park, and the concentration of ^{60}Co on suspended particles was 41 mBq/g. Since the ^{60}Co concentration on bottom sediments in the area ranges from ~30 to 45 mBq/g, the ^{60}Co concentration measured on the suspended matter (41 mBq/L) is typical for resuspended bottom sediments. Likewise, the concentrations of ^{137}Cs and plutonium in the water and on the suspended matter were not abnormally high relative to the values expected from the resuspension of river-reservoir sediments, primary productivity, and equilibrium particle:water distributions.

On December 5, 1986, the dissolved concentrations of ^{60}Co increased by an order of magnitude, and particulate concentrations of ^{60}Co increased by a factor of 4. On December 17, the dissolved concentration of ^{60}Co was lower, but particulate ^{60}Co concentrations had increased by another factor of 2. These data imply that either (1) the maximum water column concentration of ^{60}Co at Kingston, associated with the ^{60}Co release into WOC on November 26, occurred prior to the December 1 sampling or (2) that it took ~2-3 weeks before the ^{60}Co released into WOC was transported via the Clinch River into Watts Bar Reservoir. In the latter case, the delayed increase in the particulate ^{60}Co and ^{137}Cs concentrations suggests that particle deposition and resuspension processes can cause the maximum water column concentrations at Kingston to occur up to 1 month after contaminant release into WOC. Such a delay could be even longer during the summer and fall, when inflow-related resuspension events occur less frequently.

9.5 CONTAMINANT ACCUMULATION IN BOTTOM SEDIMENTS

Vertical distributions of ^{137}Cs and ^{60}Co have been measured in 35 sediment cores collected in the Clinch River and the upper half of Watts Bar Reservoir. The vertical distribution of ^{137}Cs in the sediment cores is strongly correlated with the historical record of ^{137}Cs discharges from WOL (Fig. 9.3). By using the measured particle:water distribution for ^{137}Cs (Table 9.1) and the historical ^{137}Cs profile recorded in our sediment cores (Fig. 9.2), the past history of dissolved concentrations of ^{137}Cs occurring in the reservoir water near Kingston was estimated (Table 9.2).

The thickness of radiocesium-contaminated sediments ranges from ~1.5 m in the upper portion of the reservoir (near Kingston) to ~0.5 m in the middle portion (near Cold Springs

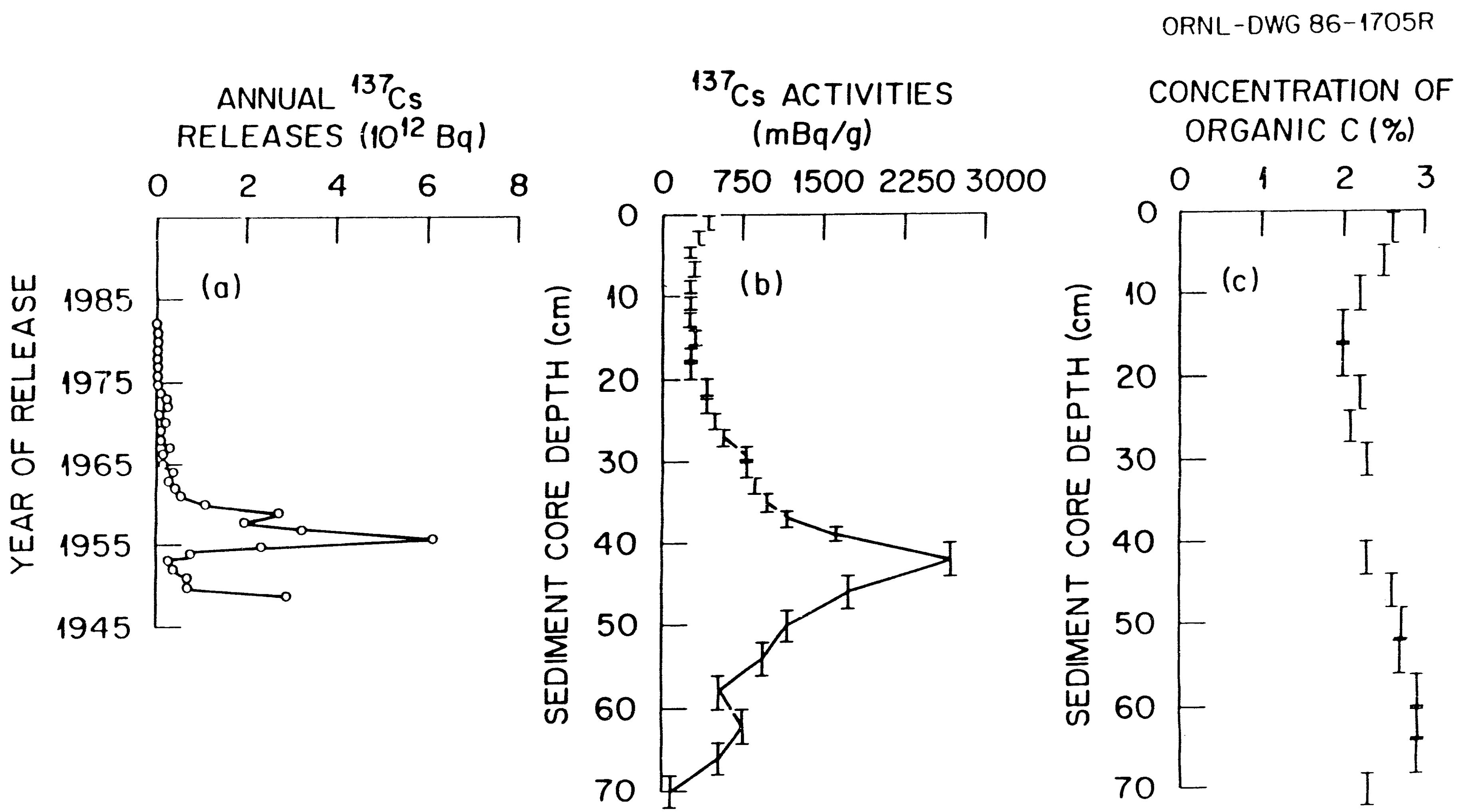

Fig. 9.3. Relationship between (a) the historical record of ^{137}Cs releases from White Oak Lake and (b) the vertical distribution of ^{137}Cs in bottom sediments of the lower Clinch River (Clinch River kilometer 1.5), with (c) concentrations of organic carbon shown for the lower Clinch River. The uniform distribution of organic carbon in this core implies that the large peak in ^{137}Cs activity at 40–44 cm does not reflect any major change in sediment size or type but reflects the extent of ^{137}Cs contamination at the time of deposition.

Table 9.2. History of dissolved concentrations of ^{137}Cs at Kingston, based on the sediment core record

Sediment depth (cm)	Sediment concentration (mBq/g)	Dissolved concentration[a] (mBq/L)	Sediment date[b]
0–2	403	1.3	1985
2–4	318	1.1	
4–6	270	0.9	
6–8	292	1.0	1980
8–10	230	0.8	
10–12	255	0.9	
12–14	230	0.8	1976
14–16	289	1.0	
16–18	307	1.0	
18–20	262	0.9	1972
20–22	390	1.3	
22–24	385	1.3	
24–26	475	1.6	1968
26–28	535	1.8	
28–30	765	2.6	
30–32	780	2.6	1963
32–34	875	2.9	
34–36	950	3.2	
36–38	1160	3.9	1959
38–40	1585	5.3	
40–44	2640	8.9	1956
44–48	1715	5.7	1955
48–52	1160	3.9	
52–56	960	3.2	
56–60	535	1.8	1951
60–64	690	2.3	
64–68	505	1.7	
68–72	85	0.3	1947
72–76	25	0.1	
76–80	ND[c]	ND[c]	

[a]Dissolved concentrations were calculated by assuming a ^{137}Cs particle:water distribution ratio of 3×10^5.
[b]Sediment dates were estimated by using a sediment accumulation rate of 1.6 cm/year.
[c]Not detectable.

Harbor). Radiocesium concentrations in the reservoir sediments commonly range from 200 to 2000 mBq/g, with the highest value (2640 mBq/g) occurring at a sediment depth of 40 to 44 cm in a core collected offshore from the City of Kingston, near the mouth of the Clinch River (Fig. 9.3). Sediment cores in the lower portion of the reservoir (near Watts Bar Dam) will be collected after winter drawdown of the water level. In addition to the sediment cores, the concentrations of ^{137}Cs and ^{60}Co have been measured in 85 surface-sediment grab samples. Concentrations of ^{137}Cs ranged from a minimum of 4 mBq/g to a maximum of 395 mBq/g, with a mean of 80 mBq/g. Concentrations of ^{60}Co in reservoir surface sediments ranged from nondetectable (<2 mBq/g) to 70 mBq/g, with a mean of 7 mBq/g.

The distribution of ^{137}Cs is being used as a cost-effective tracer to (1) identify and distinguish areas of contaminant-particle accumulation and areas of erosion (no net accumulation) in the embayment-river-reservoir system and (2) to screen sediment samples prior to other, more costly, contaminant analyses (e.g., analyses for PCBs, mercury, and other heavy metals). The discharge histories of these contaminants released from the DOE Oak Ridge Reservation are poorly known but are also recorded in the sediments of undisturbed long-term accumulation sites. Therefore, measured contaminant profiles in selected sediment cores can be used to document these histories. These histories and data on contaminant partitioning between aqueous and particulate phases will be used to estimate the dissolved concentrations of other contaminants in the reservoir waters near Kingston during the past 30 years. Finally, the results will be used to map the spatial distribution of contamination in the embayment-river-reservoir sediments, estimate the total amount of fission nuclides and other energy- and weapons-related contaminants that have been trapped within the reservoir sediments, and establish a quantitative scientific baseline for evaluating the long-term effectiveness of the ORNL RAP.

9.6 SUMMARY

Task 7 consists of an interdisciplinary combination of simulation modeling, field sampling and measurements of the water column and sediments, and laboratory analyses to determine the transport, distribution, and ecological fate of contaminants introduced to the WOC–Clinch River–Watts Bar Reservoir system as a result of waste disposal activities at ORNL. The distributions of selected radionuclides are being measured in the water, suspended particulate matter, and sediments of the embayment-river-reservoir system. These radionuclide measurements, which are performed in the laboratory by nondestructive, high-resolution gamma spectrometry, will provide

1. site-specific data on the distribution, accumulation, and ecological fate of a variety of radioactive contaminants discharged from ORNL;

2. a means of estimating water-particulate phase partitioning, sorption kinetics, water column residence times, and areas of accumulation for other particle-reactive contaminants; and

3. a cost-effective basis for selecting areas, samples, and organisms for more-costly chemical analysis for PCBs, PAHs, and heavy metals.

Linked flow models of the embayment, river, and reservoir are being developed to guide the sampling design and to permit preliminary predictions of contaminant transport and fate. The models will be continuously refined by incorporating field and laboratory results and will be used to (1) examine the interrelationships between variable physical, chemical, and biological conditions and contaminant transport, bioavailability, and fate; (2) provide guidance for additional sampling and analysis; and (3) predict the effects of alternative remedial action measures. In an initial scoping effort, sediment cores have been collected and analyzed to document the historical record of contaminant-particle accumulation and to determine the spatial extent of the contamination problem in the embayment-river-reservoir system. Ultimately, these efforts will result in a quantitative baseline of information necessary for assessing the long-term effectiveness of the ORNL RAP.

9.7 FUTURE STUDIES

Collection of sediment cores and surface sediment samples from the lower portion of Watts Bar Reservoir will be completed in 1987. Determination of the vertical distribution of ^{137}Cs in these samples, in combination with the data obtained from other cores collected in the Clinch River and the upper portion of the reservoir in 1986, will provide an inventory of ^{137}Cs accumulation in the river-reservoir sediments. Additionally, a sediment-type distribution map of the river-reservoir sediments will be developed to identify zones of ^{137}Cs accumulation. These data and the use of ^{137}Cs as an indicator of particle-reactive contaminants will allow estimation of the extent of contaminant distribution and accumulation for other contaminants of interest (especially mercury, PCBs, and other radionuclides and heavy metals) and will provide a cost-effective basis for selecting areas, samples, and organisms for further analyses. Work will continue on (1) examination of contaminant sorption-remobilization processes; (2) investigation of suspended-particle dynamics and transport; and (3) development of the ability to predict transport, distribution, and fate of ORNL-derived contaminants in the embayment-river-reservoir system through the use of an integrated simulation model.

Long-lived plutonium radioisotopes have been introduced into the river-reservoir system via fallout from atmospheric nuclear weapons testing and by releases from ORNL. The ^{238}Pu:$^{239,240}Pu$ activity ratio in global fallout is ~ 0.05. In seep area soils near a formerly used waste disposal trench (Trench 7, Fig. 2.3), Olsen et al. (1986) measured a ^{238}Pu:$^{239,240}Pu$ ratio of ~ 23.0. Surface sediments in WOL have a ^{238}Pu:$^{239,240}Pu$ ratio of ~ 0.40 (J. R. Trabalka, ORNL, personal communication, 1986). The plutonium isotopic ratio for the one suspended-matter sample collected in Watts Bar Reservoir was 0.13 (Table 9.1). These data suggest that WOL may be a source of the plutonium in the Clinch River–Watts Bar Reservoir system. Although the amount of plutonium released into the river-reservoir system from the DOE facilities poses no health or environmental risks, it provides a unique tracer for distinguishing contaminants released at some ORNL sites from other contaminant sources.

Studies conducted in Task 7 of BMAP are related to expanded future efforts to evaluate off-site contamination and to identify appropriate remedial measures. These future efforts, for which a RCRA Facility Investigation Plan is currently being prepared, will benefit directly from data provided by Task 7.

10. ABATEMENT PROGRAM

The abatement program of BMAP was designed to provide the means for initiating corrective actions if the biological monitoring identified potential problem areas in meeting the stream-use classification for waters receiving ORNL effluents. Concomitantly, ORNL has a water pollution control effort as a major task within the Environmental Restoration and Facilities Upgrade Program. The objective of this effort is to establish a sound basis for proceeding with the significant long-term commitments that will be required to give ORNL the capability for achieving and maintaining environmental compliance with state and federal water quality regulations. Thus, abatement efforts are focused on providing both short- and long-term management and technical solutions to water quality problems.

As the biological monitoring and toxicity testing progressed through the first year, a number of abatement projects or programs were initiated to address problem areas within the storm sewer system that were identified in the early testing stages (see Table 10.1). The majority of these projects addressed the areas of Fifth Creek and WOC, which were identified in Sect. 3.1.3 of this report as being of particular significance because survival of aquatic test organisms at these locations was greatly reduced. This reduction is attributed to various discharges of chlorine into the receiving streams.

The various abatement projects and programs that have been initiated are discussed in Sects. 10.1 through 10.6.

10.1 COOLING TOWER POLLUTION ELIMINATION PLAN

As discussed in Sect. 3.1.4, variance patterns in the distribution of total residual chlorine and toxicity to both the fathead minnow and *Ceriodaphnia* test species suggest that chlorine is linked to the patterns of ambient toxicity in Fifth Creek and the midreach section of WOC. One of the primary sources is the chlorine used in the cooling tower system. Historically, chlorine was added to cooling towers to prevent growth of algae and bacteria. Effluent from ORNL cooling towers was characterized through a sampling program, a review of operating procedures, and discussions with the operators of cooling towers. There is concern that chlorine concentrations in all cooling tower discharges may have exceeded specified limits. Chlorine is added at concentrations as high as 30 mg/L but is chemically reacted or evaporates during treatment to levels that are acceptable for discharge if the operating procedures, which require blowdown to be discontinued during treatment, are strictly followed. Data collected since the implementation of the NPDES permit in April 1986 indicate that the limits specified in Part III.C.1 have continuously been met, primarily as a result of (1) limiting chlorine use to control algae not controlled with biocides; (2) eliminating the use of DER-KEL DK520, which is toxic to aquatic life, and initiating the use of DK 318, which is nontoxic; (3) discontinuing blowdown during treatment; and (4) implementing administrative controls to verify the correct use of chlorine.

Table 10.1. Required actions for storm sewer system

Functional area	Required actions[a]
Generation	Expense funding Best Management Practices implementations Building inventory Water supply system characterization plan Atlas drawing upgrade and computerization Data base Tracking system
	Capital funding Wastewater Piping Replacement, 4500 area, FY 1987 GPP Wastewater Storm Drain Isolation, FY 1988 GPP
Transfer/control	Expense funding Evaluation of construction piping materials (polyethylene liner) Evaluation of alternatives for contaminated groundwater collection Interface with Remedial Action Program
	Capital funding Process Waste System Inflow/Infiltration, FY 1988 GPP Process Waste Collection System Upgrade, 3000 area, FY 1986 GPP Wastewater Piping Replacement, FY 1987 GPP
Collection	Not applicable
Treatment	Not applicable
Disposal/discharge	Expense funding Sampling programs
Monitoring	Compliance with NPDES permit requirements Interface with Environmental Monitoring

[a]GPP = general plant project; NPDES = National Pollutant Discharge Elimination System.

10.2 WATER SUPPLY SYSTEM CHARACTERIZATION PLAN

An additional, rather undefined, source of potentially toxic effluent is leakage from the water supply system. Significant inputs of chlorine from this source may be contributing to the elevated concentrations at five of the monitoring sites (Fig. 3.6, Sect. 3.1.3.4). Identification of these problem areas has led to the development of a project implementation plan that will be completed during the second quarter of CY 1987. The plan has the following subtasks that lead to a project action plan: (1) system definition, (2) leak detection evaluation, and (3) measuring and monitoring evaluation. The fourth subtask, the project action plan, will incorporate the results of the various investigations and evaluations. A recommendation and cost estimate for implementing an investigation of the water supply system will be completed in June 1987.

10.3 CONTINUOUS INSTREAM CHLORINE MONITORING

As it became evident that chlorine was the principal source of toxicity to the aquatic life of ORNL streams, plans were developed to install a system of instream monitors. The installation of this system will enable ORNL to accumulate real-time effluent concentrations that can be correlated with the results of the toxicity testing effort. It is believed that much of the chlorine toxicity may be the result of episodic events (Sect. 3.1.3.4), and this system would provide a constant tracking of the concentrations in the streams. The monitors are expected to be in place in the second quarter of CY 1987 and will have the capability of measuring chlorine concentrations in the range of 0.04 to 0.25 mg/L.

10.4 CAPITAL PROJECTS

10.4.1 Wastewater Piping Replacement

This capital project is designed to provide modifications to existing piping systems in the 4500S and 3000 areas. Preliminary investigations and sampling during CY 1986 indicated that inappropriate cross-connections between the various collection systems exist that allow process effluents to be discharged untreated to WOC. Building inventories for the 4500S complex and selected buildings within the 3000 (isotopes) area were completed. As a result of these activities, this capital project was defined and includes (1) recycling cooling water in the 4500S complex and (2) connecting Building 3500 to the process waste system.

10.4.2 Process Waste System Inflow/Infiltration and Process Waste Collection System Upgrade

A characterization of discharges from the storm sewer system to receiving streams in support of the NPDES permit application indicated that there were some Category III (containing potentially untreated process waste) wastewaters entering the storm sewer system and subsequently being discharged to the receiving streams. During CY 1986 a plan for lining process waste piping in the 3000 and 4500 areas was completed, and sampling to document prelining conditions was completed. The second phase of sampling will determine if lining process waste piping has reduced the discharge of process waste to the watershed. This phase will be initiated in the third quarter of CY 1987.

10.4.3 Wastewater/Storm Drain Isolation

Outfalls 341 and 342 on First Creek have exhibited higher than expected levels of ^{90}Sr. The data indicate that a source of ^{90}Sr is entering the storm sewer system. The accompanying high concentration of calcium is an indication that the source of the ^{90}Sr may be contaminated groundwater. Monitoring of the data during CY 1986 resulted in development of a plan to determine the source of the radioactivity through building inventories and evaluation of data from groundwater monitoring wells. This capital project will line storm drains and sanitary sewer piping to prevent the discharge of radioactively contaminated groundwater directly to the watershed.

10.5 CLEAN WATER ACT COMPLIANCE STUDY

The purpose of the Clean Water Act Compliance Study is to segregate process waste from storm drainage and sanitary sewage, thus reducing the pollutant loading on the watershed. Segregation of surface water, rainwater runoff, cooling water, and condensation from process waste will reduce the volume of process waste that requires treatment and will improve treatability of process wastewaters. Surface water and rainwater, which do not require treatment, will be discharged directly to the watershed; process waste will be transferred to the appropriate treatment plant.

The first priority in the study is segregation of process waste from storm drainage; the second and third priorities, respectively, are segregation of process waste from sanitary sewage and segregation of nonradioactive process waste and radioactive process waste. Data collected during CY 1986 have led to the initiation of an engineering study for (1) wastewater treatment of effluent from the decontamination laundry (Building 2523) and (2) transfer piping for the nonradioactive process waste from the 3000 area.

10.6 PCB AND MERCURY MONITORING PLANS

As part of the ORNL NPDES permit, plans for identifying and evaluating all sources of PCB and mercury contamination should be prepared. During CY 1986, these plans were developed and transmitted to the Tennessee Department of Health and Environment for review and approval. Implementation of these monitoring plans was initiated during the first quarter of CY 1987. Subsequent direction of abatement projects will be determined by the results of data collected during 1987.

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Appendix A

CONCENTRATIONS OF SOLUBLE METALS IN WHITE OAK CREEK WATERSHED, SEPTEMBER 1986

Table A.1. Concentrations of soluble metals (in milligrams per liter) in White Oak Creek watershed, September 1986

	Site[a]									
	WCK 6.8*	WCK 3.9	WCK 3.4	WCK 2.9	WCK 2.3	MEK 1.8*	MEK 0.6	FCK 1.0*	FFK 1.1*	NTK 1.0*
Soluble Ag	<0.05	<0.05	<0.05	<0.05	<0.05	NA	<0.05	<0.05	<0.05	NA
Soluble Al	<0.2	<0.2	<0.2	<0.2	<0.2	NA	<0.2	<0.2	<0.2	NA
Soluble As	<0.1	<0.1	<0.1	<0.1	<0.1	NA	<0.1	<0.1	<0.1	NA
Soluble B	<0.8	<0.8	<0.8	<0.8	<0.8	NA	<0.8	<0.8	<0.8	NA
Soluble Ba	<0.02	<0.02	<0.02	<0.02	<0.02	NA	<0.02	<0.02	<0.02	NA
Soluble Be	<0.002	<0.002	<0.002	<0.002	<0.002	NA	<0.002	<0.002	<0.002	NA
Soluble Co	<0.01	<0.01	<0.01	<0.01	<0.01	NA	<0.01	<0.01	<0.01	NA
Soluble Ga	<0.03	<0.03	<0.03	<0.03	<0.03	NA	<0.03	<0.03	<0.03	NA
Soluble Li	<0.2	<0.2	<0.2	<0.2	<0.2	NA	<0.2	<0.2	<0.2	NA
Soluble Ni	<0.06	<0.06	<0.06	<0.06	<0.06	NA	<0.06	<0.06	<0.06	NA
Soluble Sb	<0.2	<0.2	<0.2	<0.2	<0.2	NA	<0.2	<0.2	<0.2	NA
Soluble Se	<0.2	<0.2	<0.2	<0.2	<0.2	NA	<0.2	<0.2	<0.2	NA
Soluble Sn	<0.05	<0.05	<0.05	<0.05	<0.05	NA	<0.05	<0.05	<0.05	NA
Soluble Ti	<0.02	<0.02	<0.02	<0.02	<0.02	NA	<0.02	<0.02	<0.02	NA
Soluble V	<0.01	<0.01	<0.01	<0.01	<0.01	NA	<0.01	<0.01	<0.01	NA
Soluble Zr	<0.02	<0.02	<0.02	<0.02	<0.02	NA	<0.02	<0.02	<0.02	NA

[a]Asterisk denotes reference site; NA = not available for September.

Appendix B

MERCURY IN SUNFISH FROM WHITE OAK CREEK WATERSHED AND REFERENCE SITES

Table B.1. Mercury in sunfish from White Oak Creek watershed and reference sites

Site[a]	Dist[b]	Date	Spp[c]	Sex[d]	#	Wgt[e]	Lgth[f]	Hg[g]
WOC 3.5	3.5	11/25/86	BLUGIL	M	7275	253.4	23.0	0.52
WOC 3.5	3.5	11/25/86	BLUGIL	.	7159	123.8	18.0	0.26
WOC 3.5	3.5	11/25/86	BLUGIL	.	7119	143.8	18.5	0.16
WOC 3.5	3.5	11/25/86	BLUGIL	.	8128	125.2	18.3	0.26
WOC 3.5	3.5	11/25/86	BLUGIL	.	7175	119.0	18.0	0.22
WOC 3.5	3.5	11/25/86	BLUGIL	.	7197	161.1	18.7	0.14
WOC 3.5	3.5	11/25/86	BLUGIL	.	7597	67.6	14.8	0.21
WOC 3.5	3.5	11/25/86	BLUGIL	.	7595	47.7	13.4	0.20
WOC 2.9	2.9	12/03/86	BLUGIL	M	7554	137.1	19.3	0.52
WOC 2.9	2.9	12/03/86	BLUGIL	M	7539	93.9	16.7	0.40
WOC 2.9	2.9	12/03/86	BLUGIL	M	7538	93.0	16.6	0.30
WOC 2.9	2.9	12/03/86	BLUGIL	M	7519	49.9	14.2	0.38
WOC 2.9	2.9	12/03/86	REDBRE	M	9025	157.9	20.0	0.37
WOC 2.9	2.9	12/03/86	REDBRE	F	9029	85.3	16.6	0.58
WOC 2.9	2.9	12/03/86	REDBRE	M	9023	139.2	19.6	0.46
WOC 2.9	2.9	12/03/86	REDBRE	M	9024	167.7	20.3	0.48
WOC 2.3	2.3	2/03/87	BLUGIL	M	0008	58.0	15.1	0.40
WOC 2.3	2.3	2/03/87	BLUGIL	M	0077	162.5	21.5	0.73
WOC 2.3	2.3	2/03/87	BLUGIL	M	0067	49.8	14.4	0.20
WOC 2.3	2.3	2/03/87	BLUGIL	M	0088	68.6	15.4	0.43
WOC 2.3	2.3	2/03/87	REDBRE	M	0015	78.9	17.1	0.66
WOC 2.3	2.3	2/03/87	REDBRE	M	0106	68.9	16.6	0.64
WOC 2.3	2.3	2/03/87	REDBRE	M	0081	55.5	15.5	0.50
WOC 2.3	2.3	2/03/87	REDBRE	M	0104	42.6	13.5	0.38
MBR 0.2	0.2	12/04/86	BLUGIL	M	9035	30.0	12.0	0.11
MBR 0.2	0.2	12/04/86	BLUGIL	M	9036	82.2	16.1	0.08
MBR 0.2	0.2	12/04/86	BLUGIL	M	7469	27.1	11.7	0.11
MBR 0.2	0.2	2/05/87	BLUGIL	M	0071	28.7	12.0	0.10
MBR 0.2	0.2	2/05/87	BLUGIL	M	0017	12.0	9.4	0.14
MBR 0.2	0.2	2/05/87	BLUGIL	M	0085	9.5	8.5	0.14
MBR 0.2	0.2	2/05/87	REDBRE	M	0075	12.0	8.6	0.13
MBR 0.2	0.2	2/05/87	BLUGIL	M	0051	51.0	14.4	0.20
WOL1.0	1.0	12/05/86	BLUGIL	M	6933	65.7	15.3	0.09
WOL 1.0	1.0	12/05/86	BLUGIL	M	6950	113.1	17.7	0.09
WOL 1.0	1.0	12/05/86	BLUGIL	M	9037	101.8	17.3	0.07
WOL 1.0	1.0	12/05/86	BLUGIL	M	6968	61.4	14.5	0.09
WOL 1.0	1.0	12/05/86	BLUGIL	F	6997	56.3	14.7	0.11
WOL 1.0	1.0	12/05/86	REDBRE	M	9045	85.0	17.0	0.27
WOL 1.0	1.0	12/05/86	REDBRE	M	9043	68.7	15.3	0.18
WOL 1.0	1.0	12/05/86	REDBRE	M	9042	60.6	15.0	0.35
NWT 0.2	0.2	12/11/86	BLUGIL	M	6905	111.2	17.6	0.19
NWT 0.2	0.2	12/11/86	BLUGIL	M	6985	158.0	21.5	0.30
NWT 0.2	0.2	12/11/86	BLUGIL	M	6927	93.8	18.0	0.27

Table B.1 (continued)

Site[a]	Dist[b]	Date	Spp[c]	Sex[d]	#	Wgt[e]	Lgth[f]	Hg[g]
NWT 0.2	0.2	12/11/86	BLUGIL	M	7598	108.4	17.4	0.26
NWT 0.2	0.2	12/11/86	BLUGIL	M	7576	117.3	19.0	0.32
NWT 0.2	0.2	12/11/86	BLUGIL	F	7579	67.0	15.4	0.29
NWT 0.2	0.2	12/11/86	BLUGIL	M	7577	67.1	15.4	0.19
NWT 0.2	0.2	12/11/86	BLUGIL	M	7592	38.3	13.4	0.29
HINDSCR	.	1/08/86	BLUGIL	M	6900	82.3	16.4	0.05
HINDSCR	.	1/08/86	BLUGIL	M	6907	49.4	14.4	0.05
HINDSCR	.	1/08/86	BLUGIL	F	6916	56.5	14.9	0.06
HINDSCR	.	1/08/86	BLUGIL	M	6919	65.2	14.8	0.09
HINDSCR	.	1/08/86	BLUGIL	M	6930	83.5	16.9	0.05
HINDSCR	.	1/08/86	BLUGIL	F	6935	49.5	14.1	0.09
HINDSCR	.	1/08/86	BLUGIL	M	6970	108.1	17.9	0.06
HINDSCR	.	1/08/86	BLUGIL	M	6995	85.2	16.5	0.05
BRUSHY	.	12/10/85	REDBRE	M	8126	122.4	18.7	0.06
BRUSHY	.	12/10/85	REDBRE	F	8136	60.0	15.3	0.08
BRUSHY	.	12/10/85	REDBRE	M	8161	83.5	17.3	0.24
BRUSHY	.	12/10/85	REDBRE	M	8170	70.6	15.6	0.09
BRUSHY	.	12/10/85	REDBRE	M	8172	47.9	14.3	0.04
BRUSHY	.	12/10/85	REDBRE	F	8179	53.4	14.6	0.07
BRUSHY	.	12/10/85	REDBRE	F	8187	45.0	14.1	0.09
BRUSHY	.	12/10/85	REDBRE	M	8188	113.2	17.8	0.09

[a]WOL = White Oak Lake; WOC = White Oak Creek; MBR = Melton; Branch; NWT = Northwest Tributary; HINDSCR = Hinds Creek, Anderson County, Tennessee; BRUSHY = Brushy Fork, Anderson County, Tennessee.

[b]Distance (in kilometers) upstream from confluence with larger stream (i.e., WOC 3.5 is White Oak Creek 3.5 km upstream from the Clinch River).

[c]Species: BLUGIL = bluegill sunfish (*Lepomis macrochirus*); REDBRE = redbreast sunfish (*Lepomis auritus*).

[d]M = male; F = female.

[e]Fish weight, in grams.

[f]Fish total length, in centimeters.

[g]Total mercury in fish axial muscle, in micrograms per gram wet weight.

Appendix C

QUALITY ASSURANCE RESULTS FOR FISH BIOACCUMULATION STUDIES

Appendix C

QUALITY ASSURANCE RESULTS FOR FISH BIOACCUMULATION STUDIES

C.1 MERCURY

The results of analysis of eight pairs of blind duplicate fish samples for total mercury indicated low variability in the analytical procedure. The mean difference between duplicate samples was 0.014 ppm, and the mean standard deviation among replicates was 0.020 ppm. The mean coefficient of variation was 6.4%. The results of analyses of reference fish samples obtained from the U.S. Environmental Protection Agency (EPA) averaged 2.43 ± 0.12 ppm (mean ± 1 s.d., n = 9) vs an expected result of 2.52 ppm ; recovery was 96 ± 5 %. Mercury levels measured in fish from the uncontaminated reference site (Hinds Creek) were typical of background levels in stream fish, averaging 0.06 ± 0.02 ppm (mean ± 1 s.d., n = 9).

C.2 PCBs

The results of analyses of nine pairs of blind duplicate fish samples for polychlorinated biphenyls (PCBs) were much more variable than results of mercury analyses. The mean difference and standard deviation between duplicates was 0.22 ppm total PCBs, with a coefficient of variation of 20.6%. Measurement of PCB-1254 was more variable than measurement of PCB-1260, with mean differences between duplicates of 0.18 and 0.11 ppm respectively. Standard deviations were 0.16 and 0.10 ppm, while mean coefficients of variation were 30% and 17% for PCB-1254 and PCB-1260 respectively. Samples of uncontaminated carp (Hinds Creek) were spiked at 1 ppm each PCB-1254 and PCB-1260 and analyzed along with catfish samples. Mean recoveries were good, but variability was comparable to that observed in the analysis of blind duplicates. Recoveries averaged 104 ± 29%, 95 ± 40%, and 114 ± 29% (mean ± 1 s.d.) for total PCBs, PCB-1254, and PCB-1260 respectively. Samples of PCB-contaminated carp were homogenized and split for analysis by the Oak Ridge National Laboratory Analytical Chemistry Division (ACD) and the EPA Environmental Services Laboratory, Athens, Georgia. The mean levels of total PCBs, PCB-1254, and PCB-1260 did not differ significantly in the five samples analyzed by the two laboratories ($p > 0.05$), but results obtained by the EPA laboratory were higher, averaging 0.80 vs 0.48, 0.39 vs 0.17, and 0.41 vs 0.31 for total PCBs, PCB-1254, and PCB-1260 respectively. Additional portions of those same fish were sent to another ACD laboratory for dual-column gas chromatographic PCB analysis. Results of those analyses have not yet been received. Samples of carp known to contain only very low levels of PCBs were analyzed as analytical controls. Results reported for these five analyses were all at or below limits of detection (0.1 ppm each for PCB-1254 and PCB-1260).

Appendix D

CONCENTRATIONS OF PCBs AND ^{90}Sr IN WHITE OAK CREEK AND CLINCH RIVER CATFISH

Table D.1. Concentrations of polychlorinated biphenyls (PCBs) and ^{90}Sr in White Oak Creek and Clinch River catfish

Site[a]	Date	Spp[b]	Sex[c]	#	Wgt[d]	Lgth[e]	ΣPCB[f]	1254[g]	1260[h]	^{90}Sr[i]
WOL	8/07/86	Y.BULL	F	1	434	28.6	0.59	0.29	0.30	4500
WOL	8/07/86	Y.BULL	M	2	364	29.0	0.62	0.33	0.29	4300
WOL	8/07/86	B.BULL	M	3	415	29.8	0.90	0.46	0.44	5500
WOL	8/08/86	Y.BULL	M	4	202	24.8	1.76	0.96	0.80	4100
WOL	8/15/86	Y.BULL	F	5	500	31.0	1.00	0.55	0.45	3700
WCK0.9	8/06/86	CH.CAT	F	1	1168	49.6	0.91	0.59	0.32	1400
WCK 0.9	8/08/86	CH.CAT	M	2	594	42.4	0.68	0.38	0.30	670
WCK 0.3	8/06/86	CH.CAT	F	1	942	44.0	1.18	0.51	0.67	330
WCK 0.3	8/06/86	CH.CAT	F	2	926	45.7	2.12	1.61	0.51	1500
WCK 0.3	8/06/86	CH.CAT	F	3	836	44.5	1.70	1.01	0.69	440
WCK 0.3	8/06/86	CH.CAT	M	4	636	43.5	1.28	0.80	0.48	120
WCK 0.3	8/07/86	CH.CAT	F	5	734	42.6	2.62	1.7	0.92	1300
WCK 0.3	8/08/86	CH.CAT	M	6	668	41.8	1.33	0.80	0.53	410
WCK 0.3	8/11/86	CH.CAT	M	7	884	45.6	2.06	1.31	0.75	290
WCK 0.3	8/12/86	CH.CAT	M	8	877	45.2	0.32	0.09	0.23	820
WCK 0.3	8/12/86	CH.CAT	M	9	548	40.5	0.64	0.37	0.27	870
WCK 0.3	8/13/86	CH.CAT	M	10	436	36.0	0.73	0.46	0.27	.
CRK 33.8	7/23/86	CH.CAT	M	1	1026	47.9	0.16	0.07	0.09	430
CRK 33.8	7/23/86	CH.CAT	F	2	2058	56.2	0.59	0.22	0.37	650
CRK 33.8	7/23/86	CH.CAT	M	3	2122	57.5	0.19	0.12	0.07	.
CRK 33.8	7/23/86	CH.CAT	M	4	2242	57.0	0.10	0.02	0.08	990
CRK 33.8	7/23/86	CH.CAT	M	5	766	43.0	1.35	0.46	0.89	1200
CRK 33.8	7/31/86	CH.CAT	M	6	1358	49.5	0.86	0.42	0.44	630
CRK 33.8	8/13/86	CH.CAT	M	7	958	47.0	0.76	0.10	0.66	350
CRK 33.8	9/05/86	CH.CAT	M	8	700	44.2	0.92	0.10	0.82	690
CRK 34.8	7/23/86	CH.CAT	M	1	1125	49.0	0.27	0.19	0.08	500
CRK 34.8	7/23/86	CH.CAT	M	2	968	46.3	1.44	0.44	1.0	130
CRK 34.8	8/05/86	CH.CAT	M	3	679	47.2	1.57	0.80	0.77	40
CRK 34.8	8/12/86	CH.CAT	M	4	865	46.5	1.32	0.79	0.53	1100
CRK 34.8	8/12/86	CH.CAT	M	5	634	43.1	1.05	0.57	0.48	90
CRK 34.8	8/12/86	CH.CAT	M	6	570	41.6	0.58	0.33	0.25	430
CRK 34.8	8/12/86	CH.CAT	M	7	466	37.9	0.50	0.22	0.28	30
CRK 34.8	8/12/86	CH.CAT	M	8	442	39.7	1.84	1.1	0.74	38
CRK 34.8	8/12/86	CH.CAT	M	9	370	36.0	.	.	.	56
CRK 34.8	8/12/86	CH.CAT	M	10	266	32.0	.	.	.	48
CRK 34.8	8/12/86	BL.CAT	M	11	348	36.4	.	.	.	100
CRK 33.2	7/29/86	CH.CAT	M	1	910	46.0	1.06	0.49	0.57	2300
CRK 33.2	7/30/86	CH.CAT	M	2	567	39.8	0.52	0.30	0.22	1400
CRK 33.2	7/30/86	CH.CAT	M	3	568	40.0	0.61	0.35	0.26	2500
CRK 33.2	7/30/86	CH.CAT	M	4	658	44.0	2.4	0.1	2.3	90
CRK 33.2	7/30/86	CH.CAT	M	5	536	36.0	0.39	0.25	0.14	710
CRK 33.2	7/30/86	CH.CAT	M	6	638	40.4	0.74	0.44	0.30	1900

Table D.1 (continued)

Site[a]	Date	Spp[b]	Sex[c]	#	Wgt[d]	Lgth[e]	ΣPCB[f]	1254[g]	1260[h]	^{90}Sr[i]
CRK 32.2	8/12/86	BL.CAT	M	1	866	45.9	.	.	.	90
CRK 32.2	8/12/86	BL.CAT	M	2	957	45.7	.	.	.	120
CRK 32.2	8/12/86	FH.CAT	M	3	319	33.5	.	.	.	120
CRK 32.2	8/13/86	CH.CAT	M	4	1110	50.2	0.84	0.17	0.67	120
CRK 32.2	8/13/86	CH.CAT	M	5	568	41.5	0.47	0.17	0.30	91
CRK 32.2	8/13/86	CH.CAT	F	6	815	45.0	1.53	0.23	1.3	34
CRK 32.2	8/13/86	CH.CAT	M	7	796	45.0	1.06	0.43	0.63	17
CRK 32.2	8/13/86	CH.CAT	F	8	606	42.0	0.95	0.33	0.62	31
CRK 32.2	8/13/86	CH.CAT	M	9	434	35.0	1.88	1.6	0.28	35
CRK 32.2	8/13/86	CH.CAT	M	10	502	39.0	0.57	0.10	0.47	110
CRK 32.2	8/13/86	CH.CAT	M	11	472	38.5	0.80	0.31	0.49	15
CRK 30.2	7/29/86	CH.CAT	M	1	1722	56.8	0.88	0.43	0.45	15
CRK 30.2	7/31/86	CH.CAT	M	2	757	45.0	0.72	0.33	0.39	37
CRK 30.2	7/31/86	CH.CAT	M	3	554	37.9	0.90	0.35	0.55	25
CRK 30.2	8/01/86	CH.CAT	M	4	490	42.3	0.44	0.10	0.34	23
CRK 30.2	8/05/86	CH.CAT	F	5	1026	46.5	2.36	0.56	1.8	14
CRK 30.2	8/05/86	CH.CAT	M	6	829	44.0	0.91	0.11	0.80	16
CRK 30.2	8/06/86	CH.CAT	M	7	468	38.2	0.52	0.26	0.26	19
CRK 30.2	8/12/86	CH.CAT	M	8	459	40.0	0.32	0.16	0.16	30
CRK 30.2	8/12/86	CH.CAT	F	9	334	36.2	.	.	.	11
CRK 40	7/23/86	CH.CAT	M	1	634	41.4	0.80	0.01	0.79	1
CRK 40	7/23/86	CH.CAT	M	2	1362	52.5	0.26	0.09	0.17	15
CRK 40	7/30/86	CH.CAT	M	3	1106	50.5	0.43	0.10	0.33	44
CRK 40	8/15/86	CH.CAT	M	5	302	34.9	0.52	0.29	0.23	18
CRK 40	8/15/86	CH.CAT	M	6	412	38.3	0.20	0.10	0.10	1
CRK 40	8/15/86	CH.CAT	F	7	856	44.0	0.54	0.22	0.32	3

[a]WOL = White Oak Lake; WCK = White Oak Creek kilometer; CRK = Clinch River kilometer. CRK 40 is located in Melton Hill Reservoir.

[b]Species: CH.CAT = channel catfish (*Ictalurus punctatus*); Y.BULL = yellow bullhead (*Ictalurus natalis*); B.BULL = black bullhead (*Ictalurus melas*); BL.CAT = blue catfish (*Ictalurus furcatus*); FH.CAT = flathead catfish (*Pylodictus olivaris*).

[c]M = male; F = female.

[d]Fish weight, in grams.

[e]Fish total length, in centimeters.

[f]Total PCBs (sum of PCB-1254 and PCB-1260) in fish axial muscle, in micrograms per gram wet weight.

[g]PCB-1254 (Arochlor-1254) in fish axial muscle, in micrograms per gram wet weight.

[h]PCB-1260 (Arochlor-1260) in fish axial muscle, in micrograms per gram wet weight.

[i]Strontium-90 in catfish vertebrae, in becquerels per kilogram dry weight.

Appendix E

FISH DENSITY, BIOMASS, AND CONDITION FACTOR DATA COLLECTED IN WHITE OAK CREEK WATERSHED FROM AUGUST 1985 TO SEPTEMBER 1986

Table E.1. Fish densities (number of fish per square meter) for August–September 1985 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Species	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	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	0.10	-	NF	NF	NS	-	NF	-	-	0.02	NS	0.07	-	0.04	<0.01	-	-
Largemouth bass	-	-	-	-	-	-	-	-	-	0.01	-	<0.01	-	-	-	-	-
Redbreast sunfish	-	-	-	-	-	0.12	-	-	-	-	-	0.13	<0.01	0.03	-	-	-
Rock bass	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Warmouth	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Cottidae																	
Banded sculpin	-	0.02	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.52
Cyprinidae																	
Blacknose dace	2.27	4.08	-	-	-	<0.01	-	0.63	3.61	0.26	-	-	-	-	-	5.19	2.66
Central stoneroller	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4.28	0.03
Creek chub	0.05	0.04	-	-	-	0.30	-	0.54	1.78	-	-	-	-	-	-	0.64	0.05
Fathead minnow	0.03	-	-	-	-	-	-	-	-	0.01	-	-	<0.01	<0.01	<0.01	-	-
Ictaluridae																	
Black bullhead	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Poeciliidae																	
Mosquitofish	0.04	-	-	-	-	-	-	-	-	0.37	-	0.06	0.03	0.13	-	-	-
Number of species (N)	5	3	NF	NF	NS	3	NF	2	2	5	NS	7	3	4	2	3	4
Summed density[b]	2.49	4.14	NF	NF	NS	0.43	NF	1.17	5.39	0.67	NS	0.28	0.04	0.20	<0.01	10.11	3.26
Total density[c]	2.48	4.14	NF	NF	NS	0.43	NF	1.17	5.40	0.68	NS	0.28	0.03	0.20	<0.01	10.12	3.26

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; NF = no fish taken in sample; NS = site not sampled at this date.

[b]Sum of $\hat{N}$ for each species.

[c]From $\hat{N}$, all species combined.

Table E.2. Fish densities (number of fish per square meter) for April–May 1986 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Species	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	ECK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Centrarchidae																	
Bluegill	0.06	-	NF	-		-	NF		-	0.09	-	0.07	<0.01	0.13	-	-	-
Redbreast sunfish	-	-	-	-	-	0.18	-	-	-	-	-	0.12	-	0.01	-	-	-
Rock bass	-	-	-	-	-	-	-	-	-	-	-	0.01	-	-	-	-	-
Spotted bass	-	-		-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Warmouth	-	-	-	-	-	-	-	-	-	-	-	0.03	-	-	-	-	-
Cottidae																	
Banded sculpin	-	0.02	-	0.87	2.88	-	-	-	-	-	-	-	-	-	-	-	0.26
Cyprinidae																	
Blacknose dace	0.65	2.06	-	5.43	1.20	0.01	-	0.47	1.89	0.15	1.29	-	<0.01	<0.01	-	4.07	0.89
Central stoneroller	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.78	0.01
Creek chub	0.01	0.06	-	-	-	0.04	-	0.19	0.63	0.01	0.13	-	-	0.01	-	1.21	0.04
Fathead minnow	0.57	-	-	-	-	-	-	-	-	0.13	0.07	-	0.23	0.01	<0.01	0.06	-
Ictaluridae																	
Yellow bullhead	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Poeciliidae																	
Mosquitofish	0.01	-	-	-	-	-	-	-	-	0.50	-	0.01	0.05	0.02	-	-	-
Number of species (N)	5	3	NF	2	2	3	NF	2	2	5	3	7	4	6	1	4	4
Summed density[b]	1.30	2.14	NF	6.30	4.08	0.23	NF	0.66	2.52	0.88	1.49	0.25	0.30	0.18	<0.01	8.12	1.20
Total density[c]	1.31	2.14	NF	6.29	4.08	0.23	NF	0.65	2.52	0.88	1.49	0.24	0.29	0.18	<0.01	8.12	1.20

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; NF = no fish taken in sample.

[b]Sum of $\hat{N}$ for each species.

[c]From $\hat{N}$, all species combined.

Table E.3. Fish densities (number of fish per square meter) for August–September 1986 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Species	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	ECK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Centrarchidae																	
Bluegill	0.02	-	NF	-	-	-	NF	NS	DS	0.08	0.07	0.27	<0.01	0.16	<0.01	-	-
Largemouth bass	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Redbreast sunfish	-	-	-	-	-	0.03	-	-	-	-	-	0.16	0.18	0.51	-	-	-
Rock bass	-	-	-	-	-	-	-	-	-	-	-	0.01	-	-	-	-	-
Warmouth	-	-	-	-	-	-	-	-	-	-	-	0.06	-	-	-	-	-
Clupeidae																	
Gizzard shad	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Cottidae																	
Banded sculpin	-	-	-	1.05	1.44	-	-	-	-	-	-	-	-	-	-	-	0.11
Cyprinidae																	
Carp	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Blacknose dace	1.94	5.47	-	9.78	1.35	0.02	-	-	-	0.23	0.25	-	0.05	0.19	0.01	7.35	2.32
Central stoneroller	-	-	-	-	-	-	-	-	-	-	-	0.01	-	-	-	7.78	0.05
Creek chub	0.06	0.04	-	-	-	0.03	-	-	-	-	0.06	-	<0.01	-	<0.01	0.53	0.18
Fathead minnow	0.14	-	-	-	-	-	-	-	-	0.46	0.03	-	0.04	0.01	<0.01	0.03	-
Ictaluridae																	
Yellow bullhead	-	-	-	-	-	-	-	-	-	-	-	<0.01	-	-	-	-	-
Poeciliidae																	
Mosquitofish	0.73	-	-	-	-	-	-	-	-	3.07	0.04	0.48	1.09	0.90	<0.01	-	-
Number of species (N)	5	2	NF	2	2	3	NF	NS	DS	4	5	10	6	5	5	4	4
Summed density[b]	2.91	5.51	NF	10.83	2.79	0.08	NF	NS	DS	3.84	0.45	1.00	1.37	1.77	0.02	15.69	2.66
Total density[c]	2.89	5.50	NF	10.83	2.79	0.07	NF	NS	DS	3.84	0.44	1.00	1.36	1.77	0.02	15.71	2.66

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer, MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; NF = no fish taken in sample; NS = site not sampled at this date; DS = site dry at time of sampling.

[b]Sum of $\hat{N}$ for each species.

[c]From $\hat{N}$, all species combined.

Table E.4. Fish biomass (grams per square meter) for August–September 1985 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Species	Site[a] FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	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	0.19	-	NF	NF	NS	-	NF	-	-	0.06	NS	0.58	-	2.27	0.05	-	-
Largemouth bass	-	-	-	-	-	-	-	-	-	0.04	-	1.96	-	-	-	-	-
Redbreast sunfish	-	-	-	-	-	1.31	-	-	-	-	-	4.52	0.34	2.18	-	-	-
Rock bass	-	-	-	-	-	-	-	-	-	-	-	0.06	-	-	-	-	-
Warmouth	-	-	-	-	-	-	-	-	-	-	-	0.10	-	-	-	-	-
Cottidae																	
Banded sculpin	-	0.22	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.48
Cyprinidae																	
Blacknose dace	5.45	4.08	-	-	-	<0.01	-	0.32	0.36	0.65	-	-	-	-	-	4.16	3.93
Central stoneroller	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14.10	0.12
Creek chub	0.38	0.15	-	-	-	2.09	-	0.16	0.54	-	-	-	-	-	-	1.93	0.29
Fathead minnow	0.09	-	-	-	-	-	-	-	-	0.03	-	-	<0.01	<0.01	<0.01	-	-
Ictaluridae																	
Black bullhead	-	-	-	-	-	-	-	-	-	-	-	0.01	-	-	-	-	-
Poeciliidae																	
Mosquitofish	0.02	-	-	-	-	-	-	-	-	0.22	-	0.01	0.01	0.05	-	-	-
Total biomass	6.13	4.45	NF	NF	NS	3.40	NF	0.48	0.90	1.00	NS	7.24	0.36	4.50	0.05	20.19	6.82

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; NF = no fish taken in sample; NS = site not sampled at this date.

Table E.5. Fish biomass (grams per square meter) for April–May 1986 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Species	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	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	0.65	-	NF	-	-	-	NF	-	-	0.10	-	0.87	0.76	8.09	-	-	-
Redbreast sunfish	-	-	-	-	-	1.95	-	-	-	-	-	3.47	-	0.64	-	-	-
Rock bass	-	-	-	-	-	-	-	-	-	-	-	0.18	-	-	-	-	-
Spotted bass	-	-	-	-	-	-	-	-	-	-	-	0.13	-	-	-	-	-
Warmouth	-	-	-	-	-	-	-	-	-	-	-	0.21	-	-	-	-	-
Cottidae																	
Banded sculpin	-	0.21	-	5.30	11.82	-	-	-	-	-	-	-	-	-	-	-	1.60
Cyprinidae																	
Blacknose dace	2.42	2.47	-	16.28	3.12	0.06	-	0.51	1.32	0.46	2.58	-	0.03	<0.01	-	6.11	1.15
Central stoneroller	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	20.0	0.12
Creek chub	0.08	0.69	-	-	-	0.39	-	0.24	1.08	0.07	0.17	-	-	0.02	-	7.28	0.12
Fathead minnow	0.74	-	-	-	-	-	-	-	-	0.21	0.18	-	0.42	0.04	<0.01	0.17	-
Ictaluridae																	
Yellow bullhead	-	-	-	-	-	-	-	-	-	-	-	0.03	-	-	-	-	-
Poeciliidae																	
Mosquitofish	0.03	-	-	-	-	-	-	-	-	0.25	-	<0.01	0.03	0.01	-	-	-
Total biomass	3.92	3.37	NF	21.58	14.94	2.40	NF	0.75	2.40	1.09	2.93	4.90	1.24	8.80	<0.01	33.56	2.99

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; NF = no fish taken in sample.

Table E.6. Fish biomass (grams per square meter) for August–September 1986 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Species	Site[a]																
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 1.5	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	-	NF	-	-	-	NF	NS	DS	0.05	0.20	1.38	0.40	9.21	0.25	-	-
Largemouth bass	-	-	-	-	-	-	-	-	-	-	-	0.40	-	-	-	-	-
Redbreast sunfish	-	-	-	-	-	0.13	-	-	-	-	-	4.32	1.66	6.57	-	-	-
Rock bass	-	-	-	-	-	-	-	-	-	-	-	0.28	-	-	-	-	-
Warmouth	-	-	-	-	-	-	-	-	-	-	-	0.66	-	-	-	-	-
Clupeidae																	
Gizzard shad	-	-	-	-	-	-	-	-	-	-	-	0.43	-	-	-	-	-
Cottidae																	
Banded sculpin	-	-	-	4.32	4.33	-	-	-	-	-	-	-	-	-	-	-	0.56
Cyprinidae																	
Carp	-	-	-	-	-	-	-	-	-	-	-	NW	-	-	-	-	-
Blacknose dace	1.75	5.47	-	13.7	2.02	0.04	-	-	-	0.66	0.35	-	0.10	0.29	0.02	3.67	3.94
Central stoneroller	-	-	-	-	-	-	-	-	-	-	-	0.03	-	-	-	20.2	0.08
Creek chub	0.56	0.78	-	-	-	0.40	-	-	-	-	0.59	-	0.02	-	0.07	1.16	0.79
Fathead minnow	0.17	-	-	-	-	-	-	-	-	0.41	0.02	-	0.13	0.03	<0.01	0.13	-
Ictaluridae																	
Yellow bullhead	-	-	-	-	-	-	-	-	-	-	-	0.44	-	-	-	-	-
Poeciliidae																	
Mosquitofish	0.22	-	-	-	-	-	-	-	-	0.61	0.02	0.14	0.33	0.27	<0.01	-	-
Total biomass	2.72	6.25	NF	18.02	6.35	0.57	NF	NS	DS	1.73	1.18	8.08	2.64	16.37	0.34	25.16	5.37

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; NF = no fish taken in sample; NS = site not sampled at this date; DS = site dry at time of sampling; NW = fish not weighed.

Table E.7. 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 August–September 1985

Species	Sites[a]							
Bluegill	WCK 3.4	WCK 3.9	FCK 0.1	WCK 2.3	NTK 0.3			
	n = 15	n = 2	n = 7	n = 36	n = 2			
	(1.9941)	(1.7911)	(1.4071)	(1.3696)	(1.3663)			
Blacknose dace	MEK 0.6	MEK 2.1	NTK 0.3	FCK 0.1	MEK 1.5	WCK 5.1	FCK 0.8	WCK 6.8
	n = 1	n = 14	n = 36	n = 99	n = 63	n = 108	n = 92	n = 137
	(1.1801)	(1.0286)	(0.9305)	(0.9161)	(0.8892)	(0.8554)	(0.7947)	(0.7732)
Creek chub	FCK 0.1	FCK 0.8	MEK 0.6	MEK 2.1	WCK 6.8	WCK 5.1	MEK 1.5	
	n = 4	n = 1	n = 39	n = 7	n = 4	n = 42	n = 45	
	(1.0828)	(1.0496)	(1.0187)	(0.8743)	(0.8053)	(0.800)	(0.7869)	
Mosquitofish	WCK 3.4	WCK 2.9	NTK 0.3	FCK 0.1	WCK 2.3			
	n = 59	n = 14	n = 45	n = 2	n = 21			
	(1.0601)	(0.9995)	(0.9983)	(0.8231)	(0.6244)			
Redbreast sunfish	WCK 2.9	WCK 2.3	WCK 3.4	MEK 0.6				
	n = 2	n = 63	n = 12	n = 18				
	(2.1589)	(1.7526)	(1.6840)	(1.4511)				
Sculpin	FCK 0.8	WCK 6.8						
	n = 1	n = 50						
	(1.3741)	(0.0906)						

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; n = number of fish sampled and weighed. Values connected by the same line are not significantly different ($p > 0.05$), based on Tukey's studentized range (HSD) test.

Table E.8. 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–May 1986

Species	Sites[a]						
Bluegill	WCK 2.9	WCK 3.4	WCK 2.3	FCK 0.1	NTK 0.3		
	$n = 1$	$n = 54$	$n = 28$	$n = 3$	$n = 14$		
	(2.4992)	(2.1078)	(1.6568)	(1.6365)	(1.3280)		
Blacknose dace	FCK 0.1	WCK 2.9	WCK 5.1	NTK 1.0	MEK 1.5	WCK 3.4	MEK 0.6
	$n = 46$	$n = 2$	$n = 123$	$n = 86$	$n = 77$	$n = 1$	$n = 1$
	(1.2272)	(1.1457)	(1.1253)	(1.0716)	(1.0341)	(1.0251)	(1.0115)
	FFK 0.4	MEK 2.1	NTK 0.3	FFK 1.0	FCK 0.8	WCK 6.8	
	$n = 169$	$n = 62$	$n = 25$	$n = 31$	$n = 76$	$n = 97$	
	(1.0053)	(0.9600)	(0.9482)	(0.9355)	(0.8861)	(0.8121)	
Creek chub	MEK 0.6	FCK 0.8	FCK 0.1	WCK 5.1	NTK 1.0	NTK 0.3	
	$n = 6$	$n = 2$	$n = 1$	$n = 78$	$n = 8$	$n = 1$	
	(1.2285)	(1.1662)	(1.1358)	(1.0773)	(1.0699)	(1.0450)	
	MEK 1.5	WCK 3.4	MEK 2.1	WCK 6.8			
	$n = 35$	$n = 3$	$n = 42$	$n = 5$			
	(1.0450)	(1.0205)	(1.0089)	(0.9490)			
Fathead minnow	WCK 3.4	WCK 5.1	WCK 2.9	NTK 1.0	FCK 0.1	WCK 3.9	NTK 0.3
	$n = 5$	$n = 5$	$n = 93$	$n = 4$	$n = 40$	$n = 1$	$n = 22$
	(1.1976)	(1.0758)	(1.0607)	(0.9873)	(0.9636)	(0.9448)	(0.9097)
Mosquitofish	FCK 0.1	WCK 2.3	WCK 3.4	WCK 2.9	NTK 0.3		
	$n = 1$	$n = 4$	$n = 5$	$n = 25$	$n = 39$		
	(1.2800)	(1.2321)	(1.1868)	(1.1374)	(1.0154)		
Redbreast sunfish	WCK 3.4	MEK 0.6	WCK 2.3				
	$n = 2$	$n = 27$	$n = 51$				
	(2.2189)	(1.9013)	(1.8649)				
Stoneroller	WCK 5.1	WCK 6.8					
	$n = 166$	$n = 2$					
	(1.0702)	(0.9793)					
Sculpin	FFK 0.4	FCK 0.8	WCK 6.8	FFK 1.0			
	$n = 39$	$n = 1$	$n = 32$	$n = 68$			
	(1.3289)	(1.3137)	(1.1588)	(1.1109)			

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; n = number of fish sampled and weighed. Values connected by the same line are not significantly different ($p > 0.05$), based on Tukey's studentized range (HSD) test.

Table E.9. 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 August–September 1986

Species	Sites[a]										
Bluegill	WCK 3.9 $n = 1$ (2.5391)	WCK 2.9 $n = 1$ (2.1722)	WCK 3.4 $n = 10$ (1.7622)	WCK 2.3 $n = 86$ (1.4218)	NTK 1.0 $n = 4$ (1.3698)	FCK 0.1 $n = 1$ (1.2328)	NTK 0.3 $n = 11$ (1.1229)				
Blacknose dace	NTK 0.3 $n = 32$ (1.0126)	WCK 3.9 $n = 5$ (0.9846)	FCK 0.8 $n = 118$ (0.9745)	WCK 2.9 $n = 19$ (0.9672)	NTK 1.0 $n = 17$ (0.9637)	WCK 3.4 $n = 70$ (0.9546)	FCK 0.1 $n = 75$ (0.9315)	MEK 0.6 $n = 2$ (0.9265)	WCK 5.1 $n = 140$ (0.9260)	FFK 0.4 $n = 148$ (0.9235)	WCK 6.8 $n = 135$ (0.8771)	FFK 1.0 $n = 40$ (0.8482)
Creek chub	MEK 0.6 $n = 3$ (1.2291)	NTK 1.0 $n = 3$ (1.1824)	WCK 3.9 $n = 2$ (1.0896)	FCK 0.1 $n = 3$ (1.0595)	FCK 0.8 $n = 2$ (1.0418)	WCK 2.9 $n = 1$ (0.9269)	WCK 5.1 $n = 37$ (0.9206)	WCK 6.8 $n = 18$ (0.8964)			
Fathead minnow	WCK 2.9 $n = 18$ (1.0397)	WCK 5.1 $n = 1$ (1.0076)	NTK 0.3 $n = 56$ (0.9781)	WCK 3.4 $n = 3$ (0.9722)	NTK 1.0 $n = 1$ (0.9375)	WCK 3.9 $n = 1$ (0.9046)	FCK 0.1 $n = 8$ (0.8405)				
Mosquitofish	WCK 2.3 $n = 57$ (1.1198)	NTK 0.3 $n = 78$ (1.0614)	NTK 1.0 $n = 2$ (1.0434)	WCK 2.9 $n = 72$ (1.0404)	FCK 0.1 $n = 42$ (0.9394)	WCK 3.4 $n = 84$ (0.8178)	WCK 3.9 $n = 1$ (0.5565)				
Redbreast sunfish	WCK 2.9 $n = 65$ (1.6839)	WCK 2.3 $n = 63$ (1.6722)	MEK 0.6 $n = 3$ (1.6341)	WCK 3.4 $n = 128$ (1.6037)							
Stoneroller	WCK 2.3 $n = 2$ (0.9374)	WCK 5.1 $n = 176$ (0.9343)	WCK 6.8 $n = 4$ (0.7226)								
Sculpin	FFK 1.0 $n = 39$ (1.1796)	FFK 0.4 $n = 43$ (1.1713)	WCK 6.8 $n = 13$ (1.1456)								

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer; n = number of fish sampled and weighed. Values connected by the same line are not significantly different ($p > 0.05$), based on Tukey's studentized range (HSD) test.

Table E.10. Comparison between sampling periods for sites on White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary of mean condition factors (K) of fish species

Site[a]	Species[b]								
	Bluegill			Redbreast			Warmouth		
FCK 0.1	Spring 86 $n = 3$ (1.6365)	Summer 85 $n = 7$ (1.4071)	Summer 86 $n = 1$ (1.2328)	-			-		
MEK 0.6	-			Spring 86 $n = 27$ (1.9013)	Summer 86 $n = 3$ (1.6341)	Summer 85 $n = 18$ (1.4511)	-		
NTK 0.3	Summer 85 $n = 2$ (1.3663)	Spring 86 $n = 14$ (1.3280)	Summer 86 $n = 11$ (1.1229)	-			-		
WCK 2.3	Spring 86 $n = 28$ (1.6568)	Summer 86 $n = 86$ (1.4218)	Summer 85 $n = 36$ (1.3696)	Spring 86 $n = 51$ (1.8649)	Summer 85 $n = 63$ (1.7526)	Summer 86 $n = 63$ (1.6722)	Summer 85 $n = 3$ (1.9735)	Spring 86 $n = 11$ (1.8270)	Summer 86 $n = 23$ (1.7228)
WCK 2.9	-			Summer 85 $n = 2$ (2.1589)	Summer 86 $n = 65$ (1.6839)				
WCK 3.4	Spring 86 $n = 54$ (2.1078)	Summer 85 $n = 15$ (1.9941)	Summer 86 $n = 70$ (1.7622)	Spring 86 $n = 2$ (2.2189)	Summer 85 $n = 12$ (1.6840)	Summer 86 $n = 128$ (1.6037)			
WCK 3.9	Summer 86 $n = 1$ (2.5391)	Summer 85 $n = 2$ (1.7911)		-			-		

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

[b]Summer 85 = fish collected in August–September 1985; Spring 86 = fish collected April–May 1986; Summer 86 = fish collected in August–September 1986; n = number of fish sampled and weighed. Values connected by the same line are not significantly different ($p > 0.05$), based on Tukey's studentized range (HSD) test.

Table E.11. Comparison between sampling periods for sites on White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary of mean condition factors (K) of fish species

Site[a]	Species[b]								
	Blacknose dace			Creek chub			Fathead minnow		
FCK 0.1	Spring 86 n = 46 (1.2272)	Summer 86 n = 75 (0.9315)	Summer 85 n = 99 (0.9161)	Spring 86 n = 1 (1.3581)	Summer 85 n = 4 (1.0828)	Summer 86 n = 3 (1.0595)	Summer 85 n = 1 (0.9645)	Spring 86 n = 40 (0.9636)	Summer 86 n = 8 (0.8405)
FCK 0.8	Summer 86 n = 118 (0.9745)	Spring 86 n = 75 (0.8861)	Summer 85 n = 92 (0.7947)	Spring 86 n = 2 (1.1662)	Summer 85 n = 1 (1.0496)	Summer 86 n = 2 (1.0418)		-	
FFK 0.4	Spring 86 n = 169 (1.0033)	Summer 86 n = 148 (0.9235)			-			-	
FFK 1.0	Spring 86 n = 31 (0.9355)	Summer 86 n = 40 (0.8482)			-			-	
MEK 0.6	Summer 85 n = 1 (1.1801)	Spring 86 n = 1 (1.0115)	Summer 86 n = 2 (0.9266)	Summer 86 n = 3 (1.2291)	Spring 86 n = 6 (1.2285)	Summer 85 n = 39 (1.0187)		-	
MEK 1.5	Spring 86 n = 77 (1.0341)	Summer 85 n = 63 (0.8892)		Spring 86 n = 35 (1.0450)	Summer 85 n = 45 (0.7869)			-	
MEK 2.1	Summer 85 n = 14 (1.0286)	Spring 86 n = 62 (0.9600)		Spring 86 n = 42 (1.0089)	Summer 85 n = 7 (0.8743)			-	
NTK 0.3	Summer 86 n = 32 (1.0126)	Spring 86 n = 25 (0.9482)	Summer 85 n = 36 (0.9305)		-		Summer 86 n = 56 (0.9781)	Spring 86 n = 22 (0.9097)	Summer 86 n = 1 (0.7980)

Table E.11 (continued)

Site[a]	Species[b]								
	Blacknose dace			Creek chub			Fathead minnow		
NTK 1.0	Spring 86 $n = 86$ (1.0716)	Summer 86 $n = 17$ (0.9637)		Summer 86 $n = 3$ (1.1824)	Spring 86 $n = 8$ (1.0699)		Spring 86 $n = 4$ (0.9874)	Summer 86 $n = 1$ (0.9375)	
WCK 2.9	Spring 86 $n = 2$ (1.1457)	Summer 86 $n = 19$ (0.9672)			-		Spring 86 $n = 93$ (1.0607)	Summer 86 $n = 18$ (1.0397)	Summer 85 $n = 1$ (0.8455)
WCK 3.4	Spring 86 $n = 1$ (1.0251)	Summer 86 $n = 70$ (0.9546)			-		Spring 86 $n = 5$ (1.1976)	Summer 86 $n = 3$ (0.9722)	Summer 85 $n = 1$ (0.9111)
WCK 5.1	Spring 86 $n = 123$ (1.1253)	Summer 86 $n = 140$ (0.9260)	Summer 85 $n = 180$ (0.8554)	Spring 86 $n = 78$ (1.0773)	Summer 86 $n = 37$ (0.9206)	Summer 85 $n = 42$ (0.8000)	Spring 86 $n = 5$ (1.0758)	Summer 86 $n = 1$ (1.0076)	
WCK 6.8	Summer 85 $n = 137$ (0.7732)	Summer 86 $n = 135$ (0.8771)	Spring 86 $n = 97$ (0.8121)	Spring 86 $n = 5$ (0.9490)	Summer 86 $n = 18$ (0.8964)	Summer 85 $n = 4$ (0.8053)		-	

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

[b]Summer 85 = fish collected in August–September 1985; Spring 86 = fish collected April–May 1986; Summer 86 = fish collected in August–September 1986; n = number of fish sampled and weighed. Values connected by the same line are not significantly different ($p > 0.05$), based on Tukey's studentized range (HSD) test.

Table E.12. Comparison between sampling periods for sites on White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary of mean condition factors (K) of fish species

Site[a]	Species[b]					
	Mosquitofish			Stoneroller	Sculpin	
FCK 0.1	Spring 86 $n = 1$ (1.2800)	Summer 86 $n = 42$ (0.9394)	Summer 85 $n = 2$ (0.8231)	-	-	
FFK 0.4	-			-	Spring 86 $n = 39$ (1.3290)	Summer 85 $n = 43$ (1.1713)
FFK 1.0	-			-	Summer 86 $n = 39$ (1.1796)	Spring 86 $n = 68$ (1.1109)
NTK 0.3	Summer 86 $n = 78$ (1.0614)	Spring 86 $n = 39$ (1.0154)	Summer 85 $n = 45$ (0.9983)	-	-	
WCK 2.3	Spring 86 $n = 4$ (1.2321)	Summer 86 $n = 57$ (1.1198)	Summer 85 $n = 21$ (0.6244)	-	-	
WCK 2.9	Spring 86 $n = 25$ (1.1374)	Summer 86 $n = 72$ (1.0404)	Summer 85 $n = 14$ (0.9995)	-	-	
WCK 3.4	Spring 86 $n = 5$ (1.1868)	Summer 85 $n = 59$ (1.0601)	Summer 86 $n = 84$ (0.8178)	-	-	

Table E.12 (continued)

Site[a]	Species[b]						
	Mosquitofish	Stoneroller			Sculpin		
WCK 5.1	-	Spring 86 n = 166 (1.0702)	Summer 85 n = 138 (0.9761)	Summer 86 n = 176 (0.9343)	-		
WCK 6.8	-	Spring 86 n = 2 (0.9793)	Summer 86 n = 4 (0.7226)	Summer 85 n = 3 (0.7202)	Spring 86 n = 32 (1.1588)	Summer 86 n = 13 (1.1456)	Summer 85 n = 50 (1.0906)

[a]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

[b]Summer 85 = fish collected in August–September 1985; Spring 86 = fish collected April–May 1986; Summer 86 = fish collected in August–September 1986; n = number of fish sampled and weighed. Values connected by the same line are not significantly different ($p > 0.05$), based on Tukey's studentized range (HSD) test.

Appendix F

CHECKLIST OF BENTHIC MACROINVERTEBRATE TAXA COLLECTED FROM WHITE OAK CREEK WATERSHED, MAY THROUGH OCTOBER 1986

Table F.1. Checklist of benthic macroinvertebrate taxa collected from White Oak Creek watershed, May through October 1986

Taxon[a]	Site[b,c] FCK 0.1	FCK 0.8	FCK 1.0	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	WCK 1.1
Turbellaria																	
Planariidae	X	X	–	–	X	–	–	–	–	X	–	–	–	–	–	–	–
Nematoda	–	–	–	–	–	X	X	–	–	–	X	X	X	–	–	–	–
Oligochaeta	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Branchiura sowerbyi	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X
Crustacea																	
Isopoda	X	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–	–
Asellus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X	–	–
Lirceus	–	X	X	–	X	X	X	X	–	X	–	–	–	X	X	–	–
Amphipoda																	
Crangonyx	–	–	–	–	X	–	–	X	–	–	–	–	–	–	–	–	–
Gammarus	–	X	X	–	X	–	–	–	–	–	–	–	–	–	–	–	–
Decapoda	X	X	–	–	–	X	–	–	X	X	–	–	–	–	–	X	–
Cambarus	X	X	–	–	–	X	–	–	X	X	–	–	–	–	X	X	–
Insecta																	
Collembola	X	–	–	X	X	X	X	–	–	X	–	–	–	–	X	–	–
Isotomidae	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–
Ephemeroptera	–	–	–	–	–	–	–	X	–	–	–	–	–	–	–	–	–
Baetidae	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–
Baetis	–	X	X	–	X	–	–	X	–	X	–	–	–	–	–	X	–
Centroptilum	–	X	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Heptageniidae																	
Stenacron	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X	–
Stenonema	–	–	–	–	–	–	–	X	–	–	–	–	–	–	–	X	–
Oligoneuriidae																	
Isonychia	–	X	–	–	–	–	–	–	–	–	–	–	–	–	–	X	–
Siphlonuridae																	
Ameletus	–	–	–	–	–	–	–	–	–	X	–	–	–	–	–	–	–
Leptophlebiidae																	
Habrophlebiodes	–	X	X	–	X	–	–	–	–	X	–	–	–	–	–	X	–
Paraleptophlebia	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X	–
Ephemeridae																	
Ephemera	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X	–

Table F.1 (continued)

Taxon[a]	Site[b,c] FCK 0.1	FCK 0.8	FCK 1.0	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	WCK 1.1
Ephemeroptera (cont.)																	
Ephemerellidae																	
Ephemerella	–	–	X	–	X	–	–	–	–	–	–	–	–	–	–	–	–
Eurylophella	–	–	X	–	–	–	–	–	–	X	–	–	–	–	–	–	–
Odonata																	
Anisoptera																	
Aeshnidae																	
Boyeria vinosa	–	–	–	–	–	X	X	–	–	–	X	–	–	–	–	–	–
Gomphidae	–	–	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–
Gomphus	–	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	–
Stylogomphus albistylus	–	–	–	–	–	X	X	X	X	X	X	–	–	–	–	X	–
Cordulegastridae																	
Cordulegaster	–	–	X	–	X	X	X	–	–	–	X	–	–	–	–	–	–
Macromiidae																	
Macromia	–	–	–	–	–	–	–	–	–	–	–	–	X	–	–	–	–
Zygoptera																	
Calopterygidae																	
Calopteryx	–	–	–	X	–	–	–	–	X	–	–	–	–	–	–	–	–
Hetaerina	–	–	–	–	–	–	–	–	X	–	–	–	–	–	–	–	–
Coenagrionidae																	
Argia	–	–	–	–	–	X	X	–	–	–	X	X	X	–	–	–	–
Ischnura	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X	–
Plecoptera																	
Nemouridae																	
Amphinemura	–	–	–	–	–	–	–	X	–	X	–	–	–	–	–	X	–
Leuctridae																	
Leuctra	–	X	X	–	X	–	–	X	–	–	–	–	–	–	–	X	–
Peltoperlidae																	
Peltoperla	–	X	X	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Perlodidae																	
Isoperla	–	–	–	–	–	–	–	–	–	X	–	–	–	–	–	–	–
Chloroperlidae	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Alloperla	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–	X	–
Haploperla	–	–	–	–	–	–	–	–	–	X	–	–	–	–	–	–	–
Perlidae	X	–	–	–	–	–	–	X	–	–	–	–	–	–	–	X	–
Eccoptura xanthenes	–	X	–	–	X	–	–	X	–	–	–	–	–	–	–	X	–
Perlesta	–	–	–	–	–	X	X	–	–	X	–	–	–	–	–	–	–

Table F.1 (continued)

Taxon[a]	Site[b,c]																
	FCK 0.1	FCK 0.8	FCK 1.0	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	WCK 1.1
Hemiptera																	
Veliidae																	
Rhagovelia	—	—	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—
Megaloptera																	
Corydalidae																	
Nigronia	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Nigronia fasciatus	—	—	—	—	X	X	—	X	X	X	—	—	—	—	—	X	—
Nigronia serricornis	—	—	X	—	X	X	—	—	—	—	X	—	—	—	X	X	—
Sialidae																	
Sialis	—	X	—	—	X	—	—	X	X	X	—	—	—	—	X	X	—
Neuroptera																	
Sisyridae	X	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Trichoptera																	
Hydropsychidae																	
Cheumatopsyche	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	—
Diplectrona modesta	—	X	X	—	X	—	—	X	—	—	—	—	—	—	—	X	—
Hydropsyche	—	—	—	—	X	X	X	—	—	X	X	X	X	—	—	X	X
Philopotamidae	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Chimarra fraternus	—	X	—	—	—	X	—	X	X	—	X	—	—	—	—	X	—
Dolophilodes distinctus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Wormaldia	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Polycentropodidae																	
Cyrnellus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Polycentropus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Psychomyiidae																	
Lype diversa	—	—	—	—	X	—	—	—	—	—	—	—	—	—	—	X	—
Glossosomatidae																	
Glossosoma	—	—	X	—	X	—	—	X	—	—	—	—	—	—	—	—	—
Rhyacophilidae																	
Rhyacophila	—	X	—	—	X	—	—	—	—	X	—	—	—	—	—	X	—
Lepidostomatidae																	
Lepidostoma	—	—	X	—	X	—	—	—	—	—	—	—	—	—	—	—	—
Limnephilidae																	
Goera	—	—	—	—	X	—	—	—	—	X	—	—	—	—	—	—	—
Ironoquia	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Neophylax	—	—	—	—	X	—	—	—	—	X	—	—	—	—	—	—	—
Pycnopsyche	—	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—

Table F.1 (continued)

Taxon[a]	Site[b,c]																
	FCK 0.1	FCK 0.8	FCK 1.0	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	WCK 1.1
Trichoptera (cont.)																	
Phryganeidae																	
Ptilostomis	–	–	–	–	X	–	–	–	–	X	–	–	–	–	–	–	–
Leptoceridae																	
Triaenodes	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	-	–
Coleoptera																	
Hydrophilidae																	
Berosus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X	–	–
Psephenidae																	
Ectopria	–	–	X	–	X	X	–	–	–	–	X	–	X	–	X	X	–
Psephenus herricki	–	X	–	–	–	–	–	–	X	–	–	–	–	–	–	X	–
Ptilodactylidae																	
Anchytarsus bicolor	–	X	–	–	X	–	–	–	–	–	–	–	–	–	–	X	–
Elmidae																	
Dubiraphia	–	X	–	–	X	X	–	–	–	X	–	–	–	–	X	X	–
Optioservus	X	X	X	–	X	–	X	X	X	X	X	X	–	–	–	X	–
Stenelmis	X	X	–	–	–	X	X	X	X	X	X	X	X	–	X	X	–
Diptera																	
Tipulidae	X	X	X	X	X	X	X	–	X	X	–	–	X	X	X	X	–
Antocha	–	–	X	–	X	–	–	–	–	–	–	–	–	–	X	X	–
Hexatoma	–	–	–	–	X	–	–	–	–	X	–	–	–	–	–	–	–
Limoniinae	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–
Pseudolimnophila	–	–	X	–	X	–	–	–	–	–	–	–	–	–	–	X	–
Tipula	X	X	X	X	X	–	–	–	X	X	–	–	–	–	X	X	–
Tipula abdominalis	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–
Dixidae	–	–	–	–	–	–	–	X	–	–	–	–	–	–	–	–	–
Dixa	–	–	X	–	–	–	–	–	–	X	–	–	–	–	–	–	–
Dixella	–	X	–	–	X	–	–	–	–	–	–	–	–	–	–	X	–
Chaoboridae																	
Chaoborus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X
Ceratopogonidae	–	–	–	–	–	X	–	X	–	X	–	X	X	–	–	X	X
Chironomidae																	
Tanypodinae																	
Ablabesmyia	–	–	–	X	–	X	X	X	X	–	X	X	X	–	X	–	–
Apsectrotanypus	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	–	–
Coelotanypus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	X
Labrudinia	–	–	–	–	–	X	–	–	–	–	–	–	–	–	–	–	–

Table F.1 (continued)

Taxon[a]	Site[b,c] FCK 0.1	FCK 0.8	FCK 1.0	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	WCK 1.1
Chironomidae (cont.)																	
Larsia	—	—	—	—	—	X	—	—	X	—	—	—	—	—	—	X	—
Natarsia	—	X	—	—	—	X	—	X	X	X	X	—	—	—	X	—	—
Natarsia baltimoreus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Nilotanypus	—	X	—	—	X	X	—	X	—	X	—	X	—	—	—	X	—
Paramerina	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—	—
Pentaneura	—	—	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—
Procladius	—	—	—	—	—	X	—	—	—	—	X	X	X	X	—	—	X
Tanypus	—	—	—	—	—	—	—	—	—	—	—	—	X	X	—	—	X
Thienemannimyia gp	X	X	X	X	X	X	—	X	X	X	X	X	X	X	X	X	—
Trissopelopia	—	—	—	—	X	—	—	—	—	X	—	—	—	—	—	—	—
Zavrelimyia	—	X	—	—	—	—	—	X	X	X	—	—	—	—	—	—	—
Diamesinae																	
Diamesa	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—	—	—
Orthocladiinae	—	X	X	—	X	—	—	—	—	—	X	—	—	—	—	—	—
Cardiocladius	X	—	—	X	—	X	—	—	—	—	X	X	X	X	—	—	—
Chaetocladius	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—	—
Corynoneura	—	—	—	—	—	—	—	X	—	X	—	—	—	—	—	—	—
Cricotopus/ Orthocladius	X	—	X	X	X	X	X	X	X	X	X	X	X	X	X	X	—
Cricotopus/ Orthocladius?	—	—	X	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Eukiefferiella	X	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Heleniella	—	—	—	—	X	—	—	—	—	—	—	—	—	—	—	X	—
Heterotrissocladius	—	X	—	X	—	—	—	—	X	—	—	—	—	—	X	—	—
Krenosmittia	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—	—
Limnophyes	—	—	—	X	—	—	—	—	—	—	—	—	—	—	—	—	—
Nanocladius	—	—	—	—	—	—	—	—	—	—	—	X	X	—	—	—	—
Parametriocnemus	X	X	X	—	X	—	—	X	X	X	—	—	—	—	X	X	—
Paraphaenocladius	—	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—
Psectrocladius	—	—	—	—	—	—	—	—	—	—	—	—	X	—	—	—	—
Rheocricotopus	—	X	X	—	X	X	X	X	X	—	X	—	—	X	X	X	—
Symposiocladius	—	—	—	—	X	—	—	—	—	—	—	—	—	—	—	—	—
Thienemanniella	X	—	X	X	X	X	—	—	X	X	X	—	—	—	—	X	—
Tvetenia	—	—	X	—	X	—	—	X	X	X	—	—	—	—	—	—	—
Xylotopus par	—	—	—	—	—	X	—	—	—	—	—	—	—	—	—	—	—
Chironominae																	
Chironomini	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Chironomus	—	—	—	—	—	—	X	—	X	—	—	—	X	X	X	X	X
Cladopelma	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X

Table F.1 (continued)

Taxon[a]	Site[b,c] FCK 0.1	FCK 0.8	FCK 1.0	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	WCK 1.1
Chironomidae (cont.)																	
Cryptochironomus	X	—	—	—	—	X	—	X	X	X	—	—	—	—	X	X	X
Cryptotendipes	—	—	—	—	—	X	—	—	—	—	—	—	—	—	—	—	—
Dicrotendipes	—	—	X	—	X	—	—	—	X	—	—	—	—	—	X	—	—
Goeldichironomus	—	—	—	—	—	—	—	—	—	—	—	X	—	—	—	—	—
Microtendipes	—	—	—	—	X	—	—	X	X	X	—	—	—	—	—	—	—
Parachironomus	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Paratendipes	—	X	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Phaenopsectra	—	—	—	—	—	—	—	—	—	—	—	X	—	—	X	—	—
Polypedilum	—	X	X	X	X	X	X	X	X	X	—	X	—	—	X	X	—
Pseudochironomus	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Stenochironomus	—	—	—	—	—	X	—	—	—	—	—	—	X	—	—	—	—
Stictochironomus	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—	—
Tribelos	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	X	—
Xenochironomus	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Tanytarsini																	
Paratanytarsus	—	—	—	—	—	—	—	X	—	—	—	—	—	—	X	—	—
Rheotanytarsus	—	—	X	—	X	X	—	X	X	X	X	—	—	—	X	X	—
Stempellina	—	—	—	—	X	—	—	—	—	—	—	—	—	—	—	—	—
Stempellinella	—	—	X	—	X	—	—	X	—	—	—	—	—	—	—	X	—
Sublettea	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Tanytarsus	X	X	X	—	X	X	—	X	X	X	X	—	—	—	X	X	—
Simuliidae	X	X	X	—	X	X	—	X	X	X	X	X	X	—	X	X	—
Stratiomyidae	—	—	—	—	X	—	—	—	—	—	—	—	—	—	—	—	—
Tabanidae	—	X	—	X	—	X	X	—	X	—	—	—	X	—	—	—	—
Empididae	—	—	X	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Hemerodromia	X	—	—	—	X	—	—	—	—	—	X	X	X	—	X	—	—
Mollusca																	
Gastropoda																	
Ancylidae																	
Ferrissia	—	—	—	—	—	X	—	—	—	—	—	—	—	—	X	—	—
Lymnaeidae																	
Fossaria	—	—	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—
Physidae																	
Physella	—	—	—	X	X	—	—	—	X	—	X	—	—	—	—	—	X
Planorbidae																	
Planorbula trivolis	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Pleuroceridae																	
Elimia	—	X	X	—	X	—	—	—	—	—	—	—	—	—	—	X	—

Table F.1 (continued)

Taxon[a]	Site[b,c]																
	FCK 0.1	FCK 0.8	FCK 1.0	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	WCK 1.1
Bivalvia																	
Corbiculidae																	
Corbicula fluminea	—	—	—	—	—	—	—	—	X	—	—	—	—	—	—	—	—
Sphaeriidae	X	—	—	—	—	—	—	—	X	—	X	—	—	—	—	—	—
Pisidium	—	—	—	—	X	—	—	—	—	—	—	—	X	—	—	—	—
Sphaerium	—	—	—	—	X	—	—	X	X	—	—	—	—	—	—	—	—

[a]Lowercase *gp* denotes group.
[b]FCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.
[c]An *X* indicates that the taxon was collected at least once, a blank indicates that a lower level of classification (e.g., family, genus, or species) was possible at one or more sites, and a dash indicates that the taxon was not collected or that the taxon was identified to a lower level at one or more sites.

END

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