Y-12

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OAK RIDGE
Y-12
PLANT

MARTIN MARIETTA

First Report on the Oak Ridge Y-12 Plant Biological Monitoring and Abatement Program for East Fork Poplar Creek

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Prepared by the
Environmental Sciences Division
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MANAGED BY
MARTIN MARIETTA ENERGY SYSTEMS, INC.
FOR THE UNITED STATES
DEPARTMENT OF ENERGY

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FIRST REPORT ON THE OAK RIDGE Y-12 PLANT BIOLOGICAL MONITORING AND ABATEMENT PROGRAM FOR EAST FORK POPLAR CREEK

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LIST OF ACRONYMS

ANOVA	analysis of variance
ATDL	Atmospheric Turbulence and Diffusion Laboratory
BaP	benzo(a)pyrene
BAT	best available technology
BF	Brushy Fork
BFK	Brushy Fork kilometer
BMAP	Biological Monitoring and Abatement Program
CFS	cubic feet per second
CPCF	Central Pollution Control Facility
CPOM	coarse particulate organic matter
CV	coefficient of variation
DOE	U.S. Department of Energy
DOM	dissolved organic matter
DMSO	dimethyl sulfoxide
DTT	dithiothreitol
EDTA	ethylenediamine tetracetic acid
EFK	East Fork kilometer
EFPC	East Fork Poplar Creek
EPA	U.S. Environmental Protection Agency
EROD	7-ethoxyresorufin O-deethylase
ESD	Environmental Sciences Division
FDA	USDA Food and Drug Administration
FFCA	Federal Facility Compliance Agreement
GC/MS	gas chromatography/mass spectrometry
GLM	general linear models
HPLC	high performance liquid chromatography
IWC	instream waste concentration
LSI	liver somatic index
LTF	liquid treatment facility
MBAS	methylene-blue-reactive substances
MFO	mixed function oxidase
NADH	nicotinamide adenine dinucleotide, reduced form
NADPH	nicotinamide adenine dinucleotide phosphate, reduced form
NHP	New Hope Pond
NOEC	no observed effect concentration
NPDES	National Pollutant Discharge Elimination System
OC	organic content
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
ORTF	Oak Ridge Task Force
ORWTF	Oak Ridge Wastewater Treatment Facility
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCK	Poplar Creek kilometer
PGV	Preliminary Guidance Value

POM	particulate organic matter
QA/QC	quality assurance/quality control
SD	standard deviation
SE	standard error
SGOT	serum glutamate oxaloacetate transaminase
TCMP	Toxicity Control and Monitoring Program
TDEC	Tennessee Department of Environment and Conservation (formerly TDHE)
TDHE	Tennessee Department of Health and Environment
TRC	total residual chlorine
TVA	Tennessee Valley Authority
USGS	U.S. Geological Survey
VSI	visceral somatic index
WBR	Watts Bar Reservoir
WPCP	Water Pollution Control Program

PREFACE

On May 24, 1985, a National Pollutant Discharge Elimination System permit was issued for the Oak Ridge Y-12 Plant, a nuclear weapon components production facility operated by Martin Marietta Energy Systems, Inc., for the U.S. Department of Energy. As required in Part III(C): Special Condition No. 7 of the permit, a plan for the biological monitoring of the receiving stream, East Fork Poplar Creek (EFPC), was prepared and submitted for approval to the U.S. Environmental Protection Agency (Region IV) and Tennessee Department of Environment and Conservation (TDEC) [formerly the Tennessee Department of Health and Environment (TDHE)] in August 1985 (Loar et al. 1989). Because it was anticipated that the chemical composition of several effluent streams could be altered shortly after the permit was issued, some biomonitoring studies were initiated in May 1985 before formal approval of the plan was received from the regulatory agencies.

This document is the first volume of a series of reports that will be prepared annually on the results of the Oak Ridge Y-12 Plant Biological Monitoring and Abatement Program. This report describes studies that were conducted from May 1985 through June 1986, although additional data collected after June 1986 are included, as appropriate. The studies conducted during this first year were directed toward an ecological characterization of EFPC. Although future studies will place greater emphasis on testing various hypotheses regarding the causal factors and underlying mechanisms associated with the effects documented in the initial studies, monitoring of selected parameters at various levels of biological organization will continue. Any significant modifications in the parameters that are monitored or the frequency and location of this monitoring will also be addressed in the annual reports.

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EXECUTIVE SUMMARY

As stipulated in the National Pollutant Discharge Elimination System (NPDES) permit issued to the Oak Ridge Y-12 Plant on May 24, 1985, a Biological Monitoring and Abatement Program (BMAP) was developed for the receiving stream, East Fork Poplar Creek (EFPC). The objectives of the BMAP are (1) to demonstrate that the current effluent limitations established for the Oak Ridge Y-12 Plant protect the uses of EFPC (e.g., the growth and propagation of fish and aquatic life), as designated by the Tennessee Department of Environment and Conservation (TDEC) [formerly the Tennessee Department of Health and Environment (TDHE)], and (2) to document the ecological effects resulting from implementation of a water pollution control program that includes construction of several large wastewater treatment facilities. The BMAP consists of four major tasks: (1) ambient toxicity testing, (2) bioaccumulation studies, (3) biological indicator studies, and (4) ecological surveys of stream communities, including periphyton (attached algae), benthic macroinvertebrates, and fish. This document, the first in a series of reports on the results of the Y-12 Plant BMAP, describes studies that were conducted from May 1985 through September 1986.

BACKGROUND

Effluent discharges from the Oak Ridge Y-12 Plant enter the headwaters of EFPC above New Hope Pond (NHP), a 2.2-ha impoundment located just east of the plant boundary. The stream is 23.7 km in length from the outfall of NHP to the confluence with Poplar Creek above the Oak Ridge K-25 Site. Effluent discharges of 388 L/s from the Y-12 Plant and 227 L/s from the City of Oak Ridge Wastewater Treatment Facility (ORWTF) at East Fork kilometer (EFK) 13.4, together, constitute 39% of the mean annual flow (1436 L/s) in EFPC at EFK 5.3. The stream also receives urban and some agricultural runoff between EFK 22.7 and EFK 7.7 (that reach of EFPC located off the DOE Oak Ridge Reservation). The BMAP studies were conducted at six primary sites on EFPC (EFK 24.4 above NHP, EFK 23.4, EFK 18.2, EFK 13.8, EFK 10.6, and EFK 6.3) and a site on Brushy Fork, an off-site reference stream located just north of Oak Ridge. Other sampling sites on EFPC and several reference streams were also used depending on the specific objectives of the various BMAP tasks.

TOXICITY TESTING

Ambient (in-stream) toxicity was evaluated in March and June 1986 at 11 sites on EFPC between EFK 23.7 at the outfall of NHP and EFK 2.1 below the confluence with Bear Creek. The toxicity of EFPC water (grab samples) was determined by 7-d static-renewal tests that measured the survival and growth of fathead minnow (*Pimephales promelas*) larvae and the survival and reproduction of a small crustacean (*Ceriodaphnia dubia/affinis*). Although minnow growth was reduced at some sites, minnow survival and *Ceriodaphnia* survival and fecundity were not significantly reduced ($\alpha = 0.05$) relative to controls. The same series of tests was conducted monthly from February through September 1986 on 24-h composite water samples collected at the outfall of NHP, and no evidence of chronic toxicity was observed. Samples collected at EFK 24.0, just upstream of NHP, were tested for toxicity in August and September 1986. Water from this site clearly reduced reproduction of *Ceriodaphnia* (relative to controls and to the outfall of NHP) in both months, but there was no evidence of toxicity to fathead minnow larvae and no significant reduction in survival of *Ceriodaphnia*. These data suggest that water quality improves biologically as it passes through NHP.

The same test systems were used to evaluate the toxicity of selected waste streams, as required by the NPDES Toxicity Control and Monitoring Program for the Y-12 Plant. Those identified as problematic included two Category IV outfalls (photographic rinsewaters and the dye penetrant/emulsifier rinsewaters) and the effluent from the S-3 Pond Liquid Treatment Facility (S-3 LTF), where the toxicant was probably depleted uranium. The two Category IV outfalls are being modified to reduce toxicity, while further testing of the S-3 LTF was initiated.

BIOACCUMULATION STUDIES

Those contaminants that are known to accumulate to undesirable levels in EFPC fish, such as mercury and polychlorinated biphenyls (PCBs), are being monitored to evaluate the effectiveness of remedial actions within the Y-12 Plant. A broad suite of inorganic and organic pollutants are also monitored in EFPC biota to ensure that no other substances accumulate to unacceptable levels as new waste treatment processes are developed. The report presents data on (1) levels of mercury, PCBs, and ^{137}Cs in bluegill (*Lepomis machrochirus*), redbreast sunfish (*Lepomis auritus*), and carp (*Cyprinus carpio*)

from five sites on EFPC and several reference streams (sampling conducted in May and December 1985 and May 1986) and (2) concentrations of polycyclic aromatic hydrocarbons (PAHs) and PCBs in Asiatic clams (*Corbicula fluminea)* collected from reference streams and placed in enclosures at the outfall of NHP for 14 to 30 d (March and July 1986).

Mean concentrations of mercury in redbreast sunfish at EFK 23.4 just below NHP exceeded the Food and Drug Administration (FDA) action limit of 1 ppm in two of the three sampling periods (or 47% of the individuals collected at this site). Mean concentrations at EFK 18.2 (Route 95 bridge at Jefferson Avenue) ranged from 0.77 to 0.87 ppm, and 21% of the individuals collected at the site exceeded 1 ppm. A statistically significant (α = 0.05) decrease in mercury levels in both sunfish species with increasing distance from NHP was observed in all three sampling periods. Although no sunfishes downstream of EFK 18.2 exceeded 1 ppm, 22% of the carp collected below this site exceeded the FDA limit. Mercury levels in redbreast sunfish collected from EFPC in 1985 and 1986 were lower than those reported in previous studies conducted in 1982 and 1984 (Van Winkle et al. 1984 and TVA 1985e, respectively), but the cause of this decrease is not known. No concomitant decrease in the mean monthly concentration of total mercury in the outfall NHP was observed for the period 1980–1986.

None of the sunfish collected from EFPC exceeded the FDA action limit of 2 ppm PCBs in fish. Levels of PCBs in redbreast sunfish were highest at EFK 23.4 (mean concentration ranged from 0.48 to 0.69 ppm) and decreased with increasing distance downstream of NHP. Like mercury, PCBs in EFPC may have a continuing source upstream from NHP. The concentration of PCBs in sunfish collected in 1985 and 1986 were significantly lower (α = 0.05) than those observed in 1982. The levels of PCBs in carp, on the other hand, were elevated to much higher levels than those found in sunfish; the average concentration in 59 carp from five sites on EFPC was 1.99 ppm, and 24% exceeded 2 ppm. Background levels of PCBs were 0.05 and 0.07 ppm for sunfish and carp, respectively, in Hinds Creek and Beaver Creek. Another reference site (Brushy Fork) proved to be contaminated with PCBs to a level comparable with that in EFPC (0.64 and 2.14 ppm in redbreast sunfish and carp, respectively).

Levels of ^{137}Cs ranged from <1.9 to 11 Bq/kg at EFK 23.4 and were highest at EFK 6.3 (mean and maximum concentrations were 40 and 43 Bq/kg, respectively, in May

1986). Analyses of *Corbicula fluminea* following a 4-week exposure in the discharge from NHP did not indicate the presence of significant levels of PAHs.

Laboratory studies are also being conducted to investigate the critical factors that control the transport, fate, and bioavailability of contaminants in EFPC, focusing primarily on organic contaminants. Studies showed that dissolved organic matter (DOM) can compete with sediment for binding contaminants, but the partitioning can be predicted based on independent interactions of the dissolved contaminant with each sorbent. Failure to account for the role of DOM can result in significant errors in predictions of the transport and fate of contaminants, depending on the relative abundance of dissolved and particulate organic matter in the system. A conceptual model is proposed relating the affinity of DOM for binding contaminants to the chemical and structural properties of the sorbent.

BIOLOGICAL INDICATOR STUDIES

A suite of biological indictors, representing a gradient of short- to long-term responses to water quality changes, was measured in adult redbreast sunfish collected in the fall 1985, winter 1986, and spring 1986 from four sites on EFPC and from two reference sites: Brushy Fork (BF) and Watts Bar Reservoir (WBR). Parameters representing five general classes of bioindicators were measured: (1) blood biochemical indicators [sodium, potassium, glucose, triglycerides, cholesterol, serum glutamate oxaloacetate transaminase (SGOT), and total protein]; (2) lipid biochemical indicators (total-body lipids, total-body triglycerides, and phospholipids); (3) condition indices [liver-somatic index (LSI), visceral-somatic index, and condition factor]; (4) liver detoxification enzymes [7-ethoxyresorufin o-deethylase (EROD) cytochrome b_5, cytochrome P-450, nicotinamide adenine dinucleotide (NADH) cytochrome b_5 reductase, and nicotinamide adenine dinucleotide phosphate (NADPH) cytochrome P-450 reductase]; and (5) histopathological indicators (examination of liver, gonads, gills, and kidney). Evaluation of differences in the responses of fish in EFPC and reference areas was based on analysis of variance of individual bioindicator parameters and on canonical discriminant analysis of (1) blood, lipid, and condition index parameters and (2) liver enzymes (and LSI). These indicators provide separate measures of the integrated response of fish to their environment.

Results of the bioindicator studies provided a measure of the health (physiological condition) of individuals in the redbreast sunfish population in EFPC. Compared with BF, fish collected at EFK 23 just below NHP (and from WBR) had significantly lower (α = 0.05) levels of serum triglycerides, total body triglycerides, and total lipids, an indication of poorer nutrition and higher metabolic stress. The integrated response exhibited when the blood and lipid parameters and condition indices were considered together in a discriminant analysis showed that site EFK 23 was the most dissimilar of the EFPC and BF sites, whereas sites EFK 14 and EFK 5 were the most similar to BF. The analysis also showed the WBR site to be dissimilar to all others, possibly because of its unique habitat (only nonstream site).

Levels of four of the five liver detoxification enzymes measured in this study were significantly higher (α = 0.05) in fish from EFK 23 compared with BF and may indicate a response to elevated levels of organic contaminants in this reach of stream. Discriminant analysis of the enzyme and LSI data, which provided a measure of the integrated response of fish to water quality degradation, showed a pattern similar to that found for the other group of biochemical indicators (i.e., blood, lipid, and condition index parameters): EFK 23 was the most dissimilar site, whereas EFK 5 was most similar to BF and EFK 14. Unlike EFPC, the responses of fish from BF were closely aggregated regardless of differences in season or sex, suggesting that fish in EFPC may be exposed to greater or more variable stress than those in BF. The integrated response of the biochemical and enzyme groups of indicators each showed a gradient of decreasing response with increasing distance downstream of NHP. Results of the qualitative histopathological analyses also showed a similar longitudinal pattern in EFPC.

COMMUNITY STUDIES

The benthic macroinvertebrate and fish populations were sampled monthly and quarterly, respectively, at the six primary sites on EFPC and BF (the reference stream) from May 1985 through June 1986. Periphyton (attached algae) sampling was initiated in spring 1986 at four of the six EFPC sites and BF; samples were collected in March, May, and July 1986.

Seasonal trends in periphyton chlorophyll a, a measure of algal biomass, were associated with changes in light penetration. In early spring before canopy closure, all

sites in EFPC were similar, but after closure in May, periphyton biomass in the middle to upper reaches (EFK 23.4 and EFK 13.8) generally exceeded that in lower EFPC. On all sampling dates, all sites on EFPC had algal biomass values that were equal to, or greater than, those in BF, which in turn was similar to other, relatively undisturbed local streams. Periphyton productivity (as carbon uptake) showed a seasonal pattern similar to that for biomass, except at EFK 23.4, where biomass was high but productivity was relatively low compared with sites farther downstream. Both periphyton biomass and productivity were lower at EFK 10.6, below the ORWTF, compared with EFK 13.8, located just upstream of the facility.

The benthic macroinvertebrate community at EFK 24.4 and EFK 23.4 (above and below NHP, respectively) is significantly affected by Y-12 Plant operations. Although the source of impact is known, the cause is not clear. Toxic levels of residual chlorine could account for the depauperate community above NHP but not below it; nutrient enrichment and modification of the natural stream flow and temperature regimes may affect the benthos at both sites. Evidence of downstream recovery in EFPC was indicated by increases in the abundance, species richness, and complexity of the benthic community. Total richness was highest at EFK 13.8 and then decreased, suggesting an additional source of impact from discharges of the ORWTF at EFK 13.4. When expressed as the mean number of taxa per sample, richness at all EFPC sites was significantly lower than that in BF ($\alpha = 0.05$). The recovery rate of the benthic macroinvertebrate community in EFPC will depend on accurate identification of the causal factor(s). Because the Y-12 Plant is located in the headwaters of EFPC, the recovery rate relative to streams with unaffected upper reaches may be considerably slower because of the loss of an upstream source of organisms for recolonization.

Like the benthos community, the fish community in EFPC also exhibited a general pattern of recovery with increasing distance from NHP. No fish were collected above NHP, possibly because of toxic levels of chlorine. Although fish were relatively abundant below the pond, elevated temperatures may account for the low densities or absence of some species, such as the blacknose dace (*Rhinichthys atratulus)* and banded sculpin (*Cottus carolinae*). Species richness increased longitudinally in EFPC, as expected, because of increased habitat heterogenity from the headwaters to the mouth. However, the increased richness in EFPC primarily resulted from the addition of new,

pollution-intolerant species below EFK 18.2. The absence of these species above this site is indicative of water quality or habitat degradation in upper EFPC.

Studies of fish movement showed that only 11% of the more than 250 tagged sunfishes that were subsequently recaptured had moved more than 500 m, and 73% moved 200 m or less. Moreover, most of the extensive (>1 km) movements (71%) involved fishes moving either to or from the reach just below NHP. These findings are consistent with those of other investigators and provide validation for between-site comparisons of contaminant levels in fish that are meaningful only if movement (and thus exposure) is limited to the vicinity of the collection site. Finally, the off-site fish kill that occurred in July 1985 between EFK 22.5 and EFK 18.0 may also have contributed to the greater movement between EFK 23 and EFK 18. Although one study site (EFK 18.2) was adversely affected by the kill, the long-term impact on the BMAP was not significant. Fish population densities at this site in June 1986 were similar to the densities observed just before the kill.

FUTURE STUDIES

The results of studies conducted during the first year of the BMAP will be used to direct further monitoring efforts. For example, the ambient toxicity testing program, which included sampling at ten sites in EFPC downstream of NHP, will be modified to include more-intensive sampling at the inlet and outfall of NHP because little evidence of toxicity was found downstream of the pond. The monitoring of contaminants in biota will continue with no change in sampling frequency although additional sampling sites downstream of the confluence of EFPC and Poplar Creek will be included in future studies. Monitoring to assess ecological effects at different levels of biological organization will also be continued. A subset of the initial suite of biological indicators will be measured in adult redbreast sunfish to assess the health (physiological condition) of individual fish from the EFPC and BF populations. Periphyton sampling will be conducted monthly and will include an additional site above NHP. Benthos sampling sites and frequency will remain the same, but sampling of fish populations to obtain population estimates will be reduced from quarterly to semiannually (March and October).

In addition to continuing the monitoring phase of the BMAP, future studies will place a greater emphasis on development and testing of various hypotheses regarding the

causal factors and underlying mechanisms associated with the effects documented in this report. In some cases, the impacts characterized by the initial studies are complex, having several contributing causes. Ultimately, the rate of recovery of the stream communities, the elimination of toxicity above NHP, and the reduction in contaminant residues (mercury and PCBs) of fish below NHP will all depend on accurate identification of the causal factor(s).

1. INTRODUCTION

As a condition of the National Pollutant Discharge Elimination System (NPDES) permit issued to the Oak Ridge Y-12 Plant on May 24, 1985, a Biological Monitoring and Abatement Program (BMAP) was developed for East Fork Poplar Creek (EFPC), the receiving stream (Loar et al. 1989). The BMAP consists of four major tasks that reflect different but complementary approaches to evaluating the effects of Y-12 Plant effluent on the ecological integrity of EFPC. These tasks include (1) toxicity testing, (2) bioaccumulation studies, (3) biological indicator studies, and (4) in-stream monitoring of the benthic invertebrate and fish communities. Ecological effects are evaluated at different levels of biological organization, from tissues and organs of individuals to populations and communities. The BMAP also uses a variety of approaches, including laboratory studies, manipulative field experiments, and direct in-stream sampling of biota, to identify causal mechanisms underlying the observed effects.

The proposed BMAP was developed to meet two major objectives. First, studies were designed to provide sufficient data to determine if the effluent limitations established for the Y-12 Plant protect and maintain the classified uses of EFPC, as identified in the State of Tennessee Water Quality Management Plan for the Clinch River Basin (TDPH 1978). The two most significant uses of EFPC are (1) growth and propagation of fish and aquatic life and (2) recreation, including fishing and swimming. Primarily because of elevated levels of mercury in fish, fishing (and swimming) in EFPC have been prohibited by the Tennessee Department of Environment and Conservation (TDEC) [formerly the Tennessee Department of Health and Environment (TDHE)] since November 1982.

A second major objective of the BMAP for EFPC is to document the effects on stream biota resulting from implementation of a Water Pollution Control Program (WPCP) at the Y-12 Plant. The Program consists of strategies to (1) eliminate direct discharges of wastewaters to EFPC and (2) minimize the inadvertent release of pollutants to the environment. Significant elements of the WPCP include (1) construction of several collection/storage facilities and nine new major wastewater treatment facilities (Table 1-1) and (2) development of numerous countermeasures to reduce or preclude the release of pollutants to EFPC, including a Best Management Practices Plan; Area Source Pollution

Table 1-1. A summary of the current status of 12 wastewater treatment and waste separation/collection projects included in the Federal Facility Compliance Agreement (FFCA) of the NPDES permit for the Y-12 Plant

Project	Current status[a]	Date of construction completion	Date of initial effluent
Wastewater Treatment Facilities			
Central Pollution Control Facility	C	6-30-85	11-5-85
Central Pollution Control Facility II	UC(>95)	8-8-86	
S-3 Liquid Treatment Facility	C	3-31-85	7-17-85[b]
Waste Coolant Processing Facility	C	9-30-85	12-18-85[c]
West End Treatment Facility	UC(35)	3-31-87[d]	
East End Waste Facility	C	5-12-86	*e*
Steam Plant Wastewater Treatment Facility	UC(10)		
Plating Rinse Water Treatment Facility	UC(6)	1-15-87	
(ORNL) Biology Wastewater Treatment Facility	NC	10-30-88[d]	
Waste Separation/Collection Facilities			
Ion Resin Regeneration Waste Collection Tank (Bldg. 9201-2)	C	10-18-85	3-86
(ORNL) Isotope Separation Process Wastewater Collection Tank	C	1-29-86	*f*
(ORNL) Sump Pump Oil Separator (Bldg. 9204-3)	UC(90)	1-5-87	

[a]Current status on June 30, 1986; C = construction completed, UC = under construction (percentage of construction completed in parentheses), and NC = no construction initiated.

[b]Discharges of effluent to EFPC occurred in July–November 1985 and April–May 1986.

[c]Effluent is currently treated by the CPCF and no longer discharged to EFPC.

[d]Approval by TDHE/EPA pending.

[e]Wastewater from car washing operations, which was previously discharged to EFPC via Outfall 2 below NHP was collected while treatment options were evaluated.

[f]Wastewater collected before storage and subsequent treatment at the West Tank Farm.

Source: FFCA quarterly reports submitted by the Y-12 Plant to DOE for the period April 1, 1985 through June 30, 1986.

Control Management; a Spill Prevention, Control, Countermeasures, and Contingency Plan; and various spill prevention projects.

It is particularly important to identify any adverse effects of new wastewater treatment facilities during their initial operation so that corrective measures can be identified and alternatives for plant operation can be evaluated. As a condition of the NPDES permit for the Y-12 Plant, a Toxicity Control and Monitoring Program (TCMP) will be established to evaluate the toxicity of the effluent from these facilities. Both the TCMP and the BMAP, in turn, will be used to establish the final effluent limitations for each of the new wastewater treatment plants (Fig. 1-1).

ORNL DWG 91M-13947

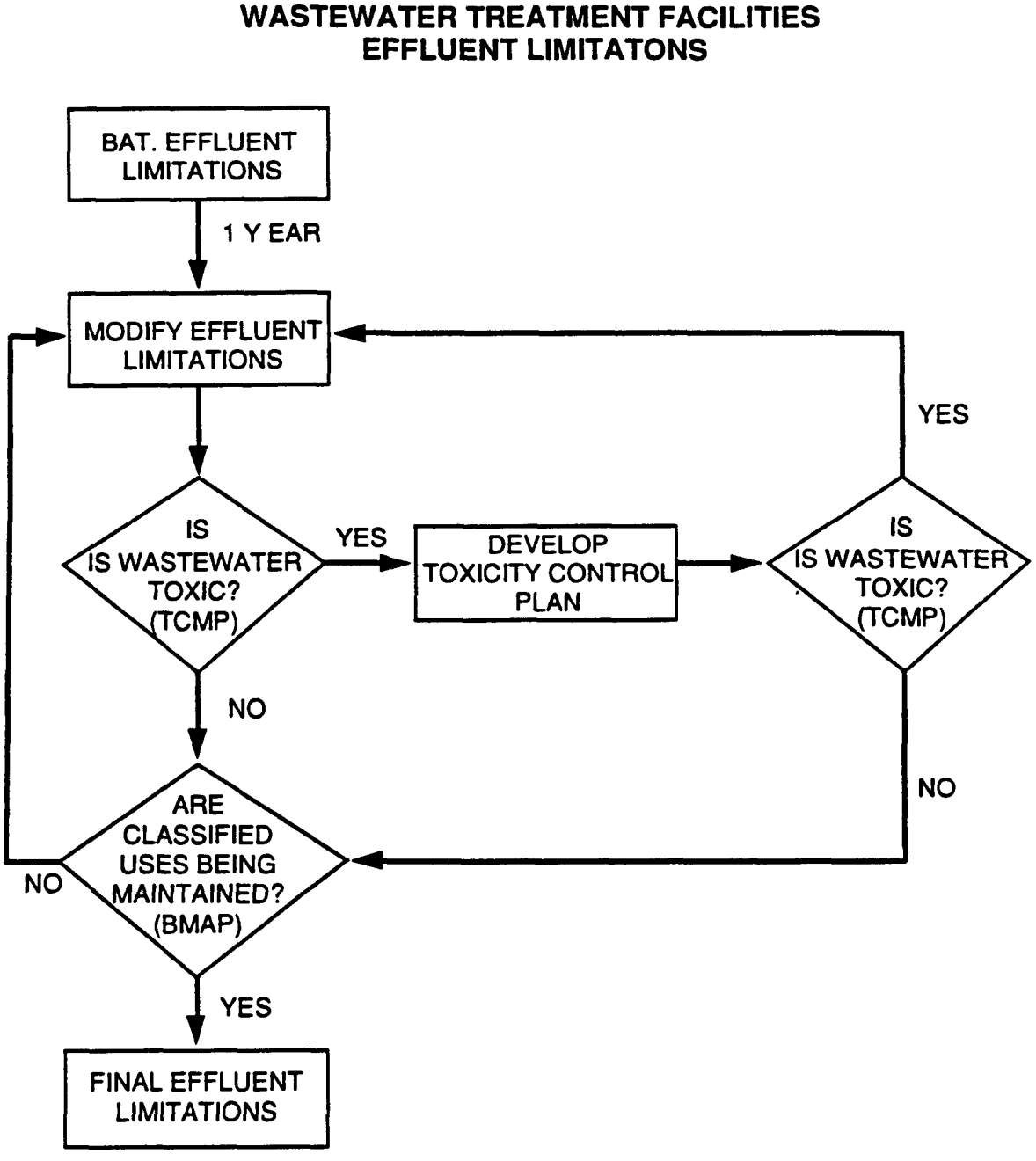

Fig. 1-1. Decision tree for establishing effluent limitations for new wastewater treatment facilities at the Y-12 Plant. BAT = Best Available Technology; TCMP = Toxicity Control and Monitoring Program; BMAP = Biological Monitoring and Abatement Program. Source: Kingrea (1986).

2. DESCRIPTION OF STUDY AREA

The EFPC drainage basin is located near the northern boundary of the U.S. Department of Energy (DOE) Oak Ridge Reservation (ORR) and has an area of 77.2 km^2 from the headwaters to the mouth at Poplar Creek kilometer (PCK) 8.7.[1] Parallel northeast-trending ridges constitute the northern (Black Oak Ridge) and southern (Chestnut Ridge) boundaries of the watershed. Elevations in the basin range from 226 to 390 m. The largest tributary is Bear Creek, which has a drainage area of 19.1 km^2 and joins East Fork Poplar Creek (EFPC) at EFPC kilometer (EFK) 2.4.[1] The Y-12 Plant is located near the watershed divide of Bear Creek and EFPC, which flow to the west and east, respectively, of the Plant (Fig. 2-1).

Because the Biological Monitoring and Abatement Program (BMAP) addresses only the effects of Y-12 Plant discharges on the biotic communities in EFPC, Bear Creek is excluded from the description of the EFPC watershed that follows. Extensive information on Bear Creek watershed is available, however, from the numerous studies conducted in 1983 and 1984 in association with the development of a remedial action plan for the Bear Creek Valley waste disposal area (e.g., Evaldi 1984, 1986; Turner and Kamp 1984; Geraghty and Miller, Inc. 1985; Loar et al. 1985; Pulliam 1985a, b; TVA 1985a-e, 1986).

2.1 GEOHYDROLOGY

The study area is located in the Valley and Ridge physiographic province of the Southern Appalachian Mountains. The ridges are composed primarily of sandstones and dolostones, and the valleys are underlain by shales, limy shales, and limestones (Geraghty and Miller, Inc. 1985). The principal groundwater-bearing formation in the Oak Ridge area is the Knox Dolomite, which comprises 25% of the surface area of the EFPC drainage basin; another 32% of the area consists of Chickamauga Limestone (Table 2-1). One of the largest springs in the basin (Crystal Spring located near EFK 9.8) is at the

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[1]Poplar Creek kilometer (PCK) 0.0 and East Fork Poplar Creek kilometer (EFK 0.0) are located at the confluence of Poplar Creek with the Clinch River and at the confluence of East Fork Poplar Creek with Poplar Creek, respectively. All distances are based on the list of key features described elsewhere in TVA (1986, Table I-1).

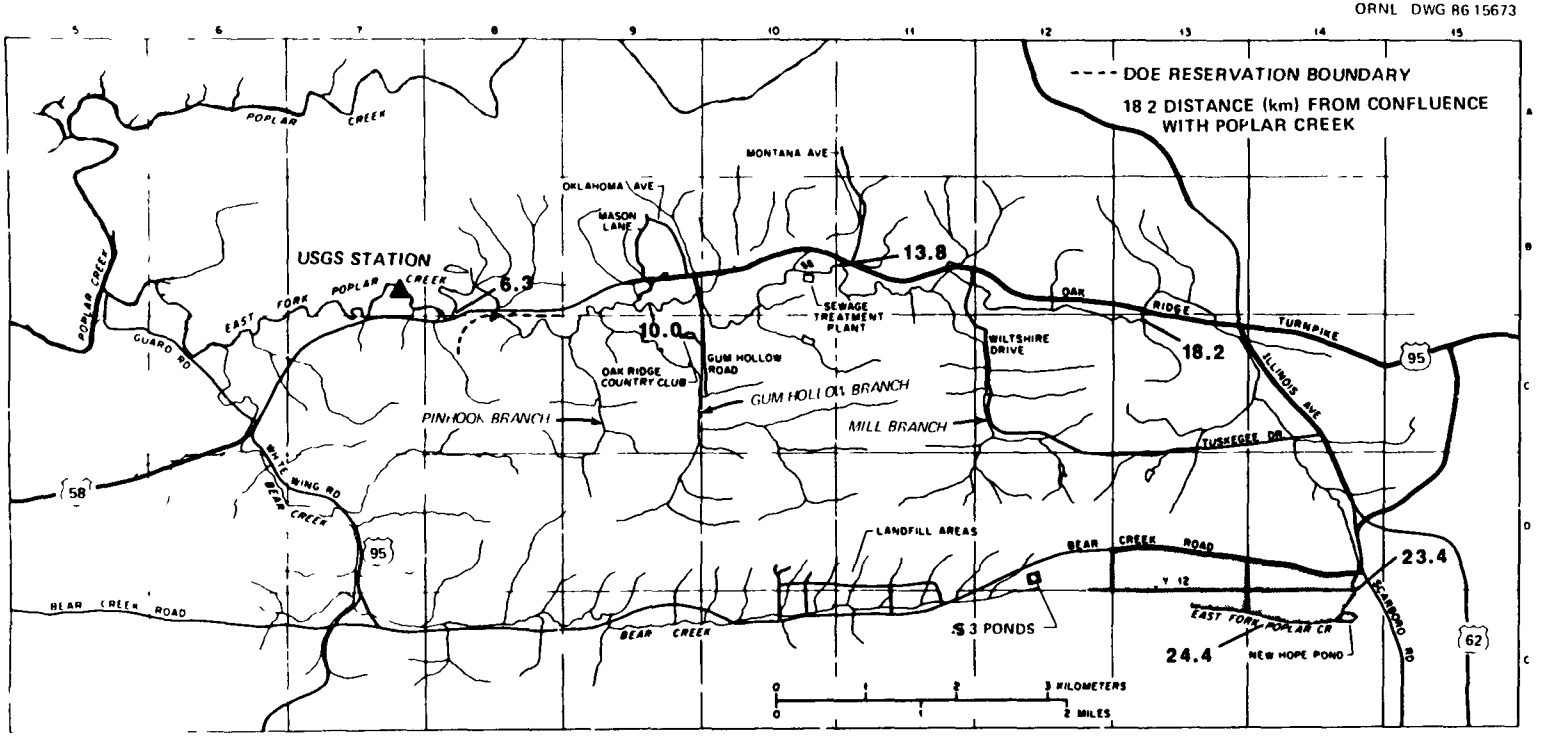

Fig. 2-1. Map of the East Fork Poplar Creek watershed showing the locations of the six primary study sites.

Table 2-1. General description of the geohydrology of tributaries of Poplar Creek,[a] East Fork Poplar Creek and Brushy Fork (a reference stream)

	East Fork Poplar Creek	Brushy Fork
Confluence with Poplar Creek	PCK 8.7[b]	PCK 29.3[b]
Station location	USGS gage	Dossett
Distance above confluence with Poplar Creek (km)	5.3	10.1
Drainage area (km^2)	50.5	26.4[c]
Distance/direction to nearest biological sampling site	1.0 km, upstream	N 2.5 km, downstream
Stream flow (L/s)[d]		
Mean annual flow (MAF)		
1936–1960 (estimated)[d]	878 (31)[e]	538 (19)
1960–1985	1456 (51.4)[f]	603 (213)[g]
7Q10[c]	48 (1.7)[e]	25 (0.88)
Low-flow per unit area (7Q10/drainage area)[h]	0.95 (0.09)	0.95 (0.09)
Geologic composition (% surface area of watershed)[c]		
Rome Formation and Conasauga Group	26	49
Knox Dolomite	25	18
Chickamauga Limestone	32	31
Late Ordovician to Mississippian age	16	

[a]See Fig. 2-2.

[b]PCK = Poplar Creek kilometer; PCK 0.0 is the confluence of Poplar Creek with the Clinch River.

[c]From McMaster (1967), Tables 5, 8-10.

[d]Flow in cubic feet per second (cfs) in parentheses.

[e]Excluding discharges from the Y-12 Plant and the City of Oak Ridge Wastewater Treatment Facility (ORWTF).

[f]Including discharges from the Y-12 Plant and the ORWTF.

[g]Estimated as $Q_{bf} = {}^{A}bf/A_{pc}$ ($Q_{pc}pc$) where A_{bf} and A_{pc} are the areas of the Brushy Fork and Poplar Creek watersheds and Q_{pc} is the stream flow at the USGS gaging station on Poplar Creek at PCK 22.2.

[h]Cubic liters per second per km^2 (cfs/square mile).

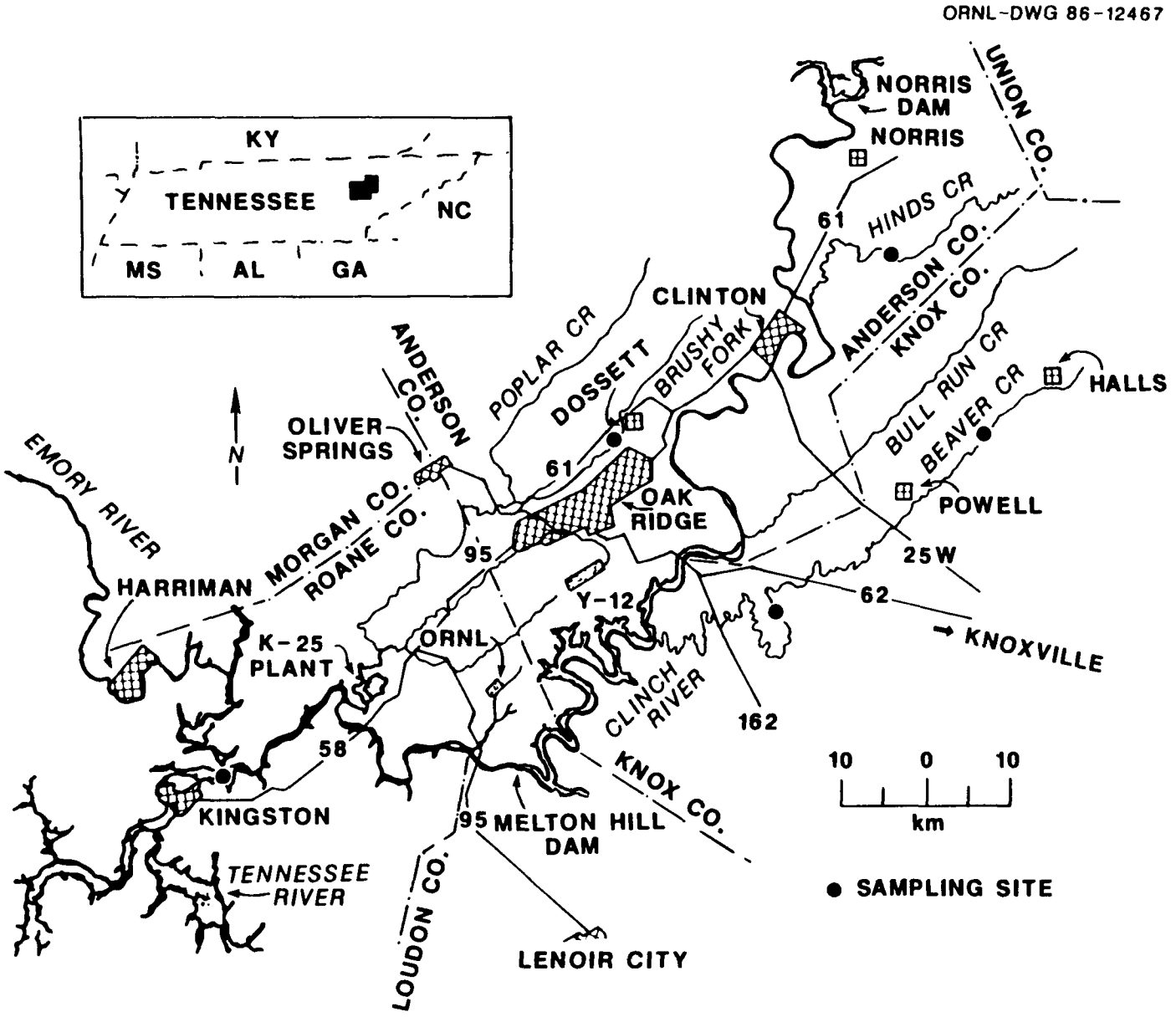

Fig. 2-2. Map of the Oak Ridge area showing locations of the reference (control) sites.

contact between the Knox and the Chickamauga Limestone and has a mean discharge of 48 L/s (1.7 cfs) with a range of 29 to 91 L/s (1.0 to to 3.2 cfs), based on monthly measurements over a 2-year period (McMaster 1967). Several smaller springs also occur along the Knox-Chickamauga contact in the EFPC basin.

The primary reference area used in the BMAP is the Brushy Fork (BF) watershed located just north of Oak Ridge and adjacent to the EFPC watershed (Fig. 2-2). The two drainages, which are separated by Black Oak Ridge, have a similar geologic composition (Table 2-1). The Knox Dolomite that underlies Black Oak Ridge is the source of three large springs that are tributaries to BF above the study site [BF kilometer (BFK) 7.6]. These springs are Bacon Spring (mean discharge of 108 L/s or 3.83 cfs), Smith Spring (52 L/s or 1.85 cfs), and Burress Spring (63 L/s or 2.22 cfs); Bacon Spring provides 8.5 L/s (0.3 cfs) to the town of Oliver Springs for domestic use (McMaster 1967). The almost identical unit-area low-flow discharges of EFPC at EFK 5.3 and BF at BFK 10.1 (Table 2-1) provide additional evidence of the similarity of their geologic composition.

The headwaters of EFPC consist of springs that originate on the northwest slope of Chestnut Ridge. The stream is contained in culverts through much of the west end of the Y-12 Plant before entering a rip-rap channel ~2.4 m wide and 2.6 m high (Kasten 1986). The creek receives discharges from more than 200 individual outfalls as it flows ~1.5 km through the plant site. These discharges include once-through cooling water and cooling tower blowdown (~64% of discharge volume), process wastewaters (32%), and coal yard runoff and storm drainage (Kasten 1986, Fig. 9). Process wastewaters include effluents from wastewater treatment facilities, photographic laboratories, fire-fighter training areas, plating operations, plant laboratories, and chemical preparation and makeup areas.

Just east of the plant boundary, EFPC flows into New Hope Pond (NHP), a 2.2-ha impoundment constructed in 1963 to equalize the pH of the effluent from the Y-12 Plant. The pond is also used for neutralization, sediment retention, and spill control (including provision for oil recovery by means of skimmers or chemical treatment). Construction of a bypass channel around NHP permits retention of hazardous chemical spills within the pond and thus provides a "last line of defense." From the outfall of NHP, EFPC flows a distance of 23.7 km to the confluence with Poplar Creek, a tributary of the Clinch River.

The average gradient between the upstream limits of the reservoir backwater area and NHP is ~1.7 m/km (TVA 1985d).

Effluent discharges from the Y-12 Plant above NHP and from the City of Oak Ridge Wastewater Treatment Facility (ORWTF) at EFK 13.4 augment the streamflow of EFPC. Mean annual discharge at the outfall of NHP was 388 L/s (13.7 cfs for the period 1980–1985 (TVA 1985d; NPDES quarterly reports for CY 1985). The mean discharge over the period just before and during the Y-12 Plant BMAP was similar to that in the past; daily flows from January 1985–June 1986 averaged 411 L/s (14.5 cfs) compared with 382 L/s (13.5 cfs) during the previous 5-year period (TVA 1985d) and rarely were less than 283 L/s (10 cfs) (Fig. 2-3). The average daily discharge of the ORWTF was 227 L/s (8.0 cfs) between January 1983 and May 1985 (TVA 1985d). Using an estimated average discharge of 59 L/s (2.1 cfs) to represent the contribution from the 3.24-km^2 drainage area above NHP (TVA 1985d), streamflow in EFPC is augmented by ~555 L/s (19.6 cfs).

Mean annual flow of EFPC at the U.S. Geological Survey (USGS) gaging station (EFK 5.3) is 1436 L/s (50.7 cfs) for the period 1960–1986; the maximum and minimum daily flows over the same period were 1.16×10^5 and 340 L/s (4100 and 12 cfs), respectively (Lowery et al. 1987). The adjusted mean annual flow [i.e., without the 555 L/s (19.6 cfs) contributed by the Y-12 Plant and the ORWTF] is 881 L/s (31.1 cfs), the same value as that estimated by McMaster (1967) based on limited flow records for the period 1961–1964 (Table 2-1). The maximum flow occurred on November 28, 1973; this storm resulted in 22 cm of rainfall in 48 h (1-in-25-year flood), based on studies conducted by studies conducted by Edgar (1978) in nearby White Oak Creek watershed. The largest known flood occurred on September 29, 1944; the peak discharge was 1.30×10^5 L/s (4600 cfs) and the recurrence interval is 50 years (TVA 1985d).

The minimum daily flow of 340 L/s (12 cfs) in EFPC is higher than that observed in much larger watersheds with little flow augmentation. Poplar Creek at the USGS gaging station (PCK 22.2), for example, has a drainage area of 213.7 km^2 (compared with 50.5 km^2 at EFK 5.3), yet the minimum discharge for the period of record (1960–1986) is 105 L/s (3.7 cfs), which occurred on July 31 and August 5 and 6, 1986 (Lowery et al. 1987). Although the creek receives discharges from a treatment facility in Oliver Springs (above PCK 22.2), the volume of these releases is probably insignificant relative to the average annual flow of 4876 L/s (169 cfs). Based on water yield for Poplar Creek

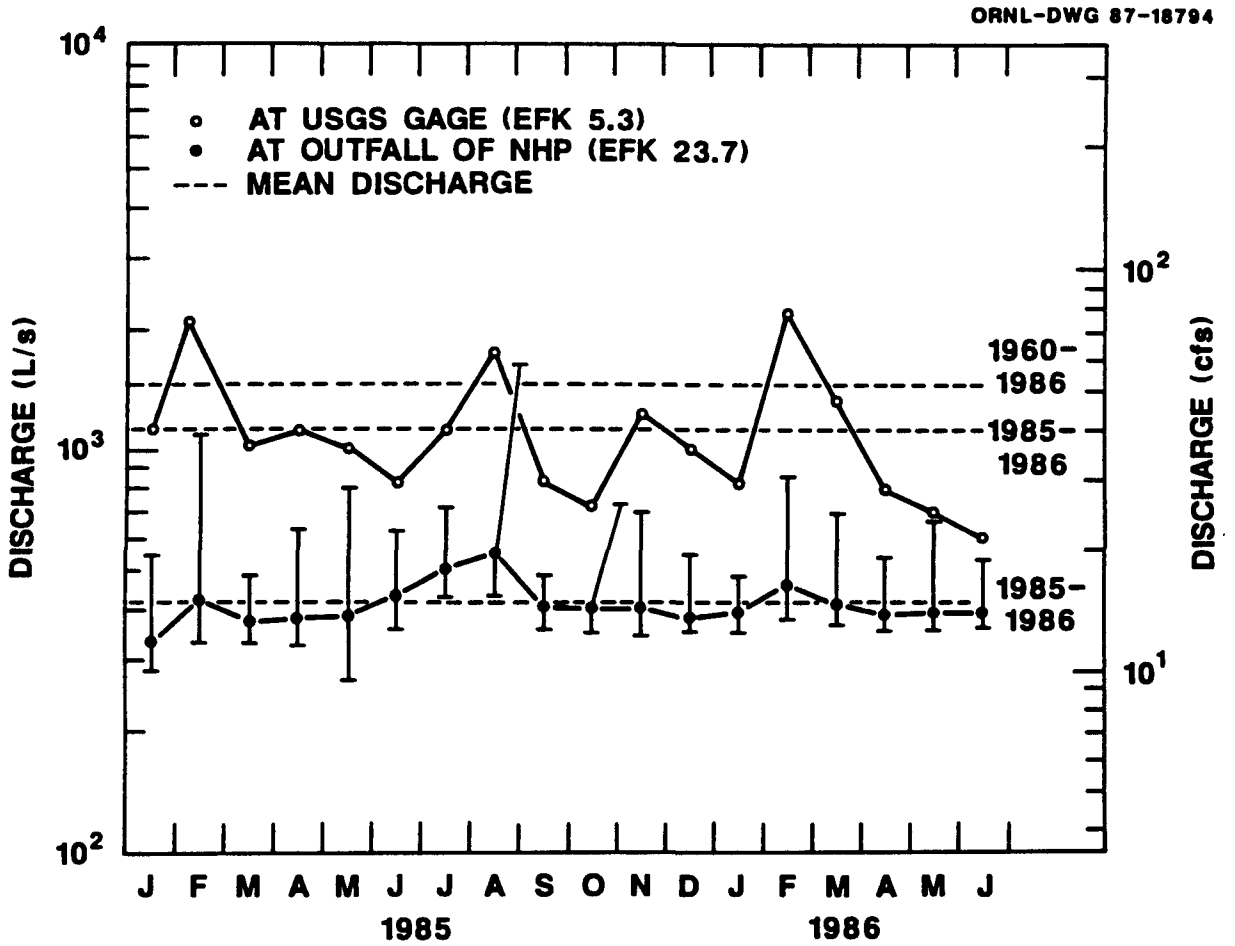

Fig. 2-3. Average monthly flows at the outfall of New Hope Pond (NHP) and at the U.S. Geological Survey (USGS) gaging station on lower East Fork Poplar Creek. Range indicated by vertical bars. Source: NPDES Quarterly Reports for EFK 23.7; Lowery et al. (1986, 1987) for EFK 5.3.

(discharge per unit area at PCK 22.2), the minimum flow of 340 L/s (12 cfs) in EFPC is at least an order of magnitude higher than the minimum flow would be without the contributions to streamflow from the Y-12 Plant and the ORWTF.

Another characteristic of streamflow in EFPC, in addition to the higher minimum flows, is decreased temporal variability resulting from the near constant average daily discharge at the outfall of NHP (Fig. 2-3). This effect is especially evident from a comparison of the hydrographs for EFPC and BF, a stream with no flow enhancement from industrial or municipal sources (Fig. 2-4). For example, the coefficient of variability (CV) based on the mean weekly discharge values plotted in Fig. 2-4 was 107.6 and 56.3% for BF and EFPC, respectively [mean discharge ± standard deviation was 368 ± 404 L/s (13.0 ± 14.3 cfs) in BF and 1107 ± 634 L/s (39.1 ± 22.4 cfs) in EFPC]. Although the increased minimum flow benefits aquatic biota by reducing streambed dewatering and thus minimizing the loss of habitat, increased flow stability reduces environmental heterogeneity (i.e., habitat diversity), which can adversely affect species richness and/or density.

2.2 LAND USE

Land use in the EFPC watershed reflects the public and private ownership of property in the basin. The creek flows less than 1 km below NHP before leaving the DOE Oak Ridge Reservation (ORR) at EFK 22.7. For the next 15 km, EFPC flows through the City of Oak Ridge, which had a population of 26,662 in 1980 (U.S. Department of Commerce 1982, as cited in TVA 1985d), before crossing the ORR boundary again at EFK 7.7 for the remainder of its course. The lower portion of the watershed within the ORR is undeveloped, consisting mostly of pine plantations and mixed hardwood stands.

Land use in Oak Ridge consists mostly of commercial and residential developments, some light industry and agriculture, and green-belt (forested) areas. Most of the industrial development is limited to the northeastern part of the basin (Fig. 2-1, grids C-15, D-15). Drainage from this area enters EFPC between EFK 22.5 and EFK 21.5. Commercial development occupies much of the floodplain and adjacent areas of the creek from EFK 22.5 to EFK 18.0. Farther downstream to the ORR boundary, residential and some agricultural development (principally livestock grazing) occur.

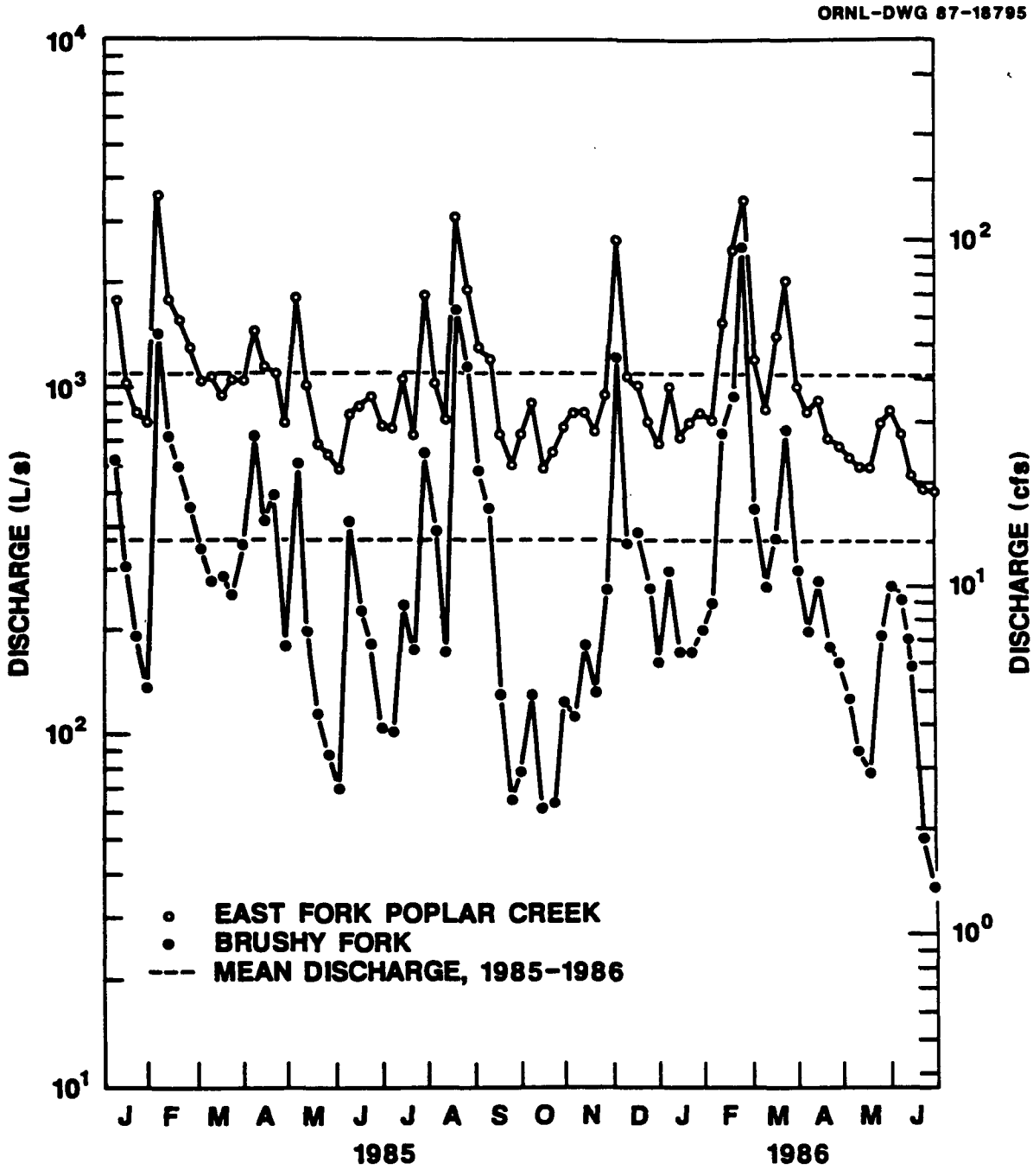

Fig. 2-4. Average weekly flows in East Fork Poplar Creek (EFPC) at EFK 5.3 and Brushy Fork (BF), a reference stream, at BFK 10.1. Values represent weekly means obtained from the average daily flows at the U.S. Geological Survey (USGS) gaging stations on lower EFPC (EFK 5.3) and on Poplar Creek (PCK 22.2) ~7 km below the confluence with BF. Estimates of flow in BF are based on extrapolation of water yield (discharge per unit area) at the gaged site on Poplar Creek (Table 2-1, footnote 'e').

Construction of single and multifamily homes between EFPC and Route 95 (Oak Ridge Turnpike), which parallels EFPC west of Oak Ridge (Fig. 2-1), has increased recently.

Approximate land use in the watershed above the USGS gaging station at EFK 5.3 is 15% urban, 28% grass (pasture, lawn, etc.), and 57% forest (TVA 1985d). Although urban development accounts for a relatively small proportion of the land use, it apparently has had a significant impact on sediment transport in EFPC. The Tennessee Valley Authority (TVA 1985d) estimated sediment transport in EFPC to be 351 t/km^2. This value exceeds the predicted sediment transport for a rural watershed the size of EFPC (50 km^2) by a factor of ~4 and is more than two orders of magnitude greater than the sediment yield from a 50 km^2 forested watershed (based on data for the Central Atlantic States, Manning et al. 1977). It is also relatively high compared with other, mostly rural, watersheds in the Tennessee Valley and is probably associated with urban development and channel realignment within the City of Oak Ridge (TVA 1985d).

2.3 WATER QUALITY

Water and sediments in EFPC downstream from the Y-12 Plant contain metals, organic chemicals, and radionuclides discharged over many years of operation. Most of the information on these contaminants was obtained in recent studies conducted by TVA for the Oak Ridge Task Force (ORTF), a multiagency group established in November 1983 to evaluate potential off-site contamination problems associated with the DOE facilities near Oak Ridge, Tennessee. Before that time, the only surveys of ambient water quality in EFPC below NHP were those conducted in 1961–1964 (McMaster 1967) and 1974–1975 (ERDA 1975).

The ORTF survey involved extensive sampling throughout the ORR and off-site. Water samples were taken at EFK 23.1 during a base-flow survey and analyzed for conventional parameters, priority pollutants (organics and metals), and radionuclides (TVA 1985a); only lithium and mercury exceeded background levels (TVA 1986, Table 3). Sediment samples were collected from EFPC and the floodplain near EFK 21.7 and EFK 2.7 and from the western and eastern ends of NHP; analyses were conducted of 114 organics, 14 metals and cyanide, and 12 radionuclides (Hoffman et al. 1984, Table I). Of these, ten priority pollutant organics [seven polycyclic aromatic hydrocarbons or PAHs, bis (2-ethyl hexyl) phthalate, total polychlorinated biphenyls (PCBs) and total phenols] and

seven metals (arsenic, cadmium, lead, mercury, nickel, silver, and zirconium) were found in EFPC at concentrations above background levels and/or above the analytical detection limit (TVA 1986, Table 4). Additional and more extensive sampling of sediments in EFPC was conducted to estimate the quantity of mercury-contaminated sediment and floodplain deposits and to assess the transport and/or stability of mercury-contaminated sediment in the EFPC watershed (TVA 1985b, d).

A review of EFPC water quality for this report is based on an analysis of NPDES data collected at the outfall of NHP from January 1, 1985–June 30, 1986. Additional information is provided from supplemental analyses conducted in various subtasks of the BMAP, including (1) routine measurements of several conventional parameters as part of the toxicity testing protocol (Sects. 3.1 and 3.2) and (2) nonroutine water and sediment sampling as part of periphyton and bioaccumulation studies, respectively (Sects. 3.3 and 4.1). More extensive ambient water quality sampling was initiated in EFPC and BF in 1986. This program will be modified, as appropriate, based on the results of future toxicity and ecological monitoring.

2.3.1 NPDES Monitoring at Outfall of NHP

Means and standard deviations of the 25 parameters monitored at NPDES station 303 (outfall of NHP) are listed in Table 2-2. The potential toxicity of the maximum observed concentrations as well as the variance in the mean concentration, as indicated by the standard deviation (SD), were evaluated in screening the data to identify possible causal links with observed ecological effects downstream. On the basis of this review, copper, ammonia, nitrogen, residual chlorine, perchloroethylene, and oil and grease were identified for further evaluation. The raw data (weekly values) for these six parameters were obtained from the Y-12 Plant Department of Environmental Management (P. M. Pritz, Y-12 Department of Environmental Management, personnel communication, 1986) and are plotted in Figs. 2-5 and 2-6.

Of the six parameters, copper, ammonia, residual chlorine, and perchloroethylene could have been toxic at the maximum concentration reported, depending on the period of exposure. Nitrogen is often associated with the adverse effects of nutrient enrichment. The oil and, grease parameter was included because of the uncertainty regarding its composition and hence toxicity. All six parameters had a relatively high SD (Table 2-2).

Table 2-2. Mean, standard deviation (in parentheses), and range of concentrations of the 25 NPDES parameters monitored at the outfall of New Hope Pond from June 1, 1985-June 30, 1986

Parameter	Type of sample[a]	Concentration (mg/L)	
		Mean (SD)[b]	Range[c]
Ammonia (as N)	1	<0.25(0.15)	<0.02-3
Beryllium, μg/L	2	<0.5(0.0)	<0.5-0.6
Biochemical oxygen demand (BOD)	1	<5.2(0.3)	<5-8
Cadmium, μg/L	1	<2.2(0.8)	<2-14
Chemical oxygen demand (COD)	1	22(31)	<5-560
Copper, μg/L	1	14(13)	<4-190
Chromium	1	<0.01(0.001)	<0.01-<0.012
Dissolved oxygen	2	8.7(1.4)	4.1-12.4
Dissolved solids	1	264(52)	150-830
Flow, MGD	3	9.62(1.11)	9.70-36.65
Fluoride	1	1.05(0.17)	0.7-1.6
Lead	1	<0.01(0.00)	<0.01-0.01
Lithium	1	<0.03(0.03)	<0.01-0.40
Mercury, μg/L	1	2.2(1.1)	0.7-8.6
Nickel	1	<0.01(0.003)	<0.01-0.04
Nitrogen	1	13.6(27.9)	1.1-410.0
Oil and grease	2	<3.1(1.8)	<2.0-24
Perchloroethylene	2	<0.10(0.26)	<0.01-3.80
pH, units	2		6.8-9.5
Residual chlorine	2	0.3(0.3)	<0.1-1.4
Settleable solids	2	<0.1(0.0)	all <0.1
Surfactants as MBAS[d]	1	<0.05(0.0)	<0.05-0.06
Temperature, °C	2	19.1(4.6)	6.7-26.7
Total suspended solids	1	<11(11)	<5-140
Zinc	1	0.05(0.01)	<0.02-0.11

[a]1 = 24-h composite; 2 = grab; 3 = continuous.

[b]Tabular values calculated from average monthly concentrations (N = 13).

[c]Tabular values are based on samples collected weekly, except settleable solids, which are collected monthly.

[d]MBAS = methylene-blue-reactive substances.

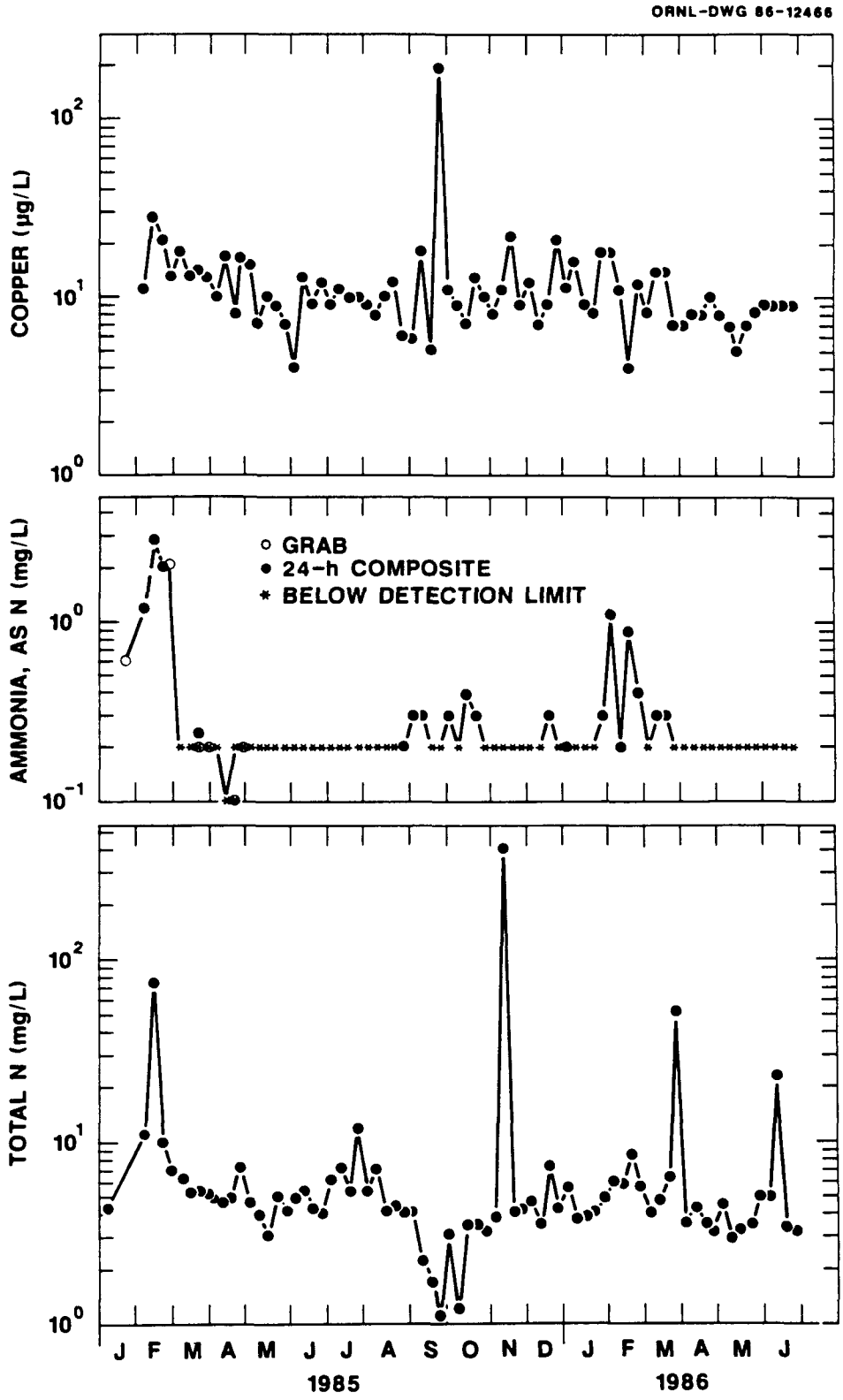

Fig. 2-5. Concentrations of copper, ammonia, and total N at the outfall of New Hope Pond (NPDES discharge station 303). Values represent 24-h composite samples collected weekly, except for that occurring before the issuance of the NPDES permit (May 24, 1985) when the frequency and type of sampling varied. Source: P. M. Pritz, Y-12 Department of Environmental Management, personnel communication, 1986.

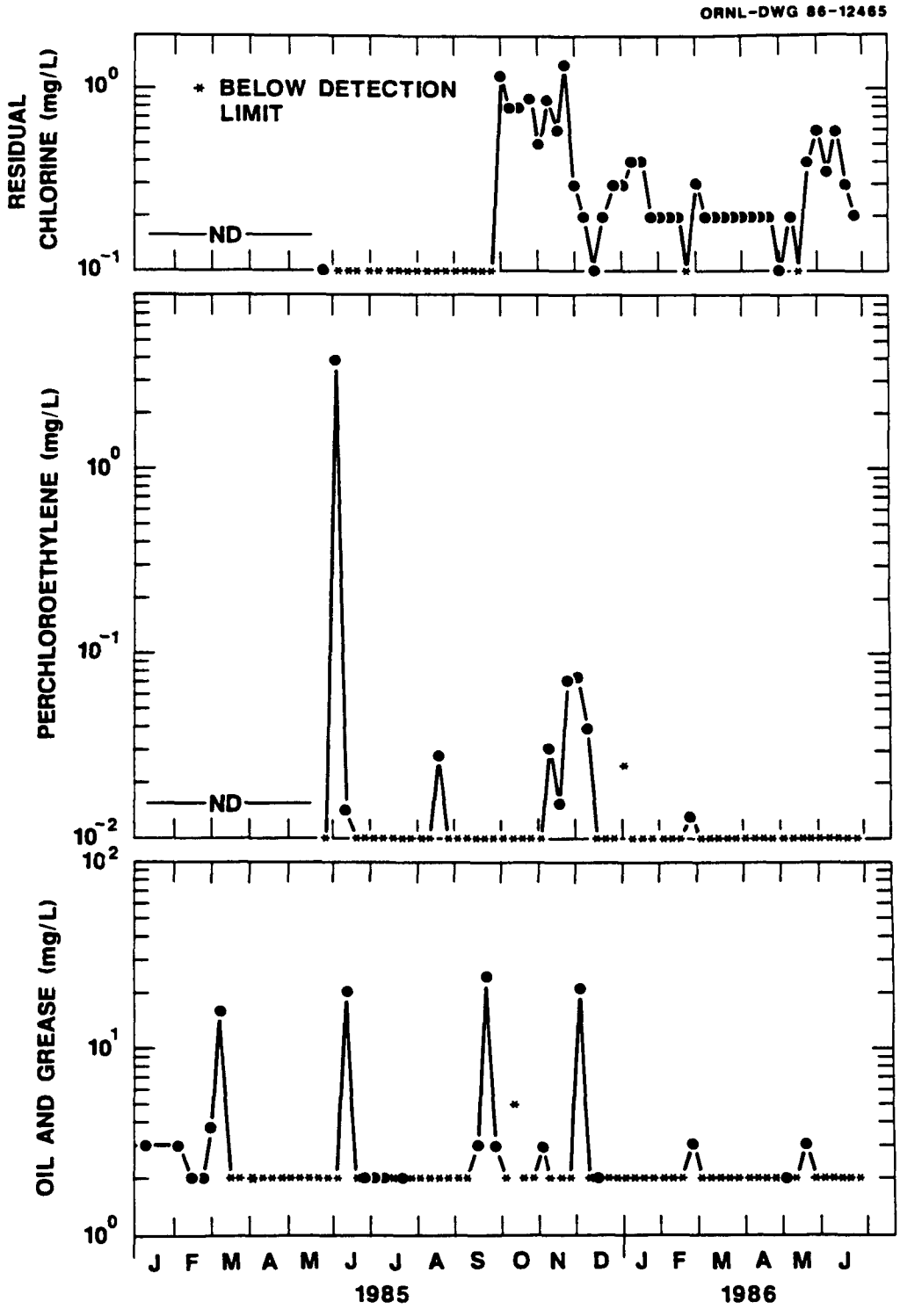

Fig. 2-6. Concentrations of residual chlorine, perchloroethylene, and oil and grease at the outfall of New Hope Pond (NPDES discharge station 303). Values represent grab samples collected weekly. ND = no data; residual chlorine and perchloroethylene were not monitored until May 24, 1985, when the new NPDES permit was issued. Source: P. M. Pritz, Y-12 Department of Environmental Management, personnel communication, 1986.

The SD of the total suspended solids concentration was also high, but a review of hourly precipitation data collected at the Oak Ridge Atmospheric Turbulence and Diffusion Laboratory (ATDL) showed that the high levels were associated with rainfall events.

The elevated levels of nitrogen and ammonia observed at the outfall of NHP in February 1985 (Fig. 2-5) were probably caused by the plant-wide use of urea, which is ~46% nitrogen, for snow removal after the supplies of bulk salt had been exhausted (T. R. Butz, Y-12 Plant Manager, personnel communication to G. H. Winebeger, 1985). The sources of other episodic increases in nitrogen concentration that occurred in November 1985 and March and June 1986 are unknown. A review of ATDL climatological data showed no snowfall or below-freezing temperatures before or on the date the samples were collected. The source of the high perchloroethylene value in June 1985 (Fig. 2-6) is also unknown. Because it would volatilize during residence in NHP, perchloroethylene concentrations above NHP were probably even greater than the concentrations measured at the outfall of the pond.

The levels of residual chlorine measured in upper EFPC were undoubtedly toxic to biota (e.g., Mattice and Zittel 1976). The dramatic increase in residual chlorine observed after September 1985 (Fig. 2-6) is probably related to a change (from colorimetric to spectrophotometric determinations) in analytical methodology (Gass 1986). Unlike the other NPDES parameters measured at the outfall of NHP, residual chlorine is measured at the inlet to the pond.

2.3.2 Ambient Temperature Regimes

Water temperatures were monitored continuously at the five primary sampling sites in EFPC below NHP (Fig. 2-1) and in BF using a Peabody Ryan Model J90 thermograph. Temperature data at 2-h intervals were keypunched into data files on the IBM 3330 system as SAS (SAS 82.4) data sets.

Annual temperature regimes at the six sites are shown in Figs. 2-7 to 2-9; data on monthly means, standard deviations, and ranges are provided in Appendix A. Water temperatures were generally 4 to 6°C higher in EFPC just below NHP than in BF. Although temperatures reached 31°C below NHP, the maximum temperature in BF was 25°C. Seasonal trends in water temperature at EFK 18.2 and EFK 13.8 were almost identical (Fig. 2-8). A longitudinal gradient of decreasing temperature was characteristic

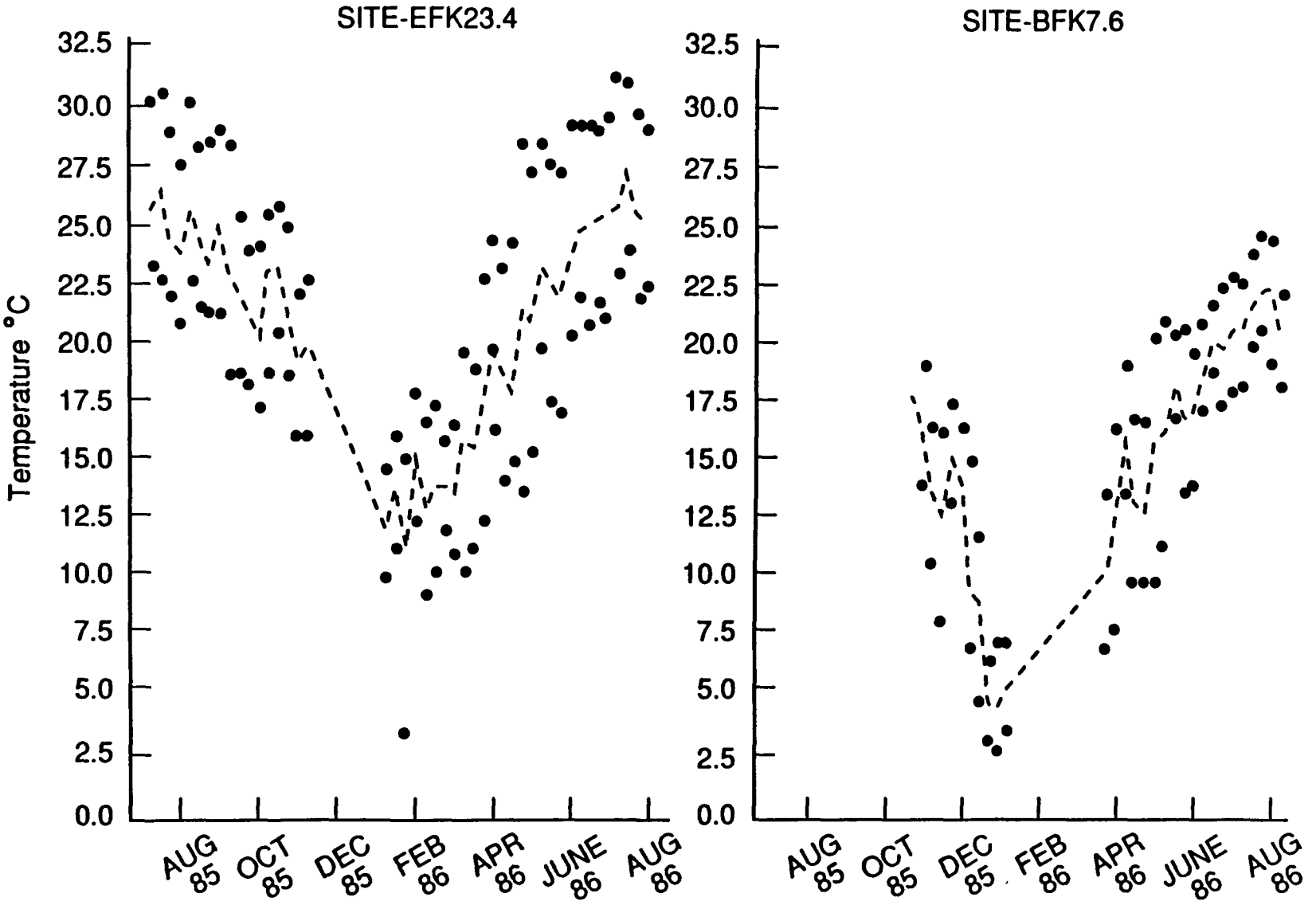

Fig. 2-7. Average weekly water temperatures (°C) in East Fork Poplar Creek below New Hope Pond and in Brushy Fork. + = maximum/minimum.

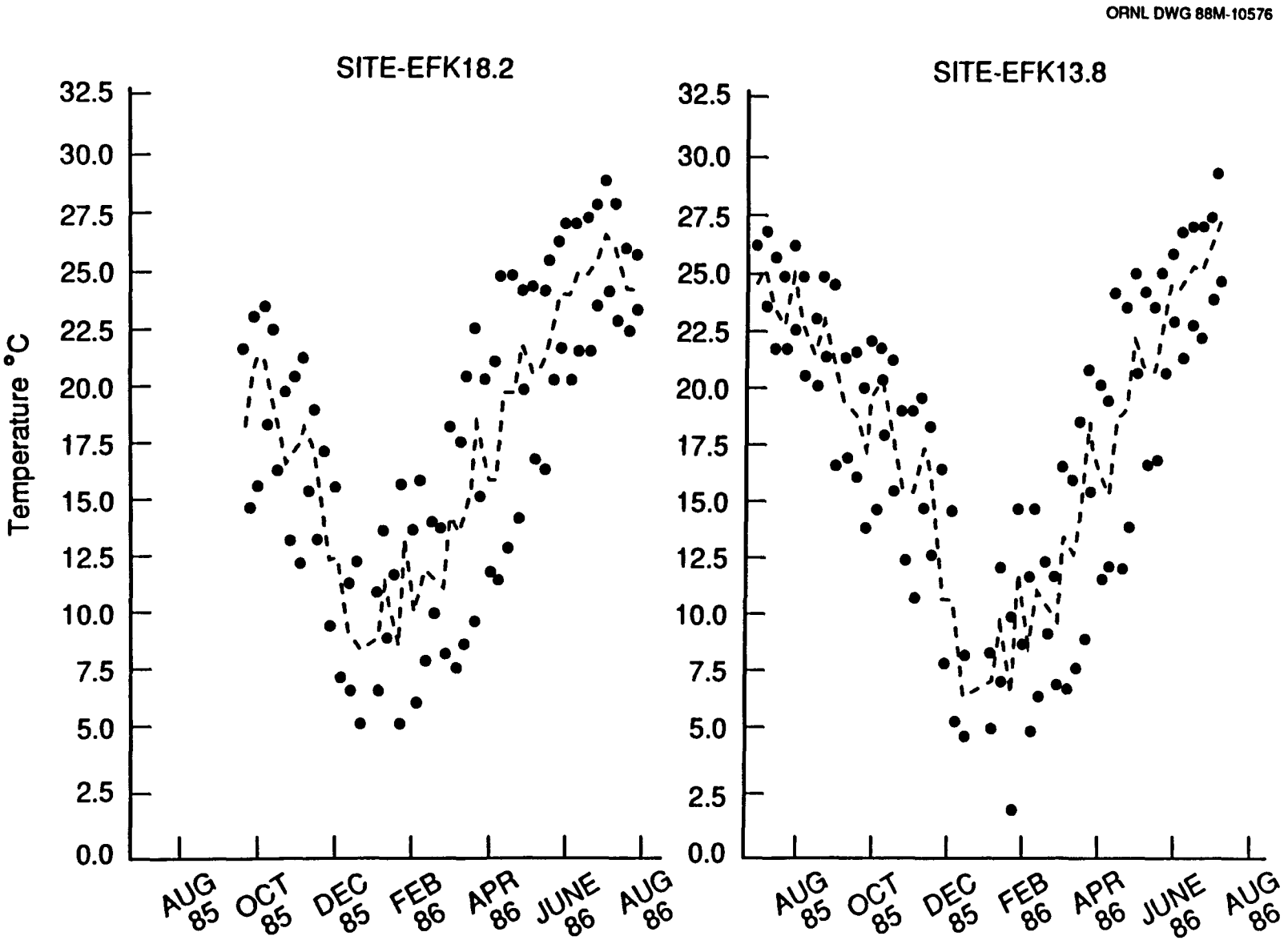

Fig. 2-8. Average weekly water temperatures (°C) in East Fork Poplar Creek at sites EFK 18.2 and EFK 13.8. + = maximum/minimum.

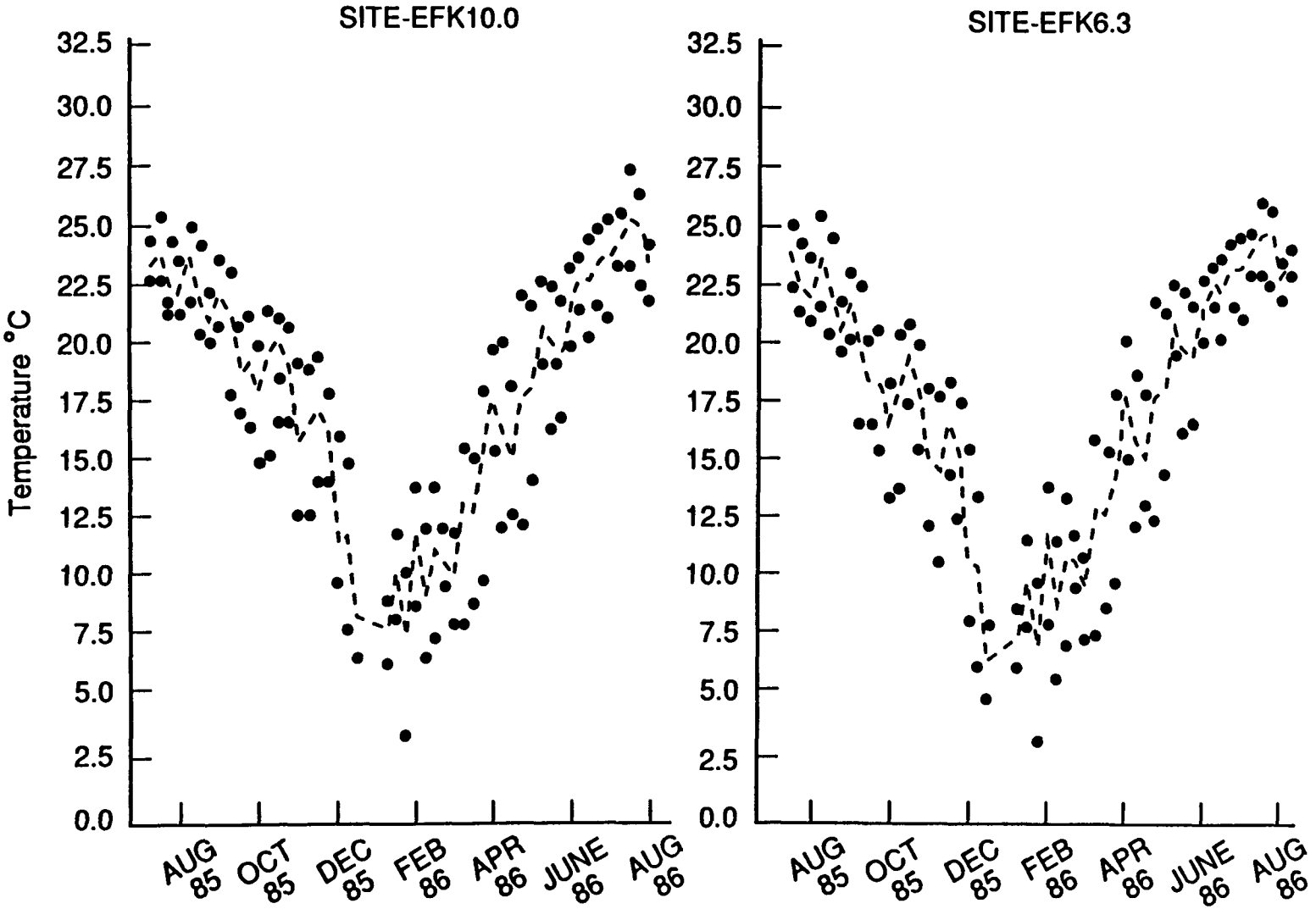

Fig. 2-9. Average weekly water temperatures (°C) in East Fork Poplar Creek at sites EFK 10.0 and EFK 6.3. + = maximum/minimum.

of EFPC; the only exceptions to this trend were the higher mean monthly temperatures in the winter at EFK 10.0 compared with EFK 13.8 and the lower maximum temperatures in the summer at EFK 10.0 compared with EFK 6.3. Such trends at EFK 10.0 may be indicative of inputs from springs, which are known to occur in this region (Sect. 2.1) and/or discharges from the ORWTF at EFK 13.4 (Fig. 2-1).

2.4 DESCRIPTION OF STUDY SITES

Six primary study sites were selected on EFPC. Criteria used in the selection of these sites included (1) location of sampling sites used in other studies, (2) known or suspected sources of downstream pollution, (3) proximity to ORR boundaries, (4) concentration of mercury in adjacent floodplain, and (5) access. The sampling sites included EFK 24.4 and EFK 23.4 (above and below NHP, respectively), EFK 18.2 located below an area of intensive commercial and limited light industrial development and just above the area of highest mercury contamination (TVA 1986, Table 11), EFK 13.8 located ~400 m above the outfall of the ORWTF, EFK 10.0 located ~900 m below the Gum Hollow Road bridge and 3.4 km below the ORWTF, and EFK 6.3 located ~1.4 km below the ORR boundary and 1.0 km above the USGS gaging station (Fig. 2-1). These sites were all routinely sampled for fish and benthic invertebrates as part of the in-stream ecological monitoring task (Sect. 6). In some cases, however, sites were excluded and/or others added, depending on the specific objectives of the various subtasks included in the BMAP. Abundances of the target fish species were also a consideration at some sites (e.g., EFK 18.2 and EFK 10.0; see Sect. 6.2).

Brushy Fork at BFK 7.6 was used as a reference stream in all four tasks of the BMAP. Additional streams off the ORR were also used as reference sites, including Beaver Creek, Hinds Creek, and the Emory River in Watts Bar Reservoir (Fig. 2-2). Extensive sampling of BF was delayed until late 1985, when preliminary results of contaminant analyses and bioindicator studies were available for all four reference sites.

The study sites on EFPC and BF can be generally described by mean width, mean depth, pool/riffle characteristics, and riparian cover. Mean width and depth are similar at the sites above and below NHP (Fig. 2-10) where EFPC flows through a rip-rap channel. At the upper site, pools are limited (except below culverts at road crossings) and velocity is generally uniform from bank to bank; vegetation is sparse, and there is no canopy.

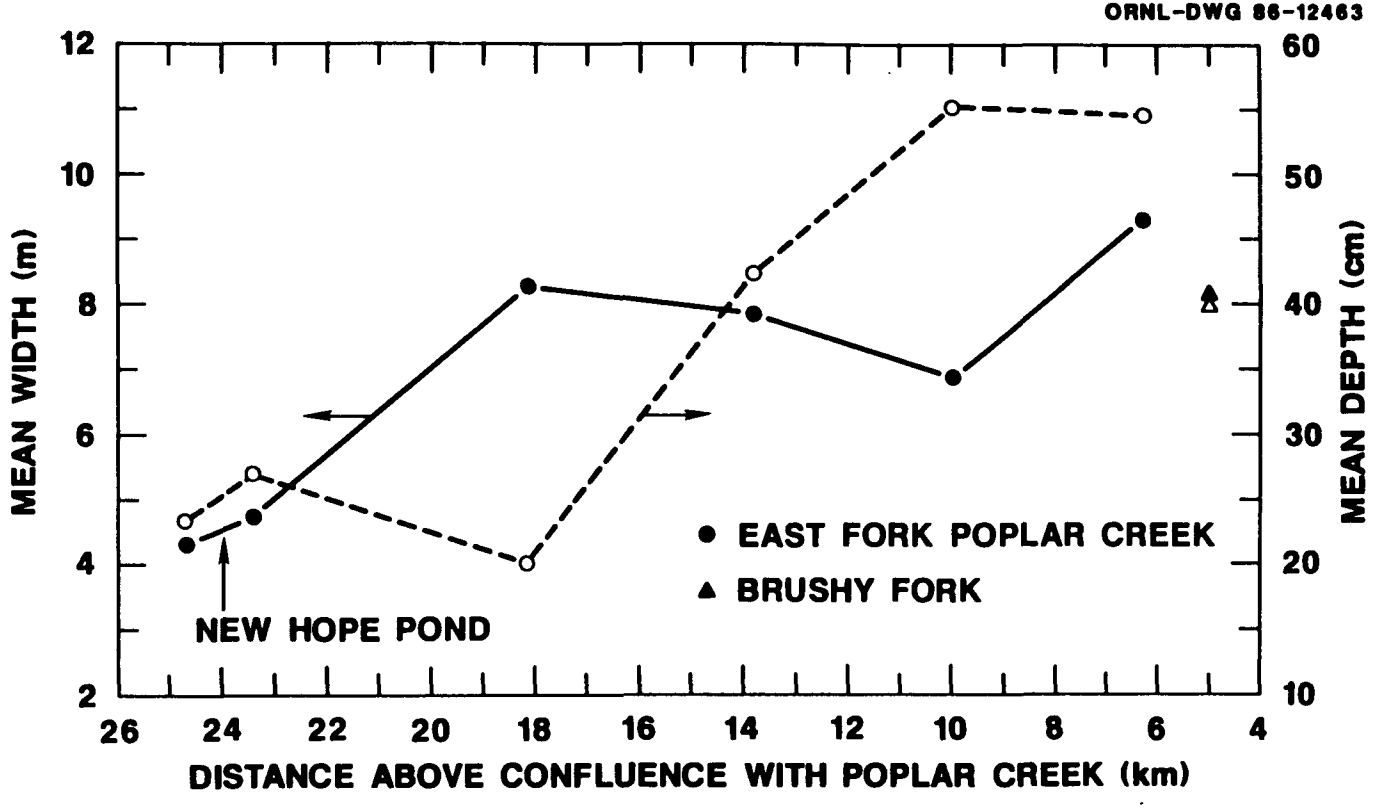

Fig. 2-10. Mean width and depth at the six primary sampling sites on East Fork Poplar Creek. Values are based on data collected from the fish sampling sections (Sect. 6.2.1).

Below NHP, velocities are relatively high, over numerous bedrock outcrops. Long riffle areas and limited pools are characteristic of the reach. Streamside vegetation is limited to a narrow strip of trees and shrubs along the stream, and the canopy varies from open to dense.

The creek at EFK 18.2 is shallow and wide, as shown in Fig. 2-10. There is a broad floodplain on the west side and high banks (2 to 3 m) on the east. Pools are very limited and a large riffle is present at the top of the reach ~25 m below the Route 95 bridge. The east bank is forested, but the west bank borders an old-field floodplain with a few scattered shrubs, primarily willows. The canopy is open except at the extreme east margins of the creek. Discarded automobile tires, bottles, and cans in the stream indicate the proximity of the site to urban development and human disturbances.

From EFK 18.2 to EFK 13.8, mean depth increases substantially. At EFK 13.8, large, deep pools are interspersed with bedrock slabs and riffles that flow over a gravel and small rubble substrate. The canopy is moderately dense with a large open area at the head of the reach. The creek above the site is bordered by large pastures, and riparian vegetation consists primarily of shrubs.

Canopy closure at EFK 10.0 is similar to that at EFK 13.8. The stream is also narrower with a greater maximum depth. The benthic invertebrate sampling site is located ~600 m upstream (EFK 10.6), where depths in June 1985 were too great for effective electroshocking of fish. A single large riffle comprises most of the riffle area at EFK 10.0. Riparian vegetation along the south bank consists of a wide strip of mixed hardwoods and shrubs adjacent to the Oak Ridge Country Club; the north bank has a narrow strip of vegetation, and single-family residences border a short segment of the reach.

The lower site (EFK 6.3) is located on the ORR and is densely forested with limited openings in the riparian canopy. Alternating riffles and deep pools between high banks are characteristic of the site. Depths are similar to those at EFK 10.0, but the mean width is greater.

Brushy Fork at BFK 7.6 is similar in size to EFK 13.8 (Fig. 2-10) and is also located adjacent to several large pastures. Narrow strips of mixed hardwoods and shrubs border the high stream banks, providing a moderately open canopy. Riffles and deep pools alternate throughout the reach.

3. TOXICITY MONITORING

3.1 EFFLUENT TOXICITY TESTING

3.1.1 Introduction

As stipulated in the National Pollutant Discharge Elimination System (NPDES) permit for the Y-12 Plant, toxicity tests are used to evaluate the acute and/or chronic toxicity of specific effluent streams discharging to East Fork Poplar Creek (EFPC) via New Hope Pond (NHP). Effluents to be monitored for toxicity include those of the cooling towers, the Central Pollution Control Facility (CPCF), the Plating Rinse Water Treatment Facility, the West End Treatment Facility, all Category IV discharges, and the Biology Wastewater Treatment Facility. Toxicity of effluent from each of these discharge points is determined using 7-d "minichronic" tests based on survival and growth of fathead minnow (*Pimephales promelas*) larvae and on survival and reproduction of a microcrustacean (*Ceriodaphnia dubia/affinis*). These tests are described in detail in Sect. 3.2.

After each test, the highest concentration of an effluent (in percentage of full-strength) causing no significant effect [the no observed effect concentration (NOEC)] is compared with the effluent's anticipated in-stream waste concentration (IWC) to determine if the effluent in question requires implementation of a Toxicity Control and Monitoring Program (TCMP). Because information on toxicity characteristics of the effluent streams given above will be available in other reports (cf. TCMP for Category IV discharges at the Y-12 Plant), only results of the program directly pertinent to biological monitoring efforts in EFPC are included here in.

3.1.2 Results and Discussion

Only two Category IV discharges were identified as particularly problematic based on considerations of NOEC + IWC. These streams, the photographic rinsewaters with a discharge of 28.4×10^3 L (7.5×10^6 gal) per year and the dye penetrant/penetrant emulsifier rinsewaters with a discharge of 7.6×10^3 L (2.0×10^6 gal) per year are being modified to reduce their toxicity. Several large treatment facilities (e.g., the Plating Rinse Water Treatment Facility and the Biology Wastewater Treatment Facility) cannot be

tested until construction is completed (Table 1-1). The CPCF also has not yet been tested.

The S-3 Pond Liquid Treatment Facility (S-3 LTF) is essentially a smaller version of the CPCF. Effluent from the S-3 LTF was tested with fathead minnow larvae and *Ceriodaphnia* and was shown to be problematic at ca. 1% of full-strength. The anticipated in-stream waste concentration of the S-3 LTF, computed at the outfall of NHP, is slightly less than 1%. A series of diagnostic toxicity tests, coupled with chemical analyses of the wastewater being treated by the S-3 LTF, showed that the toxic material in the S-3 LTF effluent was probably depleted uranium. Concentrations of uranium in effluent from the S-3 LTF were ~50 to 70 mg/L. (By comparison, wastewater entering the S-3 LTF for treatment contained about 130 mg/L.) Because reproduction of *Ceriodaphnia* was adversely affected by a uranium concentration of ~0.5 mg/L in 7-d tests, there was reason to suspect that uranium could contribute to ambient toxicity of water in the upper reaches of EFPC. Similarity in treatment schemes for the S-3 LTF and the CPCF units and similarity in composition of wastewaters being treated by these two units suggested that uranium toxicity may be more prevalent than previously recognized. Additional studies of the toxicity of effluent from the S-3 LTF and the CPCF have been initiated to evaluate this possibility.

3.2. AMBIENT TOXICITY TESTING

3.2.1 Materials and Methods

Two types of toxicity tests were used to evaluate water quality in EFPC. The first uses larvae of the fathead minnow (*Pimephales promelas*). In this test, survival and growth of newly hatched larvae in control (dechlorinated tap) water is compared with survival and growth of larvae in ambient water. The second test uses a microcrustacean (*Ceriodaphnia dubia/affinis*) and compares survival and fecundity of animals cultured in control (reconstituted hard) water with survival and fecundity of animals reared in ambient water. The fathead minnow and *Ceriodaphnia* tests are both designed to estimate chronic toxicity and are described in detail by Norberg and Mount (1985) and by Mount and Norberg (1984), respectively. Protocols for the two tests are given by Horning and Weber (1985).

Water quality parameters (pH, conductivity, alkalinity, and hardness) were normally determined for each sample collected. Some samples were also analyzed for free

and residual chlorine. Water temperature of samples that were not collected by compositer were measured in the field; temperatures were not measured in samples collected by compositors because they were cooled by ice during collection. The pH of the sample was determined with an Orion Model 811 meter equipped with a temperature-compensated combination electrode. Conductivity was determined with a YSI Model 32 temperature-correcting conductance meter, and free and residual chlorine were measured amperometrically with a Wallace and Tiernan Series A-790 titrator. Alkalinity was estimated by titrating 50 mL of unfiltered sample to pH 4.50 with 0.01 N HCl, and hardness was measured by titrating 50 mL of unfiltered sample with standard ethylenediamine tetracetic acid (EDTA) solution to an Eriochrome black T colorimetric endpoint. Alkalinity and hardness values were both expressed as mg of $CaCO_3$/L. For analyses of alkalinity and hardness, contributions resulting from suspended particulate matter were assumed to be negligible compared with the quantity of alkalinity and hardness in solution.

3.2.2 Sampling Sites and Testing Regimes

Water collected from the outfall of NHP was first tested for toxicity using fathead minnow larvae during June 26–July 3, 1985. A second series of 7-d tests was started on July 11, 1985. In the second series of tests, water samples collected from seven sites along EFPC were tested for toxicity using fathead minnow larvae. Results of the two series of tests described here were not very enlightening:

- no reduction in survival or growth of larvae was found in the first test,
- relative to controls, survival (but not growth) of larvae was significantly reduced in water collected at EFPC kilometer (EFK) 2.3 just downstream from the confluence with Bear Creek, and
- growth (but not survival) of larvae was reduced by water collected just upstream from NHP (EFK 24.0).

These two test series used water samples collected only on the first day of the test: daily renewal of the test solutions was accomplished using the "old" water, which was

stored at 7°C for the 7-d period. For these reasons, data from these tests are not included in this report.

Two more systematic testing regimes designed to determine ambient toxicity of water from EFPC were initiated early in 1986. In the first, grab samples from ten sites in EFPC downstream of NHP were collected daily for 7-d tests using *Ceriodaphnia* and fathead minnow larvae. Such tests were conducted on March 6–13 and June 26–July 3, 1986, with additional tests planned for September and December 1986. In the second regime, 24-h daily composite samples collected from the outfall of NHP were tested once each month using *Ceriodaphnia* and fathead minnow larvae. The latter testing series was initiated in February 1986. In August 1986, monthly testing at the outfall of NHP was modified to include composited samples collected both upstream from, and at the outfall of, NHP (sites EFK 24.0 and EFK 23.7, respectively). The tests of water collected by compositor from these two sites were always performed concurrently. Hence, test results in this series can be used to determine if net changes occur in toxicity of water as it passes through NHP.

3.2.3 Results

Results of the tests performed during March 6–13 and during June 26–July 3, 1986, on samples collected from 11 sites along EFPC are shown in Table 3-1; water quality measurements made on these samples are included in Table 3-2 for reference purposes. Note that samples collected from the outfall of NHP (and from EFK 24.0 upstream from NHP) were routinely taken as 24-h composites, whereas daily samples from the ten sites downstream from NHP were collected by grabs (typically at 0830 to 1000 h).

Results of monthly tests performed on water collected from the outfall of NHP are shown in Tables 3-3 and 3-4. Because the tests in February, March, and April did not show evidence of toxicity to either fathead minnow larvae or to *Ceriodaphnia* even at full strength, tests conducted in May and June used only full-strength water. In July, however, it was evident that effluent discharged from the S-3 LTF might adversely affect sensitive biota at the outfall of NHP (see Sect. 3.1). Consequently, a sampling station just upstream of NHP (at EFK 24.0) was added to the testing regime, and the August and September tests were conducted with minnow larvae and with *Ceriodaphnia* using both full-strength and 80% full-strength water from each of the two sites. Results of these tests

are shown in Table 3-4. Water quality measurements made in support of the August and September tests upstream from, and at the outfall of, NHP are shown in Table 3-5.

Table 3-1. Toxicity test results for water samples collected daily from ten sites on East Fork Poplar Creek in 1986

Site	March 6–March 13				June 26–July 3			
	Ceriodaphnia		Fathead minnow		*Ceriodaphnia*		Fathead minnow	
	Surv.[a]	Repro.[b]	Surv.	Growth[c]	Surv.[a]	Repro.[b]	Surv.	Growth[c]
EFK 23.7	100	27.6	90.0	0.564	90	27.8	97.5	0.432
EFK 22.8	100	28.9	92.5	0.382[d]	100	24.8	92.5	0.347[d]
EFK 21.9	100	30.3	95.0	0.515	100	21.5	92.5	0.355[d]
EFK 20.5	100	30.6	92.5	0.536	100	22.8	92.7	0.197[d]
EFK 18.2	100	29.8	92.5	0.425[d]	90	21.4	100.0	0.433
EFK 16.1	90	37.0	87.5	0.481	100	25.1	95.0	0.440
EFK 13.8	90	34.5	90.0	0.583	90	24.5	90.0	0.426
EFK 10.9	100	23.3	90.0	0.468	100	16.4	97.5	0.517
EFK 7.6	100	30.2	80.0	0.495	100	25.1	97.5	0.266[d]
EFK 5.1	90	28.6	85.0	0.479	100	19.2	95.0	0.533
EFK 2.1	100	28.6	87.5	0.639	100	34.0	90.0	0.384
Control	100	28.6	100.0	0.571	90	14.2	95.0	0.553

[a]Percentage survival is based on ten replicates (one animal per replicate) for *Ceriodaphnia*, and on four replicates (ten animals per replicate) for fathead minnow larvae.

[b]Reproduction of *Ceriodaphnia* is the mean number of offspring per female for the 7-d test, with reproduction of females that die before leaving offspring being set to zero.

[c]Growth of fathead larvae (average mg per animal over the 7-d test) is corrected for the initial average weight of the larvae based on a representative subsample of larvae at the start of the test.

[d]Test endpoints that differed significantly from controls [$p < 0.05$; SAS-GLM (ANOVA) followed by Dunnett's one-tailed test].

Table 3-2. Water quality (means of seven daily measurements) for samples taken from the outfall of New Hope Pond (EFK 23.7) and for ten sites farther downstream in East Fork Poplar Creek for the periods March 6–March 13 and June 26–July 3, 1986

Site	March 6–March 13				June 26–July 3			
	pH	Cond.[a]	Alk.[b]	Hard.[b]	pH	Cond.[a]	Alk.[b]	Hard.[b]
EFK 23.7	8.25	353	107.0	166.0	8.23	371	115.0	160.0
EFK 22.8	8.16	361	110.6	163.4	8.12	363	115.1	174.6
EFK 21.9	8.15	365	113.5	168.9	8.16	345	117.6	171.9
EFK 20.5	8.15	358	116.1	173.4	8.19	348	118.2	174.3
EFK 18.2	8.11	360	120.9	171.0	8.21	367	120.9	170.9
EFK 16.1	8.11	335	118.2	166.4	8.21	357	121.1	173.7
EFK 13.8	8.17	341	124.7	168.4	8.24	364	121.3	176.0
EFK 10.9	8.20	365	131.6	168.4	8.14	412	124.5	175.7
EFK 7.6	8.20	340	127.3	161.1	8.16	385	124.9	174.6
EFK 5.1	8.23	331	126.6	159.4	8.20	378	125.5	173.7
EFK 2.1	8.20	326	127.9	162.3	8.17	340	119.8	169.1

[a]Conductivity = μmho/cm corrected to 20°C.
[b]Alkalinity and hardness values are given as mg $CaCO_3$/L.

Table 3-3. Summary of toxicity test results of water from the outfall of New Hope Pond for six dates in 1986

	February 6–13	March 6–13	April 3–10	May 1–8	June 26–July 3	August 7–14
			Ceriodaphnia[a]			
Controls	24.1 + 7.7	28.6 + 5.3	30.2 + 4.7	13.4 + 4.5	14.0 + 6.4	13.2 + 3.3
10% water	17.7 + 7.3	29.4 + 3.6	30.9 + 5.8			
20% water	19.2 + 7.3	25.3 + 6.8	31.0 + 6.1			
40% water	19.8 + 6.3	27.4 + 4.8	29.6 + 9.1			
60% water	23.1 + 4.2	29.8 + 4.5	31.4 + 3.7			
80% water	20.7 + 2.1	27.7 + 5.2	31.6 + 8.1			13.2 + 3.0
100% water	19.3 + 2.4	27.6 + 3.3	36.0 + 8.0	9.6 + 8.0	27.8 + 14.3	14.8 + 2.6
			Fathead minnow[b]			
Controls	92.3	100.0	92.3	85.3	97.4	92.5
10% water	91.9	100.0	97.5			
20% water	78.0	95.0	100.0			
40% water	82.5	95.0	95.0			
60% water	53.8	95.0	90.0			
80% water	66.7	95.0	90.0			92.5
100% water	60.5	100.0	80.0	60.6[c]	99.4	97.5

[a]Data for the *Ceriodaphnia* tests are expressed as the average (± 1 SD) number of offspring per female during the 7-d test period, with mortalities (if any) of females being set to zero.

[b]Fathead minnow data are expressed as the average percentage survival of the larvae in the indicated concentration of water from New Hope Pond outfall.

[c]Values that are significantly lower ($p < 0.05$) than the control based on analysis of variance (SAS-GLM).

Table 3-4. Comparison of toxicity of water sampled daily via 24-h compositors collected upstream from, and at the outfall of, New Hope Pond

Site	August 7–14		September 4–11	
	Survival (%)	Offspring[a]	Survival (%)	Offspring[a]
	Ceriodaphnia toxicity test			
Controls	100.0	13.2 + 3.3	100.0	20.4 + 5.5
Upstream, 80%	90.0	5.9 + 4.5[b]	100.0	13.6 + 4.5[b]
Upstream, 100%	90.0	4.4 + 4.5[b]	80.0	11.7 + 4.6[b]
Outfall, 80%	90.0	13.2 + 3.0	70.0	13.1 + 9.6[b]
Outfall, 100%	100.0	14.8 + 2.6	100.0	18.0 + 3.5
	Fathead minnow larvae test			
	Survival (%)	Growth[c]	Survival (%)	Growth[c]
Controls	92.5	0.53 + 0.05	92.5	0.36 + 0.07
Upstream, 80%	92.5	0.53 + 0.10	95.0	0.29 + 0.10
Upstream, 100%	90.0	0.41 + 0.07	92.5	0.26 + 0.08
Outfall, 80%	90.0	0.45 + 0.04	85.0	0.26 + 0.08
Outfall, 100%	97.5	0.50 + 0.06	75.0	0.26 + 0.07

[a]Reproduction by adults that died before producing offspring was set to zero.

[b]Samples that significantly lowered reproduction of *Ceriodaphnia* relative to controls [ANOVA, followed by Dunnett's test; $(4,45)F = 17.16$, $p < 0.001$].

[c]Seven-day growth (mean dry weight per larva $\pm$ 1 SD), corrected for weight of larvae at the start of the test. No significant differences were found ($p = 0.101$; $(4,45)F = 2.35$).

Table 3-5. Comparison of water quality parameters for samples collected upstream from, and at the outfall of, New Hope Pond

	August 8–14, 1986, analyses[a,b]		September 4–10, 1986, analyses[a,b]	
	Upstream	Outfall	Upstream	Outfall
pH	7.97–8.26 (8.17 ± 0.10)	8.07–8.74 (8.43 ± 0.28)	8.05–8.22 (8.14 ± 0.07)	8.02–8.62 (8.34 ± 0.22)
Conductivity[c]	294–422 (343 ± 45)	301–363 (336 ± 23)	352–384 (368 ± 12)	335–443 (386 ± 34)
Alkalinity[d]	97.5–116.0 (109.1 ± 5.7)	100.5–114.0 (110.5 ± 5.3)	104.0–116.5 (111.0 ± 3.8)	99.0–117.0 (110.3 ± 6.2)
Hardness[d]	116.0–184.0 (159.1 ± 23.1)	152.0–180.0 (166.3 ± 12.3)	148.0–190.0 (168.6 ± 14.0)	152.0–202.0 (176.0 ± 15.7)
Total chlorine[e]	0.01–0.07 (0.04 ± 0.02)		0.06–0.24	0.00–0.00 (0.13 ± 0.07)
Free chlorine[e]	0.00–0.04 (0.02 ± 0.02)		0.00–0.20 (0.06 ± 0.08)	

[a]Water sampled daily via 24-h compositors.

[b]Ranges are given for each parameter; means (± 1 SD) for each factor are shown in parentheses beneath the range.

[c]μmho/cm, corrected to 25°C.

[d]mg/L, expressed as $CaCO_3$.

[e]mg/L, determined amperometrically.

3.2.4 Discussion

Water collected by compositor at the outfall of NHP was not chronically toxic to *Ceriodaphnia* or to fathead minnow larvae in the tests conducted monthly from February through September 1986 (Tables 3-3 and 3-4), and there was little systematic evidence for chronically toxic conditions farther downstream in EFPC in tests that used daily renewal of water from each of ten sites (Table 3-1).

Grab samples from some sites downstream from the outfall of NHP (e.g., EFK 22.8, EFK 21.9, EFK 20.5, and EFK 7.6 during June 26–July 3; EFK 22.8 and EFK 7.6 during March 6–13) reduced growth of fathead minnow larvae relative to controls, but did not reduce survival of the larvae or fecundity or survival of *Ceriodaphnia*. Curiously, even when water from EFPC sites downstream from the outfall of NHP was "toxic" to fathead

larvae (as evidenced by reduced rates of growth for larvae reared in water from sites EFK 22.8, EFK 21.9, and EFK 20.5 in the June-July tests), no such reduction was noted for samples taken directly from the outfall of NHP, where water should have been at least equally toxic. Three possibilities could explain this finding: (1) the composited sample may lose toxicitybecause it is chilled during collection, (2) the sample may lose toxicity as it "ages" during the 24-h compositing period, and/or (3) water from the outfall of NHP may be more toxic at night than during the day. In the latter case, 24-h compositing would dilute toxicity relative to grab samples because grab samples were routinely collected in early morning. Because of the time lags involved with the movement of water through NHP (and downstream in EFPC), early morning grab samples would include a larger proportion of "night" water exiting from NHP. Alternatively, the "toxicity" downstream in EFPC (evidenced by reductions in growth of fathead minnow larvae) may be a statistical artifact originating from one or more unknown sources of bias within the test procedures.

Water collected at EFK 24.0 upstream of NHP was more toxic to *Ceriodaphnia* (relative to both controls and to water taken at the outfall of NHP) during August 7–14 than it was during similar tests conducted in September (Table 3-4). In both August and September, however, water that clearly lowered reproduction of *Ceriodaphnia* (relative to both controls and to water collected from the outfall of NHP) showed no evidence of toxicity to fathead minnow larvae and did not significantly reduce survival of *Ceriodaphnia*. These data suggest that water quality improves biologically as it passes through NHP. Results of water quality measurements above and below NHP (Table 3-5) showed that values for chlorine (free and total residual) decline and that pH increases by passage through the pond. At the site upstream from NHP (EFK 24.0), however, chlorine concentrations were higher in September than in August, but toxicity of the water to *Ceriodaphnia* was lower in September than it was in August. Chlorine is therefore unlikely to be the agent that reduced reproduction of *Ceriodaphnia* at EFK 24.0.

The slight elevation in pH of water at the outfall of NHP (average of 0.26 and 0.20 units in August and September, respectively, relative to water entering the pond; Table 3-5) was probably the result of high net productivity of submersed aquatic plants in the pond. Increases in pH of this magnitude are certainly not great enough to markedly

affect reproduction of *Ceriodaphnia* and are probably not large enough to indirectly alter toxicity by affecting speciation of potential toxicants.

Acute and chronic tests performed in support of the TCMP showed that one or more materials in the effluent from the S-3 LTF affected reproduction of *Ceriodaphnia* at low concentrations (at about 1% of full-strength effluent) but affected survival and growth of fathead minnow larvae and survival of *Ceriodaphnia* only at much higher concentrations. The August and September tests comparing toxicity of water upstream and downstream from NHP also affected *Ceriodaphnia* reproduction more than either *Ceriodaphnia* survival or fathead minnow larvae growth or survival. However, the toxic agent(s) responsible for reducing fecundity of *Ceriodaphnia* upstream from NHP in August may have no relation to those discharged from the S-3 LTF: concentrations of uranium in EFPC upstream from NHP in August were ~0.010 to 0.013 mg/L, whereas the 7-d chronic no observed effect concentration (NOEC) of this metal for *Ceriodaphnia* reproduction and for fathead minnow larvae survival was higher (~0.5 and 5.2 mg/L, respectively; Kszos and Stewart, unpublished data). The *Ceriodaphnia* bioassay is more sensitive than the fathead minnow larvae test to many toxicants, and reproduction is on average a more sensitive indicator of chronic toxicity than growth or survival (cf. Suter et al. 1987). Finally, the statistical power of the *Ceriodaphnia* test is considerably greater than that associated with the fathead minnow test by virtue of the number of replicates used. For these reasons, the *Ceriodaphnia* test system frequently can "see" toxicity that remains obscure or invisible to the larval fathead minnow test system.

The toxicity tests of ambient stream water reported here provide interesting challenges with respect to their collective interpretation. When viewed in conjunction with results of toxicity tests of effluent streams entering EFPC upstream from NHP and those of the water quality analyses and the other biological monitoring parameters measured in EFPC downstream from NHP, the ambient tests are more definitive. For example, toxicity of water upstream from NHP in August (Table 3-4) was clearly not the result of chlorine levels or changes in pH, alkalinity, hardness, or conductivity (Table 3-5). Similarly, results of the ambient toxicity tests, the biological monitoring of invertebrates and fish in EFPC (Sects. 6.1 and 6.2, respectively), and the periphyton studies (Sect. 3.3) are in general agreement in other areas: (1) the major impacts on EFPC are more obvious upstream from NHP than downstream, (2) based on fathead minnow test results

and on those of biological monitoring of fish, invertebrates, and periphyton, some reduction in toxicity occurs with distance downstream from the outfall of NHP, and (3) a secondary source of impacts to the stream (based on slight reductions in fecundity of *Ceriodaphnia*, reductions in carbon uptake and chlorophyll analyses of periphyton, and benthic invertebrate sampling) occurs between EFK 13.8 and EFK 10.9, probably in association with discharges to the stream by the City of Oak Ridge Wastewater Treatment Facility (ORWTF). A major advantage of ambient toxicity tests made obvious through the Y-12 Plant Biological Monitoring and Abatement Program (BMAP) is that a dozen or more sites can be evaluated simultaneously at frequent intervals with test systems having considerable sensitivity and statistical power; the data from such tests are also available soon after the tests are completed. When compared with invertebrates and fish inhabiting the stream, the test systems have the major disadvantage of being somewhat less sensitive indicators of ecologically adverse conditions, possibly because 7-d exposure periods are used in the tests, whereas in-stream communities are exposed to potential toxicants for considerably longer periods.

3.3 PERIPHYTON STUDIES

3.3.1 Introduction

Algae and microbes attached to benthic surfaces act as energy transducers in streams: algae convert radiant energy and nutrients into biomass, and bacteria and fungi release nutrients that would otherwise be tied up into organic matter. Stream microbiota, therefore, regulate fluxes of energy and nutrients to higher trophic levels.

To date, monitoring of the periphyton/microbial communities in EFPC has focused largely on evaluating and modifying the methodology used to monitor the algal component of the periphyton. The use of uniform substrates to monitor algal growth rates, standing crop, and production has been abandoned in favor of natural substrates. Although uniform substrata (ceramic tiles) control for substratum type and facilitate the removal of algal biomass for other measurements, it was found that (1) ceramic tiles had to be incubated in situ for relatively long periods (>6 weeks) before they developed substantial standing crops of algae (measured as chlorophyll *a*) and (2) the algal community that developed on the artificial surfaces often differed from that found on natural substrates at the same site. Methodological difficulties involved in the quantification of periphyton

production in situ have been overcome by using short-term laboratory measurements of carbon assimilation. These measurements are made at monthly intervals for substrates collected from each study site. Recent modifications in the lighting and water circulation systems in the laboratory have further improved the utility of the carbon assimilation approach.

Concentrations of potential toxicants in EFPC water downstream from NHP may be biased by procedures routinely used for sampling in which water samples are collected by compositors. The sampling port of the compositors consists of a plastic sleeve perforated with many holes (~5 mm in diameter). These holes are designed to exclude coarse particulate organic matter (CPOM) that would, if allowed to enter the compositor, clog and disrupt the flow of water in the sampling line. Sampling by compositors may underestimate some water quality parameters when there is considerable downstream drift of CPOM. Collections and analyses of CPOM at the outfall and downstream from NHP were made to evaluate this possibility.

3.3.2 Materials and Methods

A survey of ten sites between EFK 23.4 and EFK 6.3 was conducted in April 1986. This survey suggested that the algal biomass at four monitoring sites (EFK 23.4, EFK 13.8, EFK 10.6, and EFK 6.3) was reasonably representative of major regions of the stream. These four sites coincide with fish and invertebrate sampling stations and were selected for routine monitoring of periphyton. A periphyton/microbial community monitoring site was also established in Brushy Fork (BF) at BFK 7.6.

For the measurement of algal biomass and production, small relatively flat rocks (10 to 60 cm^2) were collected from each site and transported to the laboratory in plastic containers filled with water from the site. In the laboratory, four rocks from a given site were placed in an incubation chamber containing water from the site and 10 mCi of $NaH^{14}CO_3$. During the subsequent 2-h incubation period, water in each incubation chamber was recirculated rapidly with a submersible pump to simulate natural flow regimes. Overhead lighting (~300 $\mu E/m^2/s$ as photosynthetically active radiation) was provided throughout the incubation period via 500-W metal halide and sodium vapor lamps. Water temperature in the chambers during the incubation phase was maintained close to that of the ambient stream with a water bath. After incubation, the rocks were

rinsed twice in distilled water to remove residual inorganic carbon and each was placed into a separate container with 30 mL of dimethyl sulfoxide (DMSO). Chlorophyll and other soluble organic compounds were extracted into the DMSO for 24 h in darkness (Filbin and Hough 1984). The contents of each container 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 chlorophyll in the acetone-DMSO mixture was determined spectrophotometrically (Jeffrey and Humphrey 1975). Chlorophyll values were corrected for phaeopigments (Strickland and Parsons 1972). A 0.5-mL aliquot of each extract was also radioassayed for ^{14}C by liquid scintillation spectrophotometry. The surface area of each rock was determined by covering the upper surface of the rock with aluminum foil and then weighing the foil; the foil weight per unit area was determined separately for each test to permit reliable estimates of the rock surface area. Chlorophyll *a* and the rate of carbon incorporation were expressed per unit of rock surface area.

To evaluate the possibility that the sampling method underestimated the quantity of heavy metals being exported from NHP in particulate form, CPOM was collected from four sites downstream from NHP on September 3, 1986. The sites were EFK 23.7 (30 m below the outfall of NHP), EFK 22.8, EFK 21.9, and EFK 18.2. At each site, two drift-net samplers (mesh size = 363 μm) were placed in the stream for 1.5 to 8.0 min, depending on the flow and the amount of material being collected. The volume of water filtered was computed from the water velocity (estimated by timing floating objects over 5- or 10-m lengths of stream), the cross-sectional area of the net opening, and the length of the collection period. The CPOM sample was rinsed into containers, dried at 105°C, weighed, and analyzed for various metals. Bulk water samples were also collected at sites EFK 23.7, EFK 21.9, and EFK 18.2 and filtered (0.5 μm, glass fiber) prior to analysis of the same suite of metals. Finally, triplicate periphyton samples were scraped from the surfaces of rocks at EFK 23.4 km on the same date. Two of the samples were analyzed for metals (ICP scans plus uranium); the third was used to provide an estimate of organic vs inorganic material in the periphyton community by weight loss on ignition.

3.3.3 Results and Discussion

Periphyton chlorophyll *a* provided a measure of algal biomass at each of the monitoring sites (Fig. 3-1). Seasonal trends were associated with changes in light penetration. In March, for example, all sites received full sun because riparian vegetation was largely leafless, but by May, only site EFK 23.4 received full sun; the others were well shaded because of nearly closed canopies. Before canopy closure, all sites in EFPC were similar with respect to algal biomass. Decreases in algal biomass at some sites between the March and May sampling periods were probably the result of canopy closure, which increased shading. After canopy closure, periphyton biomass in upstream reaches of EFPC was greater than the biomass at the downstream sites which, in turn, was similar to that observed in BF, the reference site. On all sampling dates, all of the EFPC monitoring sites had algal biomass values that were equal to, or greater than, those in BF. Algal biomass in BF was similar to that of other, relatively undisturbed local streams, including upper White Oak Creek, Melton Branch, and First Creek near ORNL (H. L. Boston, unpublished data).

Periphyton productivity (as carbon uptake) showed a seasonal pattern similar to that of algal biomass (Fig. 3-2). Production rates in EFPC were similar to, or greater than, production rates of algae in other local streams described above (H. L. Boston, unpublished data). Although biomass at EFK 23.4 was relatively high, production by periphyton at this site was relatively low compared with sites farther downstream. By midsummer, productivity and biomass of periphyton at the various sites in EFPC showed similar patterns, with both parameters being higher in the more upstream reaches. Chlorophyll *a* and carbon uptake per unit area were substantially lower at EFK 10.6 downstream from ORWTF compared with EFK 13.8, located upstream of the facility.

The quantity of CPOM collected with drift nets declined rapidly downstream from the outfall of NHP (Table 3-6), and differences in CPOM composition between the four EFPC sites were visually striking. At the outfall of NHP, *Potamogeton*, *Najas*, and filamentous algae contributed approximately 50%, 35%, and 10%, respectively, to CPOM total dry weight; small snails, leeches, feathers (from geese that frequent the pond), and

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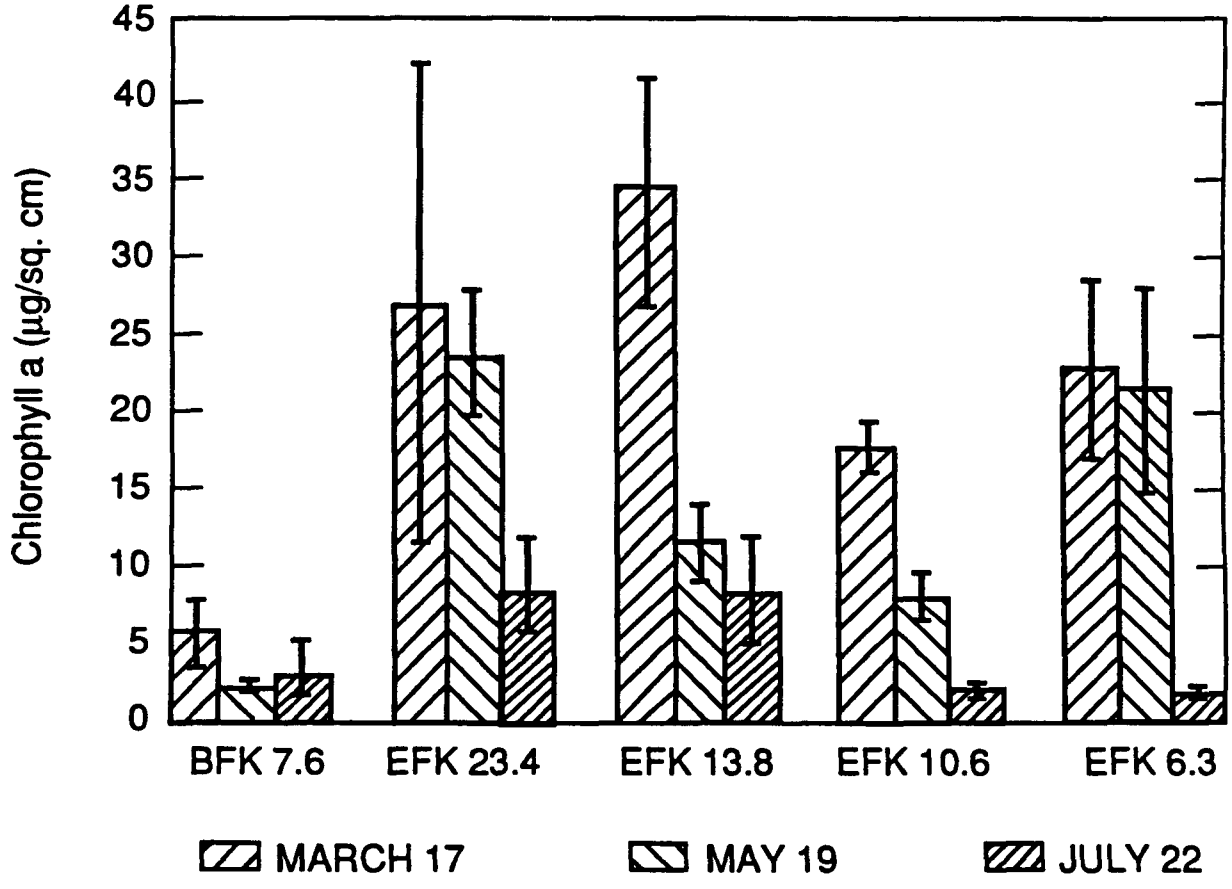

Fig. 3-1. Periphyton chlorophyll *a*, corrected for phaeopigments, on small rocks collected on three dates in 1986 from four sites in East Fork Poplar Creek and a reference site in nearby Brushy Fork Creek. Values shown are the mean ± 1 S.D., for four rocks, as mg/cm^2 of chlorophyll *a* per upper rock surface. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

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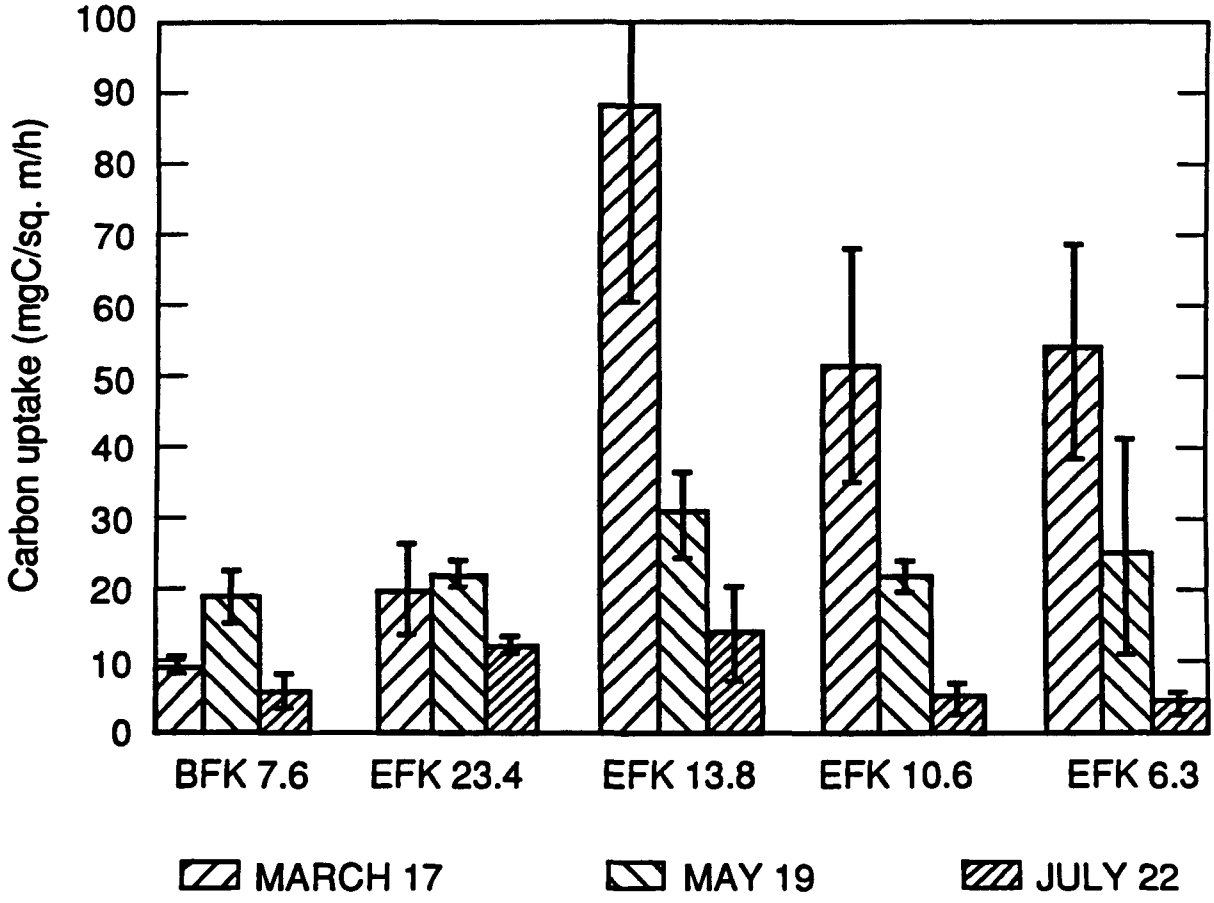

Fig. 3-2. Periphyton carbon uptake rates from laboratory ^{14}C uptake studies for periphyton on small rocks collected on three dates in 1986 from four sites in East Fork Poplar Creek and a reference site in nearby Brushy Fork Creek. Values shown are the mean $\pm$ 1 S.D., for four rocks, as mg/m^2 of carbon per upper rock surface per hour. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

Table 3-6. Mean concentrations of coarse particulate organic matter (CPOM <63-mm particle size) and selected metals in CPOM from four sites in East Fork Poplar Creek. Means were computed from duplicate water samples taken at each site. Concentrations of CPOM are expressed in mg/m^3, dry wt and concentrations of metals are expressed as mg/g CPOM, dry wt. Samples of CPOM were collected with drift nets on September 3, 1986

Site	CPOM	Cadmium	Chromium	Copper	Nickel	Zinc
EFK 23.7[a]	1,356	6.9	8.8	145	21.5	820
EFK 22.8	198	5.1	5.9	87	16.0	475
EFK 21.9	72	20.0	60.5	315	84.0	1,285
EFK 18.2	51	8.0	35.5	120	39.0	4,502

[a]Outfall of New Hope Pond.

fragments of various emergent aquatic plants contributed an estimated 5%. At EFK 18.2 (5.5 km downstream from NHP), *Potamogeton* and *Najas* were absent and CPOM was dominated instead by allochthonous matter, primarily tree leaves and twigs. Concentrations of metals of toxicological interest in the CPOM collected from the four sampling sites are also given in Table 3-6. These analyses showed that (1) the types and quantities of autochthonous material available to invertebrates and decomposers in a 2.0-km reach of EFPC immediately downstream from NHP is, at least seasonally, very much influenced by NHP, (2) the types and quantities of CPOM available to decomposers and invertebrates in this segment of the stream change rapidly with distance downstream, and (3) sampling procedures that routinely exclude CPOM may in certain circumstances generate biases that might complicate our ability to interpret patterns of community structure and/or function.

3.4 FUTURE STUDIES

Results of the ambient toxicity tests on EFPC to date suggest that the utility of such tests can be maximized if they are used largely to monitor toxicity at the outfall of, and upstream from, NHP where a toxicity gradient is more evident. For this reason, evaluations of ambient toxicity will be conducted more frequently (at monthly or biomonthly intervals) above and immediately below NHP. Sites farther downstream on EFPC will be tested quarterly using both *Ceriodaphnia* and the fathead minnow.

Additional emphasis will be given to the use of periphyton communities, both as monitoring and experimental units, because these communities have the advantage of being logistically tractable, encompass natural exposure regimes, and serve as a major food source to higher trophic levels. Finally, it is clear that biota in NHP can influence water quality parameters (e.g., CPOM and pH) and thus affect biota farther downstream. Therefore, sampling of CPOM will be continued to determine the seasonal variation in the amount and quality of CPOM available to biota in the upper reaches of EFPC.

4. BIOACCUMULATION STUDIES

4.1 ACCUMULATION OF CONTAMINANTS BY BIOTA IN EAST FORK POPLAR CREEK

4.1.1 Introduction

The problems of chemical contamination of water, sediments, and biota in East Fork Poplar Creek (EFPC) have been examined in a number of recent reports and studies (Van Winkle et al. 1984; TVA 1985a-e). Management of the Y-12 Plant is committed to reducing releases of substances, such as mercury and polychlorinated biphenyls (PCBs), to levels that will ensure that biota in EFPC are safe for human consumption. The bioaccumulation component of the Biological Monitoring and Abatement Program (BMAP) (Loar et al., 1989) is designed to monitor those substances (mercury, PCBs) known to accumulate to undesirable levels in EFPC fish to evaluate the effectiveness of remedial actions taken within the Y-12 Plant. A broad suite of inorganic and organic pollutants is also monitored in EFPC biota to ensure that no other substances accumulate to unacceptable levels in biota as new treatment processes are developed and new treated waste streams are discharged to the creek. This first annual report presents monitoring data on mercury, PCBs, ^{137}Cs, and polycyclic aromatic hydrocarbons (PAHs). The monitoring was conducted to provide a record of the time course of these contaminants in fish. A broad spectrum of inorganic and organic priority pollutants in fish (*Lepomis macrochirus*) and invertebrates (*Corbicula fluminea*) was also monitored in 1986, and results of these analyses are discussed in this report.

4.1.2 Methods

Fish were collected for contaminant analyses from five sites in EFPC and three reference streams (Brushy Fork, Hinds Creek, and Beaver Creek; Fig. 2-2). Four of the EFPC sites coincided with sites used for the quantitative fish population surveys: EFPC kilometer (EFK) 23.4, EFK 18.2, EFK 13.8, and EFK 6.3. The fifth site was located at EFK 2.1, ~300 m downstream from the mouth of Bear Creek. Sites were selected to describe the longitudinal pattern of accumulation of contaminants throughout the length of EFPC and to coincide with sample locations used in studies conducted in 1982 (Van Winkle et al. 1984) and 1984 (TVA 1985e) of mercury contamination in EFPC fish.

Several reference streams were included in the study design to obtain more-representative estimates of background levels of mercury and PCBs in rural streams similar in size to EFPC and to identify a single stream where all three species were abundant.

Fish were collected for contaminant analyses by electrofishing, beginning at the upper boundary of the population survey reach (Sect. 6.2.2.1) and continuing upstream until a suitable number of fish were collected or a distance of ~800 m was covered. If adequate numbers of fish were not obtained in this effort, electrofishing was initiated ~400 m downstream from the lower boundary of the population survey reach and proceeded upstream to that boundary.

Collections were initiated in May 1985, and were made at approximately 6-month intervals thereafter. The three sets of fish collections presented in this report are identified by the month and year the collections were initiated–May 1985, December 1985, and May 1986. Eight fish of each of three species (redbreast sunfish, *Lepomis auritus*; bluegill sunfish, *L. macrochirus*; and carp, *Cyprinus carpio*) were collected at each site, if possible. Redbreast sunfish were generally collected within the first 400 m of electrofishing at each site. Densities of bluegill in the middle reaches of the stream and of carp in the upper and lower reaches of stream were low, resulting in less than eight individuals of those species being taken at many sites. An attempt was made to restrict the collections of sunfish to only individuals of a size likely to be taken by sport fisherman (>50 g). However, at sites where fish were not abundant, it was necessary to collect smaller fish for analysis to obtain an adequate sample size.

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 hold fish from each site when more than one station was sampled on a given date. In the laboratory, a unique four-digit tag was attached by wire to the lower jaw of each fish. Each fish was then weighed and measured, and scale samples were taken for age determination. The fish was filleted, and the skin from each of the two fillets was removed. For small fish (sunfish), a 2- to 4-g sample of the anterior dorsal portion of the axial muscle fillet was excised for mercury analysis, and the remainder of the fillet was used for PCB analysis. The other fillet was saved for archival purposes or used to provide duplicate analyses for mercury and PCB determinations. In a limited number of samples, the remaining fillet was freeze-dried, pulverized with a mortar and pestle, and submitted for ^{137}Cs analysis. For

large fish (carp), one fillet was ground in a hand meat grinder, and subsampled for mercury and PCB analyses; the other fillet was wrapped in heavy-duty aluminum foil and archived. All samples were packaged for analysis by wrapping the sample in heavy-duty aluminum foil and writing the tag number on an adhesive paper label affixed to the outside of the foil package. The tag number was also written directly on the aluminum foil in case the adhesive label fell off. All samples were stored at -20°C in a locked freezer until they were submitted for analysis.

Mercury and PCBs in EFPC sediments were analyzed in July 1986 by the same procedures used in 1982 by Van Winkle et al. (1984). Surface sediments (upper 1 cm or less) were scooped from three to ten areas of active deposition within 50 m of each of nine sampling stations on EFPC, ranging from EFK 24.0, immediately above NHP, to EFK 2.1, below the mouth of Bear Creek. Samples were wet-sieved in the field to <2 mm and returned to the laboratory. A subsample of the <2 mm sediment was oven dried at 40°C and analyzed for mercury. The remaining sample was wet-sieved to $\leq$125 μm, air-dried, and analyzed for total mercury and PCBs. Sediment collections and analyses were performed in cooperation with R. R. Turner of the ORNL Environmental Sciences Division so that results could be utilized in this project and in the Oak Ridge Task Force (ORTF) investigation of the role of NHP as a source of contamination to EFPC.

Even though many organic compounds have physical and chemical properties favoring substantial bioaccumulation, many such compounds are not accumulated by fish because of their rapid metabolic alteration and subsequent excretion (Southworth et al. 1980). However, many invertebrates, including clams, snails, and zooplankton are not as capable of metabolizing organic compounds accumulated from water and thus are useful for monitoring substances, such as PAHs, which do not accumulate appreciably in fish (McCarthy et al. 1985). The Asiatic clam (*Corbicula fluminea*), which has been previously used as a bioaccumulation monitor (Graney et al. 1983, Tatem 1986), was used to monitor for PAHs in the effluent from New Hope Pond (NHP) in March 1986 and to monitor for a broad suite of organic priority pollutants in subsequent exposures (July 1986). Because neither *Corbicula* nor any other bivalve species was abundant enough in EFPC to use as a biomonitor, *Corbicula* were collected from Beaver Creek (Fig. 2-2) and maintained in cages suspended in EFPC immediately below the NHP discharge and in Brushy Fork, the

reference stream. After 4 weeks exposure, the cages were removed and returned to the laboratory. Individual clams were removed from the shells (~1 to 2 g wet wt) and analyzed for PAHs using high-performance liquid chromatography (HPLC) with fluorescence detection.

Mercury determinations in fish and sediment samples were performed by the ORNL Analytical Chemistry Division using 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.

Analyses of PCBs in fish and sediment samples collected in May 1985 were performed using procedure EC-440 (Martin Marietta Energy Systems 1983), which was also used in previous PCB monitoring near the three DOE facilities in Oak Ridge (Martin Marietta Energy Systems 1984, 1985) and in EFPC in 1982 (McElhaney 1982). This procedure utilizes a hot potassium hydroxide in methanol solution to digest the tissue, followed by extraction into ethyl ether and concentration of the extract by evaporation. Analysis is conducted by packed-column gas chromotography (referenced to commercial PCB mixtures: Arochlor 1242, 1254, 1260) using electron capture detection.

The December 1985 and May 1986 samples were analyzed for PCBs using procedure EPA 600-4-81-055 (EPA 1980). This procedure utilizes a Soxhlet extraction with methylene chloride, followed by adsorption column cleanup, solvent exchange, and evaporative concentration prior to analysis by packed-column gas chromatography with electron capture detection. This procedure was adopted (1) because of the poor recovery of PCB 1254 observed in procedure EC-440 in the quality assurance/quality control (QA/QC) program and (2) to standardize analytical procedures with those used by the Environmental Protection Agency (EPA). The initial use of the EC-440 procedure for PCB analysis, however, provides a direct basis for comparison of the results of the 1982 and 1985 studies.

Analyses of PAHs in clam tissue were conducted using procedure EPA-PPB 12/83 (EPA 1984a). The method involves extraction and cleanup similar to that used for PCBs, followed by analysis using HPLC with a fixed wavelength fluorescence detector.

Cesium-137 was determined by counting the characteristic 0.662 Mev gamma emission on a gamma spectrometer equipped with a germanium-lithium (GeLi) crystal

scintillation detector, using procedure EMSL-LV-0539-17 (EPA 1979). A preweighed 5 to 10-g (wet wt) sample of freeze-dried fish was counted in a 6.8-cm-diam plastic dish placed on top of the GeLi crystal.

Statistical evaluations of the data were made using SAS procedures (SAS 1985a) for analysis of variance, Duncan's multiple-range test, linear regression analysis, t-test for comparison of paired observations, and the calculation of means, standard deviations (SD), standard errors (SE), and coefficients of variation.

Quality assurance was maintained using a combination of (1) blind duplicate analyses, (2) split sample analyses between the EPA Environmental Services Laboratory, Tennessee Valley Authority (TVA) Analytical Laboratories, the Y-12 Plant Analytical Laboratory, and the ORNL Analytical Chemistry Division, and (3) analysis of fish reference standards. Results of these analyses are summarized in Appendix B.

4.1.3 Results

4.1.3.1 Mercury

1985/1986 Sampling

Fish collected from EFPC in 1985 and 1986 contained higher levels of mercury than fish collected from reference streams in all three sampling periods. Redbreast sunfish (*Lepomis auritus*), which is ubiquitously distributed throughout EFPC, averaged 0.67 ppm (fresh wt) total mercury, approximately eight times the 0.08 ppm observed in *L. auritus* from reference streams. Bluegill sunfish (*Lepomis macrochirus*) are abundant only at the uppermost and lowermost sites in EFPC, whereas carp (*Cyprinus carpio*) are abundant year-round only in the middle reaches of the stream. Mercury levels averaged 0.61 ppm and 0.71 ppm in bluegill and carp from EFPC, respectively, and 0.06 ppm and 0.15 ppm in fish from reference streams. Results of all analyses are listed in Appendix C, Tables C-1 through C-6.

The U.S. Food and Drug Administration (FDA) established a limit of 1 ppm mercury (as methyl mercury) in fish and shellfish sold for human consumption (FDA 1984a). Virtually all mercury in fish is methyl mercury (Hildebrand et al. 1980), hence total mercury levels measured in this study are a conservative estimate of methyl mercury levels. Most fish collected from EFPC in 1985 and 1986 did not exceed 1 ppm total mercury (Table 4-1). Overall, 5% of the redbreast sunfish, 7% of the bluegill, and 20% of

Table 4-1. Proportion of fish collected from East Fork Poplar Creek with a total mercury concentration greater than 1 ppm, May 1985–May 1986

Site	Species		
	Bluegill (*Lepomis macrochirus*)	Redbreast sunfish (*L. auritus*)	Carp (*Cyprinus carpio*)
EFK 23.4	4/29	13/28	0/2
EFK 18.2	0/2	5/24	0/3
EFK 13.8	0/2	0/24	4/20
EFK 6.3	0/4	0/24	7/23
EFK 2.1	0/24	0/23	0/8

the carp collected contained mercury in excess of 1 ppm. The highest level observed was 2.0 ppm in a carp, whereas the highest level observed in either species of sunfish was 1.7 ppm. Most fish containing more than 1 ppm mercury were collected in the reach immediately below NHP. At this site, 46% of the redbreast exceeded 1 ppm mercury, as did 14% of the bluegill.

The level of mercury in sunfish from EFPC was highest at the site immediately downstream of NHP and decreased steadily with distance downstream from that site (Tables 4-2 and 4-3, Figs. 4-1 and 4-2). This pattern was also observed in previous studies of mercury contamination in EFPC (Van Winkle et al. 1984; TVA 1985e). The downstream decrease in mercury levels in fish was most pronounced in redbreast sunfish, which was collected in adequate numbers at all sites. The pattern is obvious in bluegill data as well, but only a few fish were collected from sites in the middle portions of the stream. The decrease in mercury levels in fish with increasing distance from NHP was statistically significant ($\alpha = 0.05$) for bluegill and redbreast sunfish in all sampling periods. No such relationship was observed in carp; however, very few fish were collected at sites in the upper reaches of the stream, and these generally had lower mercury levels than carp collected further downstream (Table 4-4). It is likely that carp found in the upper reaches of EFPC are temporary residents that move upstream during the spawning season and then return to downstream sites.

An anomalously low level of mercury in redbreast sunfish was noted at EFK 23.4 in May/June 1985 (Table 4-2). Mercury levels in this species at other sites were

Table 4-2. Total mercury in redbreast sunfish (*Lepomis auritus*) from East Fork Poplar Creek, 1985–1986

Site	Total mercury[a] (ppm, wet wt)		
	May 1985	December 1985	May 1986
EFK 23.4	0.62 ± 0.08 (12)	1.26 ± 0.12 (8)	1.19 ± 0.16 (8)
EFK 18.2	0.77 ± 0.09 (9)	0.83 ± 0.09 (8)	0.87 ± 0.09 (8)
EFK 13.8	0.65 ± 0.07 (8)	0.73 ± 0.06 (8)	0.59 + 0.04 (9)
EFK 6.3	0.38 ± 0.02 (8)	0.56 ± 0.05 (8)	0.41 ± 0.03 (8)
EFK 2.1	0.45 ± 0.12 (7)	0.36 ± 0.06 (8)	0.34 ± 0.03 (8)

[a]Values are mean ± 1 SE (sample size in parentheses). Background level (reference stream fish) is 0.08 ± 0.01 ppm (N = 24).

Table 4-3. Total mercury in bluegill sunfish (*Lepomis macrochirus*) from East Fork Poplar Creek, 1985–1986

Site	Total mercury[a] (ppm, wet wt)		
	May 1985	December 1985	May 1986
EFK 23.4	0.83 ± 0.06 (12)	0.86 ± 0.08 (8)	0.74 ± 0.12 (9)
EFK 18.2	NC	0.96 (1)	0.40 (1)
EFK 13.8	0.37 (1)	NC	0.47 (1)
EFK 6.3	0.65 ± 0.05 (2)	0.40 ± 0.25 (2)	NC
EFK 2.1	0.48 ± 0.11 (8)	0.38 ± 0.10 (8)	0.29 ± 0.06 (8)

[a]Values are mean ± 1 SE (sample size in parentheses). Background level (reference stream fish) is 0.06 ± 0.02 ppm (N = 12). NC = No fish collected.

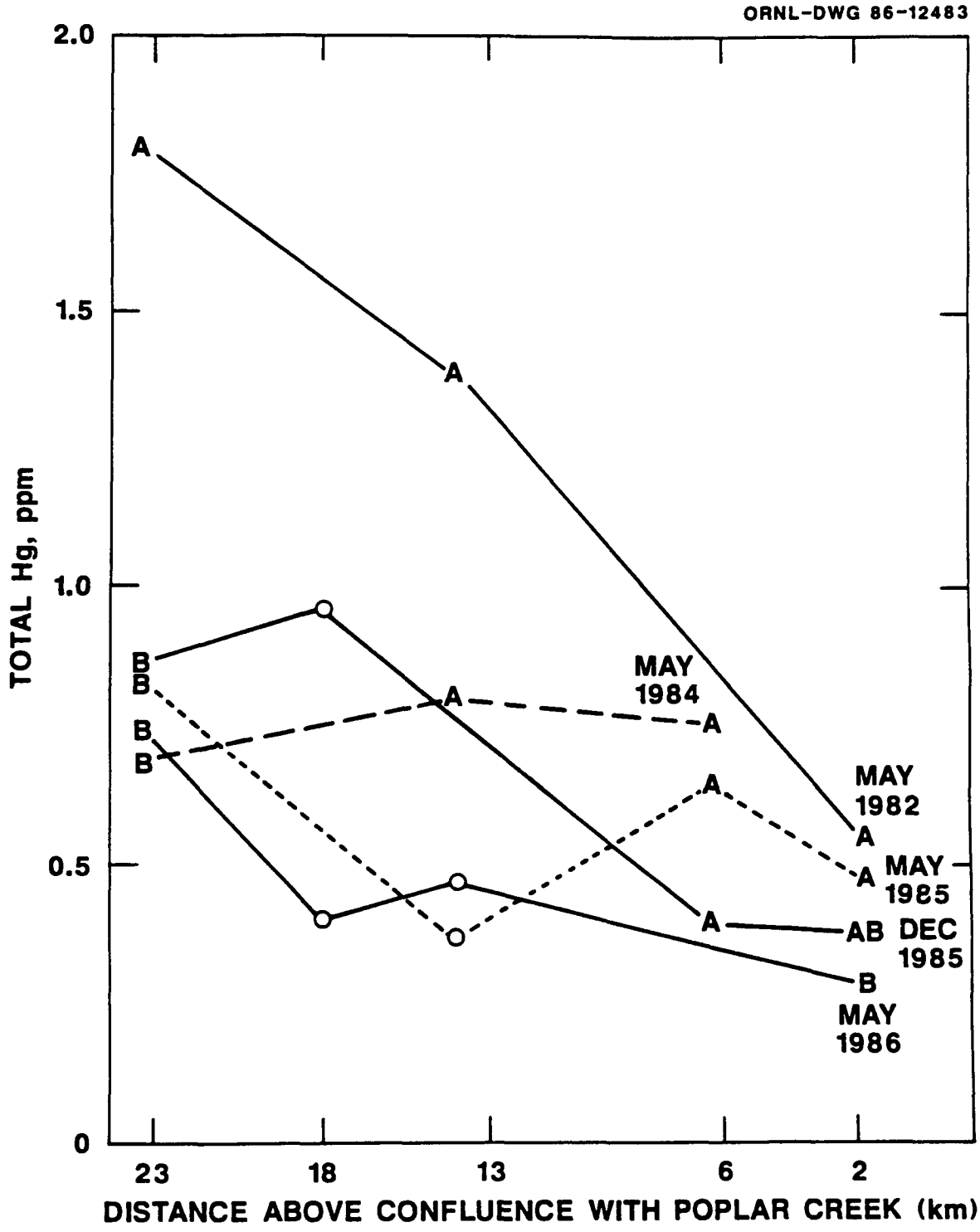

Fig. 4-1. Mean total mercury in axial muscle (ppm, wet w) from bluegill (*Lepomis macrochirus*) collected in East Fork Poplar Creek in 1982, 1984, 1985, and 1986 vs distance above confluence with Poplar Creek. Means at a specific site denoted by the same letter are not significantly different. Open circles indicate sample size (one to two fish) was too small for meaningful comparisons.

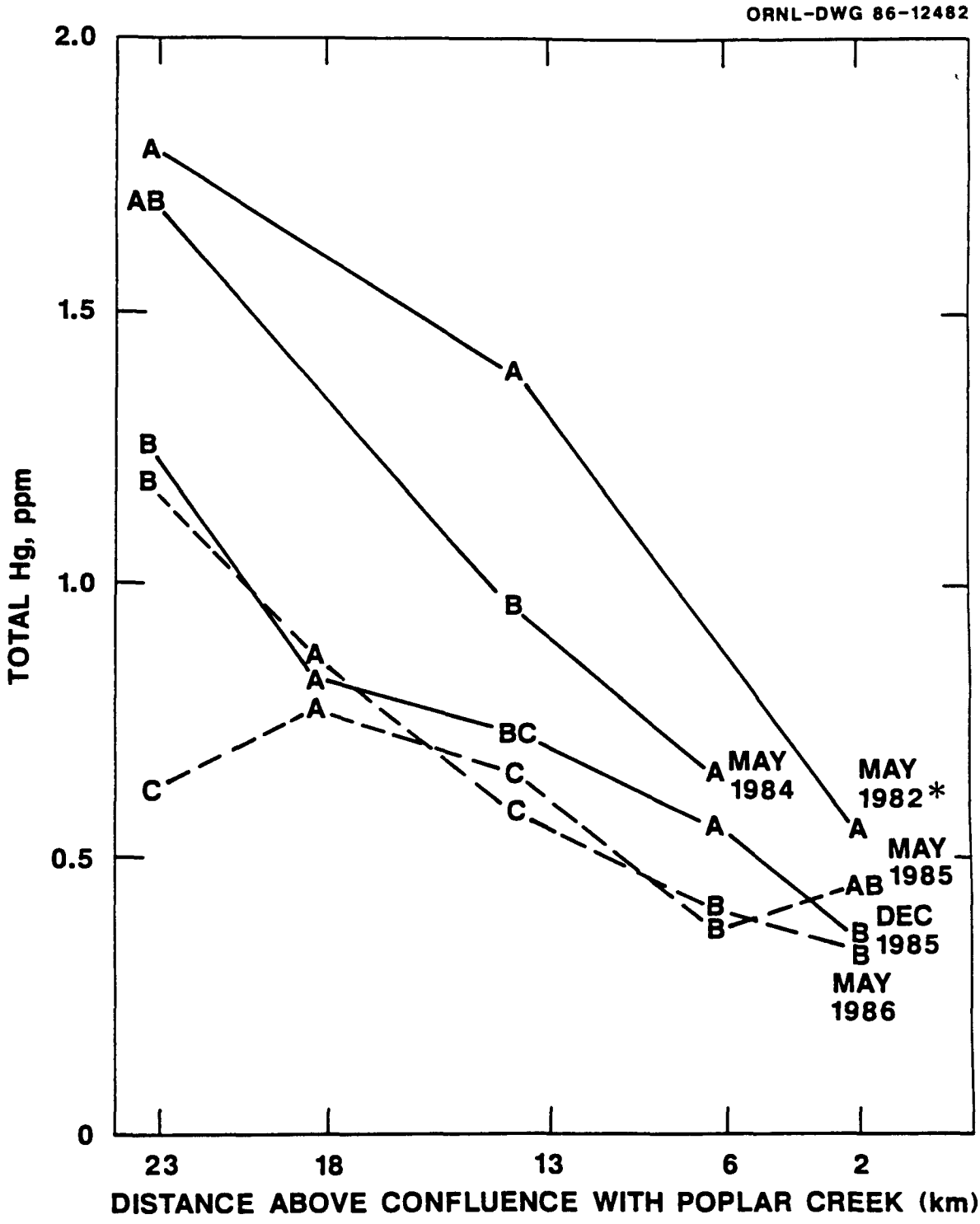

Fig. 4-2. Mean total mercury in axial muscle (ppm, wet wt) from redbreast sunfish (*Lepomis auritus*) collected in East Fork Poplar Creek in 1982, 1984, 1985, and 1986 vs distance above confluence with Poplar Creek. Means at a specific site denoted by the same letter are not significantly different. Open circles indicate sample size (one to two fish) was too small for meaningful comparisons.

Table 4-4. Total mercury in common carp (*Cyprinus carpio*) from East Fork Poplar Creek, 1985–1986

Site	Total mercury[a] (ppm, wet wt)		
	May 1985	December 1985	May 1986
EFK 23.4	0.29 ± 0.10 (2)	NC[b]	NC
EFK 18.2	NC	NC	0.34 ± 0.08 (3)
EFK 13.8	1.12 ± 0.26 (5)	0.66 ± 0.07 (8)	0.54 ± 0.07 (7)
EFK 6.3	0.80 ± 0.05 (8)	0.89 ± 0.09 (8)	0.80 ± 0.10 (8)
EFK 2.1	0.61 ± 0.02 (2)	NC	0.54 ± 0.07 (8)

[a]Values are mean ± 1 SE (sample size in parentheses). Background level (reference stream fish) is 0.15 ± 0.02 ppm (N = 16).
[b]NC = No fish collected.

similar among sampling periods, but at EFK 23.4, the mean total mercury concentration in spring 1985 was significantly lower ($\alpha = 0.05$) than the concentrations observed in the two later sampling periods and typical of the levels found in fish 5 to 10 km downstream. A possible explanation for this observation may be recent colonization of this reach of stream by redbreast sunfish from downstream sites. Such a phenomenon could occur if the species moved downstream because of adverse environmental factors during the previous fall/winter, and the site was recolonized by individuals from downstream reaches of the stream in the spring.

Historical changes in mercury levels in EFPC fish

The systematic sampling of bluegill sunfish for mercury analysis conducted in 1982 (Van Winkle et al. 1984) found average levels to approach 2 ppm at sites nearest NHP (Fig. 4-1) and to decrease to nearly 0.5 ppm at 21 km downstream. Unfortunately,

although these data provide a crucial benchmark for referencing later trends in mercury levels in fish, bluegill abundance in EFPC appears to have decreased dramatically since 1982. The redbreast sunfish is now the dominant sunfish species in the stream, and bluegill are abundant only in the uppermost and lowermost reaches (Sect. 6.2.3.2). Comparisons of the 1982 data on mercury levels in bluegill with later data are constrained by the small sample sizes of bluegill in the 1984, 1985, and 1986 collections and the possible alteration of habitat use and feeding habits of bluegill resulting from greater competition with redbreast sunfish. When such a comparison is made, a striking decrease in mercury levels in bluegill is evident between 1982 and 1985/1986 (Fig. 4-1). Most of this decrease appears to have occurred between 1982 and 1984. The mercury levels in bluegill at EFK 23.4 and EFK 2.1 in 1985/1986 are significantly ($\alpha = 0.05$) lower than those observed in 1982 in five of six possible comparisons. The 1985/1986 data showed a downstream decrease in mercury levels similar to that observed in 1982. Such a trend was not observed in 1984, probably becuase of the very small sample size available in 1984.

The data for redbreast sunfish provide a more complete data set for evaluating historical changes in mercury contamination of EFPC sunfishes. If the data on bluegill collected in 1982 are assumed to be representative of the mercury levels in redbreast sunfish at that time, then a substantial decrease in mercury contamination in sunfish occurred between 1982 and 1984 and again between 1984 and 1985. The concentration of total mercury in redbreast sunfish collected in 1985–1986 was significantly ($\alpha = 0.05$) lower than that observed in bluegill in 1982 in eight of nine possible comparisons (Fig. 4-2). The decrease between 1984 and 1985–1986 is less clear, with four of nine possible comparisons being significantly lower in 1985/1986. Since 1985, mercury levels in EFPC redbreast sunfish have remained relatively constant.

A review of data on the average monthly concentrations of total mercury in water exiting NHP showed no striking decrease in mercury concentrations (Fig. 4-3). Although the 1982 and 1984 samplings were preceded by excursions in total mercury concentrations in water exiting NHP within the previous year, these were of relatively short duration. A change in mercury speciation or solid phase partitioning could result in an increase or decrease in its availability without a concomitant change in total mercury concentrations. At this time, whether or not such a change occurred is strictly speculative. Results of mercury analyses of the <125-mm particle size surface sediments indicate little or no

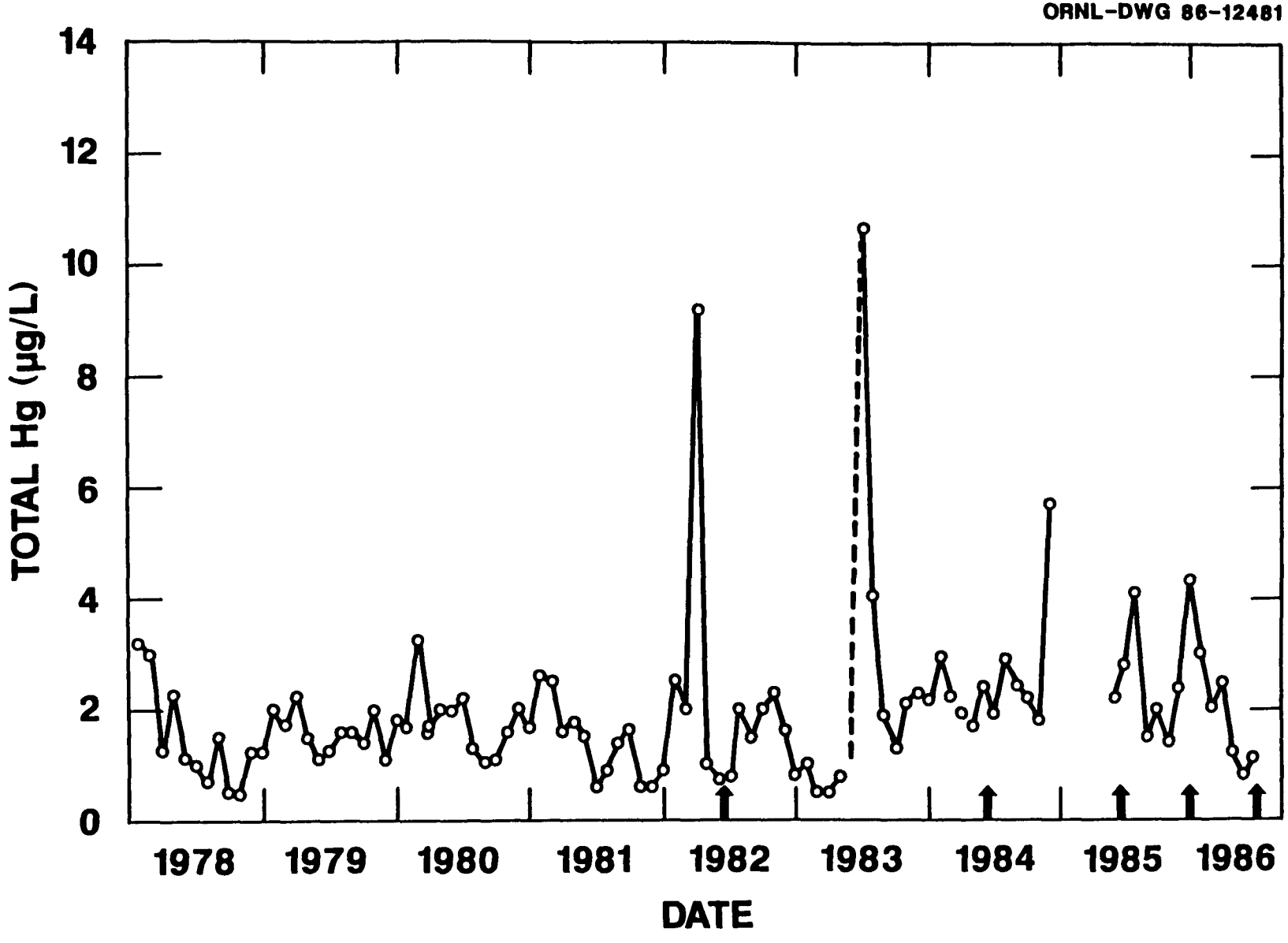

Fig. 4-3. Monthly average mercury concentrations in New Hope Pond outfall from 1978–1986. Samples are weekly grab samples analyzed for total mercury. Arrows denote dates of fish collections for mercury analysis.

change in mercury levels between 1982, 1984, and 1986 (Fig. 4-4). However, as was the case with the total aqueous-phase mercury, such measurements may have little relationship to the biologically available mercury species.

Mercury vs fish weight

Most studies have shown a positive correlation between fish weight/age and mercury levels in fish tissue in both contaminated and uncontaminated waters (Hildebrand et al. 1980; Elwood 1984; Van Winkle et al. 1984). This relationship complicates comparisons among sampling dates and sites. In this study, an attempt was made to minimize the effect of the mercury vs age relationship by selecting fish of similar age at each site. By restricting our collections to fish of a size likely to be kept by sport fisherman, it was hoped that most fish would be age IV, V, and VI and that no relationship between mercury content and size would be observed. Least squares regression of mercury vs fish weight for each species at each site in EFPC found only 2 of 29 regressions tested to have positive slopes that were significantly different from zero ($\alpha = 0.05$). It was concluded that no significant relationship between mercury concentration and fish weight existed in the fish collected, and no normalization procedure was required.

Differences among species

The most important systematic difference in mercury levels among fish species in this study is the possible difference between bluegill and redbreast sunfish because analysis of historical trends requires the comparison of these species. Comparison of the mercury levels in these two species for given site-date combinations revealed significantly ($\alpha = 0.05$) higher levels of mercury in redbreast sunfish at site EFK 23.4 in December 1985 and May 1986 but not in May 1985. (The anomalous nature of the May 1985 redbreast data was discussed previously.) At EFK 2.1 there were no significant differences in mean mercury levels between the two species. Similarly, no differences existed in reference streams. Thus, it appears that mercury levels in the two species are comparable in lower EFPC but that true differences may exist in the upper EFPC below NHP (EFK 23.4). Assuming that mercury levels in 1982 in redbreast sunfish at this site may have

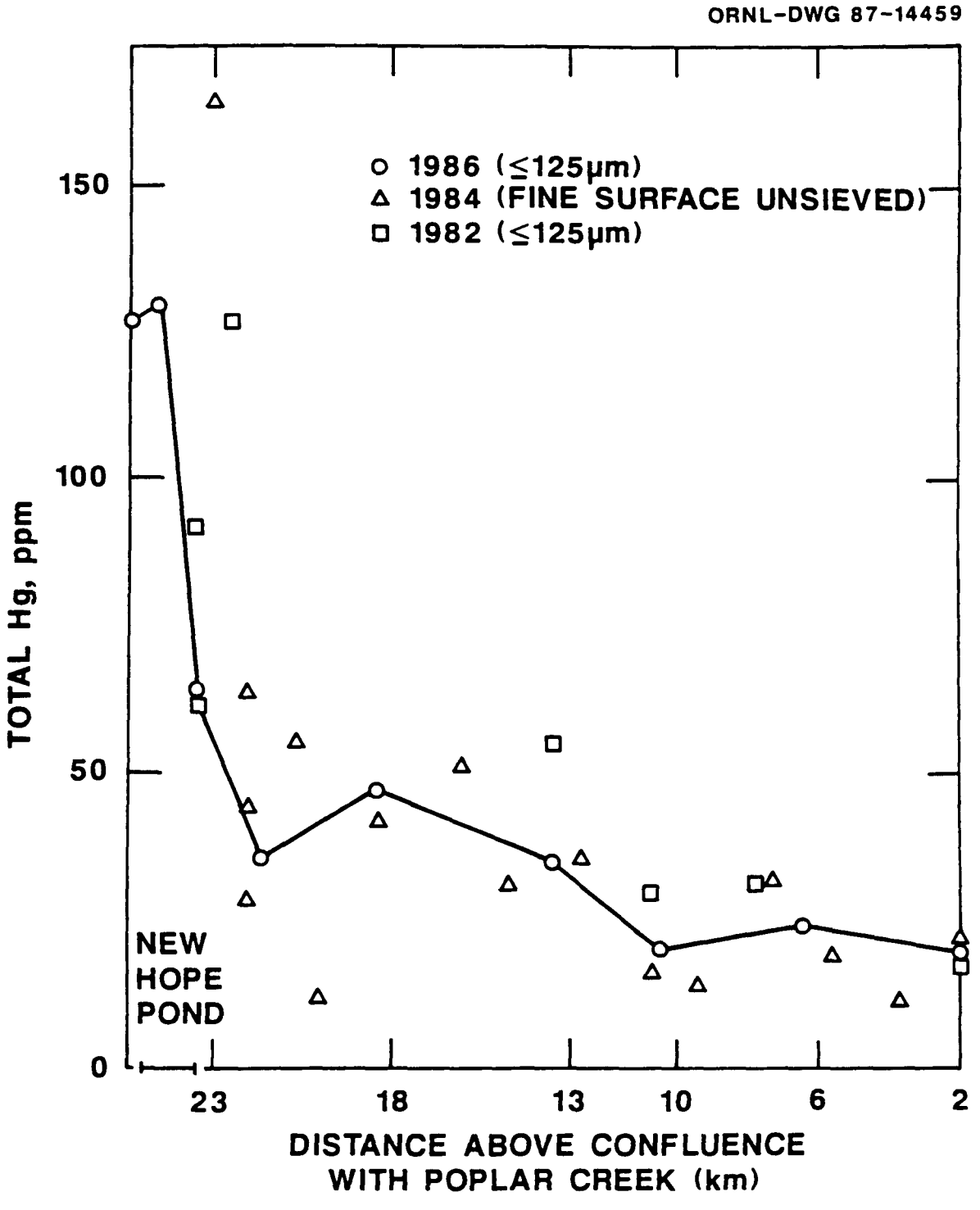

Fig. 4-4. Total mercury in fine particulate surface sediments in East Fork Poplar Creek vs distance above confluence with Poplar Creek, 1982, 1984, and 1986.

been about 1.5 times higher than the levels observed in bluegill, the apparent decrease in mercury in redbreast at that site between 1982 and 1986 is even more dramatic.

Mercury levels in carp were similar to those in redbreast and bluegill sunfish. Direct comparisons are difficult because mercury in sunfish varied as a function of distance from NHP, whereas mercury in carp did not. However, when mercury levels for carp and the sunfish species were compared for given site-date combinations, no significant differences were noted in seven of ten comparisons; whereas mercury in carp was significantly higher than mercury in sunfish at EFK 6.3 and EFK 2.1 in May 1986 and significantly lower at EFK 23.4 in May 1985. These observations are consistent with the finding of a relatively homogeneous distribution of mercury vs distance below NHP in carp and a strong decrease in mercury with distance in sunfishes. Thus, mercury in sunfish would likely exceed that in carp in the upper reaches of the stream, approximate that in carp in the middle reaches, and be lower than that in carp in the downstream reaches of the stream.

4.1.3.2 Other metals

Sunfish collected from EFK 23.4 in June 1986 contained levels of all metals (other than mercury) that were similar to those found in fish from Hinds Creek, a reference stream (Table 4-5, Appendix C-4). The levels of metals in EFPC fish in 1986 were also similar to those observed in 1984 by TVA in fish from Melton Hill Reservoir (Table 4-5), a reference site, and from EFPC (TVA 1985c, 1986). The geometric mean levels of metals (arsenic, cadmium, copper, lead, selenium, and zinc) observed in fish in the National Contaminant Biomonitoring Program (Lowe et al. 1985) were also generally similar to levels observed in EFPC fish (Table 4-5). A comparison of the levels of metals in EFPC fish with Preliminary Guidance Values (PGVs), which were derived to screen for contamination that potentially threatened human health (Hoffman et al. 1984; Travis et al. 1986), indicated that only arsenic, beryllium, and mercury approach this threshold (Table 4-5). Neither arsenic nor beryllium was shown to be elevated by these data; however, the PGV is set at a level below background as a result of the carcinogenicity of these two metals. The PGV screening approach is very conservative and is designed to eliminate from concern any substances not exceeding the PGV (Hoffman et al. 1984).

Table 4-5. Metal concentrations in fish from East Fork Poplar Creek below New Hope Pond (EFK 23.4) and from Hinds Creek, a reference stream

Metal	Concentration (ppm, wet wt)				
	EFK 23.4[a,b]	Hinds Cr[b,c]	TVA[d]	USFWS[e]	PGV[f]
Antimony	<0.3	<0.3			
Arsenic	<0.05	<0.05	<0.03	0.16	0.0007
Beryllium	<0.04	<0.05	<1		0.004
Cadmium	0.016 (0.011)	0.015 (0.014)	0.007	0.04	1.0
Chromium	<0.1	<0.1	0.06		1.8
Copper	2.6 (3.8)	0.45 (0.56)	0.4	0.86	36
Lead	0.04[g]	<0.02	0.21	0.19	1.8
Lithium	<0.5	<0.5			
Nickel	<1.0	<1.0	<1.0		5.2
Selenium	0.78[h] (0.09)	0.55[h] (0.12)	0.7	0.46	12
Silver	<0.1	<0.1	<0.3		0.29
Thallium	<0.2	<0.2	<1		0.66
Zinc	7.6 (2.5)	6.2 1.0	8.4	25.6	180

[a]Fish from EFK 23.4 were bluegill (*Lepomis macrochirus*) and redbreast sunfish (*L. auritus*), N = 5 each.

[b]Tabular values for these two sites are the mean and SD (in parentheses).

[c]Fish from Hinds Creek were bluegill (N = 3) and redbreast sunfish (N = 1).

[d]TVA 1985e (N = 13 for Cd; N = 9 for Cr, As, Ni; N = 4 for others).

[e]Lowe et al. 1985.

[f]Preliminary Guidance Values (from Hoffman et al. 1984; Travis et al. 1986).

[g]Most values below detection limit (included in calculation of mean as if present at that level).

[h]Significantly different at $\alpha = 0.05$.

4.1.3.3 PCBs

1985/1986 Sampling

Fish collected from EFPC in 1985 and 1986 contained detectable residues of PCBs in all three sampling periods. Redbreast sunfish and bluegill contained similar levels, averaging 0.33 and 0.27 ppm total PCBs, respectively (Tables 4-6 and 4-7), whereas carp contained much higher concentrations, averaging 1.99 ppm (Table 4-8). Unlike mercury, there is no natural background level of PCBs in aquatic organisms; hence all residues arise from human activity. Because of their widespread use in the past several decades and their susceptibility to atmospheric transport, PCBs contaminate many U.S. waterways (Schmitt et al. 1985). Levels found in organisms from reference streams are therefore of interest as a basis for comparison. Redbreast and bluegill sunfish from the Hinds Creek and Beaver Creek reference sites contained low levels of total PCBs, averaging 0.05 ppm. Carp from Hinds Creek contained higher levels, averaging 0.07 ppm total PCBs. The Brushy Fork reference stream proved to be contaminated with PCBs to a level comparable with EFPC, with redbreast sunfish averaging 0.64 ppm and carp 2.14 ppm total PCBs. This site is discussed in more detail later in this subsection.

The FDA established a limit of 2 ppm total PCBs in fish and shellfish sold for human consumption (FDA 1984b). All sunfish collected from EFPC contained less than 2 ppm total PCBs. The maximum levels observed were 1.46 ppm in redbreast sunfish and 0.75 ppm in bluegill. In May 1985, 41% (7/17) of the carp from EFPC exceeded 2 ppm total PCBs, and a similar fraction (44%, 7/16) exceeded the limit in December 1985. However, no carp collected from EFPC in May 1986 exceeded 2 ppm total PCBs.

The longitudinal pattern of decreasing contaminant levels with increasing distance from NHP that was observed for mercury in redbreast sunfish was repeated for PCBs (Fig. 4-5). Levels of PCBs in redbreast sunfish were highest at EFK 23.4 (immediately below NHP) and decreased steadily downstream, a pattern similar to that observed for PCBs in bluegill in 1982 (Fig. 4-5) (W. Van Winkle, ORNL Environmental Sciences Division, unpublished data). The relationship between PCB levels in redbreast sunfish and distance downstream from NHP was statistically significant ($\alpha = 0.05$) in all sampling periods. Bluegill were collected in adequate numbers only at sites EFK 23.4 and EFK 2.1, and the decrease in PCB concentrations observed between these two sites was statistically

Table 4-6. Total PCBs in redbreast sunfish (*Lepomis auritus*) from East Fork Poplar Creek, 1985-1986

Site	Total PCBs[a] (ppm, wet wt)		
	May 1985	December 1985	May 1986
EFK 23.4	0.48 ± 0.09 (12)	0.49 ± 0.09 (8)	0.69 ± 0.15 (9)
EFK 18.2	0.34 ± 0.04 (8)	0.27 ± 0.08 (8)	0.21 ± 0.04 (8)
EFK 13.8	0.39 ± 0.09 (8)	0.22 ± 0.02 (8)	0.26 ± 0.05 (8)
EFK 6.3	0.25 ± 0.03 (8)	0.21 ± 0.02 (8)	0.16 ± 0.04 (8)
EFK 2.1	0.19 ± 0.01 (7)	0.26 ± 0.05 (8)	0.22 ± 0.06 (8)

[a]Values are mean ± 1 SE (sample size in parentheses). Background level (reference stream fish) is 0.04 ± 0.01 ppm (N = 15).

Table 4-7. Total PCBs in bluegill sunfish (*Lepomis macrochirus*) from East Fork Poplar Creek, 1985-1986

Site	Total PCBs[a] (ppm, wet wt)		
	May 1986	December 1985	May 1986
EFK 23.4	0.26 ± 0.04 (8)	0.36 ± 0.06 (8)	0.34 ± 0.10 (8)
EFK 18.2	NC[b]	0.75 (1)	0.07 (1)
EFK 13.8	NC	NC	0.16 (1)
EFK 6.3	0.20 ± 0.16 (2)	0.24 ± 0.01 (2)	NC
EFK 2.1	0.31 ± 0.05 (8)	0.20 ± 0.00 (8)	0.18 ± 0.04 (8)

[a]Values are mean ± 1 SE (sample size in parentheses). Background level (reference stream fish) is 0.06 ± 0.01 ppm (N = 10).

[b]NC = No fish collected.

Table 4-8. Total PCBs in carp (*Cyprinus carpio*) from East Fork Poplar Creek, 1985-1986

Site	Total PCBs[a] (ppm, wet wt)		
	May 1985	December 1985	May 1986
EFK 23.4	2.52 ± 2.08 (2)	NC	NC
EFK 18.2	NC[c]	NC	0.83 ± 0.03 (3)
EFK 13.8	2.94 ± 0.81 (5)	1.95 ± 0.38 (8)	0.75 ± 0.14 (7)
EFK 6.3	1.70 ± 0.42 (8)	1.29 ± 0.19 (8)	0.44 ± 0.06 (8)
EFK 2.1	3.25 ± 2.05 (2)	NC	0.44 ± 0.06 (8)

[a]Values are mean ± 1 SE (sample size in parentheses). Background level (reference stream fish) is 0.07 ± 0.01 ppm (N = 7).

[b]Excludes two fish containing 3.2 and 0.78 ppm PCBs that were collected from the reference stream (Hinds Creek). PCB analyses of sediments and sunfish from this stream did not indicate the presence of contamination. The source of PCBs found in these two fish is not known; however, they do not appear to have acquired such PCB levels at the site where they were collected.

[c]NC = fish collected.

significant ($\alpha = 0.05$) in December 1985 and May 1986 but not in May 1985. Like mercury, PCB residues in carp showed no relationship with distance from NHP.

If the anomalously low mercury levels observed in redbreast sunfish in May 1985 at EFK 23.4 were the result of the recent movement of downstream fish into that site, as was hypothesized earlier, a similar trend might be expected for PCBs. Although the difference was less than that for mercury, nevertheless, the decrease in PCBs in redbreast sunfish between EFK 23.4 and EFK 13.8 was much less pronounced in May 1985 than that observed in December 1985 and May 1986, lending support to the colonization hypothesis. The lack of a significant decrease in PCB levels in bluegill between EFK 23.4 and EFK 2.1 in May 1985 is also consistent with dispersal of bluegill out of the uppermost study reach and redistribution of those fish into the lower reaches of EFPC.

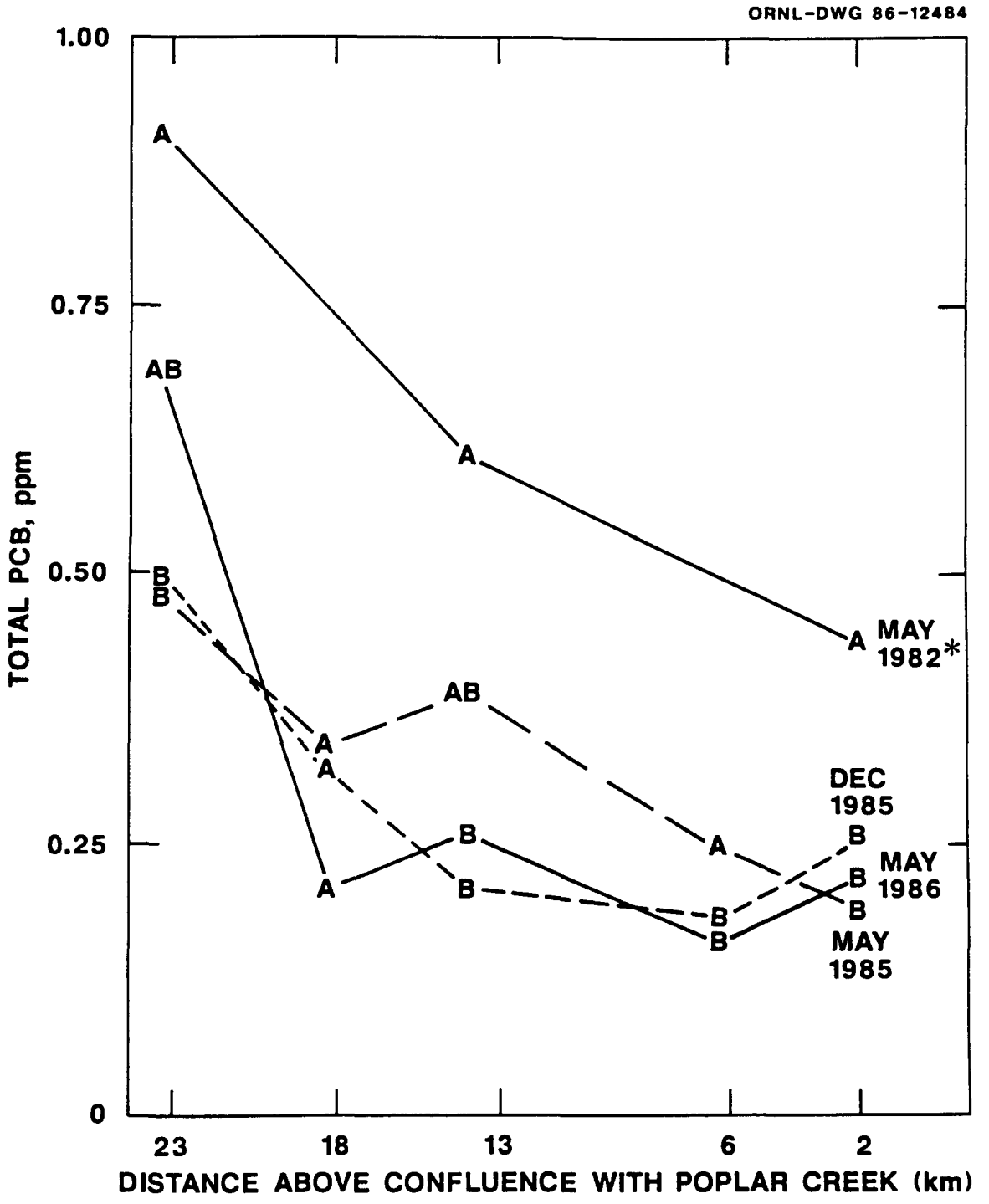

Fig. 4-5. Mean total PCBs in axial muscle (ppm, wet weight) in redbreast sunfish (*Lepomis auritus*) collected in East Fork Poplar Creek in 1982*, 1984, 1985, and 1986 vs distance above confluence with Poplar Creek. Means at a specific site denoted by the same letter are not significantly different ($\alpha = 0.05$). *1982 fish are bluegill (*Lepomis macrochirus*).

PCBs are mixtures of chlorinated biphenyls containing varying numbers of chlorine atoms arranged in different isomeric positions. Those containing higher numbers of chlorine atoms have higher bioaccumulation potential; the PCBs found in fish tissue commonly are dominated by chlorinated biphenyls containing four to seven chlorine atoms. Quantification of PCBs in fish is generally referenced to commercial PCB mixtures, which are identified by numbers such as PCB-1248, PCB-1254, and PCB-1260. The final two digits in this number refer to the chlorine content of the PCB mixture by mass, thus PCB-1254 consists of 54% chlorine and contains predominantly compounds having five and six chlorine atoms per biphenyl molecule. PCB-1260 contains primarily compounds with six and seven chlorine atoms per molecule.

The PCBs found in fish from EFPC were characteristic of those isomers and congeners found in PCB-1254 and PCB-1260. Roughly equal amounts of the two commercial mixtures were found in sunfish at all sites in EFPC except EFK 23.4, where PCB-1254 predominated in a ratio of approximately 2:1. The ratio of PCB-1254:PCB-1260 in carp from EFPC was roughly the reverse of this (1:2) and comparable with the ratio of PCB-1254 to PCB-1260 observed in fine particulate sediments. Total PCBs are a more reliable measure of environmental PCB levels than are any of the commercial mixtures quantified individually because of variations in the abilities of different analyses to resolve and quantify chromatographically overlapping mixtures such as PCB-1254 and PCB-1260 (Schmitt et al. 1985). Therefore, total PCBs were generally used to evaluate trends and make comparisons in these data.

Historical changes in PCB levels in fish in East Fork Poplar Creek

Many of the bluegill analyzed for mercury in 1982 (Van Winkle et al. 1984) were also analyzed for PCBs. The results were summarized by McElhaney (to W. Van Winkley, ORNL-ESD, personnel communication, 1982) and provide an important benchmark for evaluating historical trends in PCB contamination in EFPC. In 1982, PCBs averaged 0.8 ppm in bluegill collected from EFPC immediately below NHP and decreased downstream to 0.38 ppm at EFK 2.1 (Fig. 4-5). In 1984, the TVA (TVA 1985e) sampled sunfish and rock bass from EFPC, but none of the samples contained PCBs above the detection limit of 0.1 ppm. The apparent decline in the bluegill populations in the middle reaches of EFPC after 1982 and the difficulty of comparing the results of the 1982 study with more

recent data on both bluegill and redbreast sunfish have been discussed previously. Comparison of the 1982 and 1985/1986 data sets on bluegill (Fig. 4-5 and Table 4-7, respectively) revealed a substantial decrease in PCB levels. Concentrations of PCBs in bluegill collected at EFK 23.4 and EFK 2.1 in 1985 and 1986 were significantly lower (α = 0.05) than those observed in bluegill at the same sites in 1982, with the single exception of May 1985 at EFK 2.1. A similar, but less dramatic, decrease was observed when the 1985/1986 redbreast sunfish data were compared with the 1982 bluegill data (Fig. 4-5). At each site, mean PCB levels in bluegill from 1982 exceeded those observed in redbreast sunfish from 1985/1986 on all dates, and significant (α = 0.05) differences occurred in six of nine possible comparisons (Fig. 4-5).

Little change in total PCB concentrations in bluegill and redbreast sunfish was observed from May 1985 to May 1986. The patterns of mean PCB levels vs distance shown in Fig. 4-5 overlapped considerably for redbreast and bluegill in the May 1985, December 1985, and May 1986 collections. The only significant difference occurred between May 1985 and December 1985 for redbreast sunfish at EFK 6.3 and between May 1985 and May 1986 for bluegill at EFK 2.1.

Mean PCB levels in carp from EFPC decreased from 2.3 ppm in May 1985 to 1.6 ppm and 0.6 ppm in December 1985 and May 1986, respectively. Both decreases were statistically significant (Duncan's Multiple Range Test, α = 0.05). Such a decline in PCB levels in carp, if real, would indicate that the high levels observed previously were more the result of past releases of PCBs than to present discharges from NHP. The absence of a similar decrease in PCB levels in sunfish is not necessarily inconsistent with the observed decline in carp, but it does indicate that one of the data sets should be regarded with a degree of skepticism. Archived carp tissue from May 1985, December 1985, and May 1986 will be re-analyzed for PCBs along with carp collected in December 1986 to verify this apparent decrease.

Levels of PCBs in fine surface sediments from EFPC (Fig. 4-6) showed a pattern similar to that of mercury (Fig. 4-4). Levels observed in 1986 were similar to those observed by TVA in 1984 (TVA 1985b,c), especially in the upper reaches of the stream. Higher values observed in lower EFPC in 1986 may be the result, in part, of comparing the PCB content of sieved (<125 mm) sediment fractions in 1986 with unsieved (bulk) sediments collected in 1984. The high levels of PCBs (4 to 5 ppm) observed in surface

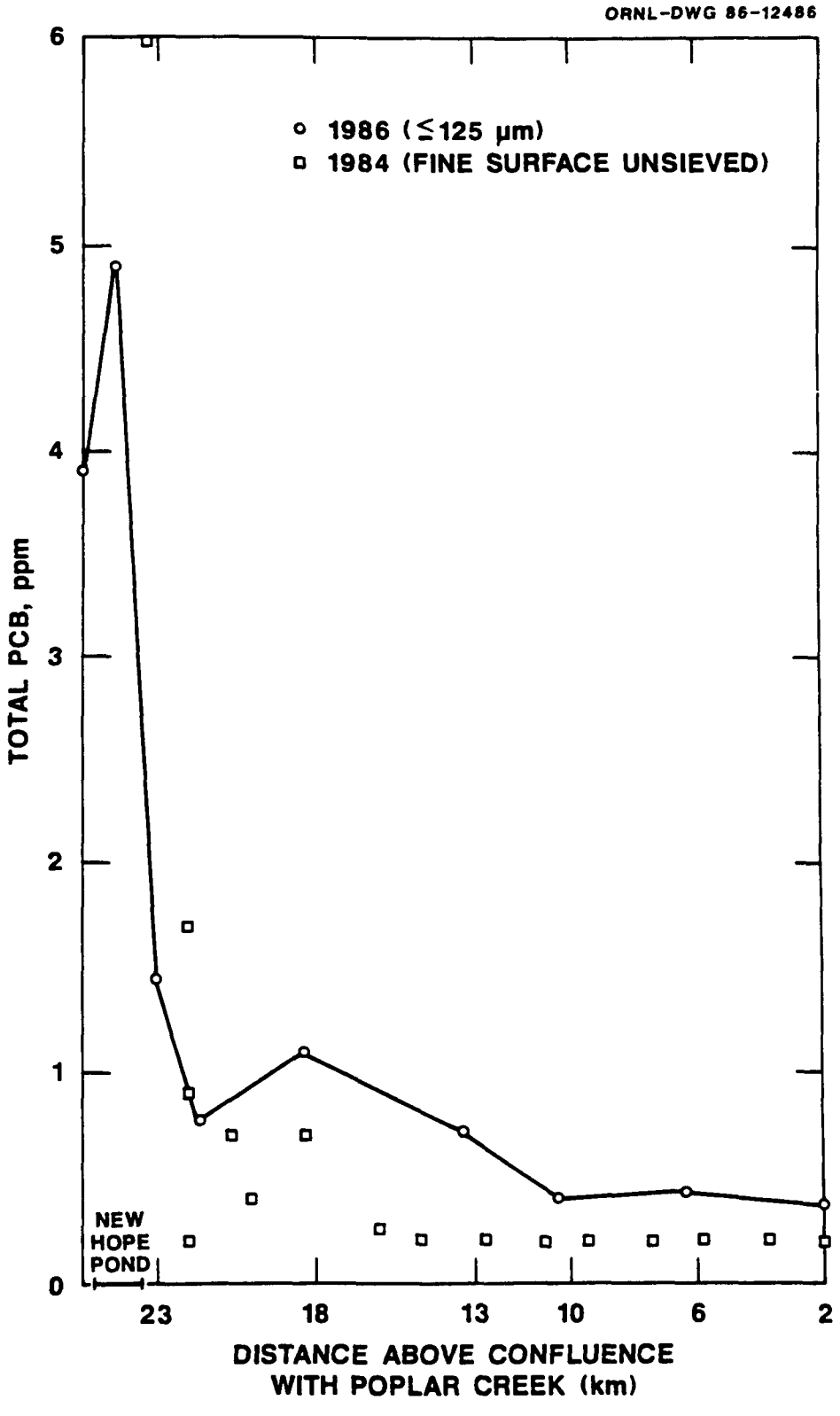

Fig. 4-6. Total PCBs in fine particulate surface sediments in East Fork Poplar Creek vs distance above confluence with Poplar Creek, 1984 and 1986.

sediments from NHP and EFPC immediately above NHP indicate a continuing source of PCBs upstream of these sites.

PCB vs fish weight

No significant positive relationship was observed between the concentration of PCBs in axial muscle and fish weight in 20 of 25 possible comparisons. Significant positive relationships were observed for redbreast sunfish at EFK 13.8 in May 1985 and at EFK 23.4 and EFK 18.2 in December 1985, and a significant negative relationship was observed for redbreast at EFK 2.1 in May 1986. Significant positive relationships were also observed for carp collected at sites EFK 13.8 and EFK 6.3 in May 1985, but no significant relationships were found in December 1985. No significant relationships between PCB levels and weight were observed in bluegills.

When fish from all three sampling periods were combined, significant positive relationships were only observed for redbreast sunfish at EFK 13.8 and carp at EFK 6.3 and significant negative relationships were noted for bluegill at EFK 23.4 and EFK 13.8. Because carp did not exhibit a significant trend in PCB concentration with distance below NHP, all EFPC carp data were combined to evaluate PCB levels vs fish weight. No significant relationship was observed. The relationship between PCB levels and weight in sunfish species from EFPC was weak, so the data were not normalized prior to making comparisons as was done for mercury in 1982 (Van Winkle et al. 1984). However, the few significant relationships that were observed demonstrate the need to exclude fish smaller than 50 g in future collections.

Differences among species

No significant differences ($\alpha = 0.05$) were observed in the mean total PCB concentration between bluegill and redbreast sunfish for any sampling date or site where both species were abundant. In all instances except one, PCB residues in carp were significantly higher than the levels observed in sunfish collected at the same time and location. Such differences are not unexpected. Carp contained >2 ppm total PCBs in the 1984 survey of EFPC (TVA 1985e) and are generally longer-lived and richer in lipids than sunfish; both of these characteristics act to increase the levels of hydrophobic contaminants.

PCBs in caged clams

Asiatic clams (*Corbicula fluminea*) in polypropylene cages were suspended in EFPC at the NHP outfall in July 1986. Clams were removed at intervals of 2 and 4 weeks and analyzed for PCBs and PAHs. The clams rapidly accumulated PCBs in the NHP outfall, attaining levels averaging 0.34 and 0.45 μg/g total PCBs after 14 and 30 d, respectively (Table 4-9). PCB-1254 dominated the PCB mixture found in the clams, accounting for 95% of the total; PCB-1260 accounted for the remainder. These results indicated that PCBs are currently being released in the NHP discharge and that levels observed in fish in the upper reaches of EFPC reflect current, rather than past, contamination.

PCB contamination in Brushy Fork

Carp collected in Brushy Fork in July 1985 contained what appeared to be elevated levels of PCBs, averaging 2.14 ± 1.23 ppm (mean ± 1 SE, N = 7), 80% of which was PCB-1254 and the remainder, PCB-1260. Although only one fish exceeded 2 ppm total PCBs, it contained 9.4 ppm PCBs. Because Brushy Fork drains a rural agricultural watershed having little industrial development, these results aroused skepticism. A subsample of the most highly contaminated carp was analyzed by gas chromatographic mass spectroscopy (GC/MS) by the ORNL Analytical Chemistry Division. This analysis confirmed the presence of PCBs in the fish. A subsample of archived tissue from the same fish was included in split samples analyzed by the ORNL and EPA-Athens laboratories. Each laboratory found PCBs in the sample; EPA reported 3.3 ppm PCB-1254 and <2.4 ppm 1260 and ORNL reported 0.78 and 0.84 ppm of the two commercial mixtures, respectively.

Because Brushy Fork is the ecological reference stream for the BMAP, sunfish were collected from the stream (near BFK 7.6) in December 1985 and analyzed for PCBs for comparison with sunfish from EFPC. Mean total PCB concentrations were 0.68 ± 0.28 ppm (N = 4) and 0.64 ± 0.20 ppm (N = 8) in bluegill and redbreast sunfish, respectively). More than 80% of the total PCBs measured was PCB-1254. A final confirmation of PCB contamination at the Brushy Fork site was obtained by analyzing fine particulate sediments for PCBs in July 1986. The sediment sample contained 0.33 ppm PCB-1254 and 0.01 ppm PCB-1260, a level of PCB-1254 comparable to that observed in

Table 4-9. Concentrations in ppm wet wt of polycyclic aromatic hydrocarbons and polychlorinated biphenyls (PCBs) in clams[a] (*Corbicula fluminea*) collected from Beaver Creek, a reference stream, and maintained in East Fork Poplar Creek below New Hope Pond for 14 and 30 d, July 1986

Compound	East Fork Poplar Creek						Beaver Creek		
	14 d			30 d					
	1	2	3	1	2	3	1	2	3
Naphthalene	<450	0.836	<0.450	1.222	1.053	0.650	1.025	<0.450	<0.450
Acenaphthene	<0.120	<0.120	<0.120	<0.120	<0.120	<0.120	<0.120	<0.120	<0.120
Phenanthrene	<0.120	<0.120	<0.120	0.284	<0.120	<0.120	0.188	<0.120	<0.120
Anthracene	1.750	<0.450	4.585	<450	<0.450	<0.450	1.690	<0.450	<0.450
Fluoranthene	0.264	0.084	0.062	0.148	0.110	0.082	<0.050	0.145	<0.050
Pyrene	<0.070	<0.070	<0.070	<0.070	<0.070	<0.070	<0.070	<0.070	<0.070
Benzo(*a*)anthracene	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.056	<0.020
Benzo(*b*) fluoranthene	0.032	0.027	0.052	<0.020	<0.020	<0.020	<0.020	0.078	<0.020
Benzo(*k*) fluoranthene	0.015	<0.007	0.044	0.007	0.012	<0.007	<0.007	0.039	<0.007
Benzo(*a*)pyrene	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007	0.011	0.045	0.017
Dibenz(*a,h*) anthracene	<0.020	<0.020	0.067	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
Benzo(*g,h,i*) perylene	<0.020	<0.020	0.132	<0.020	<0.020	<0.020	<0.020	0.070	<0.020
Indeno(1,2,3-*c,d*) pyrene	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090
PCB 1254	0.320	0.390	0.250	0.440	0.350	0.500	0.060	0.030	0.080
PCB 1260	0.030	0.020	0.010	0.030	0.020	<0.010	0.020	0.020	0.020
Total PCBs	0.350	0.410	0.260	0.470	0.370	0.510	0.080	0.050	0.100

[a]Each of the three samples is a composite of six clams (~10 g).

the middle reaches of EFPC and a level of total PCBs comparable to that found in the lower reaches of EFPC.

Subsequent analyses of fish from other reference streams considered for this project (Hinds Creek, Beaver Creek, and Bull Run) generally showed <0.1 ppm total PCBs (G. R. Southworth, unpublished data). These results indicate that PCB contamination of reference streams is not a ubiquitous problem.

4.1.3.4 Other Organics

Clams (*Corbicula fluminea*) maintained in the NHP discharge for 4 weeks in March 1986 did not accumulate high levels of polycyclic aromatic hydrocarbons (PAHs) (Table 4-10). Only fluoranthene was detected in quantifiable levels in all four of the individual clams analyzed (0.8 ± 0.1 ppm wet wt, mean ± SE), whereas no fluoranthene was found in clams maintained in Brushy Fork. Other PAHs above detection limits in one or more EFPC clams were naphthalene, phenanthrene, pyrene, benz[*a*]pyrene. All of these compounds were very close to the analytical detection limit, and all but naphthalene and pyrene were found at similar levels in clams maintained in Brushy Fork.

Another set of clams was maintained in the NHP discharge for 4 weeks in July 1986. All were dead on retrieval. The clams were replaced, and sampled after 2- and 4-week exposures in the NHP discharge. Most clams survived, and six-clam (~10 g) composite samples were analyzed for PAHs, PCBs, and organic priority pollutants. Although the larger sample size provided lower detection limits for PAHs in clam tissue, little evidence of PAH accumulation was noted (Table 4-10). Naphthalene, anthracene, fluoranthene, benzo(*b*)fluoranthene, dibenz(*a,h*)anthracene, and benzo(*g,h,i*)perylene were detected in some clams maintained in the NHP discharge, but similar levels were noted in clams from Beaver Creek, the collection site for the clams utilized in this study. Only anthracene and naphthalene exceeded 1 ppm. The presence of only one or two PAHs at significant levels in such samples is unlikely if their source is petroleum or coal-derived hydrocarbon mixtures. Such mixtures would contain a large suite of PAHs, and the more-water-soluble PAHs, such as fluoranthene, pyrene, phenanthrene, and benzo(*a*)anthracene, would be likely to appear in aquatic biota. GC/MS analysis of the *Corbicula* extracts will be used to confirm or refute the presence of anthracene and naphthalene in these samples.

Table 4-10. Concentrations of polycyclic aromatic hydrocarbons in clams (*Corbicula fluminea*) maintained in East Fork Poplar Creek below New Hope Pond and Brushy Fork, a reference site, for 4 weeks, March 1986. Each of the four samples represents a single individual

Compound	Concentration (ppm, wet wt)							
	East Fork Poplar Creek				Brushy Fork			
	1	2	3	4	1	2	3	4
Naphthalene	2	<2	<2	<2	<2	<0.8	<1	<3
Acenaphthene	<0.3	<0.7	<0.5	<0.6	<0.7	<0.3	<0.4	<1
Fluorene	<2	<4	<3	<3	<4	<2	<2	<5
Phenanthrene	0.5	<0.7	<0.6	<0.7	<0.8	<0.4	<0.4	<1
Anthracene	<1	<3	<2	<3	<3	<1	<2	<5
Fluoranthene	0.7	0.5	0.9	0.9	<0.4	<0.1	<0.2	<0.5
Pyrene	0.2	<0.4	<0.3	<0.3	<0.4	<0.2	<0.2	<0.5
Benzo[*a*]anthracene	0.05	<0.06	<0.06	<0.06	<0.07	<0.03	0.05	<0.09
Chrysene	<0.2	<0.4	<0.3	<0.3	<0.4	<0.2	<0.2	<0.6
Benzo(*b*)fluoranthene	0.08	<0.09	<0.08	<0.08	<0.1	0.05	0.1	<0.1
Benzo(*k*)fluoranthene	0.03	<0.04	<0.03	<0.05	<0.05	0.03	0.07	<0.06
Benzo[*a*]pyrene	0.03	<0.05	<0.04	<0.04	<0.05	0.03	0.04	<0.07
Divenz(*a,h*)anthracene	<0.05	<0.1	<0.09	<0.1	<0.1	<0.05	<0.07	<0.2
Benzo(*g,h,i*)perylene	<0.05	<0.1	<0.1	<0.1	<0.1	<0.05	<0.07	<0.2
Indeno(1,2,3-*c,d*)pyrene	<0.3	<0.6	<0.5	<0.7	<0.7	<0.3	<0.4	<0.9

No organic priority pollutants were detected in fish in screening by GC/MS analyses. High-performance liquid chromatography (HPLC)/fluorescence indicated the presence of very low levels of individual PAHs appearing occasionally in single samples (Table C-5). As noted previously, the presence of single PAHs in samples is unlikely if the source of the PAHs is fossil-fuel derived hydrocarbons. The ability of HPLC/fluoresence to detect very low levels of PAHs also makes it susceptible to interpreting interferences as very low levels of PAH, which probably accounts for their reported presence in several fish samples.

4.1.3.5 Cesium-137

Mean concentrations of ^{137}Cs in bluegill and redbreast sunfish collected from EFPC are shown in Table 4-11. Values below the limits of detection were assumed to be equal to those limits in calculating means. Levels of ^{137}Cs ranged from <1.9 to 11 Bq/kg at EFK 23.4. These levels are comparable to those observed in fish from the Clinch River downstream of the mouth of Poplar Creek and are roughly a factor of 5 lower than levels observed in fish from the Clinch River near the mouth of White Oak Creek (Martin Marietta Energy Systems 1984, 1985).

In both the May and December 1985 collections, no significant differences were observed in mean ^{137}Cs levels in fish from EFK 23.4 and EFK 2.1. However, in May 1986, fish from the lower reaches of EFPC contained significantly ($\alpha = 0.05$) higher levels of ^{137}Cs (Table 4-11). The highest level (40 ± 1.7 Bq/kg) was observed at EFK 6.3, ~7 km downstream from the City of Oak Ridge Wastewater Treatment Facility (ORWTF), and is roughly comparable to values for Clinch River fish collected near White Oak Creek (Martin Marietta Energy Systems 1984, 1985). The mean ^{137}Cs levels in fish collected in

Table 4-11. Cesium-137 in bluegill (*Lepomis macrochirus*) and redbreast sunfish (*L. auritus*) from East Fork Poplar Creek, 1985/1986

Site	Concentration (Bq/kg, wet wt)		
	May 1985	December 1985	May 1986
EFK 23.4	4.9 ± 0.9 (5)	6.0 ± 0.8 (8)	7.4 ± 1.5 (4)
EFK 18.2	NS[b]	NS	5.1 ± 2.7 (2)
EFK 13.8	NS	NS	34 (1)
EFK 6.3	NS	NS	40 ± 1.7 (3)
EFK 2.1	2.2 ± 0.9 (7)	6.7 ± 1.3 (4)	17 ± 4.0 (2)

[a]Values are mean ± 1 SE error (sample size in parentheses). Background level (reference stream fish) is <7.7 ± 1.2 Bq/kg (N = 2).

[b]NS = Not sampled.

May 1986 were also significantly greater (α = 0.05) than those of May and December, 1985. Several fish from EFPC were analyzed for ^{137}Cs by TVA in 1984. Bluegill and redbreast sunfish averaged 19 $\pm$ 9.5 Bq/kg dry wt (N = 3) in the TVA collection, which corresponds to about 4 Bq/kg on a wet wt basis. This level is similar to those observed in these species in the upper reaches of EFPC in 1985 and 1986.

4.1.4 Discussion

4.1.4.1 Mercury

Significance of observed levels of mercury in fish

The results of mercury monitoring of EFPC fish in 1985 and 1986 were consistent with the findings of previous studies (Van Winkle et al. 1984; TVA 1985e). Fish from all sites in EFPC remain contaminated with abnormally high levels of mercury. The levels of mercury observed in EFPC are elevated relative to fish collected from reference streams in this study and from other local lakes and streams, including Poplar Creek above the confluence with EFPC (Loar et al 1981a; TVA 1985e), Melton Hill Reservoir (Loar et al 1981b; Elwood 1984), and the Clinch River upstream from the mouth of Poplar Creek (Martin Marietta Energy Systems, Inc. 1984, 1985). Mercury in fish from EFPC exceeded the geometric mean concentration (0.11 μg/g) observed nationwide in the National Contaminant Biomonitoring Program (Lowe et al. 1985). The levels of mercury observed in fish from the lowermost sites on EFPC were similar to those found in fish from lower Poplar Creek and the Clinch River near the mouth of Poplar Creek in previous studies dating back to 1976 (Loar et al. 1981a; Elwood 1984; TVA 1985e; Martin Marietta Energy Systems 1984, 1985). Mercury levels in fish from lower EFPC also approximated those observed in fish from reservoirs downstream from sources of mercury contamination, such as Pickwick and Cherokee Reservoirs (Elwood 1984). Mercury levels in fish from the uppermost reaches of EFPC were similar to levels found previously in highly contaminated sites, such as the North Fork Holston River (Hildebrand et al. 1980). However, total mercury in water and sediments in EFPC are roughly a factor of 10 higher than levels observed in the North Fork Holston River at sites where mercury in fish was ~1 ppm.

In the reach of stream immediately below NHP, nearly half of the redbreast sunfish large enough to be taken by fishermen exceed 1 ppm total mercury (Table 4-1). Assuming that over 90% of the total mercury is methyl mercury, many of these fish would

exceed the current FDA action limit of 1 ppm methyl mercury (FDA 1984a). No sunfish exceeding 1 ppm mercury were collected at EFK 13.8 or any sites further downstream; however, many carp collected from the middle and lower reaches of East Fork exceeded 1 ppm total mercury (Table 4-1). Mercury contamination thus continues to result in excessive levels of mercury in fish throughout the length of EFPC.

Significance of observed patterns of mercury contamination

Mercury vs distance. The 1985/1986 data collected in this study support the conclusion of Van Winkle et al. (1984) that the pattern of mercury contamination in fish and sediments in EFPC is consistent with a continuing discharge of mercury from NHP and subsequent downstream dilution. Levels of mercury in fine-particle sediments immediately upstream from NHP (Fig. 4-4) were similar to those observed in surface sediments of NHP in 1982, indicating a continuing source of mercury within the Y-12 Plant itself. The absence of fish in NHP and EFPC above NHP (Sect. 6.2.3.2) precluded monitoring mercury contamination in fish from those sites. There was no indication of an increase in mercury levels in fish in the vicinity of highly contaminated floodplain areas (upstream of site EFK 13.8) or in fish below the ORWTF (EFK 13.4), suggesting that these are not major sources of biologically available mercury to fish. Levels of mercury that currently exist in EFPC fish are probably determined by the rate of release of biologically available mercury species from the NHP discharge.

The unambiguous pattern of decreasing mercury levels with distance from NHP observed in redbreast sunfish over most sampling periods suggests the presence of relatively discrete populations of fish associated with specific localities in the stream. Thus, contaminant levels in fish can be assumed to be representative of exposure to the contaminant at that locality. Conversely, the presence of relatively homogeneous levels of mercury in fish among sampling sites may be indicative of fish movement. These inferences regarding fish distribution patterns based on the analysis of mercury in fish tissues are consistent with the results obtained from independent studies of fish movements in EFPC (see Sect. 6.2.3.4).

Mercury vs weight. Little relationship was observed between fish weight and mercury levels in this study. In previous studies, a wider range of fish sizes was sampled and a relatively weak but statistically significant relationship was observed between fish

weight and mercury content in EFPC fish (Elwood 1984; Van Winkle et al. 1984). By restricting sampling in the current study to fish large enough to enter the sport fishery, mercury levels in fish likely to be consumed by humans could be measured directly and not extrapolated from regressions of fish weight vs mercury concentration. The former procedure simplified statistical comparisons and reduced possible errors introduced by extrapolating from regressions derived from small sample sizes.

Mercury vs time. Despite the large decrease in mercury levels in sunfish from EFPC between May 1982 and May 1985, mercury levels in fish currently appear to be stable. The record of mercury concentrations in water discharged from NHP showed no indication of a decreasing trend from 1980 to the current time, and mercury levels in readily transported surficial sediments in EFPC do not appear to have decreased since 1982. The decrease in mercury levels in fish may result from changes in the levels of biologically available mercury species or the fact that sediment and water column monitoring are less sensitive measures of temporal changes in the release rates of biologically available mercury than are biological monitors. Another possible explanation for the observed historical differences could be the presence of a systematic difference in measurements between the ORNL analytical laboratory and the laboratories that performed the earlier analyses (i.e., one laboratory always reporting higher or lower results on samples containing the same amount of mercury). However, the quality control procedures utilized in this study and earlier studies do not support this explanation. Although differences in performance on split samples were observed between individual laboratories (Appendix B), they were not large enough to account for the observed changes between 1982 and 1985/1986. Although some of the difference in mercury levels over time may be the result of systematic differences between analytical laboratories, the data most likely reflect actual changes in the level of mercury in fish in EFPC. Data on mercury levels in fish collected from EFPC since this report was prepared indicate no decreasing trend and substantial variation in mercury levels between sampling periods.

4.1.4.2 Other metals

The levels of metals (other than mercury) found in EFPC fish in 1986 agreed with findings of the 1984 ORTF study conducted by TVA (TVA 1985e); that is, fish from EFPC contain levels of these metals typical of fish from local reference sites. A

comparison of the levels of metals in EFPC fish in 1986 with screening levels used by ORNL scientists to evaluate possible health risks also indicated no cause for concern (Hoffman et al. 1984; Travis et al. 1986).

4.1.4.3 PCBs

Significance of observed levels

Contamination of EFPC fish by PCBs was clearly evident in the current study. Levels in redbreast and bluegill sunfish were distinctly elevated above those found in sunfish from uncontaminated reference sites. The pattern of highest residue levels in fish nearest NHP with decreasing levels at sites further downstream was similar to that observed for mercury, reflecting the apparent common source and similar environmental partitioning of the two contaminants. The PCB levels in EFPC fish were not unusual for streams flowing through industrialized areas. Fish contained PCBs at 94% of the stations sampled nationwide in the U.S. Fish and Wildlife Service National Pesticide Monitoring Program. The geometric mean total PCB concentration (wet weight, whole fish) in those collections was 0.53 ppm (Schmitt et al. 1985).

The levels of total PCBs observed in EFPC sunfish were well below the 2 ppm FDA action limit (FDA 1984b) but may be of concern if viewed on the basis of recent carcinogenicity test results and commonly accepted risk analysis protocols (Hoffman et al. 1984; Travis et al. 1986). Such protocols result in the identification of extremely low levels of PCBs in fish as warranting concern.

The levels of total PCBs in carp from EFPC are elevated to much higher levels than those observed in sunfish, with many fish approaching or exceeding the FDA limit. The 1984 TVA survey of EFPC fish collected only three carp, but all were found to equal or exceed the FDA limit. The 1985/1986 data confirm the suggestion of the earlier study that a significant proportion of carp from EFPC contains residues of PCBs in excess of 2 ppm.

The redbreast sunfish (*Lepomis auritus*) and common carp (*Cyprinus carpio*) appear to be useful biomonitors of PCB contamination in streams. Although the sunfish does not accumulate PCBs to nearly as high levels as the carp, its movements are restricted enough to monitor exposure conditions at specific collection sites. Moreover, the sunfish has low background levels of PCBs in uncontaminated areas (yielding good

analytical sensitivity despite lower concentrations) and is short lived, thus providing a measure of recent exposure. The carp, on the other hand, accumulates hydrophobic contaminants to higher levels; therefore low residue levels in this species would imply that all other fish contain low PCB levels. The Asiatic clam (*Corbicula fluminea*) also holds promise as an effective monitor. Because it accumulates detectable levels of PCBs rapidly and can be placed in specific sites, it should be valuable in identifying the location of PCB sources above NHP.

PCBs vs distance

As noted previously, the relationship between PCBs in redbreast sunfish vs distance downstream from NHP was similar to that found for mercury. The same pattern of PCB concentration vs distance was also noted in 1982 (McElhaney 1982) in bluegill sunfish. Like mercury, PCBs in EFPC appear to have a source upstream from the NHP outfall. The similarity between the levels of PCBs in surface sediments upstream of NHP and surface sediments in the pond suggests a continuing source of PCB contamination within the Y-12 Plant complex. The pattern of PCBs in sunfish vs distance was also similar to the pattern of PCBs vs distance in fine surface sediments and is in general agreement with theoretical expectations of PCB partitioning among water, sediment, and biota (Kenaga and Goring 1980).

The data did not show a significant increase in PCB levels in redbreast sunfish below the ORWTF or adjacent to the highly contaminated floodplain between sites EFK 18.2 and EFK 13.8, indicating that these are not currently major sources of PCB contamination relative to the NHP discharge.

PCBs vs time

The decrease in PCBs in EFPC sunfish since 1982 is similar in pattern and magnitude to that of mercury over the same period. No further decrease was apparent between May 1985 and May 1986. No PCBs were detected in bluegill, redbreast sunfish, or rock bass from EFPC in 1984, suggesting that levels have increased from 1984 to the present. However, PCB levels in surface sediments of the upper reaches of EFPC in 1984 were comparable to the levels observed in 1986, and PCB concentrations in carp from EFPC in 1984 were comparable to those observed in 1985/1986. Although it seems

unlikely that PCB contamination in EFPC was lower in 1984 than in 1982 or 1985/1986, it is not known if wide temporal variations occur because the nature of the source is unknown.

The high variability observed in PCB analyses and possible systematic losses of lower chlorinated congeners in KOH digestion procedures makes comparison of historical levels of PCBs in fish problematic (see Appendix B). There is no doubt, however, that PCBs were present in the NHP outfall in the recent past (1982) and continue to enter EFPC somewhere above that point.

Variation in PCBs among species

Carp contained much higher levels of PCBs than did sunfish in EFPC. This result was not unexpected; carp, especially large carp such as those collected in EFPC, tend to be older and contain higher intramuscular lipid levels than do sunfish. Because hydrophobic contaminants (such as PCBs) accumulate in lipids, higher levels would be expected in fattier fish. It is also possible that PCB levels in carp reflect a historical exposure to much higher PCB levels 5 to 10 years ago. Levels of PCBs in sediment cores from NHP suggest that much higher levels of PCBs entered NHP in the years soon after it was dredged in 1973 (H. L. Boston, ORNL Environmental Sciences Division unpublished data) than enter the pond currently.

The higher proportion of hexachlorobiphenyl isomers (PCB-1260) found in carp relative to sunfish also suggests that a component of the PCB residues in carp reflects a previous exposure to higher levels of PCBs. The more highly chlorinated PCBs are less rapidly excreted by fish (Niimi and Oliver 1983). Consequently, PCB residues acquired in previous years would now be depleted of the less chlorinated isomers, leaving a higher proportion of PCB-1260 to PCB-1254.

Results of the current study indicate declining levels of PCBs in EFPC carp, but analytical uncertainties prevent the assessment of this apparent decline with certainty. If current levels are a result of historical exposure, then PCBs in carp should decrease in the future as older fish are replaced in the population.

4.1.4.4 Other organics

Analyses of *Corbicula fluminea* following a 4-week exposure in the discharge from NHP did not indicate the presence of significant levels of PAHs. However, PAHs were found in significant levels in NHP sediments and sedimentary deposits within the EFPC floodplain (TVA 1985b,c), indicating a source in the Y-12 Plant complex. The possibility exists that PAHs in these sediments are associated with coal fines and, thus, are biologically inert. However, biochemical changes in fish from EFPC immediately downstream from NHP were typical of those found in fish exposed to aromatic hydrocarbons (Sect. 5.3.1.4).
Therefore, it is advisable to continue using introduced clams to monitor PAHs in the upper reaches of EFPC.

Analyses of organic priority pollutants in fish agreed with findings of the ORTF Study (TVA 1985e) that none of these substances were accumulated in fish in EFPC.

4.1.4.5 Cesium-137

The levels of ^{137}Cs observed in fish from EFPC were all below the maximum permissible concentrations in fish of 100 and 63 Bq/kg used by Hoffman et al. (1984) and Travis et al. (1986), respectively. Levels of ^{137}Cs were occasionally higher in the lower reach below NHP, suggesting a possible source of intermittent contamination in the vicinity of the ORWTF. In 1984, sludge from the ORWTF was found to be contaminated with ^{137}Cs and other radionuclides (Oakes et al. 1984). Levels of ^{137}Cs in fine surface sediments analyzed by TVA in 1984 were also highest in this lower portion of EFPC (TVA 1985b,c).

Cesium is relatively labile in fish compared with mercury and PCBs (Kolehmainen and Nelson 1969). If episodic discharges are a primary source, levels in biota may fluctuate widely over the course of a year and typical maximum levels of ^{137}Cs in fish might not be observed in a biannual monitoring program. If more detailed studies on releases and possible sources of ^{137}Cs to EFPC are conducted, they should focus on the accumulation of ^{137}Cs in sieved sediments, which irreversibly bind cesium and are representative of specific locations in the stream (Cerling and Spalding 1982).

4.1.5 Future Studies

4.1.5.1 PCB and mercury monitoring

The biannual monitoring of PCBs and total mercury in fish throughout the length of EFPC will continue. Data from these sampling programs will be used both to assess the trend of declining levels of mercury and PCBs in EFPC fish and to provide early detection of any increase in contaminant levels. Bluegill abundance in the middle reaches of the stream (EFK 18.2, EFK 13.8, and EFK 6.3) is not sufficient to provide adequate sample sizes for statistical evaluation. Therefore, collection of bluegill for mercury and PCB analysis will be discontinued at these sites unless populations increase substantially.

Caged clams (*Corbicula fluminea*) were demonstrated to be effective monitors of PCB contamination. Their use will be expanded into upper EFPC above NHP to identify possible sources of PCBs to this reach of the stream. Such studies will be carried out in conjunction with clam exposures at sites throughout EFPC.

4.1.5.2 PCB and mercury uptake studies

High levels of mercury and PCBs are found in sediments above, in, and below NHP. Elemental mercury is found in Y-12 Plant pipes, sumps, and drains that drain to EFPC. Studies have been initiated to investigate the importance of NHP and EFPC sediments and dissolved elemental mercury as sources of mercury and PCB contamination in EFPC fish. In the initial phase of this study, blacknose dace (*Rhinichthys atratulus*) fish are being exposed to water overlying sediments from NHP and EFPC and to water continuously passing over elemental mercury in 30-L aquaria. Simultaneously, fish are maintained in cages below NHP. Levels of mercury accumulated after 2, 4, and 8 weeks of exposure will be measured and compared. At the conclusion of this experiment, a similar study using fish or clams will be conducted that will look at the accumulation of PCBs from sediments.

4.1.5.3 Broad spectrum monitoring of metals and organics

Sunfish and clams will again be collected for broad-spectrum contaminant analysis in 1987, ~6 months after discharges are initiated at the West End Treatment Facility. Such monitoring should detect any increases in contaminant levels in fish associated with new effluent streams.

4.1.5.4 Re-analysis of carp samples for PCBs

The decrease in PCBs in carp observed between 1985 and 1986 coupled with wide variation in results of PCB analyses, indicates a need to re-analyze selected archived carp samples to verify the observed change in PCB levels. This analysis will be conducted concurrently with the December 1986 fish sampling, so that fish from all four sampling dates can be analyzed in a single submission to the analytical group.

4.2 CRITICAL FACTORS IN TRANSPORT, FATE, AND BIOAVAILABILITY

4.2.1 Introduction

This task investigates the critical factors that control the transport, fate, and bioavailability of contaminants in EFPC, with a focus on the organic contaminants. The conceptual model describing the distribution of contaminants in EFPC is shown in Fig. 4-7. The goal of this task is to (1) quantify the rate coefficients associated with the different arrows on Fig. 4-7 and (2) understand how seasonal factors and other environmental variables modulate these rates. These data will eventually be incorporated into a dynamic model to predict the response of biota to various remedial actions at the Y-12 Plant.

Studies in the past year were conducted in five major areas

1. quantifying bioavailability of contaminants from water and sediments;
2. measuring the efficiency with which contaminants in food are assimilated by fish;
3. determining the efficiency with which contaminants in water are extracted by the gills of fish;
4. identifying the significance of dissolved organic matter (DOM) as a sorbent for contaminants; and
5. identifying a simple predictor of the affinity of DOM for binding contaminants in EFPC.

4.2.2 Bioavailability of Contaminants from Water and Sediments

The accumulation of a contaminant from water and sediment by fathead minnows (*Pimephales promelas*) was measured under controlled laboratory conditions. The PAH,

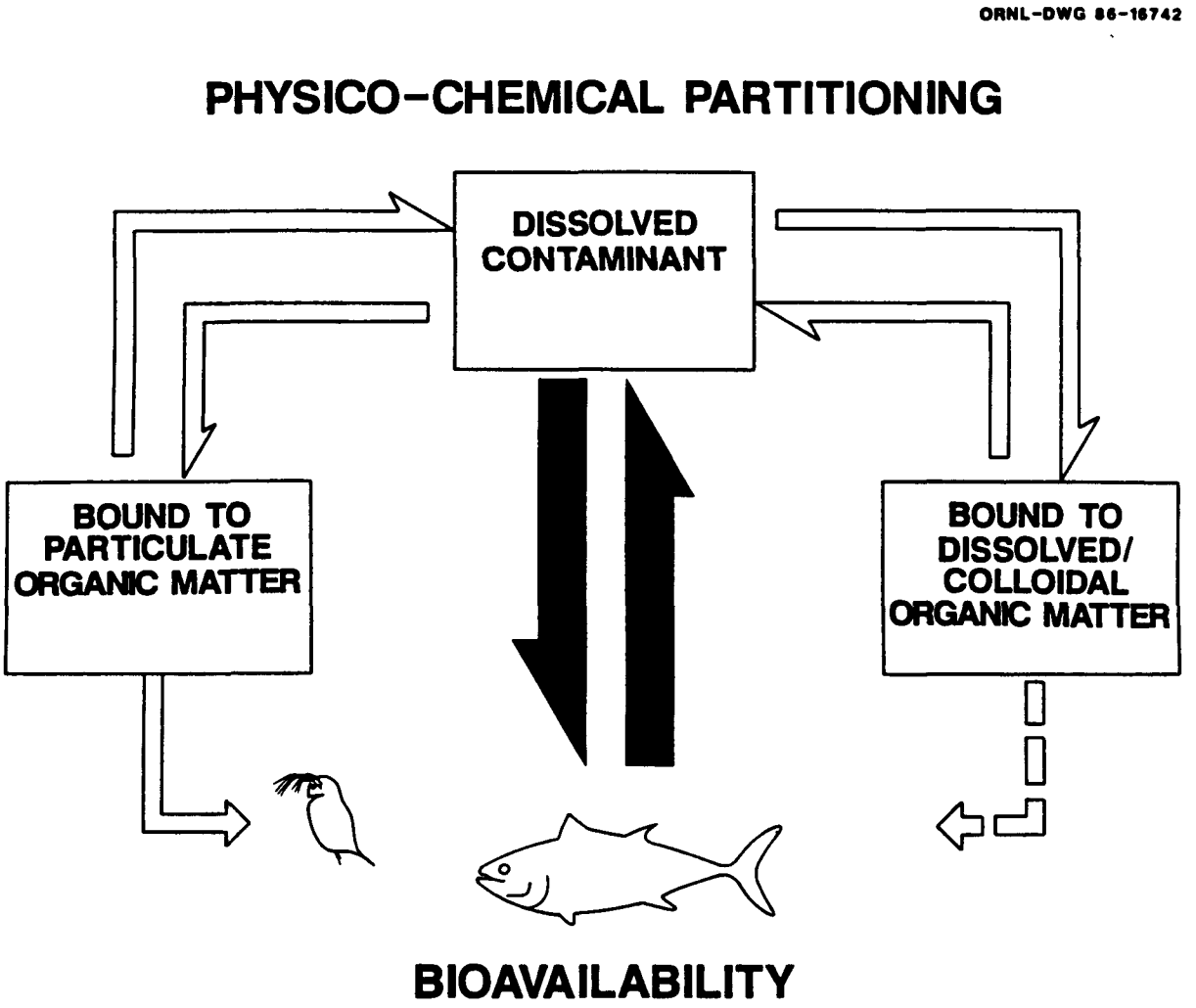

Fig. 4-7. Conceptual model illustrating the environmental compartments with which contaminants can be associated in aquatic systems. Contaminants that are sparingly soluble in water tend to bind to particles and sediments and dissolved organic matter. Biota can accumulate contaminants most readily if the contaminants are dissolved in water. Organisms can also accumulate by ingestion of particles or sediments to which compounds are bound or contaminated organisms at lower trophic levels.

benzo(*a*)pyrene (BaP), was used as a prototypical contaminant. Like PCBs, BaP has very limited solubility in water and a high affinity for accumulation in animals and for sorption to sediments and DOM. It is a ubiquitous environmental contaminant and a known carcinogen and mutagen. The behavior of BaP in the environment should be very similar to that of other PAHs and other classes of hydrophobic contaminants, such as PCBs.

The accumulation of radiolabeled BaP by minnows was measured in the absence of sediment or particles to determine the rate coefficients for uptake and elimination of the dissolved contaminant. The uptake from sediment contaminated with BaP was also measured using sediments with high and low organic content (OC). Sediments with high OC have a higher affinity for binding BaP and other PAHs and PCBs (Karickhoff et al. 1979; Means et al. 1979). The rate of ingestion of the two sediment types by fish might also be different, resulting in different levels of exposure.

4.2.2.1 Water exposure

The exposure of fathead minnows to ^{14}C-BaP was conducted using a flowing water system. Details of the exposure system have been described previously (McCarthy and Jimenez 1985a; Jimenez et al. 1987). At various times over a 10-d exposure period, animals were removed from the tank and analyzed for accumulated radioactivity by (1) combustion in a Packard Model Oxidizer and (2) liquid scintillation spectrometry.

The rate coefficients for uptake and elimination of BaP were estimated by an iterative least-squares fitting procedure (Knott 1979), using the following equation

$$dC_f/dt = (k1 \times C_w) - (k2 \times C_f) \ , \qquad (4\text{-}1)$$

where C_f and C_w are the concentration of BaP in the fish and water (pg/g or pg/mL), respectively, and k1 and k2 are the rate coefficients for uptake and elimination, respectively. The rate coefficients estimated by this procedure are presented in Table 4-12.

4.2.2.2 Sediment exposures

The two sediments used in these experiments differed in their OC content: the high OC sediment contained 8% OC (determined using a Leco Total Carbon Analyzer),

Table 4-12. Rate coefficients (asymptotic standard error in parentheses) for uptake from water, k_1, and for elimination, k_2, from the fish are indicated for animals exposed to BaP in water alone and in high- and low-organic sediment. The sediment ingestion rate and the best estimate of k_3, the rate coefficient for uptake of contaminant attributable to ingestion of sediment, are given

	Water-only exposure	High-organic sediment	Low organic sediment
k_1,[a] mL g^{-1} h^{-1}	43 (+3)	39 (+3)	92 (+7)
k_2,[a] h	0.030 (+0.003)	0.031 (+0.008)	0.033 (for 0–72 h) (+0.007) 0.17 (for 72–240 h) (+0.02)
Sediment ingestion,[b] g g^{-1} h^{-1}		1.8×10^{-4}	6.2×10^{-4}
k_3[b], g g^{-1} h^{-1}		$<10^{-4}$	$<10^{-4}$

[a]The rate coefficients were estimated by fitting the data on the time course of accumulation of ^{14}C-BaP and the concentration of BaP in the water to Eq. (4-1).

[b]The sediment ingestion rate was estimated from the slope of a plot of log (^{141}Ce) accumulated by the fish over time (first order rate coefficient). The estimate of k_3 was obtained by fitting data on the time course of accumulation of ^{14}C-BaP by fish in the sediment exposures to Eq. (4-2).

and the other contained 2% OC. Sediments were double-labeled with ^{14}C-BaP and ^{141}Ce. Cerium-141 has a very high affinity for binding to the inorganic matrix of the sediment and served as an easily monitored marker to measure the amount of sediment ingested by the fish and to help ensure that measurements of the ^{14}C radioactivity in the water were not compromised as a result of the presence of suspended particles. The amount of ^{141}Ce was measured using a gamma counter. The sediment was rinsed several times with water to remove excess ^{14}C-BaP that did not bind to the sediment. The sediment was placed in aquaria, water was added, and the distribution of BaP between water and sediment was allowed to equilibrate before fathead minnows (~1 g wet wt per fish) were introduced. The aquaria were aerated, but the water was not changed during the 240-h (10-d) exposure.

At various times, groups of animals were removed from the aquaria. Whole minnows were weighed and counted for ^{141}Ce to determine the rate of ingestion of sediment. At 240 h, a group of 5 fish was dissected and the gut and carcass measured in the gamma counter; all of the ^{141}Ce was in the gut and none was present in the rest of the animal, indicating that ^{141}Ce was not absorbed by the fish. The fish were analyzed for ^{14}C-radioactivity, as described for the water exposures. The sediment and water were also analyzed during the time course of the exposure. Samples of the water were centrifuged to remove particulates and counted for ^{14}C-radioactivity. Centrifuged water samples were also analyzed for ^{141}Ce to ensure that all suspended particles were removed; the presence of suspended particles would erroneously inflate estimates of the concentration of dissolved ^{14}C-BaP in the water.

4.2.2.3 Results

The concentration of BaP in the water overlying the sediment was greatly influenced by both the OC of the sediment and activity of the fish. The water concentrations in the low-OC sediment were approximately twice that in the high-OC sediment tanks (Fig. 4-8a). This result was expected because the low-OC sediment has a lower affinity for binding BaP, so more was released to the water. The presence of fish also increased the amount of BaP in the water compared with identical aquaria without fish (data not shown; $p < 0.003$). The feeding activity of the fish mixed the sediment with the water and enhanced the release of BaP from the sediment. The OC of the sediment also affected the rate of ingestion of sediment by the minnows. Fish ingested more of the low-OC sediment ($p < 0.01$) than the high-OC sediment. This result agrees with the observation of Cammen (1980) that fish adjust their feeding rates in response to the amount of organic material present in their food.

The accumulation of ^{14}C-BaP by the minnows was greater in the aquaria having low-OC sediment than in those having high-OC sediment (Fig. 4-8b). This finding is consistent with the fact that fish in the low-OC sediment are exposed to higher concentration of dissolved BaP (Fig. 4-8a) and that they ingest more of the contaminated low-OC sediment (although the low-OC sediment had a lower concentration of BaP).

The patterns of uptake by fish was obviously different in the two sediments (Fig. 4-8b). The amount of BaP in the fish in the low-OC-sediment aquaria increased

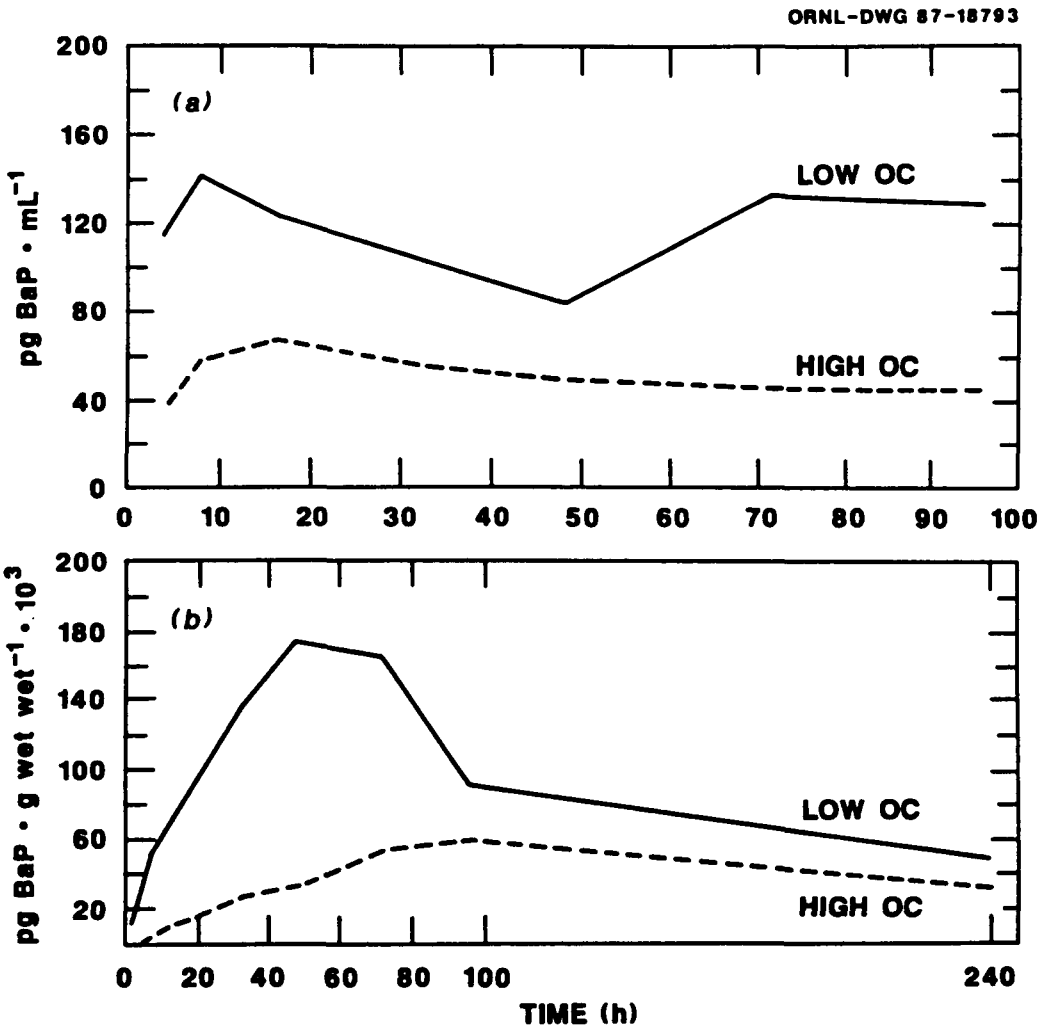

Fig. 4-8. The concentration of BaP in (a) water and (b) fish over the 240-h exposure period. The solid lines are data from exposure to low-organic-content sediment and dotted lines are from tanks containing high-organic-content sediment.

rapidly, then declined after 72 h to a lower steady state during the remainder of the exposure. These results can be interpreted in terms of an induction of the enzyme systems for detoxification and elimination of PAHs (mixed function oxidase/P450 system). In animals receiving a sufficient dose of PAH, the enzyme systems are induced by increased activity of existing enzymes and/or by production of new types of enzymes. In bluegill sunfish (*Lepomis macrochirus*), induction occurs within 2 to 4 d after exposure to BaP (B. D. Jimenez, ORNL, unpublished data). The fish in the low-OC sediment tanks appear to have received a sufficiently high dose of BaP to induce the detoxification enzymes, resulting in a five-fold increase in the elimination rate after 72 h of exposure (Table 4-12).

An attempt was made to quantify the rate coefficient for accumulation of BaP from the sediments using the equation

$$dC_f/dt = (k1 \times C_w) - (k2 \times C_f) + (k3 \times C_s) \ , \qquad (4\text{-}2)$$

where C_s is the concentration of BaP in the sediment (pg/g of sediment) and k3 is the rate coefficient for uptake of BaP from the sediment. The k3 was too small, relative to k1, to be accurately estimated by the current model (Table 4-12). The results did demonstrate that the BaP in the water was by far the most readily available source of contaminant and that ingestion of sediment would be less important as a route of uptake.

It would be inappropriate to interpret these results as indicating that contaminants bound to sediment are totally unavailable for uptake by biota. The exposure system used in these experiments made it difficult to estimate the rate of uptake from the sediment because the concentration of BaP in the water was relatively high. The experimental design should be modified to allow flow-through of uncontaminated water to minimize the concentration of BaP in the water, thus avoiding the complications from rapid uptake from the water. In an environmental context, even a slow rate of uptake can lead to significant accumulation of pollutants by animals. Furthermore, the results of this study demonstrate that sediment can be a source of contaminant released to the water, where it becomes readily available for bioaccumulation. Feeding activity by fish enhanced this transfer by increasing contact between the sediment and water.

4.2.3 Assimilation of Anthracene by Fish

Because of the difficulty in interpreting the previous experiment, an attempt was made to measure the availability of PAH ingested on contaminated food by gavage (forced-feeding) experiments. Trout chow (Purina) was labeled with ^{14}C-anthracene (a three-ring PAH) and ^{141}Ce. The ^{141}Ce served as an inert marker to follow the passage of the contaminated bolus through the fish without requiring quantitative determination of the amount ingested or quantitative recovery of the fecal material. This method also eliminated the need to make corrections for the water content of the food and feces. Assimilation efficiency was determined by measuring the ratio of ^{14}C-Anth and ^{141}Ce in the contaminated food and in the feces:

$$\% \text{ assimilation} = \frac{(^{14}C/^{141}Ce \text{ in food}) - (^{14}C/^{141}Ce \text{ in feces})}{(^{14}C/^{141}Ce \text{ in food})} . \quad (4\text{-}3)$$

Because hydrophobic PAHs, such as anthracene, preferentially associate with lipids, the influence of the lipid content of the food on the availability of Anth for uptake across the gut membrane was tested. Some of the Trout Chow was extracted in hexane prior to labeling with anthracene and ^{141}Ce to remove the lipids in the food.

Rainbow trout (*Salmo gairdneri*) were anesthetized with MS-222 (trican methane sulfonate). A glass tube packed with contaminated food was inserted into the gut, and the food was extruded into the gut using a glass plunger. Fecal material was collected from the bottom of the tank in which the fish were held and from the large intestine. Results are shown in Table 4-13. As expected, the food from which the lipids were extracted contained less anthracene (lower ^{14}C/^{141}Ce ratio). Regardless of the type of food, the fish assimilated ~84% of the anthracene in the food.

These results indicate that anthracene, and probably other organic contaminants, in food are readily assimilated by fish. To a first approximation, it can be assumed that all of the contaminant ingested will be incorporated by the organism. Assimilation may be different if the contaminant is naturally incorporated into the food (e.g., by uptake of pollutants by lower-trophic-level animals, which are ingested by predators). These factors will be explored in future studies.

Table 4-13. Percentage assimilation of ^{14}C-anthracene in food with high- and low-lipid content that was force-fed to rainbow trout

Food	$^{14}C/^{141}Ce$ in food	Fecal source	Percentage assimilation	Mean (+S.E)
High lipid	26.8	large intestine	91.8	84.6 (+8.2)
		large intestine	68.3	
		bottom of tank	93.7	
Low lipid	17.9	large intestine	93.1	84.0 (+6.6)
		large intestine	85.5	
		large intestine	64.7	
		bottom of tank	92.6	

4.2.4 Efficiency of Extraction of Contaminants by Fish Gills

Studies described previously indicated that a contaminant dissolved in water is readily available for uptake by fish. This uptake occurs by absorption of the compounds by the gills as the contaminated water is pumped past the gills during the normal ventilatory activity of the fish. The amount of contaminant accumulated via this route is dependent on the respiratory demand of the animal, which changes with temperature, feeding activity, reproductive demands, etc. Experiments were initiated to measure the efficiency with which contaminants are extracted from water pumped over the gills, thus enabling changes in exposure to be predicted over a seasonal cycle.

Extraction efficiency was measured using a metabolic chamber (Fig. 4-9). Fish were immobilized by spinal transection, and a piece of latex rubber was sutured around their mouths. After an animal was placed in the chamber, the latex was mounted around the support between chambers A and B to create a water-tight seal between the chambers (Fig. 4.9). Another piece of latex rubber was fitted around the animal's midsection to create a water-tight seal between chambers B and C. Contaminated water entered chamber A (arrow 1 in Fig. 4-9) and was pumped into chamber B by the normal ventilatory activity of the fish (arrows 2 and 3 in Fig. 4-9). The difference in the concentration of contaminant in chambers A and B was used to calculate an extraction efficiency.

Initial experiments measured the efficiency with which BaP was extracted by the gills of the rainbow trout. Extraction efficiency for oxygen and BaP was measured in animals acclimated to 18°C. The change in the extraction efficiency resulting from the

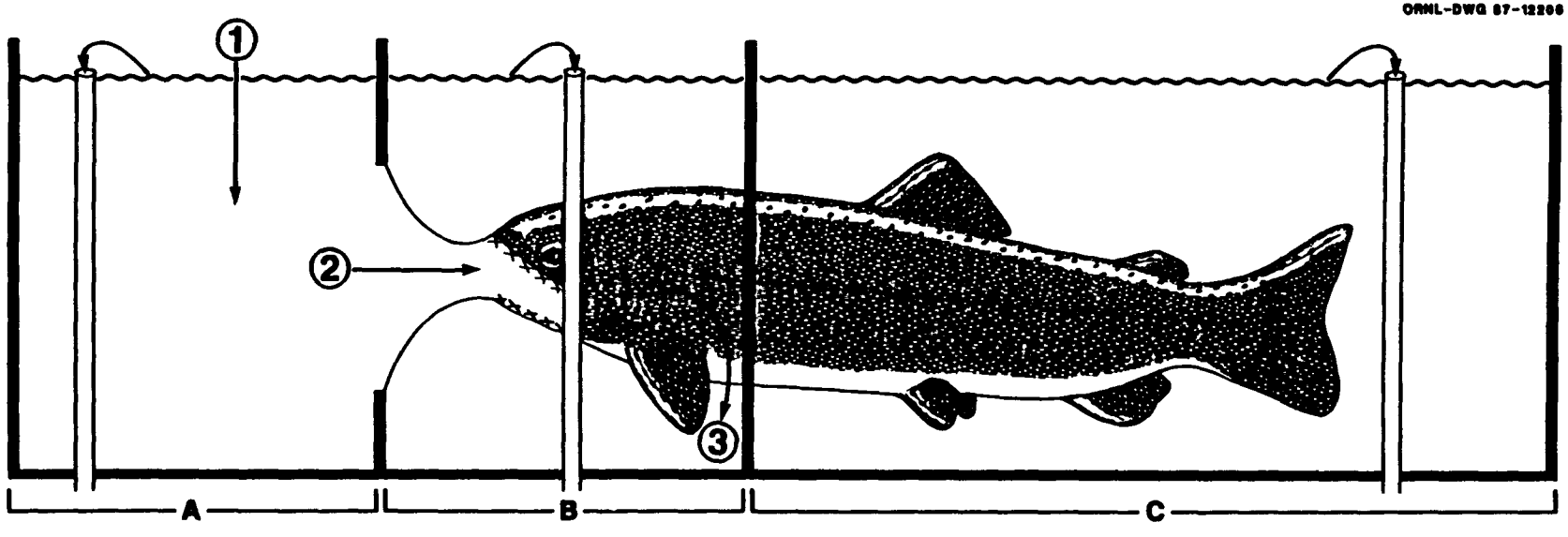

Fig. 4-9. Metabolic chamber used to measure the extraction efficiency of gill. Contaminated water entered (arrow 1) chamber A and was pumped into chamber B by the normal ventilatory activity of the fish (arrows 2 and 3). A piece of latex rubber fitted around the animal's midsection created a water-tight seal between chambers B and C. The difference in the concentration of contaminant in chambers A and B was used to calculate an extraction efficiency.

presence of DOM was also examined. The source of the DOM was humic acid (Aldrich Chemical Co.). Results are indicated in Table 4-14. Oxygen extraction efficiency, ventilation rate, and ventilation volume remained the same regardless of the presence of DOM. The efficiency of extraction of BaP decreased by 85% because of the presence of 5 mg C/L of DOM. Bioaccumulation in free-swimming fish and aquatic invertebrates is also reduced in the presence of DOM (McCarthy and Jimenez 1985a; McCarthy et al. 1985).

Table 4-14. Extraction efficiency for oxygen and BaP by gills of rainbow trout

DOM (mg C/L)	N[a]	Oxygen extraction efficiency	Ventilation rate[b] (per min)	Ventilation volume[b] (mL/min)	BaP extraction efficiency
0	8	52.6% (+5.2)	100.2 (+2.6)	213.5 (+14.2)	49.0% (+1.4)
0.5	4	58.3% (+1.4)	102.8 (+1.9)	246.2 (+33.9)	14.2% (+1.0)
1.0	5	53.1% (+6.2)	116.4 (+1.7)	255.2 (+21.2)	10.3% (+0.9)
5.0	5	61.0% (+0.7)	102.0 (+1.9)	180.0 (+14.0)	7.4% (+0.5)

[a]The number of fish tested under each condition (N) is noted.
[b]The ventilation rate and volume are indicated for fish exposed in the presence and absence of DOM.

The extraction efficiency may change with temperature and ventilation rate. The efficiency of oxygen extraction is inversely related to the ventilation volume, presumably because of the reduced contact time between the water and the gill at higher flow rates. A similar relationship is expected for extraction of contaminant. This hypothesis will be examined in future studies.

4.2.5 Significance of DOM on Transport and Bioavailability

Naturally occurring DOM is capable of binding organic contaminants with an affinity comparable to that of particulate organic matter in sediment (McCarthy and

Jimenez 1985b; Morehead et al. 1986). The potential competition between DOM and sediment particles for binding of contaminants was examined. The results of these studies are described in detail in McCarthy and Black (1987). The affinity of a contaminant for associating with a sorbent can be quantified as an association coefficient K_p,

$$K_p = C_p / (C_d \times P) \quad , \tag{4-4}$$

where C_p and C_d are the concentration of contaminant bound to the sorbent particles or dissolved in water, respectively, and P is the concentration of particulate sorbent.

In practice, measurements of K_p are routinely made by centrifuging or filtering particles and determining the concentration of the contaminant in the particulate vs nonparticulate (dissolved) phases. However, contaminant bound to DOM remains in the nonparticulate phase and can lead to significant errors in predicting the amount of contaminant bound to particles. Figure 4-10 illustrates the exponential decline in the apparent K_p (K_{app}) for the binding of BaP to yeast cells when increasing amounts of DOM are present in the water. Yeast cells were used as a model particle because of their uniform size, shape, and surface characteristics. Figure 4-10 demonstrates that the presence of DOM can reduce the binding of contaminant to particles by orders of magnitude.

If the dissolved contaminant binds independently and noninteractively with each sorbent, then the multiple sorbent system can be analyzed

$$K_p \text{ for BaP-yeast} = K1 = C_p/C_d{*}P \quad , \tag{4-5}$$

$$K_p \text{ for BaP-DOM} = K2 = C_{dom}/C_d{*}DOM \quad , \tag{4-6}$$

$$K_{app} = C_p/(C_d+C_{dom}){*}p = K1/1+K2{*}DOM \quad , \tag{4-7}$$

where C_{dom} = pmol contaminant bound to DOM/mL and DOM = concentration of DOM (g C/mL). K1 and K2 were measured individually and in combination using equilibrium dialysis. Two dialysis bags were suspended in a jar containing water, DOM, yeast particles, and ^{14}C-BaP. One dialysis bag contained only water, and the other

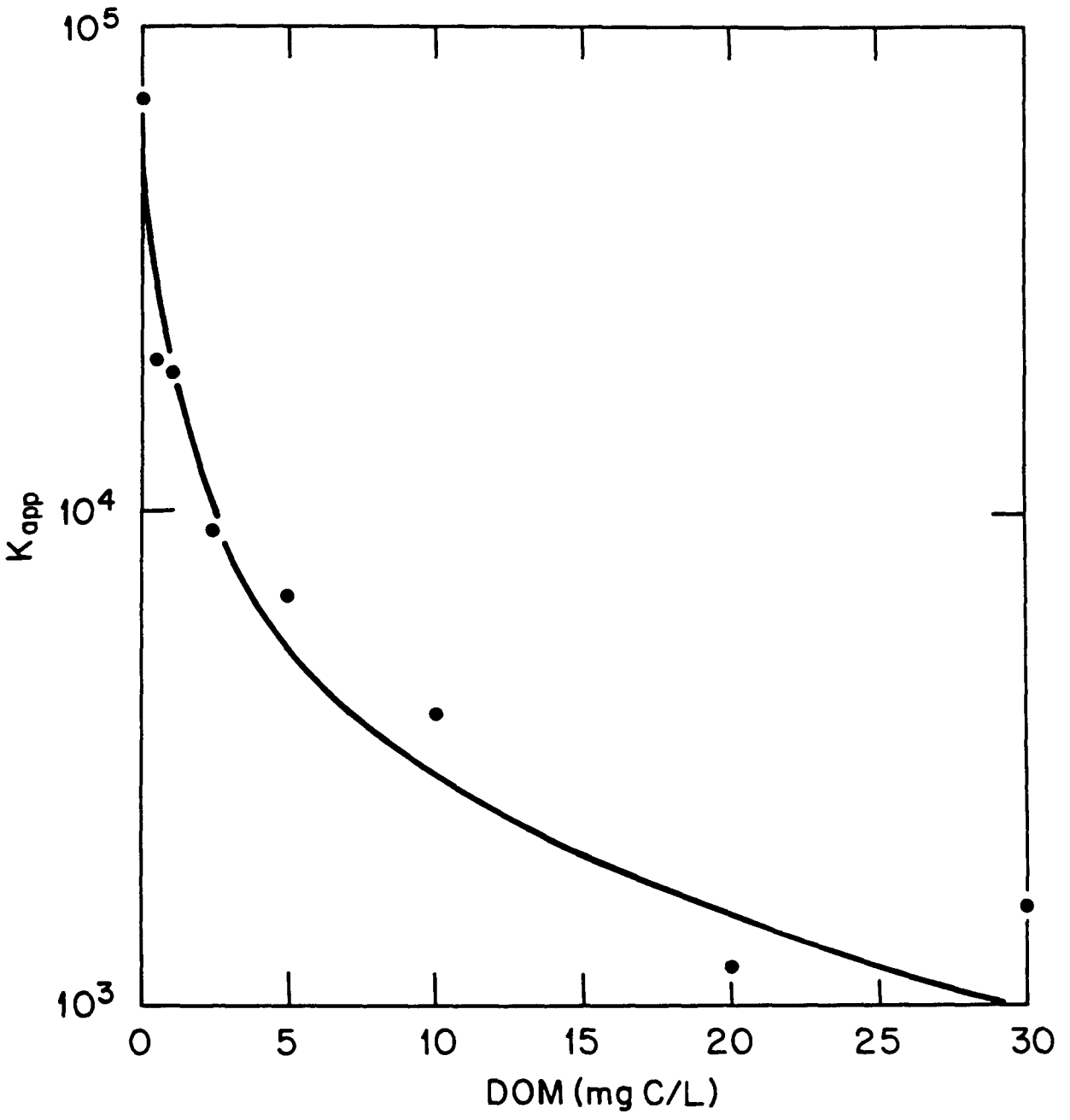

Fig. 4-10. The decrease in the apparent K_{pf} (K_{app}) for binding of BaP to particles in the presence of DOM. Circles are data derived from the ratio of BaP in the particulate and nonparticulate phases of the centrifuged water. The curve represents model predictions based on fitting the data to Eq. (4-7).

contained water plus DOM (same DOM concentration as in the bulk solution outside the bags). After equilibration, the contents of the dialysis bags were analyzed for radioactivity. The water in the jar was centrifuged, and the pellet and supernatant were analyzed for radioactivity. This system provided information on the concentrations of BaP (1) dissolved in water (from the dialysis bag containing only water), (2) bound to DOM (from the dialysis bag containing DOM), (3) bound to particles (from the pellet), and (4) in the nonparticulate phase (from the supernatant of the water in the jar). From these data, K1 was determined to be 7.2×10^4 (+0.6 SE) and K2 was 1.6×10^6 (+0.04 SE).

Substituting these values for K1 and K2 into Eq. (4-7) resulted in a line that was in excellent agreement with the data for K_{app} in Fig. 4-10. To further test the appropriateness of this model, the data in Fig. 4-10 were fitted to Eq. (4-7) using a nonlinear least-squares fitting procedure to estimate K1 and K2 based on observed values of K_{app}. The measured K1 and K2 fall within the 95% asymptotic confidence intervals for the estimates from the statistical fitting procedure. These results indicate that the competitive binding of DOM and particles with contaminants can be predicted from the association coefficients for the individual sorbents.

Hassett and Anderson (1982) suggested the possibility of an interaction between the contaminant-DOM complex and the particle by

$$K3 = C_{p,dom}/C_{dom}*P \ , \tag{4-8}$$

and

$$K_{app} = K1 + K2*K3*DOM \,/\, K2*DOM + 1 \ , \tag{4-9}$$

where $C_{p,dom}$ = pmol of DOM-bound contaminant binding to particles per gram of particle. The data in Fig. 4-10 were fitted to Eq. (4-9) and the values of K1, K2, and K3 were estimated. The value for K3 was very low, compared with those for K1 and K2, with a 95% asymptotic confidence interval that included zero. Comparison of the residual sums of squares (incremental F-test) for the two-parameter and three-parameter models [Eqs. (4-7) and (4-9), respectively] indicate that the extra parameter did not significantly improve the fit of the data to the model ($p > 0.05$). In this system, the interaction between the DOM and the particles exerts little influence on contaminant partitioning.

This result is also consistent with the observation that the DOM concentration in the supernatant of the centrifuged bulk water before the addition of yeast did not differ significantly from that remaining at the end of the incubation period ($p > 0.05$), suggesting that there was no measurable binding of DOM to the particles.

Results of these experiments suggest that DOM can compete independently with particulate sorbents for binding of hydrophobic organic contaminants. Further research is needed to test this result using other natural waters and a wider variety of particles. There may be more interaction of particulate and dissolved organic matter (K3) in natural systems than in this laboratory proof-of-principal experiment.

The preceding results demonstrate that DOM affects binding of contaminants to particles and also alters the bioavailability of contaminants. For a variety of PAHs, PCBs, and other hydrophobic compounds, binding to DOM makes the compounds unavailable for uptake by animals (see Sect. 4.2.4; Leversee et al. 1983; McCarthy and Jimenez 1985a,b; McCarthy et al. 1985). Predictive fate and transport models and simpler equilibrium models used to assess the environmental impact of contamination generally account for the partitioning of hydrophobic contaminants between a particulate and nonparticulate phase. In some situations, failure to include a third phase (a dissolved macromolecular or colloidal sorbent, such as DOM) can lead to serious errors in estimates of the form and distribution of contaminants. Errors could include (1) overestimation of the amount of contaminant bound to particles, (2) overestimation of contaminant removal from the water column resulting from sedimentation of particles, and (3) underestimation of the extent of downstream advection of contaminants stabilized in the water column by DOM.

McCarthy and Black (1987) demonstrated that the error in transport or bioaccumulation predictions resulting from failure to account for the presence of DOM in the system can be predicted:

$$\% \text{ error} = (1 + K_{oc}*POM / 1 + K_{oc}*POM + K_{oc}*DOM) * 100\% \ , \qquad (4\text{-}10)$$

where K_{oc} is the carbon-referenced association coefficient for the contaminant (Karickhoff et al. 1979) and POM is the concentration of particulate organic matter in the system (mg C/L). The shaded areas of Fig. 4-11 illustrate the ranges of POC and DOM

concentrations for which failure to consider the effect of DOM would lead to a 50% (Fig. 4-11a) or 90% (Fig. 4-11b) error in predictions of the fractions of contaminants in the dissolved and particulate phases for contaminants having a K_{oc} of 10^7 to 10^3. When POM levels are very high and DOM levels are low, binding to POM dominates and DOM has little effect (lower right-hand corner of figures). As POM concentrations decrease and DOM levels increase, DOM becomes increasingly important for accurate predictions.

The POM concentrations in EFPC vary along the length of the stream and over the course of the year, although only limited data are available currently (Sect. 3.3). Concentrations of DOM also vary seasonally and with distance downstream. Because many sections of EFPC are poor in organic-rich sediments, DOM may play an important role in contaminant transport in this system. Significant errors can be introduced in predictions of the transport and bioaccumulation of very hydrophobic compounds ($K_{oc} > 10^{4.5}$) if DOM is not measured and incorporated into predictive models. Future work will include measurements of POM and DOM in EFPC over a seasonal cycle.

4.2.6 Predicting the Binding Affinity of DOM for Contaminants

The preceding analysis of the significance of DOM assumes that all DOM has the same affinity for binding contaminants. However, accurate prediction of the environmental fate of contaminants is hampered by the fact that DOM from different sources of water can vary by orders of magnitude in their affinity for binding contaminants (Morehead et al. 1986; McCarthy et al., 1989). The structural and chemical properties of DOM, which determine the capacity of the DOM for binding contaminants, were examined, and a simple and easily measured predictor of binding affinity was identified.

The affinity of DOM from ten diverse sources (several sites in EFPC, commercial humic acid, groundwaters, and ponds) were analyzed by equilibrium dialysis for their ability to bind BaP. The hydrophobic acid and hydrophobic neutral composition of the DOM was analyzed by chromatography on XAD-8 macroreticular resins. The molar volume of the DOM was analyzed using fluorescence polarization. The extinction coefficient of the DOM at 270 nm (E_{270}) was measured by absorbance spectroscopy.

The DOM (Fig. 4-12) having a high affinity for binding BaP can be characterized as an open, loose, spongy macromolecule with not only a high percentage of hydrophobic

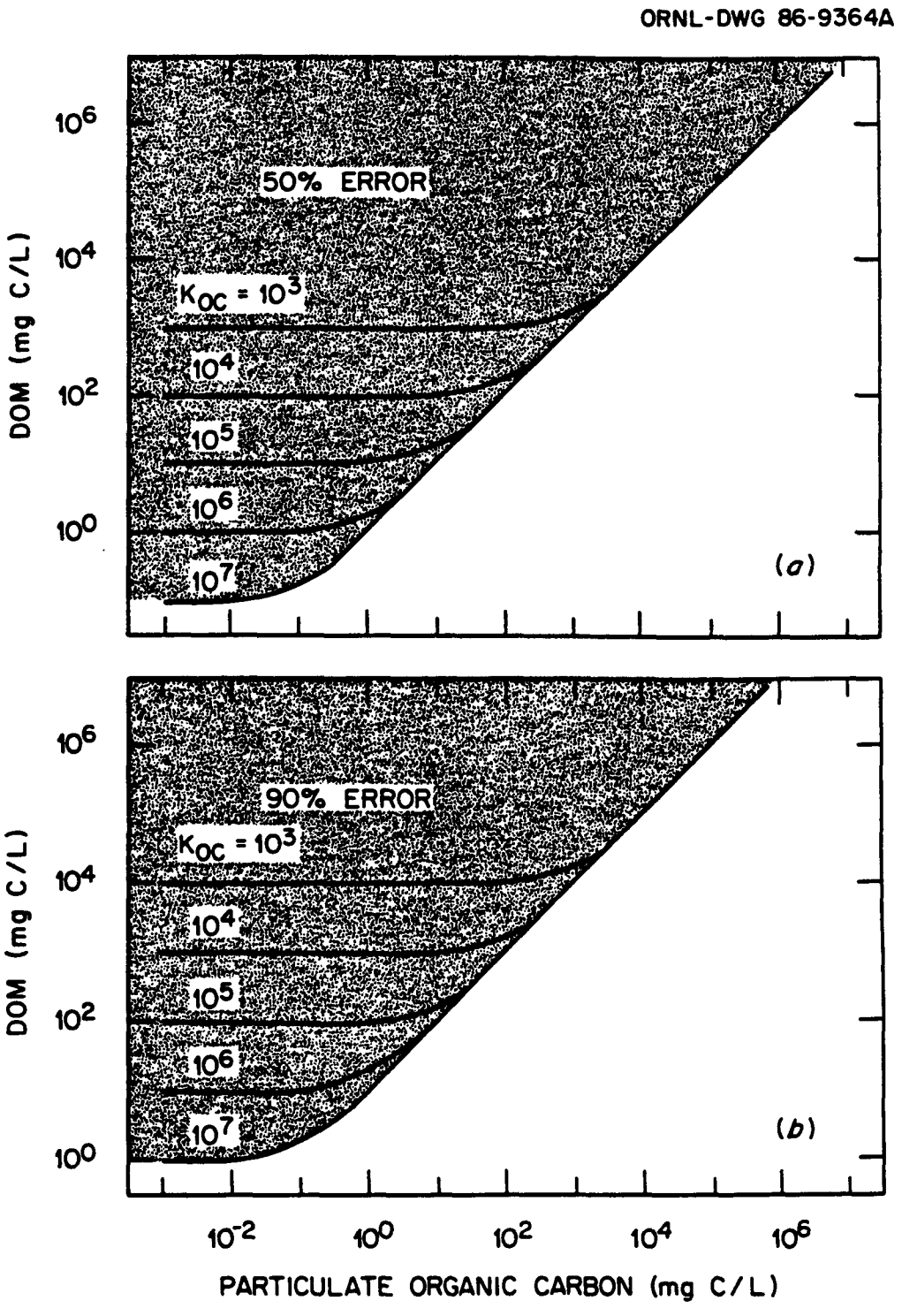

Fig. 4-11. Error associated with failure to account for the role of DOM as a sorbent in predictions about exposure of aquatic organisms. The shaded areas indicate the range for which interactions between dissolved organic matter (DOM) and particulate organic carbon (POC) would lead to a (a) 50% or (b) 90% error in estimates of the fraction of contaminant truly dissolved or bound to particles if the role of DOM as a competing sorbent were ignored. The 50% or 90% error regions are shown for contaminants having a K_{oc} from 10^3 to 10^7; DOM-POC combinations above or to the left of the line indicate the error region for each K_{oc}.

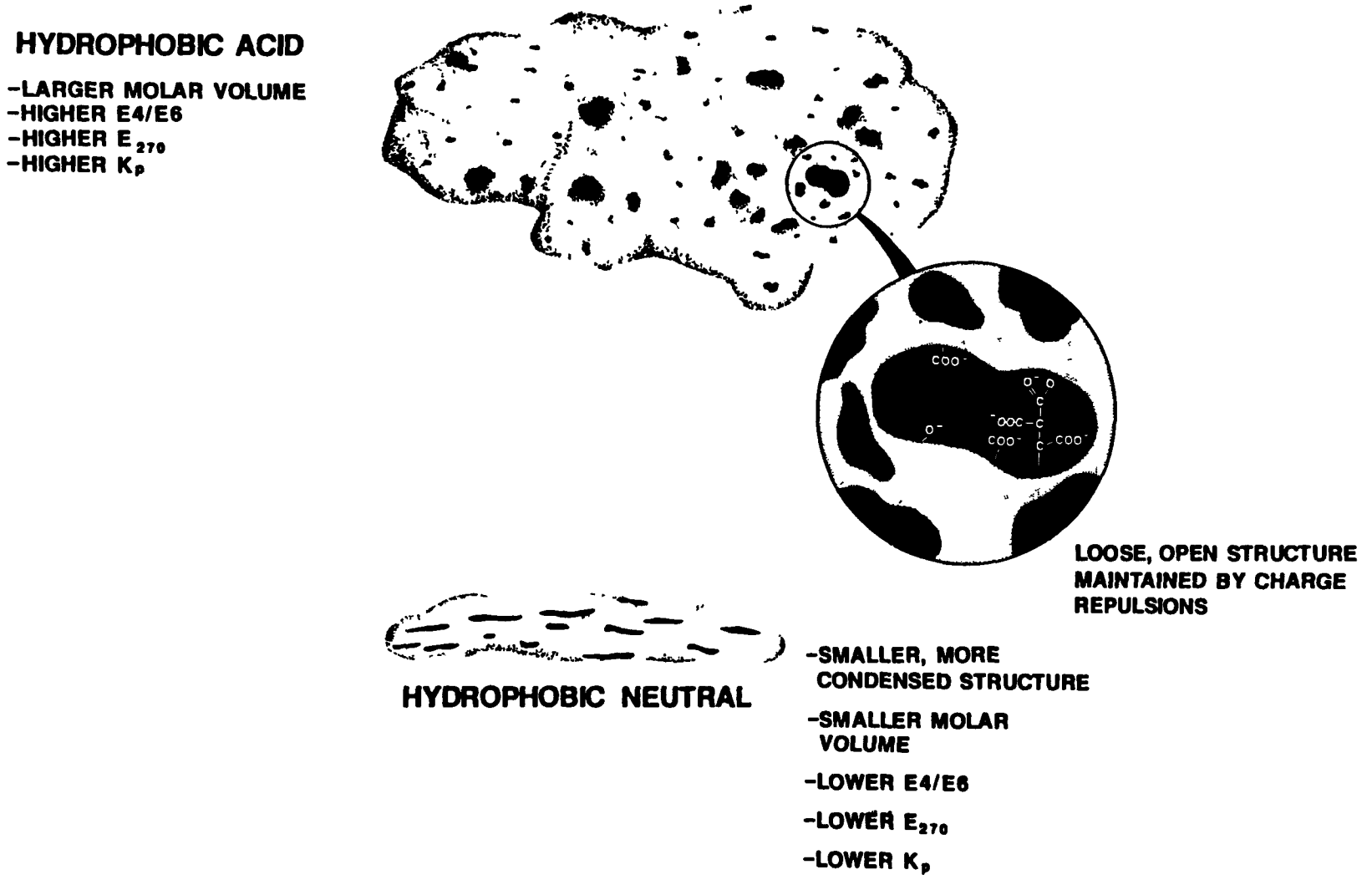

Fig. 4-12. A conceptual model of the structural and chemical properties of DOM related to its affinity for binding hydrophobic contaminants and the effect of aging or humification of the DOM on these properties. The drawings of the macromolecules are for illustrative purposes and are not intended to represent actual molecular shapes.

constituents, but also a sufficient number of acidic functional groups to maintain an open structure resulting from charge repulsions between carboxyl and other negatively charged groups. The DOM from other sources of water have a much lower affinity for binding BaP. This DOM is hydrophobic but lacks acidic functional groups and has a low aromatic content and a smaller molar volume. This more mature macromolecule has a lower affinity for binding BaP (Fig. 4-12). An excellent predictor of binding affinity is the E_{270} of the DOM (Fig. 4-13). This parameter is a powerful predictor because it probes the same chemical and structural features that determine the affinity of the DOM for binding BaP. An open, loose DOM macromolecule provides an accessible hydrophobic structure that readily binds BaP. In this configuration, the aromatic chromophores (absorbing at 260 to 280 nm) efficiently absorb light. As the structure ages and condenses, the hydrophobic aromatic constituents fold in on themselves, becoming less efficient at absorbing light and less accessible for binding BaP.

This conceptual model of the structure of DOM needs to be extended to other contaminants and to a larger number of water samples from EFPC over a seasonal cycle during the leaching and subsequent humification of DOM from leaf-fall through the summer. However, this approach promises to provide a simple and easily measured predictor of the capacity of DOM to bind organic contaminants and thus will significantly improve the accuracy of predictions of organic contaminant transport and bioaccumulation in aquatic systems.

4.2.7 Summary of Results and Conclusions

The bioavailability of hydrophobic contaminants from water, sediment, and food was investigated, and significant conclusions were drawn.

1. Contaminants dissolved in water are rapidly incorporated by fish.
2. Contaminants bound to sediment are far less available for uptake, but dissociation of contaminants from the sediments can be a source for bioaccumulation.
3. Contaminants in food are efficiently assimilated by fish.
4. The extraction efficiency for uptake of dissolved contaminants by fish gills can be measured using a metabolic chamber.
5. The presence of DOM reduces uptake of contaminants through gill membranes.

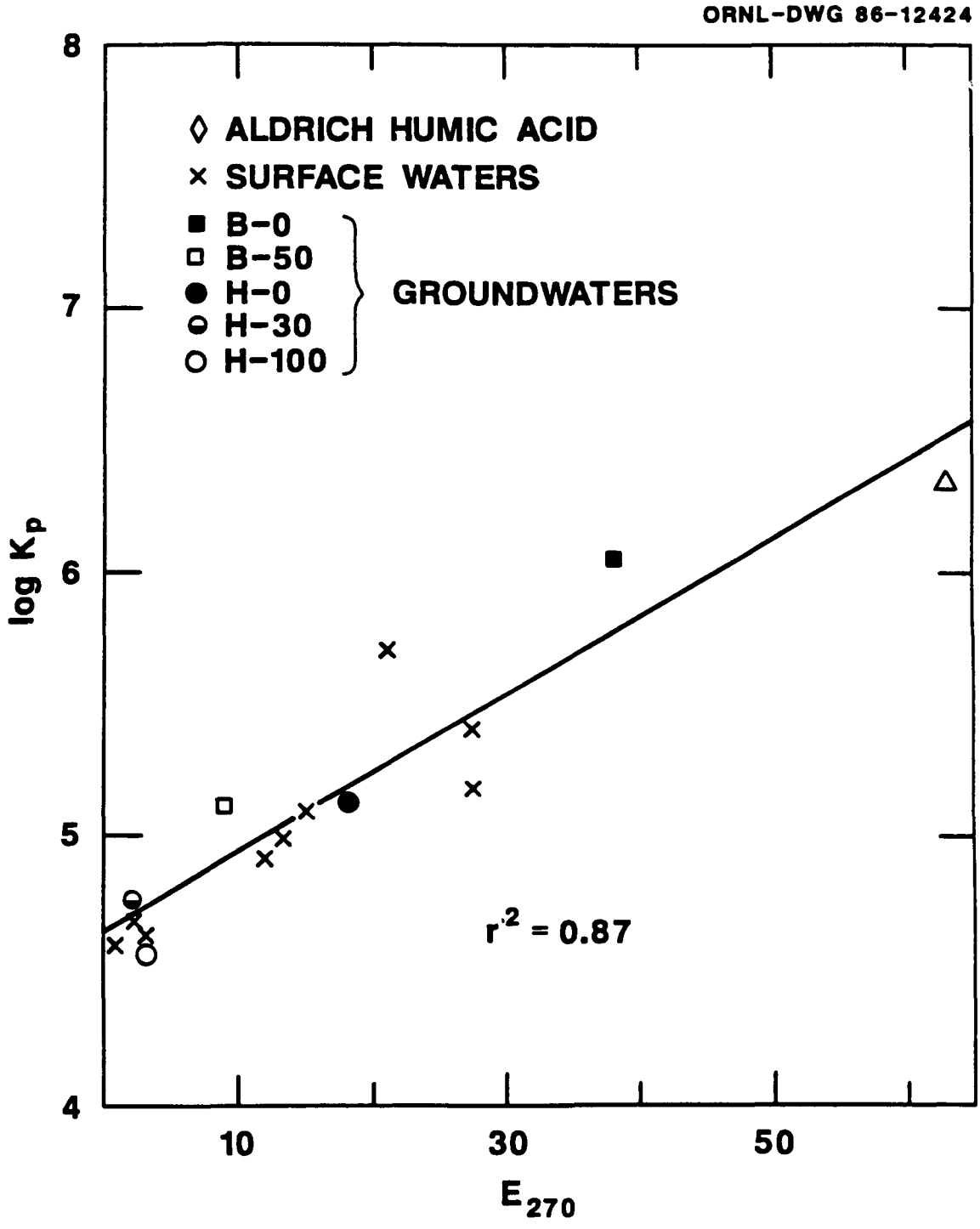

Fig. 4-13. The log of the K_p for binding of BaP to DOM for different waters is directly related to the E_{270} of the water. The symbols for the different water samples are indicated on the figure.

6. DOM can compete with sediment for binding contaminants, but the partitioning can be predicted in terms of independent interactions of the dissolved contaminant with each sorbent.
7. Failure to account for the role of DOM can lead to significant errors in predictions of the transport and fate of contaminants, depending on the relative abundance of dissolved and particulate organic matter in the system.
8. A conceptual model is proposed relating the affinity of DOM for binding contaminants to the chemical and structural properties of the sorbent.
9. The E_{270} of the water can be used to estimate the affinity of the DOM in the water for binding contaminants.

4.2.8 Future Studies

During the next year, the factors controlling the transport and bioavailability of organic contaminants will be examined in greater detail. Studies will be focused on the three major areas that follow.

4.2.8.1 Role of dissolved sorbents in EFPC

The affinity of DOM in EFPC for binding organic contaminants, such as PAHs and PCBs, will be examined quarterly throughout the year. Seasonal changes in organic input to the system should result in significant variations in the nature of the DOM and, thus, the capacity of that DOM for binding contaminants. This research should greatly improve understanding of the potential significance of DOM in altering the transport of contaminants in EFPC and lead to more-accurate predictions of contaminant fate.

4.2.8.2 Sediments as a source or sink for contaminants

The capacity of sediment for binding contaminants and the reversibility of this process will be examined. Sediments may sequester contaminants in the system and make them less available for bioaccumulation, but conversely, desorption of the bound contaminants may form a persistent source of future contamination. These processes will be examined, with special reference to the possible role of DOM as a competitive sorbent and potential facilitator for release of contaminants bound to sediments.

4.2.8.3 Seasonal and physiological factors in contaminant bioaccumulation

The metabolic fish chamber will be used to explore the influence of variables such as temperature, reproductive status, and presence of dissolved sorbents on the uptake of contaminants by fish. These results will be interpreted in the context of bioenergetic models to predict changes in bioaccumulation over an annual cycle.

5. BIOLOGICAL INDICATORS OF CONTAMINATION-RELATED STRESS

5.1 INTRODUCTION

This task involves the development, screening, and application of various biological indicators to evaluate the responses of fish populations in the East Fork Poplar Creek (EFPC) to effluent discharges from the Y-12 Plant. Biological indicators are valuable because they (1) can provide early warning signals of potential ecological effects because of their sensitivity to environmental change, (2) can be used to identify causal mechanisms underlying observed effects at higher levels of biological organization, and (3) are generally easy to measure and cost effective for long-term biomonitoring.

During the first year (1985/1986) of the Biological Monitoring and Abatement Program (BMAP), two major subtasks were addressed (1) selection/screening of bioindicators and (2) application of bioindicators in the field for long-term monitoring. A suite of indicators, representing a series of biological responses along a gradient of relatively short-term to long-term responses, was selected and measured in the field. Each indicator was evaluated by the criteria (1) through (3) in the preceding paragraph, and a subset of the most relevant indicators was selected for long-term field monitoring studies. Evaluation of the bioindicators was based on comparisons of both individual and integrated biological responses of fish from various areas of EFPC to the responses of fish from reference (control) areas.

5.2 METHODS

The biological indicator task was initiated in the spring of 1985 with a preliminary screening study. Initially, the strategy was to sample three species of fish at each of four EFPC sites and two reference sites (Brushy Fork and Watts Bar Reservoir). The three target species, bluegill (*Lepomis machrochirus*), redbreast sunfish (*L. auritus*), and rockbass (*Ambloplites rupestris*), were sampled for blood chemistry, liver enzyme activities, and various measures of overall condition. However, several problems were encountered with this initial sampling strategy: (1) all three species did not occur at all sites, thus severely limiting comparisons of responses among sites, (2) adequate sample sizes for statistical analyses could not be obtained for all three species at all sites, and (3) a study of all three species at all sites was cost prohibitive. Based on the information obtained in this

preliminary screening study, the redbreast sunfish was selected for intensive study. It was the only large, sport fish species that occurred in sufficient numbers at all sites to provide adequate sample sizes for a wide range of bioindicators.

5.2.1 Sampling Procedures

Sampling was conducted during the fall of 1985 and the winter and spring of 1986 at four sites on EFPC: (1) immediately below New Hope Pond, EFPC kilometer (EFK) 23, (2) near the Route 95 bridge at Jefferson Avenue (EFK 18), (3) above the City of Oak Ridge Wastewater Treatment Facility (ORWTF) (EFK 14), and (4) near the U.S. Geological Survey (USGS) gaging station (EFK 5) (see Fig. 2-1). Fish were also collected from reference sites on Brushy Fork (BF) near BF kilometer (BFK) 7.0 and Watts Bar Reservoir (WBR) near the Clinch River arm (see Fig. 2-2).

A minimum of 12 adult redbreast sunfish were collected by electroshocking at each site during each season. Blood samples were taken from each fish within 2 to 3 min following capture with unheparinized syringes and placed on ice. Each fish was identified with a numbered tag for future reference and transported alive to the laboratory.

5.2.2 Analytical Procedures

Total lengths and weights were recorded before the fish were sacrificed, then the liver, gonads, gills, and kidney were removed for analysis. The liver and ovaries were weighed and sections of each organ were preserved for histopathological analysis. The remainder of the liver was used for enzyme assays. Blood samples taken in the field were centrifuged, and the serum was frozen for subsequent biochemical analysis. Blood analysis was performed on a centrifugal fast analyzer by the ORNL Health Division following the procedures describes in Adams et al. (1985). Organs removed for histopathological studies were shipped to the Department of Pathology at the University of West Virginia School of Medicine for analysis. Qualitative evaluations of each organ were performed as described in Hinton et al. (1973).

Following removal of all organs, the carcass was analyzed for total lipid content according to the procedures of Bligh and Dyer (1959). A subsample of the lipid extract from the carcass was shipped to the Technical University of Nova Scotia for lipid

biochemical analysis, using the methods described by Harvey and Patton (1981) and Rigler (1983).

For enzyme assays, livers removed from fish were homogenized in ice-cold 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 mM ethylenediaminetetraacetic acid (EDTA), and 10% glycerol by sonication. Protein was 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), 80 mM with 5 mM magnesium acetate, 1 μM 7-ethoxyresorufin, 0.25 mJ nicotinamide adenine dinucleotide phosphate (NADPH), and 1 mM EDTA (Egan et al. 1983). The electron transfer enzyme, NADPH-cytochrome c reductase, was assayed spectrophotometrically by a modification of Phillips and Langdon (1962), using cytochrome c as an electron donor. Reduced cytochrome c was determined using an extinction coefficient of 21.1 cm^{-1} mM^{-1}. The reaction mixture contained 50 mM tris buffer (pH 7.4), 20% glycerol, 1 mM dithiothreitol (DTT), 1 mM EDTA, 1.1 mg/mL horse heart cytochrome c, 0.175 mM NADPH, and 2 to 10 mg of microsomal protein. The activity of nicotinamide adenine dinucleotide (NADH)-cytochrome b_5 reductase was assayed in a manner identical to that for NADPH cytochrome c reductase, except that 0.25 mM NADH was used. 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 NADH.

5.2.3 Statistical Procedures

Analysis of variance (ANOVA) procedures were used to test for differences in individual bioindicators among sites and sexes for each seasonal data set. Interaction effects between site and sex were also included in the ANOVA model. If the ANOVA procedure rejected a multisample (site or sex) null hypothesis of equal means, then the Tukey multiple range test was used to identify significant differences among pairs of

variables (e.g., sites). The Tukey test was used because it is fairly robust with respect to departures of the data from normality and homogeneity and also because it is the most widely accepted and commonly used multiple comparison test (Zar 1984).

To determine the integrated response of fish to the environmental conditions at each sampling site, all the bioindicator variables were considered jointly within a multivariate context by using canonical discriminant analysis (Seal 1968). This method provides a graphical representation of the positions and orientations of the various integrated site responses relative to each other.

5.3 RESULTS AND DISCUSSION

Blood biochemistry, liver enzymes, and lipid biochemistry along with various integrated measures of overall body and histopathological condition were quantified in fish collected from EFPC and two reference areas during the fall of 1985 and winter and spring of 1986. Data are presented first for individual bioindicators and then for integrated
bioindicator responses.

5.3.1 Individual Bioindicator Responses

Individual bioindicator responses can be conveniently grouped into six functional categories: (1) blood biochemical parameters [serum sodium, potassium, glucose, triglycerides, cholesterol, serum glutamate oxaloacetate transaminase (SGOT), and total protein], (2) lipid biochemistry (total body lipids, body triglycerides, phospholipids, and steriols), (3) condition indices [liver-somatic index (LSI), visceral-somatic index (VSI), and condition factor], (4) liver detoxification enzymes, and (5) histopathological responses (liver, gonads, gills, and kidney). The individual response parameters in each group are discussed in the following sections.

5.3.1.1 Blood biochemical parameters

Indicators of electrolyte homeostasis

Serum sodium and potassium were used as indicators of electrolyte homeostasis or ionoregulatory function. Because sodium and potassium behaved similarly in fish from all sampling sites, only sodium responses are presented here. Levels of this electrolyte were

generally lowest in the spring and highest in the winter (Fig. 5-1). No clear pattern in sodium levels was observed among sites. Sodium levels in fish from BF were significantly lower than the levels observed in fish from EFK 23 and EFK 18 in the fall and from EFK 18 and EFK 5 in the spring (Table 5-1). Additional monitoring is needed to adequately evaluate the usefulness of serum sodium as an indicator of ionoregulatory dysfunction in EFPC fish.

Table 5-1. Statistical comparisons between sampling sites, by season, for individual bioindicators measured in redbreast sunfish

	BF vs EFK 23			BF vs EFK 18			BF vs EFK 14			BF vs EFk 5			BF vs WBR[a]		
	F[b]	W[b]	S[b]	F	W	S	F	W	S	F	W	S	F	W	S
Sodium	--[c]			--		--						--			
Glucose	-														
Serum triglycerides	+[d]		++			++							+		+
Chloresterol		-								--					--
Serum protein			+							--		--	++		++
SGOT									-						+
Total lipid			++	--						--			++		++
Phospholipid	--												++		
Body triglycerides	++												++		
Condition factor	--			-		-			-				++		++
Visceral-somatic index		--											++		
Liver-somatic index	--	--		-		-						-	++		++

[a]WBR = Watts Bar Reservoir.

[b]F = fall 1985; W = winter 1986; S = spring 1986.

[c]-- and - = value for BF significantly lower at the 99 and 95% confidence levels, respectively, than the comparison site.

[d]++ and + = value for Brushy Fork (BF), the reference, significantly higher at the 99 and 95% confidence levels, respectively, than the comparison site.

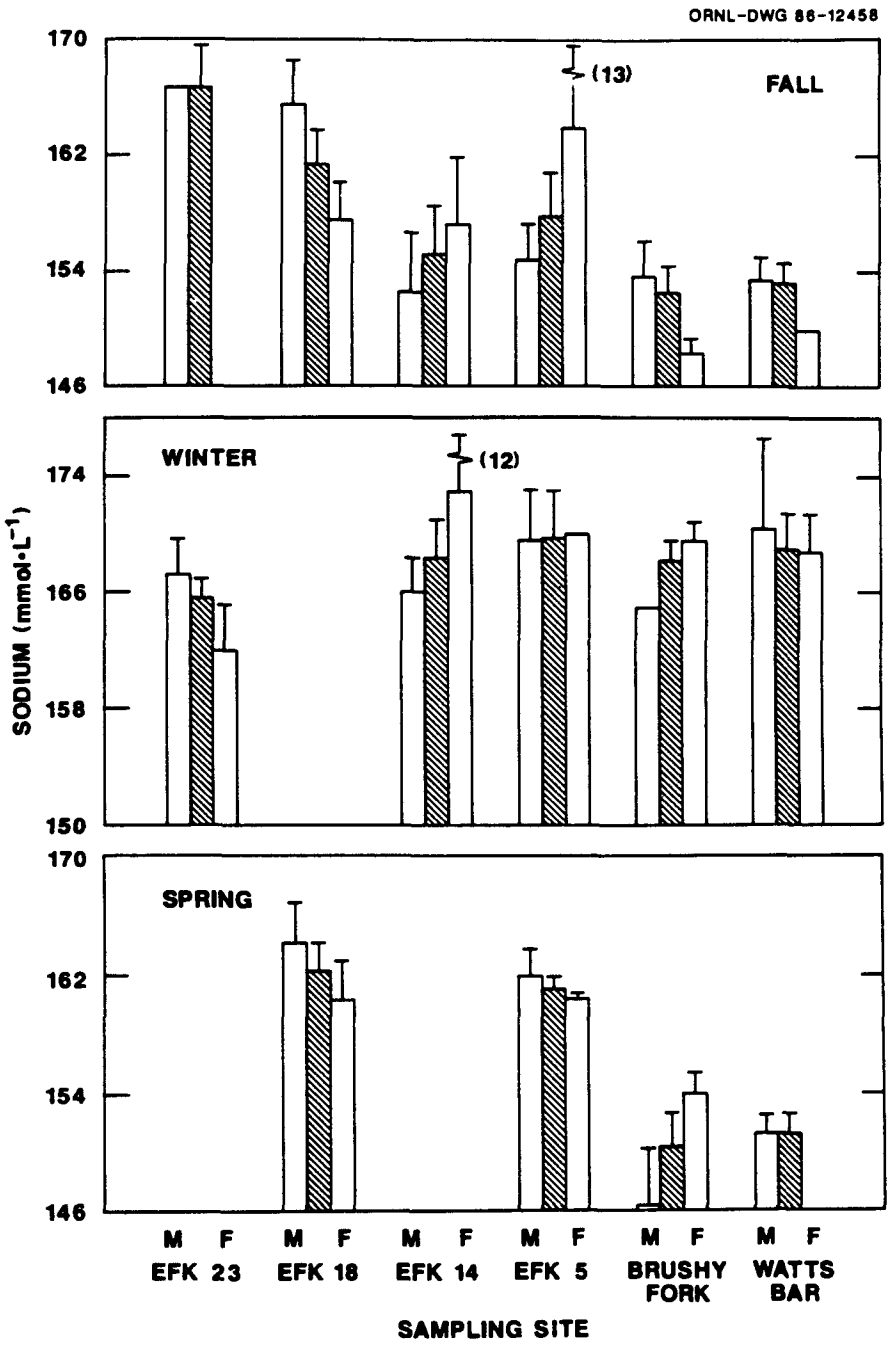

Fig. 5-1. Levels of sodium in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for male (M), females (F), and the two sexes combined (hatched bars).

Indicators of carbohydrate metabolism

Serum glucose was measured as an indicator of carbohydrate metabolism. Few differences were found in the glucose levels of fish collected from the various sites (Table 5-1), although some seasonal differences were noted (Fig. 5-2). For example, levels of glucose in fish from most sites was generally highest in the fall. Hyperglycemia (elevated glucose levels) in fish, however, is a generalized stress response to a broad spectrum of environmental perturbations (Silbergeld 1974) and may not be a good indicator for assessing toxicant stress in fish populations of EFPC.

Indicators of nutrition and lipid metabolism

Both serum triglycerides and cholesterol provide measures of nutritional status and metabolism (mobilization) of energy reserves during stress. Lipids can be mobilized by fish to partially mediate the effects of stress (Lee et al. 1983). In the fall and spring, fish from BF had significantly higher levels of serum triglycerides than fish from EFK 23 and WBR (Table 5-1). Cholesterol patterns between sites and seasons were more ambiguous than those observed for the triglycerides (Figs. 5-3 and 5-4, respectively). The level of triglycerides in the blood appears to be a good short-term indicator, whereas, cholesterol is a long-term indicator of nutrition and chronic stress.

Indicators of protein metabolism

Levels of SGOT (a transaminase enzyme) and total serum proteins were used as indicators of protein metabolism. Because of its role in protein catabolism, high levels of SGOT may indicate tissue damage in organs such as the liver. However, no distinctive patterns or differences in SGOT levels among sites were observed for any season (Fig. 5-5 and Table 5-1). In contrast to SGOT, total serum protein levels varied among sampling sites, especially in the spring (Fig. 5-6). Statistically significant differences occurred between BF and (1) EFK 23 in the spring, (2) EFK 5 the fall and spring, and (3) WBR in the winter and spring (Table 5-1). Total serum protein levels may be indicative of changes in overall protein metabolism between populations experiencing different levels of environmental stress. Preliminary studies on reproductive success indicate that levels of vitellogenin, a high density lipoprotein, were also high at EFK 5 (S. M. Adams,

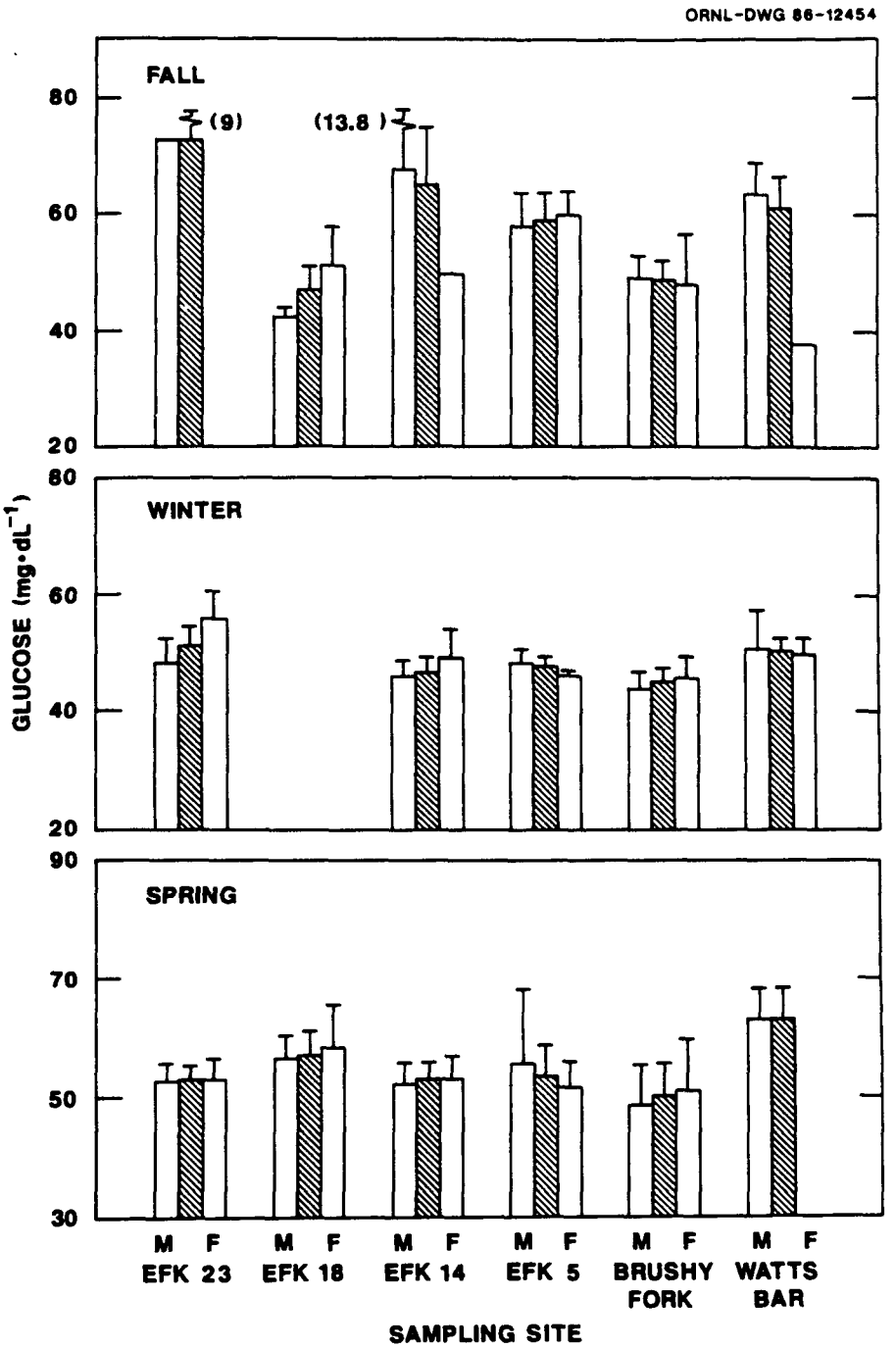

Fig. 5-2. Levels of glucose in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

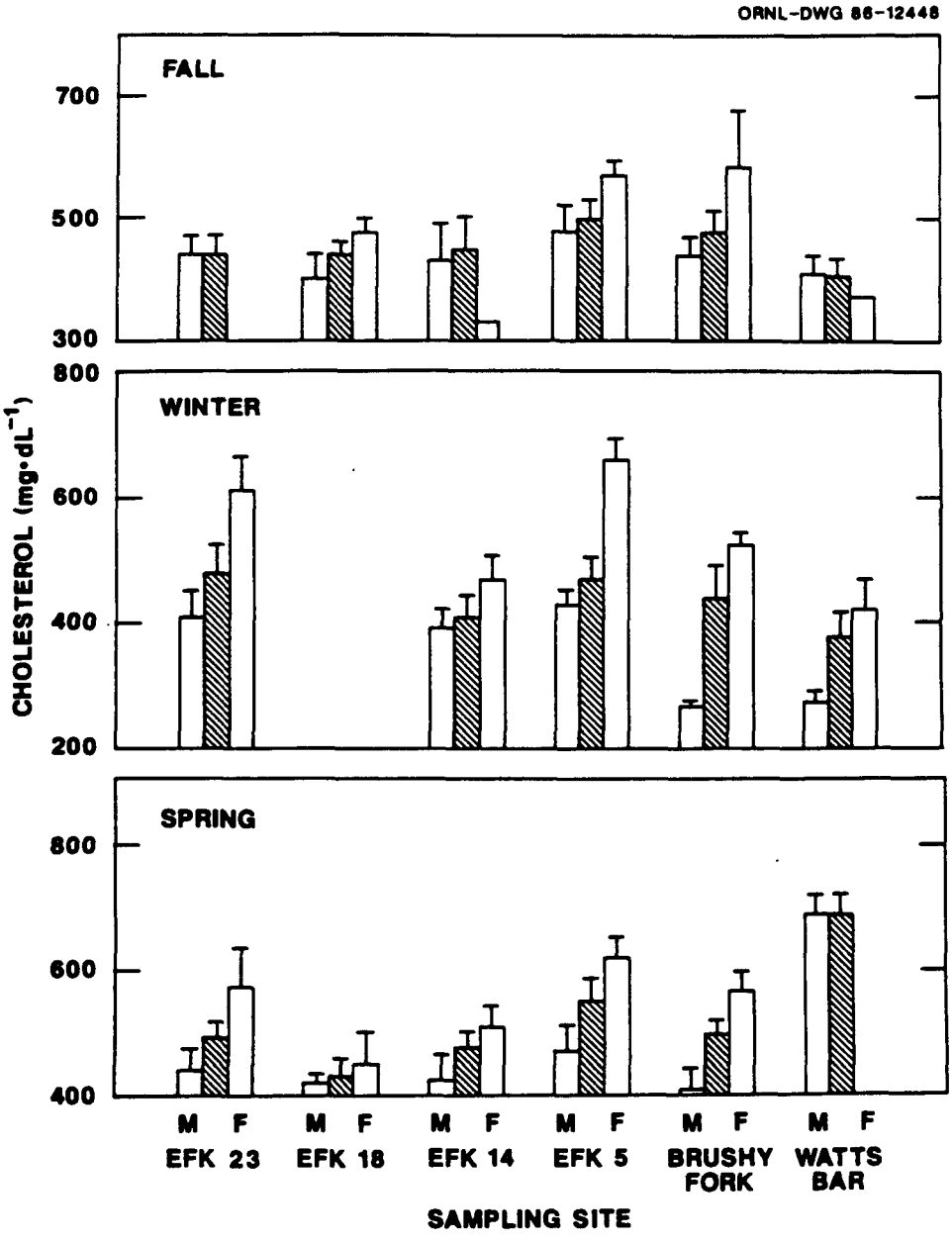

Fig. 5-3. Levels of cholesterol in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

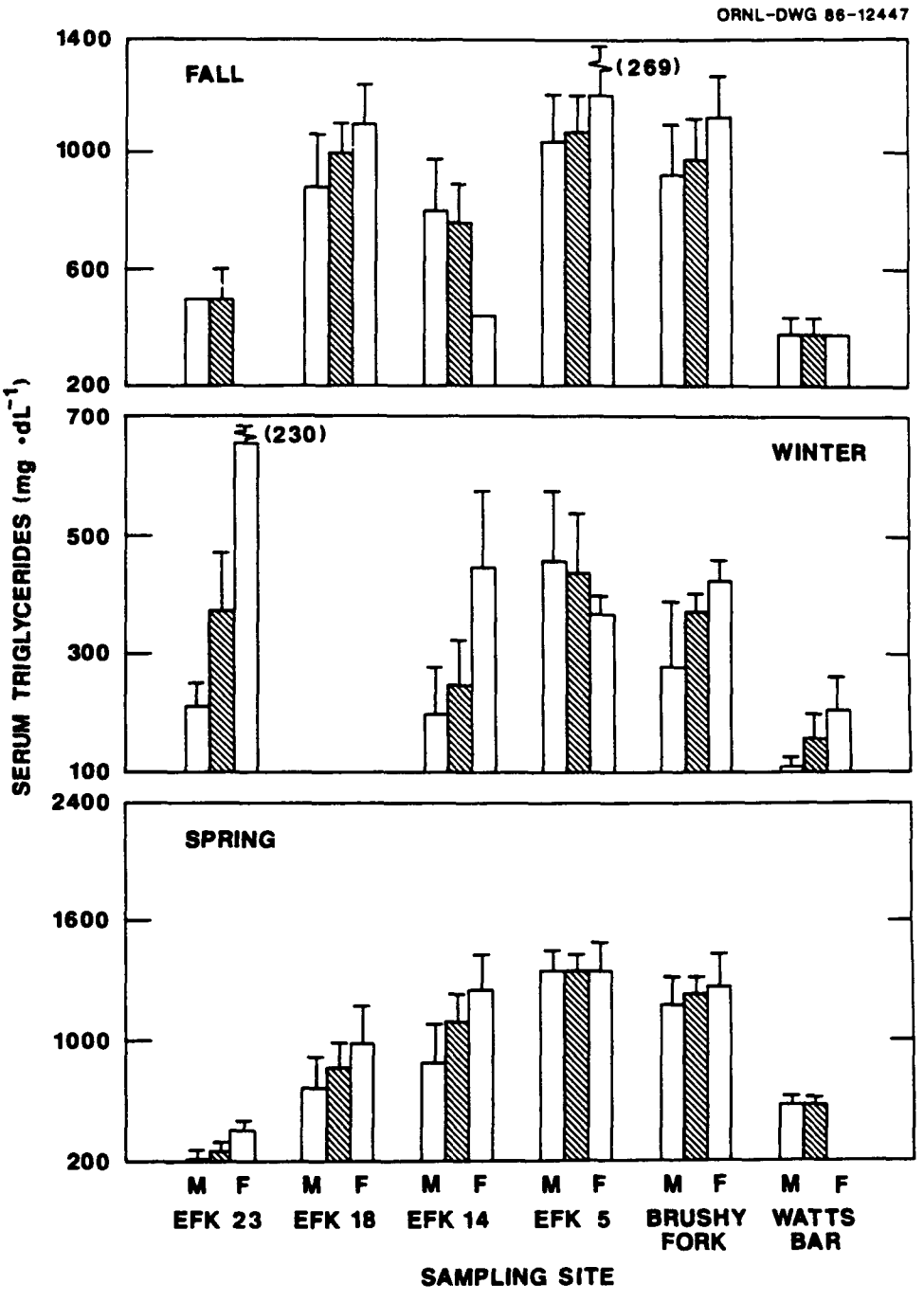

Fig. 5-4. Levels of serum triglycerides in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The means and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

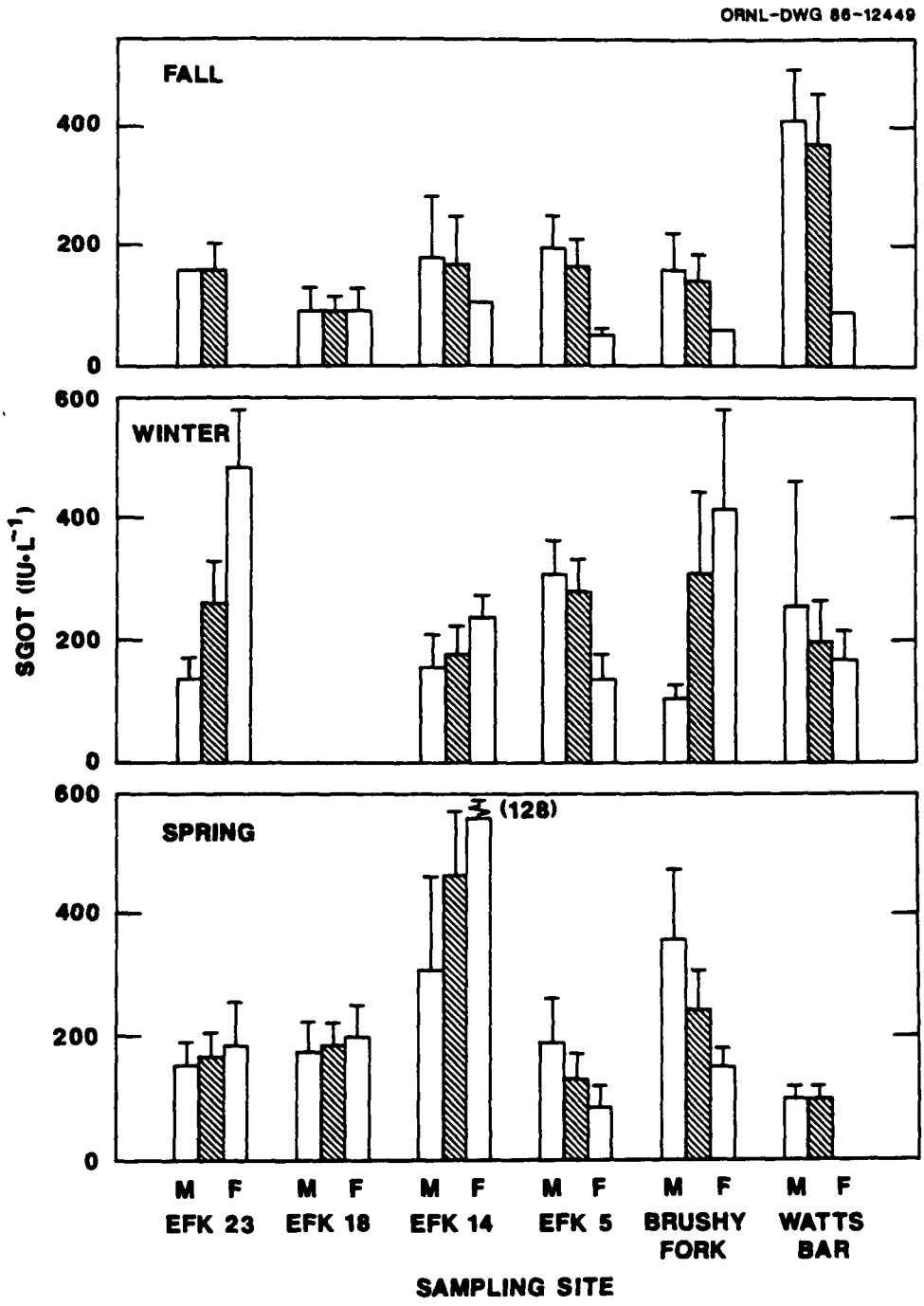

Fig. 5-5. Levels of SGOT, a transaminase enzyme, in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

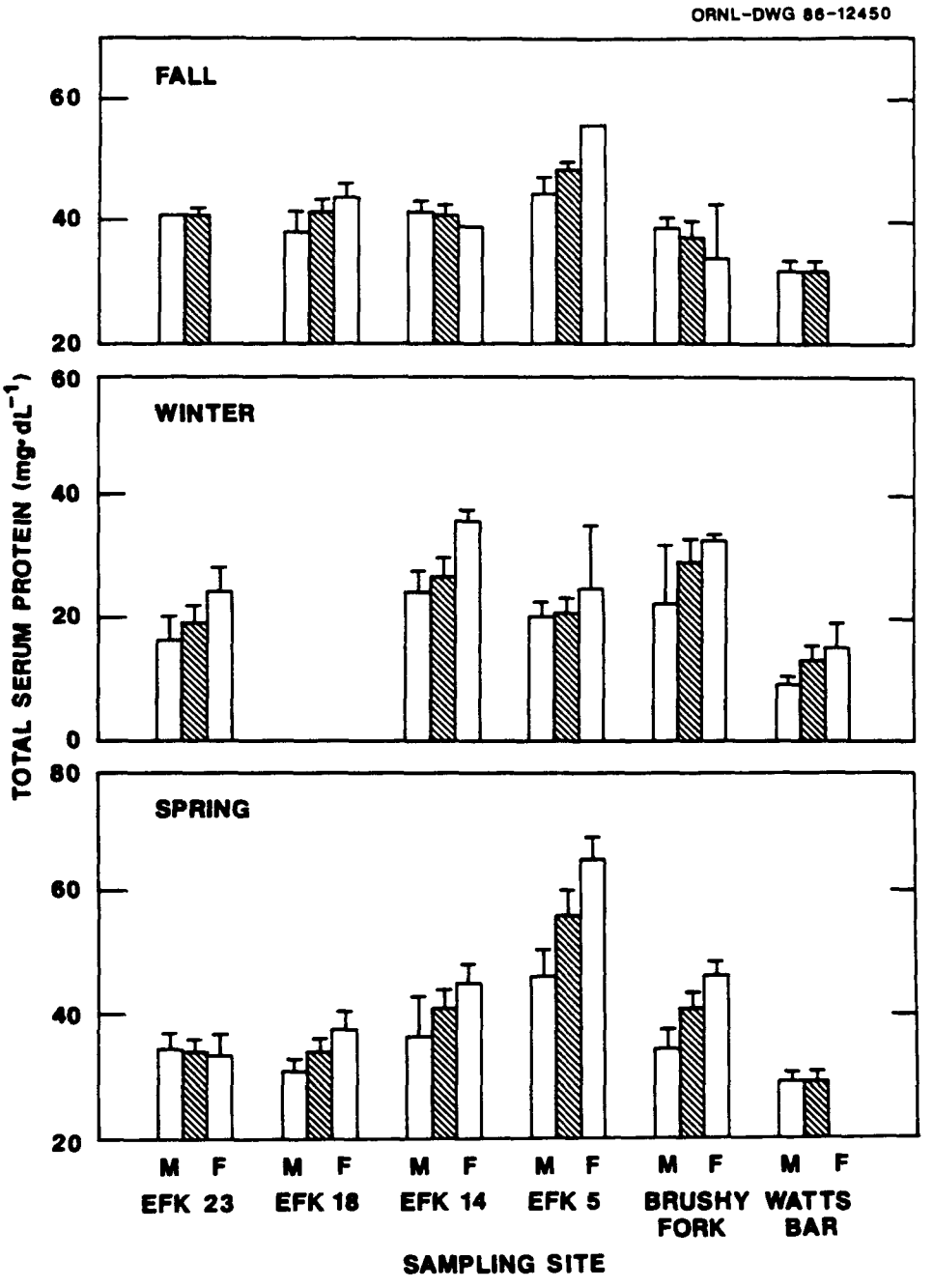

Fig. 5-6. Levels of total serum protein in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

unpublished data). Serum protein and vitellogenin levels are, therefore, closely linked and may serve as surrogates for each other.

5.3.1.2 Lipid biochemistry

Total body lipids

Total lipid content reflects the combined effect of many processes within an organism and, therefore, can serve as an integrative indicator of both nutrition (feeding status) and metabolic stress responses. Even though serum triglycerides also reflect the nutritional status and levels of metabolic stress in fish, total lipid content has the advantage over triglycerides of being a long-term integrator of many stresses acting on an organism. In the fall, levels of total lipids in BF fish were significantly lower than the levels found in fish from EFK 18 and EFK 5 (Table 5-1). In the winter, lipid levels of WBR fish were significantly lower than those of fish from all other sites. During the three months between the winter and spring collections, total lipid levels of fish from EFK 23 declined by almost 50%, whereas levels in fish from all other sites approximately doubled during this period (Fig. 5-7). Lipid levels in fish from EFK 23 and WBR were significantly lower than all other sites in the spring (Table 5-1). The large difference between males and females in the spring at EFK 18 compared with other sites indicates that this area may be a preferred spawning site for redbreast sunfish in EFPC. The condition of females of many fish species increases several weeks prior to spawning (Delahunty and deVlaming 1980; Adams et al. 1982), and this was reflected by the high lipid levels (Fig. 5-7) and high condition indices (Sect. 5.3.1.3) of females from EFK 18 in the spring.

Body triglycerides and phospholipids

These two components of total body lipids represent, respectively, neutral lipids, which are directly available for bioenergetic use, and polar lipids, which are not physiologically available for energy use. Total body triglyceride levels were significantly higher in BF fish than in both EFK 23 and WBR fish in the fall and winter, respectively (Fig. 5-8, Table 5-1). Phospholipid levels in fish from the various sites exhibited a pattern opposite that for triglycerides (Fig. 5-9, Table 5-1), and the two parameters were inversely related.

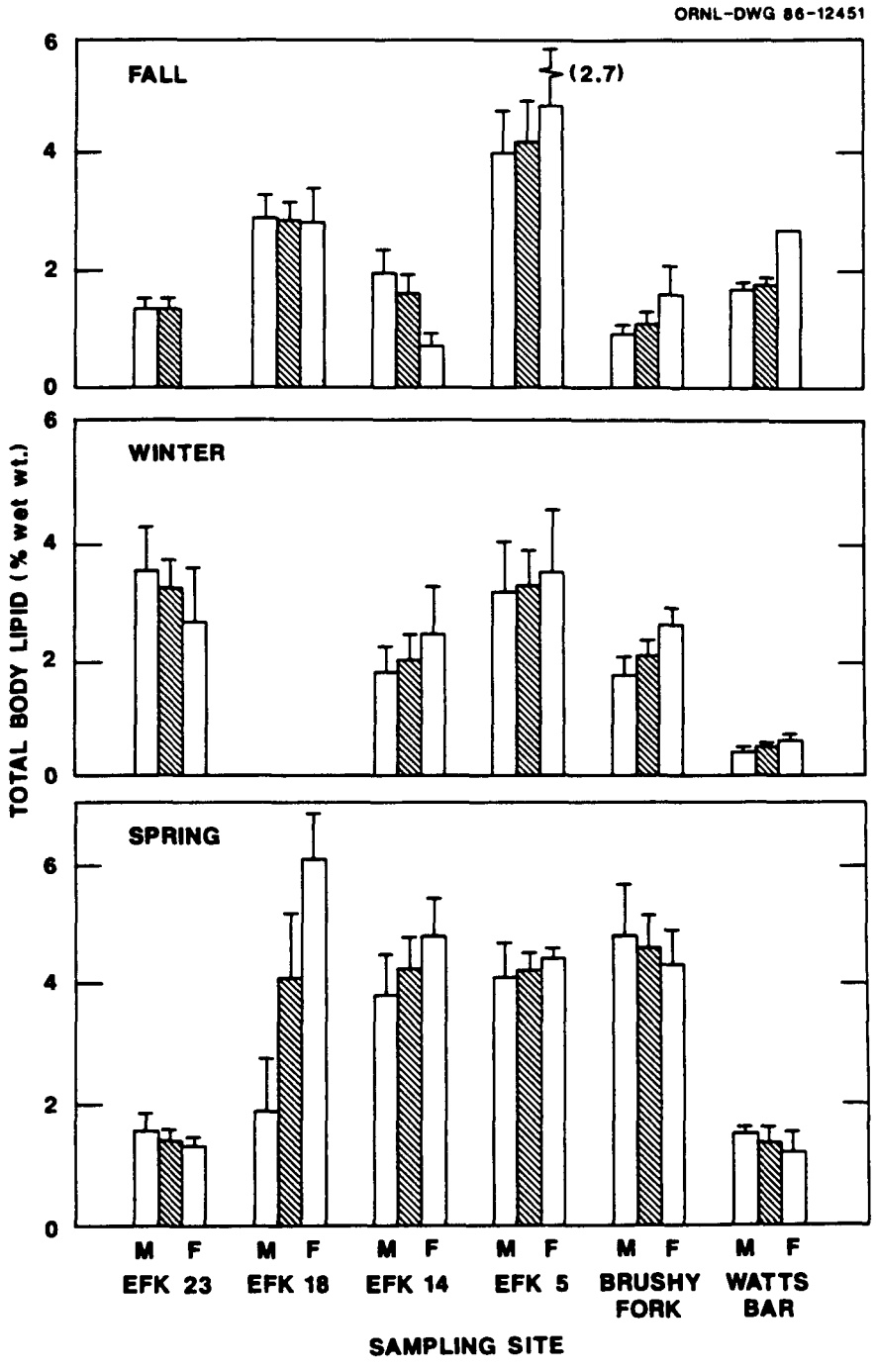

Fig. 5-7. Levels of total-body lipids in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

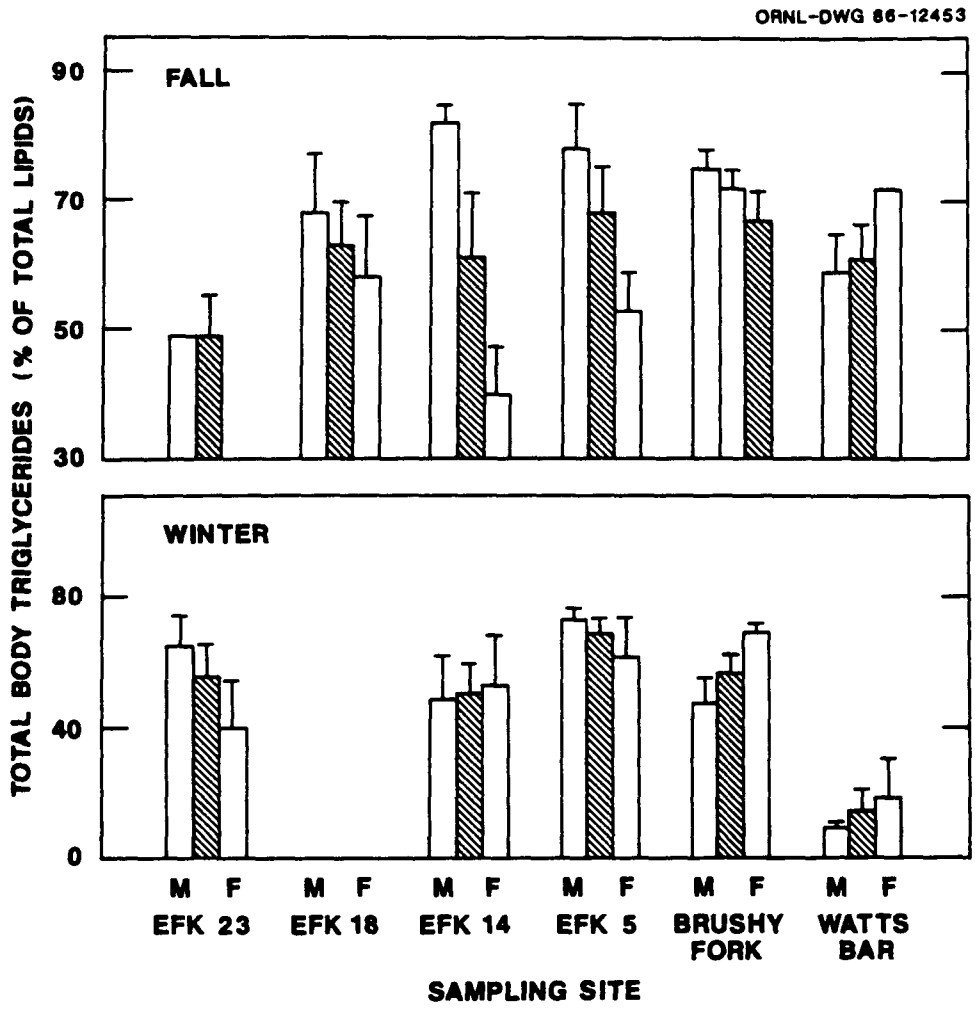

Fig. 5-8. Levels of total-body triglycerides in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985 and winter 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

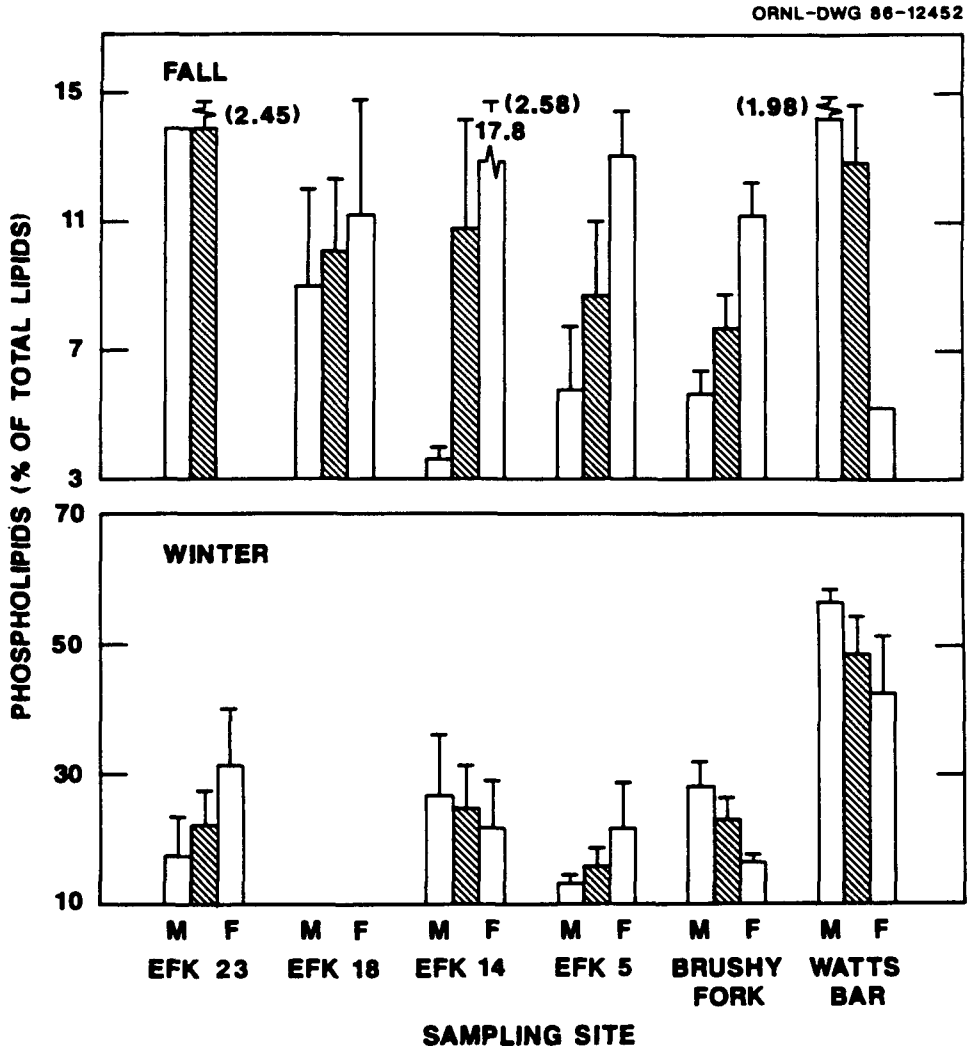

Fig. 5-9. Levels of phospholipids in redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985 and winter 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

Total-body triglycerides are good indicators of the energy available to an organism to partially mediate the effects of stress and for critical physiological functions, such as gonadal development. Body triglycerides also act as energy buffers in periods of food shortages. Preliminary results of phospholipid analyses indicate that the ratio of two major types of phospholipids (phosphaticylethanolamine and phosphatidylcholine) may reflect direct effects of contaminants on cell membrane integrity.

5.3.1.3 Condition indices

Condition factor

The condition factor (total-body length/total-body weight^3) is a generalized indicator of overall body fitness or "plumpness" of a fish. It reflects the effects of both feeding and metabolic stress caused by contaminant exposure. Fish at EFK 18 generally appeared to be in better condition than fish from other sites, including BF (Fig. 5-10, Table 5-1). As demonstrated by many of the other indicators, redbreast sunfish from WBR were in extremely poor condition (Fig. 5-10), probably because this species is ecologically adapted to streams but living in a lacustrine environment.

Visceral-somatic index (VSI)

This index, which is calculated as (total visceral weight – stomach contents)/(total-body weight), is used as a general indicator of fat levels or storage in the fish. Because most centrarchids store their lipid reserves in the mesenteries of the viscera, the VSI can be a cost-effective indicator for monitoring energy reserves. In the winter, the VSI of fish from BF was significantly lower than that of fish at EFK 23 and significantly higher than the VSI of fish from WBR (Fig. 5-11, Table 5-1). No significant differences in the VSI were observed between sites in the spring.

Liver-somatic index (LSI)

Of all the indicators measured in this study, the LSI was one of the most useful and informative. Not only did the LSI vary with sex and season (Fig. 5-12), but large differences in LSI were observed between sites (Table 5-1). The LSI reflects both short-term nutritional status and metabolic energy demands (Heidinger and Crawford 1977, Adams and McLean 1985). In addition, the LSI is sensitive to toxicant stress and

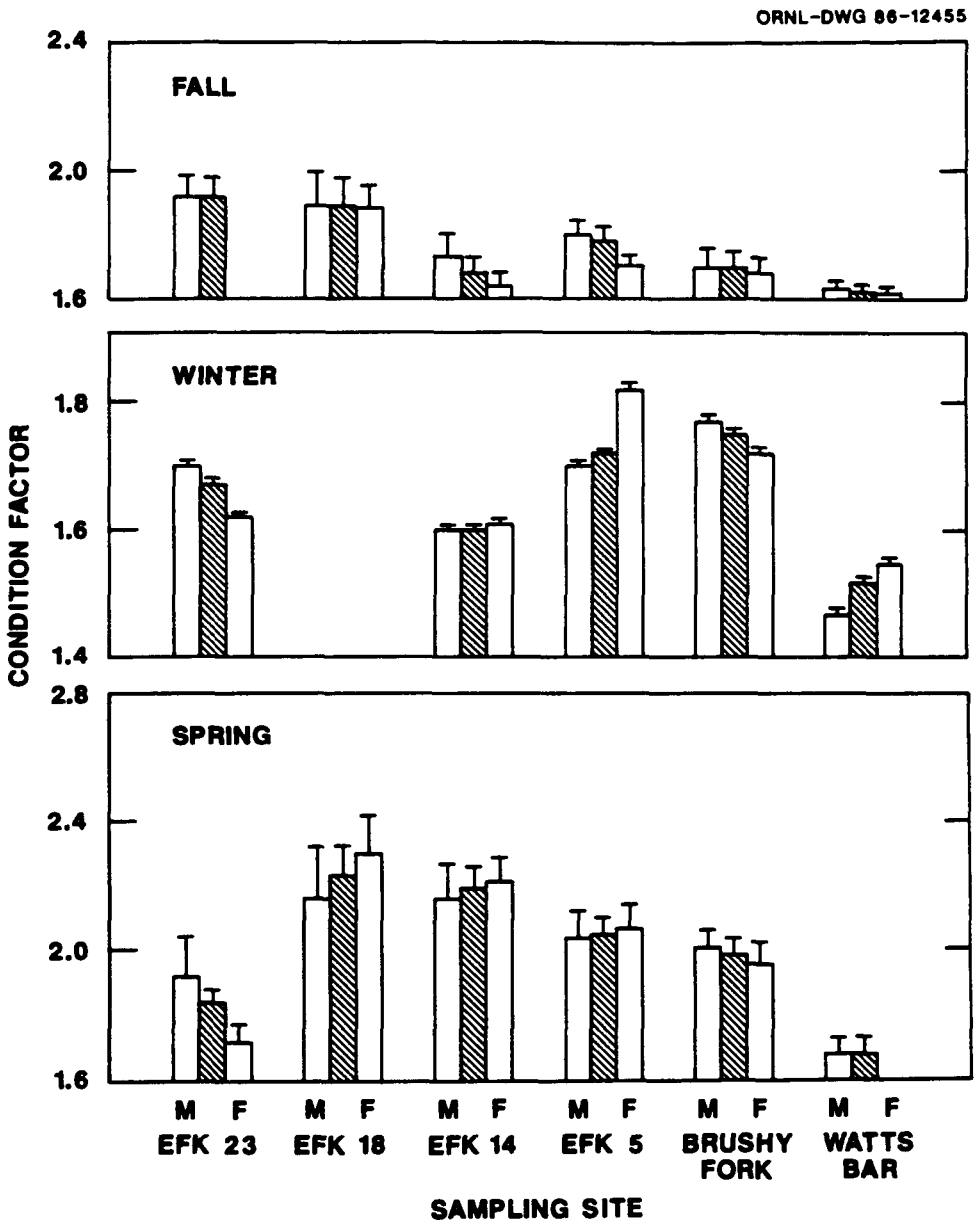

Fig. 5-10. Condition factors for redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

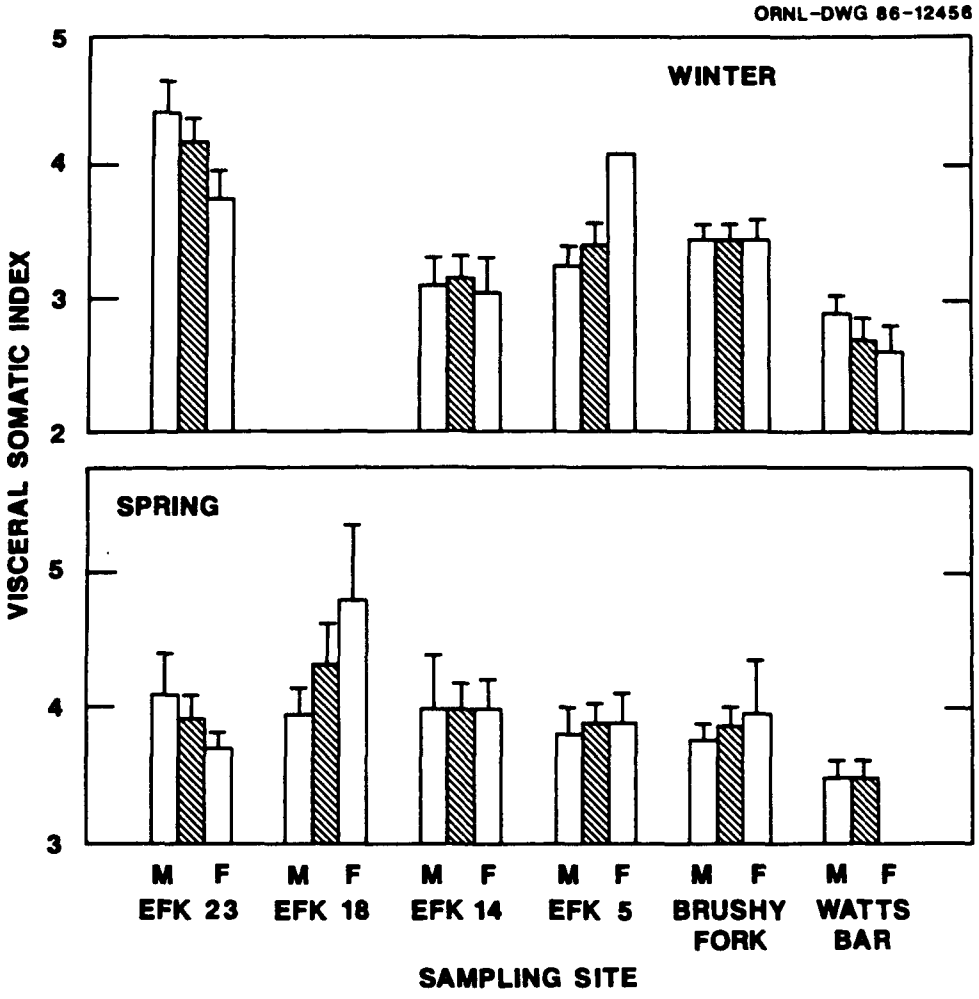

Fig. 5-11. Visceral-somatic index values for redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

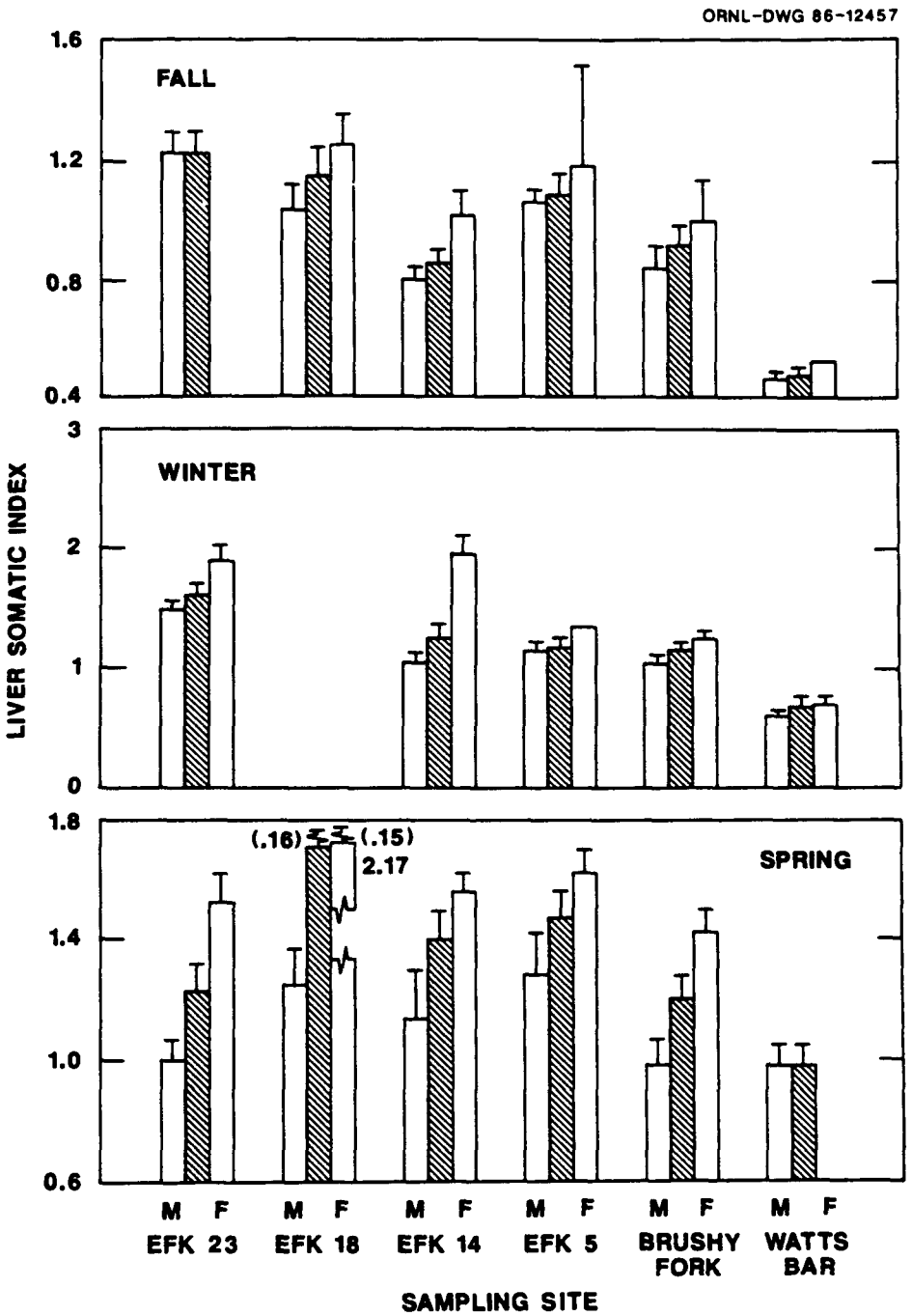

Fig. 5-12. Liver-somatic index values for redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985–spring 1986. The mean and standard error (vertical line) are shown for males (M), females (F), and the two sexes combined (hatched bars).

liver enlargement (hyperplasia) has been reported in fish exposed to various pollutants (Poels et al. 1980, Slooff et al. 1983, Chambers 1979). Liver enlargement may well have occurred in fish at EFK 23 and EFK 18 as indicated by the significantly higher LSI values at these sites compared with the reference site in BF (Fig. 5-12, Table 5-1). The significantly lower LSI of fish from WBR compared with BF was probably indicative of the poor nutritional status of the WBR fish.

5.3.1.4 Liver detoxification enzymes

Liver detoxification enzymes, such as cytochrome P-450 dependent mono-oxygenases, play a major role in the biotransformation or metabolism of xenobiotics (e.g., pesticides, hydrocarbons, and chlorinated hydrocarbons) and numerous biological molecules, including steroid hormones. Induction of hepatic cytochrome P-450 mono-oxygenases in fish by xenobiotics is a well-known phenomenon (Payne and Penrose 1975; Lidman et al. 1976; Elcombe and Lech 1978; 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 oxidase (MFO) system. Other enzymes associated with this detoxification system are the electron transfer enzymes. These enzymes are responsible for the transfer of electrons to cytochrome P-450 and therefore could play an important role in the oxidation of xenobiotics. The electron transfer enzymes measured in this study were cytochrome b_5, NADH cytochrome b_5 reductase, and NADPH cytochrome P-450 reductase.

EROD

Significant differences in 7-ethoxyresorufin o-deethylase (EROD) activities were observed between BF and all EFPC sites except EFK 23 (Fig. 5-13, Table 5-2). Although mean EROD activity at this site was higher than BF in the fall, the difference was not statistically significant because of the high variability among individual fish. In winter, however, EROD activity in fish from EFK 23 was significantly higher than BF. Differences in EROD activity between seasons is probably related to seasonal variations in water temperature. Higher acclimation temperatures are associated with higher EROD activity in bluegill sunfish (Jimenez et al. 1988). Mean monthly temperatures in the fall and winter were generally 4 to 7°C higher in EFPC below NHP compared with BF.

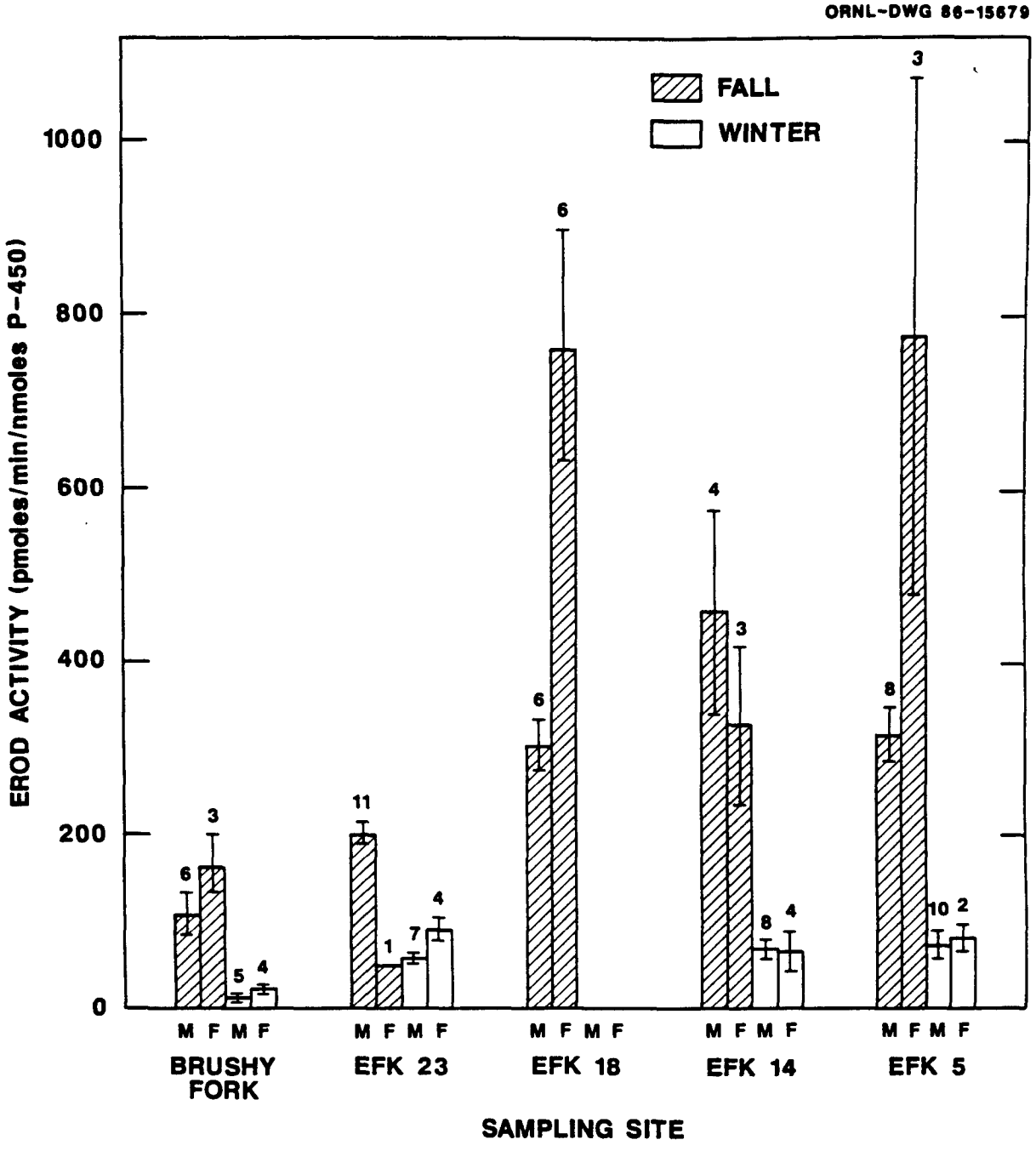

Fig. 5-13. EROD activities in male (M) and female (F) redbreast sunfish from four sampling sites in East Fork Poplar Creek and Brushy Fork, the reference stream, fall 1985 and winter 1986. The mean, standard error (vertical line), and sample size are shown.

Table 5-2. Summary of analysis of variance for liver detoxification enzymes in redbreast sunfish across seasons, sex (M = males), and sites

	EROD[a]	P-450	b_5	NADH	NADPH
		Main effects			
Site[b]					
BF-M vs EFK 23-M		++	++	++	++
BF vs EFK 18	++	++			
BF vs EFK 14	++			+	+
BF vs EFK 5	++				++
Sex	+				
Season	++	+		++	
		Interactions			
Site vs season					
Site vs sex					
Season vs sex		+			
Site vs sex vs season					

[a] + = $0.01 < P < 0.05$; ++ = $P < 0.01$.
[b] BF = Brushy Fork; EFK = East Fork kilometer.

Significant differences in EROD activity also occurred between sexes (Table 5-2). These differences were especially marked in the fall when females generally exhibited higher EROD activity than males. Although sex differences in cytochrome P-450 have been reported in fish (Williams et al. 1986), few studies have examined the differences between sexes in EROD activity. Research on enzymes important in fatty acid catabolism in fish, however, has demonstrated higher levels of activity in females during sexual maturation (Henderson et al. 1984).

Cytochrome P-450

Cytochrome P-450 exhibited seasonal responses between sites that were similar to EROD activity (Fig. 5-14). However, no significant differences in activity were observed between fish from BF and fish from EFK 14 and EFK 5 (Table 5-2). The effect of season was not the same at all sites, as indicated by the significant site vs season interaction.

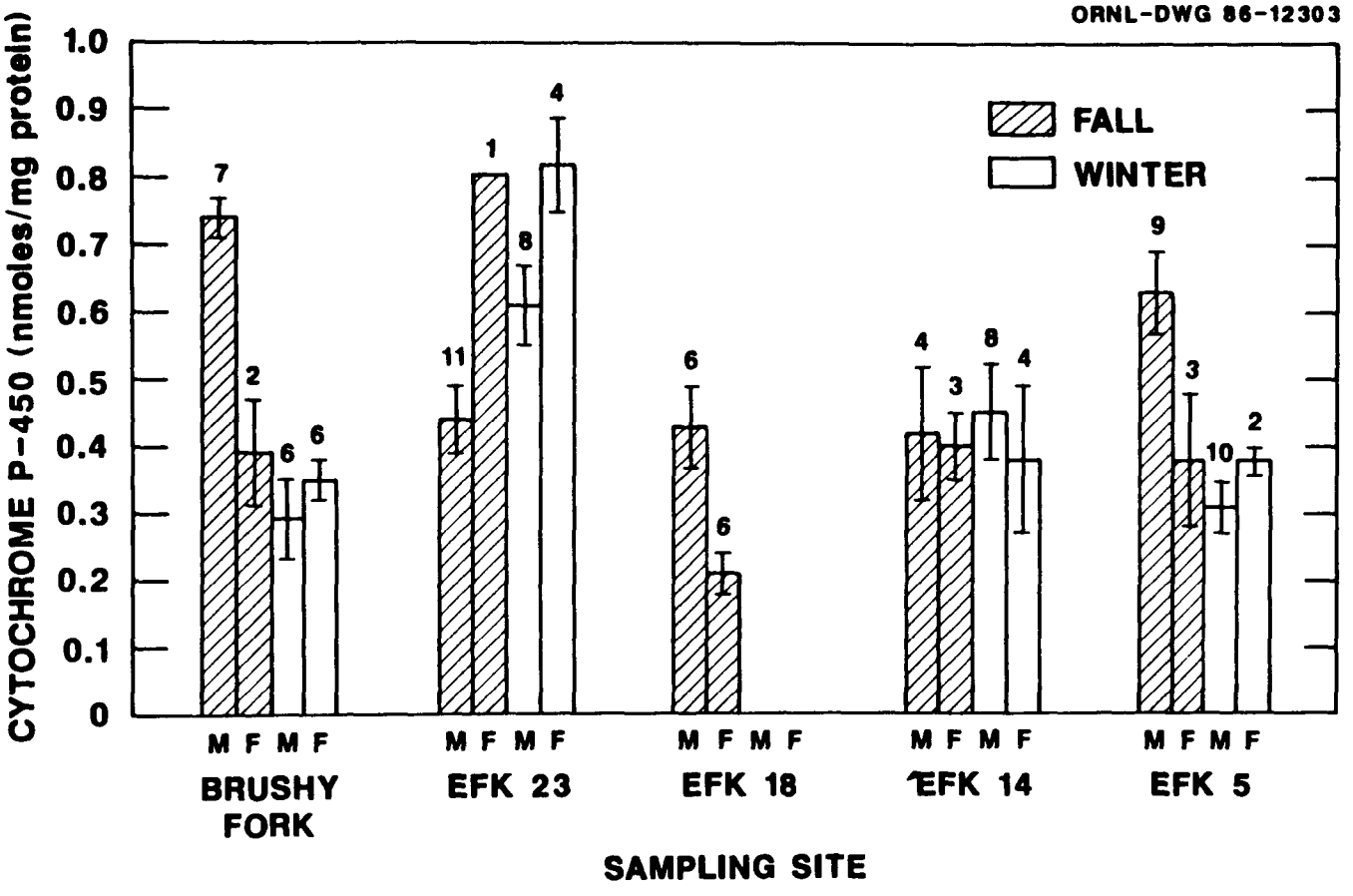

Fig. 5-14. Cytochrome P-450 activities in male (M) and female (F) redbreast sunfish from four sampling sites in East Fork Poplar Creek and Brushy Fork, the reference stream, fall 1985 and winter 1986. The mean, standard error (vertical line), and sample size are shown. EFK = East Fork kilometer.

In the fall, P-450 levels in males from BF were substantially higher than the levels found in males from EFK 23, EFK 18, and EFK 14 (Fig. 5-14). The higher levels of EROD activity in fish from EFK 23 compared with that in fish from BF in the winter were associated with an increase in cytochrome P-450 content of fish at EFK 23 from fall to winter. At BF and the other EFPC sites, P-450 levels in males generally decreased from fall to winter.

Cytochrome b_5

The highest levels of the cytochrome b_5 enzyme were found in fish from EFK 23 (Fig. 5-15). No other EFPC sites were significantly different from the reference site (BF), and there were no significant
two-way interaction effects of sex or season with site (Table 5-2).

NADH and NADPH reductase

Levels of both of these electron transport enzymes were significantly higher in fish from EFK 23 and EFK 14 compared with that in fish from BF (Figs. 5-16 and 5-17; Table 5-2). Levels of NADPH were also significantly higher in fish from EFK 5 compared with that in fish from BF. Levels of NADH were significantly different between fall and winter, but season vs site or sex vs site interactions were not significant.

Summary

The activities of five enzymes of the liver detoxification system were measured in redbreast sunfish from four sites in EFPC and BF, the reference stream. Except for EROD, enzyme levels were significantly higher at EFK 23 below NHP compared with that in BF and generally showed a gradient of decreasing response with increasing distance downstream. Such a trend may suggest a response to elevated levels of organic contamination in the upper reaches of the stream. Of the enzymes measured, cytochrome P-450 and NADPH were the most sensitive to environmental conditions in EFPC and, therefore, are the most relevant enzymes (of the MFO system) to monitor in future studies. Care must be taken when applying and interpreting these parameters as indicators of environmental stress because of the confounding effects of season (i.e., temperature and feeding level) and sex.

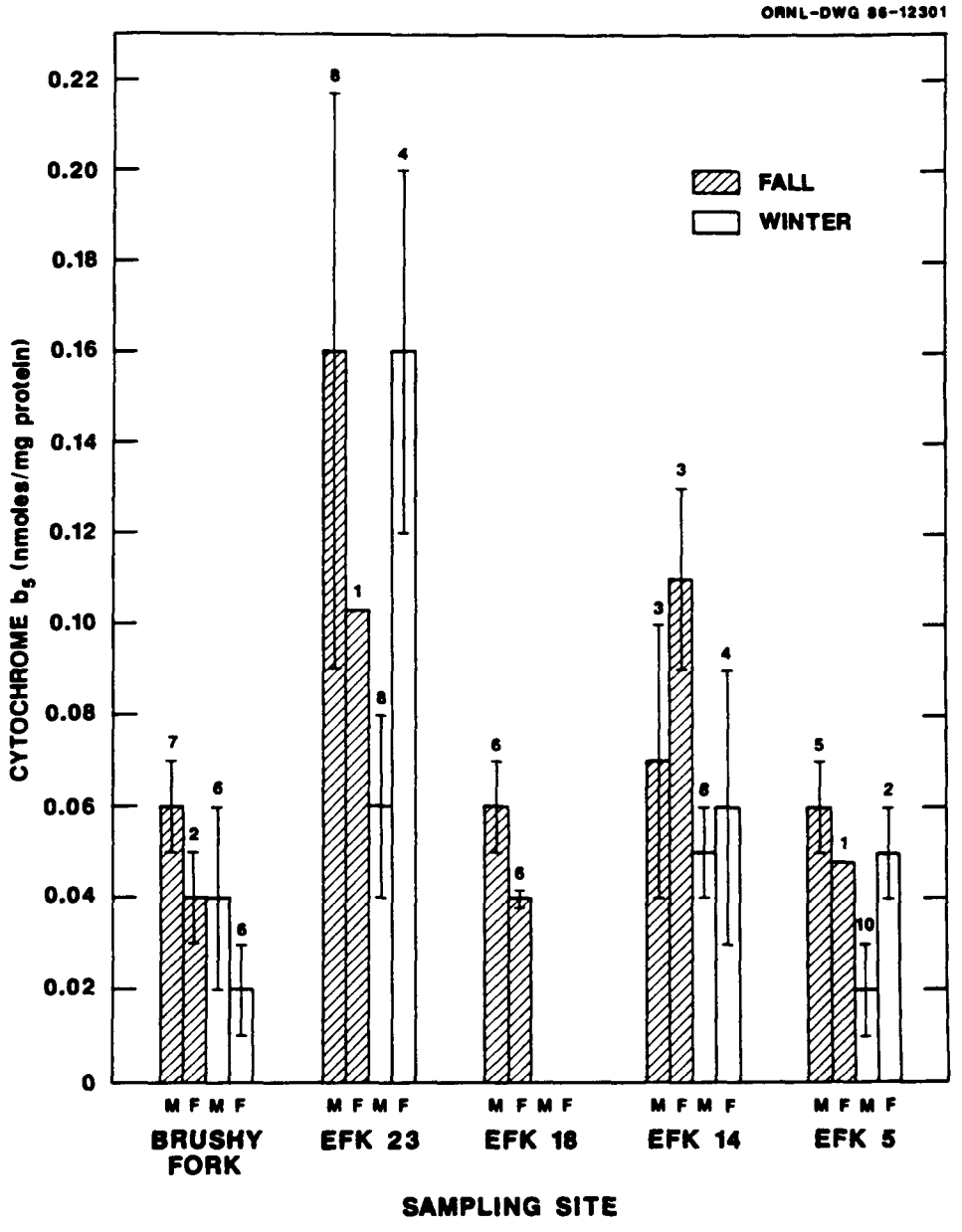

Fig. 5-15. Cytochrome b_5 activities in male (M) and female (F) redbreast sunfish from four sampling sites in East Fork Poplar Creek and Brushy Fork, the reference stream, fall 1985 and spring 1986. The mean, standard error (vertical line), and sample size are shown. EFK = East Fork kilometer.

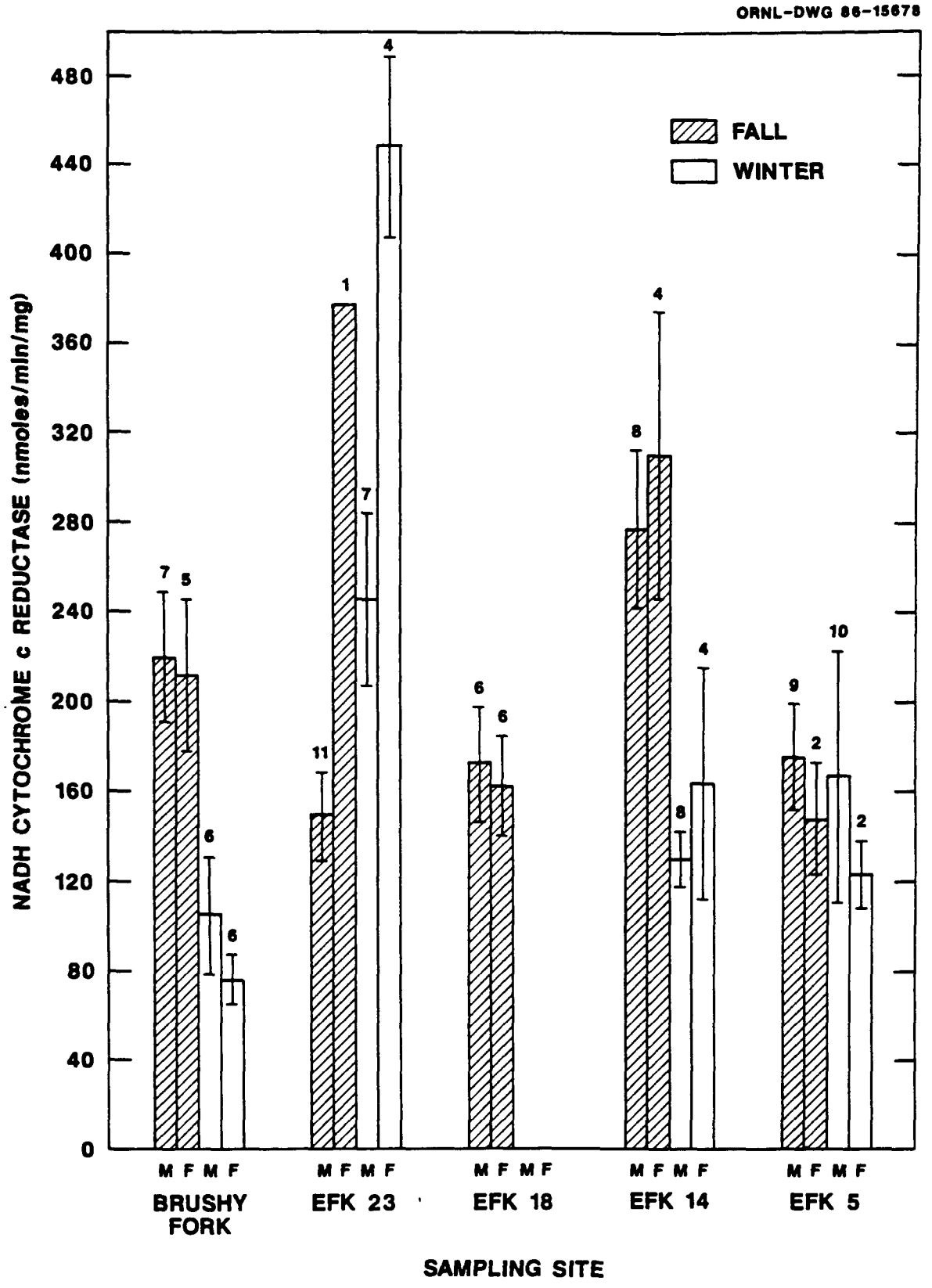

Fig. 5-16. NADH activities in male (M) and female (F) redbreast sunfish from four sampling sites in East Fork Poplar Creek and Brushy Fork, the reference stream, fall 1985 and winter 1986. The mean, standard error (vertical line), and sample size are shown. EFK = East Fork kilometer.

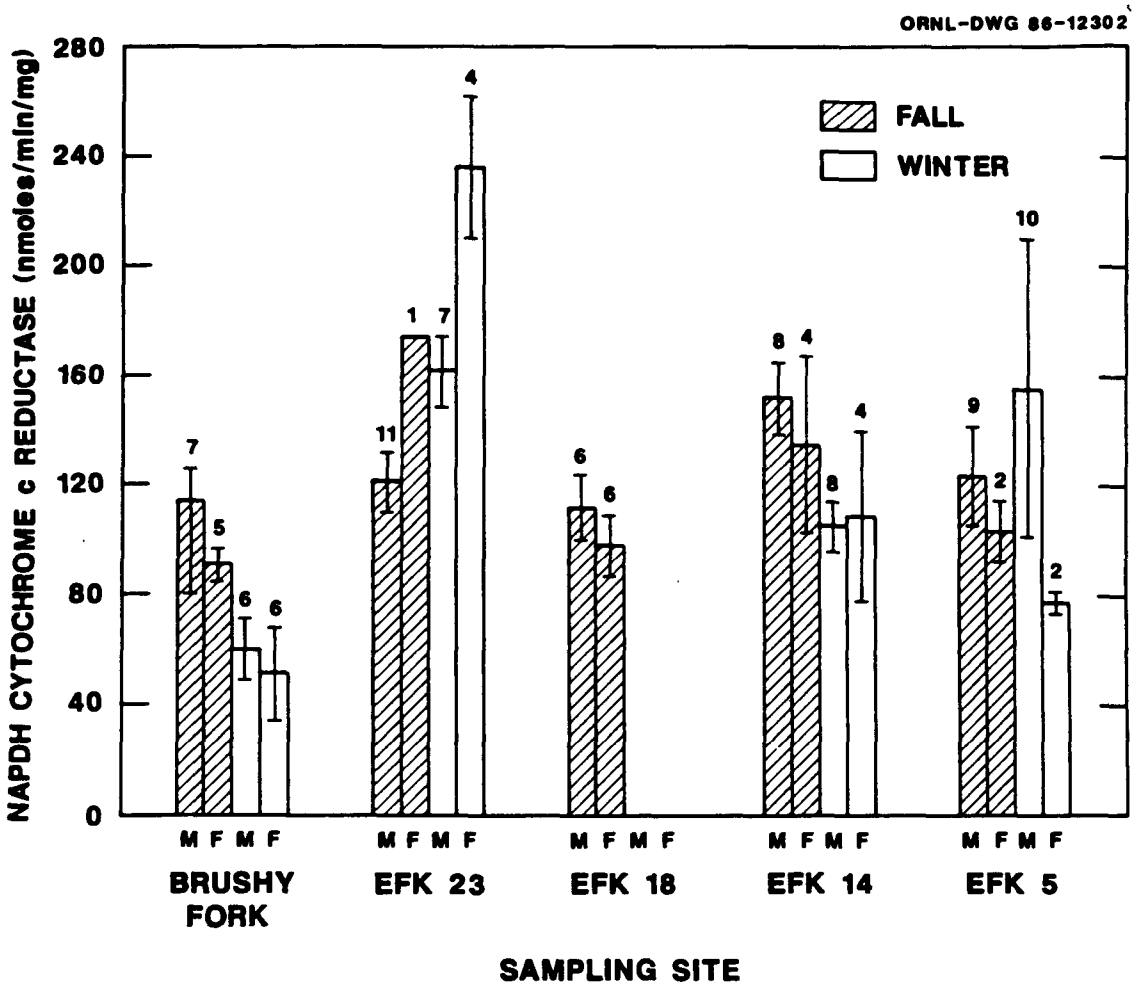

Fig. 5-17. NADPH activities in male (M) and female (F) redbreast sunfish from four sampling sites in East Fork Poplar Creek and Brushy Fork, the reference stream, fall 1985 and winter 1986. The mean, standard error (vertical line), and sample size are shown. EFK = East Fork kilometer.

5.3.1.5 Histopathological Indicators

Qualitative histopathological examinations were conducted to (1) evaluate the usefulness of histopathology as an indicator of environmental stress on EFPC fish populations and (2) screen several major organs for tissue damage and identify the primary target organs affected by chronic stress. Quantitative examination of appropriate target organs is planned for future studies.

For both the liver and kidney of fish from all sites, a range of effects was observed, ranging from mild parasitic infections to hepatic tumors (Table 5-3). In the fall, the livers and kidneys of fish from EFK 23 and EFK 18 showed a higher incidence and more-severe pathological changes than were found in fish from other sites. In the winter, fish from EFK 14, WBR, and EFK 23 had the highest degree of pathology, whereas in the spring, fish from EFK 23, EFK 18, and EFK 14 were the most affected (Table 5-3). Most of the effects observed in the gonads and gills were related to parasitism (e.g., parasitic lesions).

When pathological effects from all seasons were combined, fish from EFK 23 appeared to be the most affected and fish from the WBR reference site were in worse overall condition than fish from lower EFPC (EFK 5) or BF (Table 5-4). A gradient of pathological effects extending from the outfall of NHP (EFK 23) to at least EFK 14 was also observed. The similarity in the results from fish collected at EFK 5 and BF suggests that water quality in EFPC may have improved by this point downstream. Fish from WBR had high levels of parasitism and disease and, as shown by several other bioindicators (Sects. 5.3.1.1 through 5.3.1.3), were in extremely poor condition, probably, in part, because of starvation (i.e., nutritional deficiencies). Redbreast sunfish in WBR demonstrated many of the classical symptoms of starvation, which weakens fish and renders them more susceptible to disease (Shulman 1974; Glebe and Leggett 1981).

5.3.2 Integrated Bioindicator Responses

To examine the integrated response of fish to their environment during each season, blood biochemical parameters, lipid biochemical parameters, and condition indices (Sects. 5.3.1.1 through 5.3.1.3, respectively) were considered together within a multivariate context using the canonical discriminant analysis procedure. Similarly, all the responses related to liver detoxification enzymes (Sect. 5.3.1.4), including the liver somatic index (Sect. 5.3.1.3), were examined in an integrated context by the same procedure. Using

Table 5-3. Summary of qualitative histopathological examinations of four organs of redbreast sunfish from East Fork Poplar Creek and two reference areas, fall 1985–spring 1986

Histopathology of organ	Fall	Winter	Spring
	Liver		
Parasitic lesions	1,[a] 2	5, 6, 4	All
Hepatic tumors	2, 3	*b*	*b*
Fatty vaculation	1	None	*b*
Foci of putative preneoplastic lesions	1, 2, 3	None	5, 4, 3
Granuloma	1, 2, 6	*b*	*b*
Melanomacrophage aggregates	1, 2	3, 6	*b*
Adenofibrosis	*b*	*b*	1, 2, 3
	Kidney		
Parasitic lesions	1, 2	1, 5, 3	1, 2, 5
Renal interstitial inflammation	1, 2, 5	*b*	*b*
Necrosis of tubular epithelium	1, 2	5, 3	1
Chronic glomerular lesions	*b*	3, 1	*b*
	Gonads		
Parasitic infestation	All sites	All sites	All sites
	Gills		
Parasitic infestation	All sites	2, 1, 6 sites	All

[a]Numbers represent those sites with the highest levels of a particular pathology and, for each season, are arranged in order of decreasing severity from right to left. 1 = EFK 23, 2 = EFK 18, 3 = EFK 14, 4 = EFK 5, 5 = Watts Bar Reservoir, 6 = Brushy Fork.

[b]Parameter not measured.

Table 5-4. Relative ranking of sites based on the severity of the pathological condition observed in the liver, kidney, gonads, and gills of redbreast sunfish collected from East Fork Poplar Creek and two reference areas, Brushy Fork (BF) and Watts Bar Reservoir (WBR)

Season	Ranking[a]					
	1	2	3	4	5	6
Fall	EFK 23	EFK 18	EFK 14	WBR	BF	EFK 5
Winter	EFK 14	WBR	EFK 23	BF	EFK 18	EFK 5
Spring	EFK 23	EFK 18	EFK 14	WBR	EFK 5	BF
All seasons combined	EFK 23	EFK 18	EFK 14	WBR	BF	EFK 5

[a]Ranking: 1 = most severe; 6 = least severe.

canonical discriminant analysis, the mean values of the first two canonical variables (the two variates that account for most of the ability to separate integrated fish responses at each site) are plotted for each site along with the statistical confidence region (90 and 95%). Sites were considered to be significantly different if the 90% (or 95%) confidence radii of the site means did not overlap.

5.3.2.1 Blood, lipid, and condition index responses

Fall 1985

Fish at EFK 23, EFK 18, EFK 5, and WBR separated on the first two canonical variables (Fig. 5-18). The response of fish from BF and EFK 14 was similar, as indicated by their overlapping confidence radii, and intermediate between that of fish from WBR and the other EFPC sites. Because the integrated response of fish from WBR and BF was strikingly different and because WBR was a distinct statistical outlier, WBR is not a representative reference site and will be excluded in future monitoring.

Winter 1986

Fish collected from sites EFK 5, EFK 14, and BF all had similar response patterns at both the 90 and 95% confidence levels (Fig. 5-19). Redbreast sunfish from EFK 23 and WBR were again distinct in their integrated responses compared with other sites.

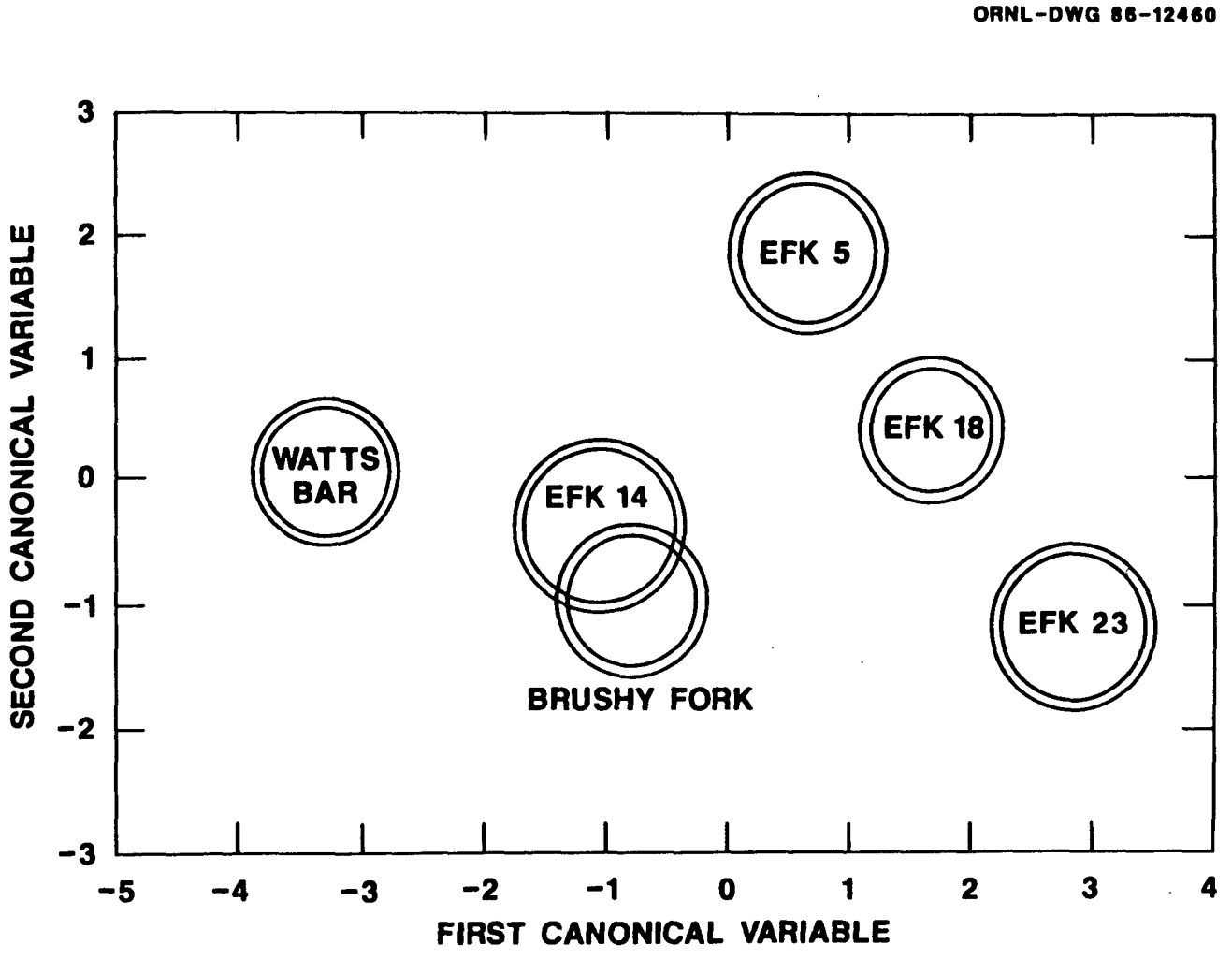

Fig. 5-18. Segregation of integrated bioindicator responses (blood chemistry, lipid biochemistry, and condition indices) for redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), fall 1985. Circles represent site means and the 90% (inner circles) and 95% (outer circles) confidence radii of the site means. EFK = East Fork kilometer.

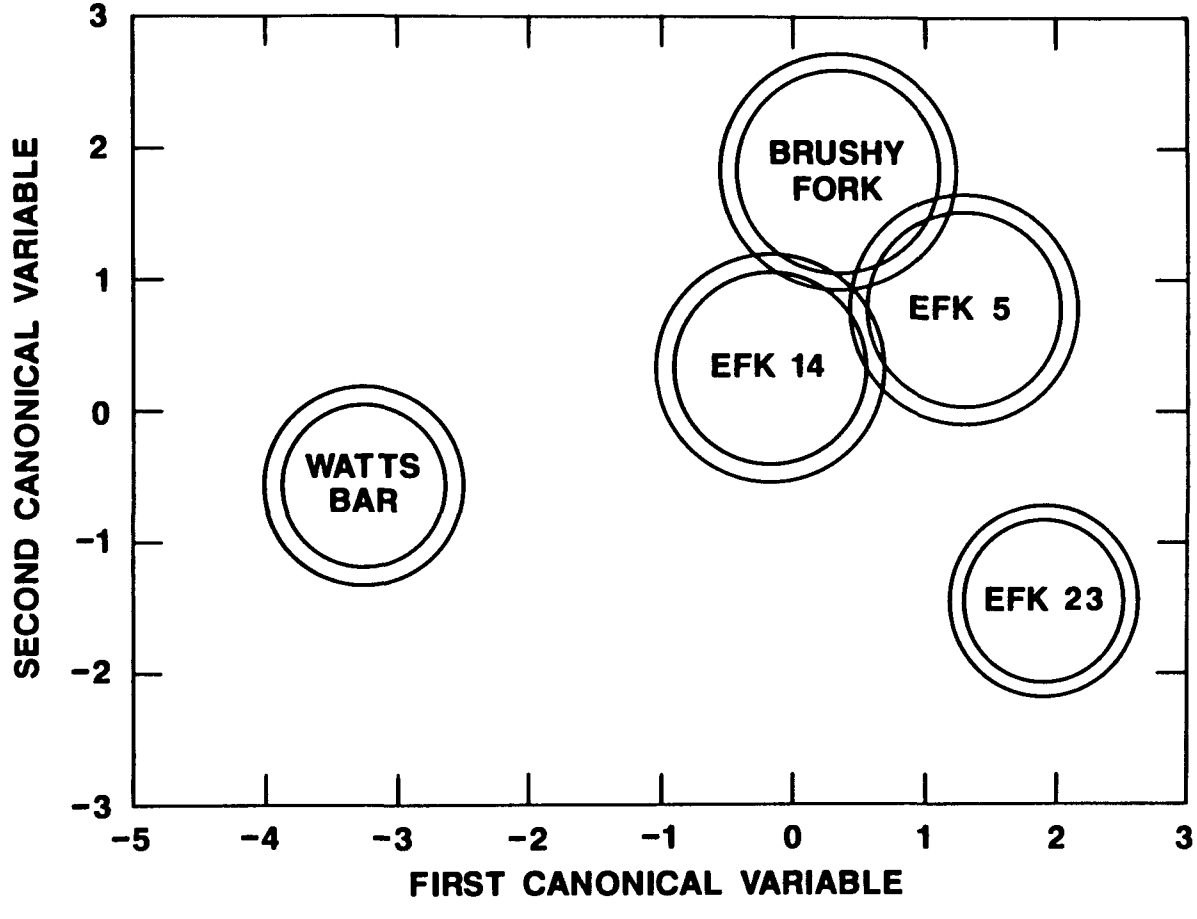

Fig. 5-19. Segregation of integrated bioindicator responses (blood chemistry, lipid biochemistry, and condition indices) for redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), winter 1986. Circles represent site means and the 90% (inner circles) and 95% (outer circles) confidence radii of the site means. EFK = East Fork kilometer.

Spring 1986

In this season, also, fish from BF and EFK 5 had similar integrated responses (Fig. 5-20). However, site EFK 14 was segregated from EFK 23, whereas WBR fish continued to demonstrate their large distance separation from the other sites. The primary difference in the overall integrative response patterns in the spring vs fall was the switch in the position of EFK 14 and EFK 5 relative to BF. All other sites remained separated from BF throughout the year.

Summary

When considering the integrative responses of redbreast sunfish over all seasons, sites EFK 14 and EFK 5 were the most similar to BF (Table 5-5) and the similarity of EFPC sites to BF increased with increasing distance downstream of New Hope Pond (EFK 23.7). This same pattern of downstream response was observed for the pathological condition of fish (see Table 5-4). Other conclusions that emerged from this analysis were (1) WBR was not an appropriate reference site, (2) fish at EFK 23 were the most dissimilar of any EFPC site compared with BF, and (3) interpretation of fish responses to environmental conditions or stress may differ, depending on whether individual or multiple responses are being evaluated. For example, if many of the biochemical parameters measured in this study, such as SGOT, glucose, and cholesterol, were used separately to

Table 5-5. Relative ranking of sites based on the similarity of integrative bioindicator responses in redbreast sunfish from East Fork Poplar Creek and Watts Bar Reservoir (WBR) compared with Brushy Fork (BF), the reference stream.

Season	Ranking[a]				
	1	2	3	4	5
Fall	EFK 23	WBR	EFK 5	EFK 18	EFK 14
Winter	WBR	EFK 23	EFK 14	EFK 5	
Spring	WBR	EFK 23	EFK 18	EFK 14	EFK 5
All seasons combined	WBR	EFK 23	EFK 18	EFK 14	EFK 5

[a]Ranking: 1 = low similarity; 5 = high similarity.

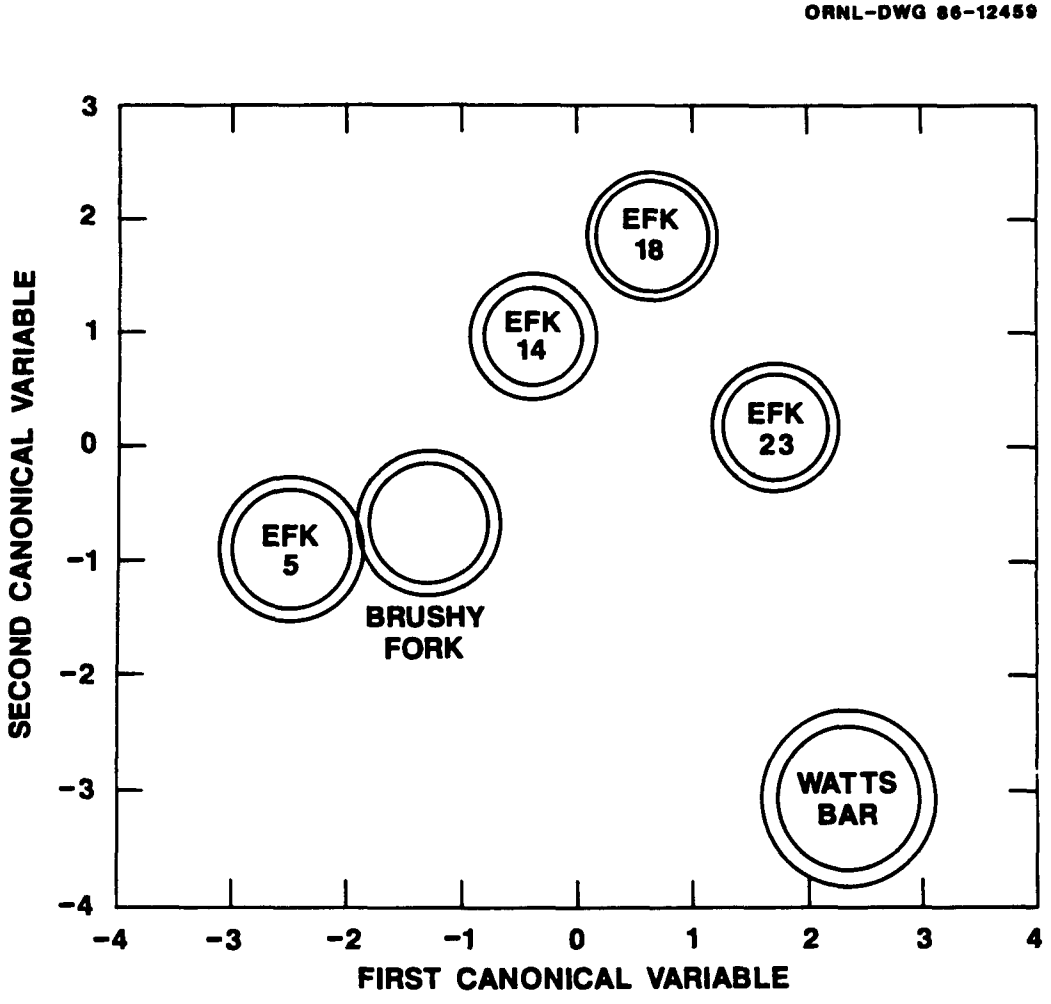

Fig. 5-20. Segregation of integrated bioindicator responses (blood chemistry, lipid biochemistry, and condition indices) for redbreast sunfish from four sampling sites in East Fork Poplar Creek and two reference areas (Brushy Fork and Watts Bar Reservoir), spring 1986. Circles represent site means and the 90% (inner circles) and 95% (outer circles) confidence radii of the site means. EFK = East Fork kilometer.

evaluate the stress responses of fish populations in EFPC, then entirely different conclusions would be reached regarding chronic effects compared with the integrated response approach. Because many biochemical and physiological variables are interrelated and may affect each other in various ways, investigations of fish response to environmental stress are more effectively evaluated within an integrated context rather than on an individual biochemical basis (Adams et al. 1985).

5.3.2.2 Liver enzyme responses

As shown in Table 5-2, significant differences in both season and sex existed for many of the liver enzyme variables, necessitating a separate discriminant analysis for season and for sex (Fig. 5-21).

Fall 1985 (males)

The integrated enzyme response of males at all EFPC sites was different from that of the reference (BF) males. However, the responses of fish at EFK 23, EFK 18, and EFK 5 were all similar. Site EFK 14 segregated from all other EFPC sites and appeared to be the most different from the reference site.

Fall 1985 (females)

No females were collected at EFK 23 during the fall. Females from EFK 14 were the most similar to BF, whereas females from EFK 18 and EFK 5 were different from BF but similar to each other.

Winter 1986 (males)

Only the integrated enzyme response of fish at EFK 23 was significantly different from BF. Sites EFK 14 and EFK 5 were similar to each other and to BF.

Winter 1986 (females)

Like the males, females at EFK 23 were segregated from the other EFPC sites and from BF. Even though EFK 5 and EFK 14 fish were similar to each other, only EFK 5 fish were similar to the reference site on BF.

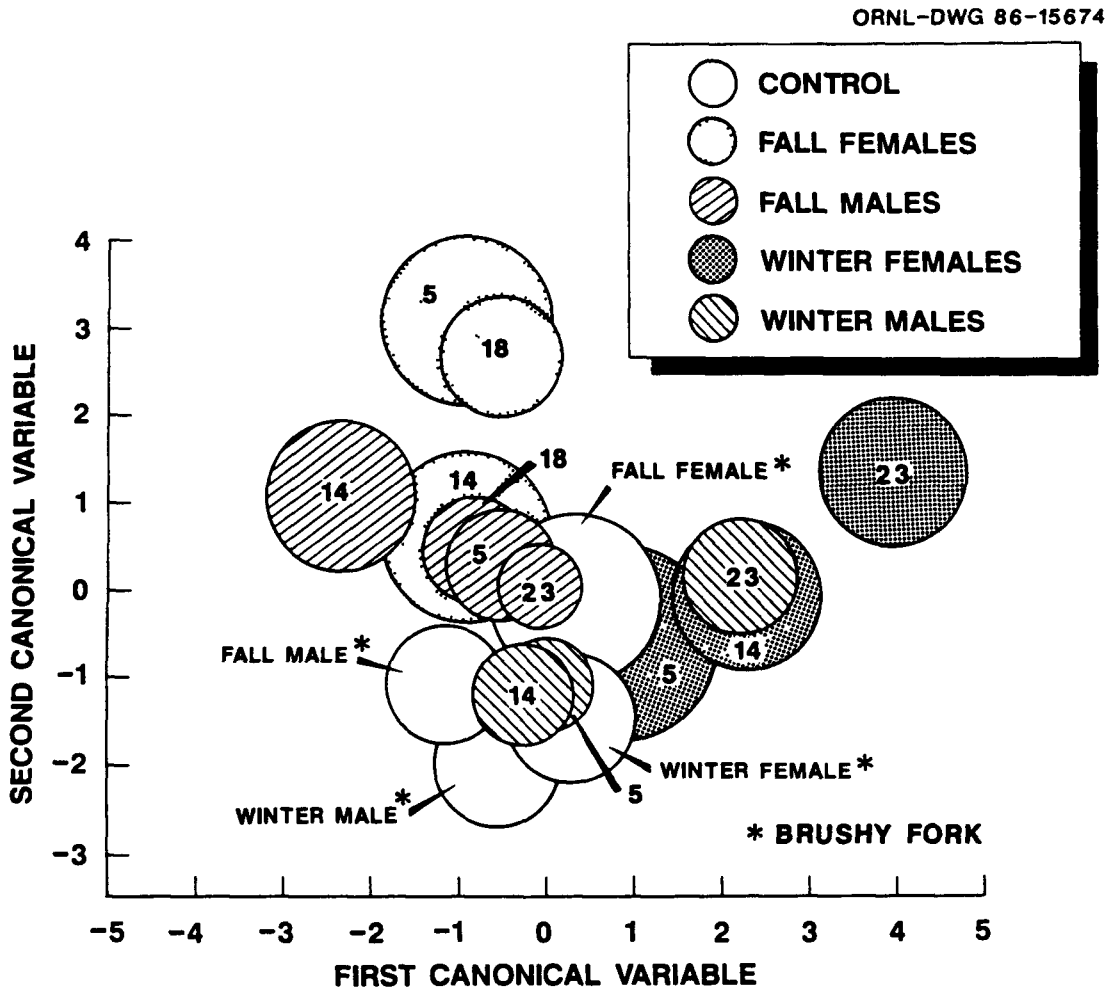

Fig. 5-21. Segregation of integrated liver enzyme responses for male and female redbreast sunfish from four sampling sites in East Fork Poplar Creek and Brushy Fork, the reference stream, fall 1985 and winter 1986. Circles represent site means and the 90% confidence radii of the site means.

Summary

Unlike the fish from EFPC, fish from BF were aggregated in close proximity to each other irrespective of season or sex. This pattern of aggregation suggests that EFPC fish may be subjected to a greater magnitude of, or variability in, environmental stress, and the responses differ for males and females.

Of all the sites sampled in the winter, EFK 23 exhibited the least similarity to BF. In the fall, females at sites EFK 18 and 5 exhibited the greatest dissimilarity from females collected in BF. Female fish were not very abundant in EFPC in either of the two seasons, and no females were collected at EFK 23 in the fall.

5.4 CONCLUSIONS

A suite of biological indicators, representing a gradient of short- to long-term responses to contaminant-related stress, was measured in redbreast sunfish from four sites in EFPC and two reference sites, Brushy Fork and Watts Bar Reservoir. The bioindicators that provided the most ecologically relevant and cost-effective information were identified from this initial suite of indicators. These include liver enzymes, serum triglycerides, total-serum protein, total-body lipids, total-body triglycerides, phospholipids, and selected histopathological analyses.

Based on this selection/screening study, the bioindicators that will be retained in future studies can be categorized into three general response groups based on causal mechanisms (1) indicators that reflect primarily nutritional status of fish or indirect effects of environmental conditions mediated through the food chain, (2) those indicative of water quality or the direct effects of contaminant stress, and (3) those that integrate (1) and (2). Bioindicators representative of these three groups are listed in Table 5-6.

In addition to providing information that was used to screen/select relevant bioindicators, studies conducted in the past year provided information on the nature of the responses by fish to conditions in EFPC. The following general conclusions are based on these initial findings.

1. Evaluation of fish responses based on an integrated bioindicator approach, histopathological analyses, and liver enzyme activities all demonstrated the same pattern of increasing similarity to BF fish with increasing distance downstream of

NHP (Table 5-7). Responses of fish from EFK 5 and EFK 23 were most similar and least similar, respectively, to those of fish from BF.

2. Determination and interpretation of fish response to environmental conditions or stress may differ, depending on whether individual or multiple responses are being evaluated. Because many bioindicators are interrelated and may affect each other in various ways, investigations of fish response to environmental stress are more effectively evaluated within an integrated context rather than on an individual bioindicator basis.
3. In evaluating the effects of chronic stress on fish populations, indicators should be used that are representative of several physiological functions, such as carbohydrate-protein metabolism, lipid metabolism, enzyme response, and condition indices.

Table 5-6. Major categories of bioindicators based on differences in the causal mechanisms that are ultimately responsible for each response

Indicators of nutrition or indirect effects via the food chain	Indicators of water quality or direct effects of contaminants	Indicators that integrate direct and indirect effects
Serum triglycerides	Liver detoxification enzymes	Growth
Total lipids	Serum sodium	
Total-body triglycerides	Serum protein	Total lipids
Liver-somatic index (if liver enzymes are low)	Phospholipid ratios Liver-somatic index (if liver enzymes are high)	Condition factor Liver-somatic index Visceral-somatic index
	Certain histopathological parameters	Certain histopathological parameters (kidneys)

Table 5-7. Relative qualitative ranking of sites based on the similarity of three groups of bioindicator responses measured in redbreast sunfish from East Fork Poplar Creek and Brushy Fork (BF), the reference stream

Measure of response	Ranking[a]			
	1	2	3	4
Integrated bioindicators	EFK 23	EFK 18	EFK 14	EFK 5
Pathological condition	EFK 23	EFK 18	EFK 14	EFK 5
Liver enzymes	EFK 23	EFK 18	EFK 14	EFK 5

[a]Ranking: 1 = low similarity; 4 = high similarity.

5.5 FUTURE STUDIES

In addition to the parameters retained from the initial suite of bioindicators (Sect. 5.4), several new parameters will be measured to improve the ability to evaluate, interpret, and predict the effects of environmental stress on fish populations in EFPC. These parameters include (1) quantitative histopathological measurements, (2) RNA/DNA ratio in fish tissue (indicators of growth), (3) ratio of phospholipid types (indicators of cell membrane integrity), and (4) measures of immune system competence.

5.5.1 New Initiatives

5.5.1.1 Food habits

Studies of food habits will be initiated to evaluate how biological effects at the biochemical and physiological level are manifested at higher levels of biological organization. Also, these studies will aid in evaluating the relative importance of indirect effects of contaminant stress (via the food chain) compared with direct (toxicant) effects on organisms (Table 5-7). Food habits studies will include identification of major organisms consumed by fish and the nutritional value of this food as determined by carbon-nitrogen analysis.

5.5.1.2 Manipulative caging experiments

These experiments will be initiated to evaluate the effects of chronic stress under semicontrolled conditions. Healthy, uncontaminated fish will be placed in large cages

located in EFPC and BF. Fish also will be released outside the cages at both sites. The response of these fish to environmental conditions in EFPC will be determined by applying the suite of indicators selected in this and other studies (Loar et al. 1991). Differences in responses between caged (starved) and uncaged (well-fed) fish should indicate the relative importance of direct (water borne) vs indirect (food-chain) effects of chronic stress on fish in EFPC.

5.5.1.3 Studies of reproductive success and competence

Preliminary studies will be initiated to evaluate the effects of chronic stress on the reproductive competence or success of fish in EFPC. The initial studies will primarily involve the design, development, and screening of various bioindicators of reproductive success, such as steroid hormones, vitellogenin production, and egg quality and quantity.

5.5.1.4 Histopathological studies

The histopathological studies will be continued with a focus on quantitative rather than qualitative analyses. Because the liver appears to be the organ of major importance not only in indicating pathological responses of the organism but also for detoxifying contaminants and performing other critical functions, the pathological studies will focus on the liver. Quantitative histopathological measurements of the liver will include functional volume, mean hepatocyte volume, and number of parasites per unit volume. These data will enhance the researcher's ability to relate the effects of stress on individual fish to meaningful fish population responses, such as growth and reproductive success.

5.5.2 Sampling Strategies

As discussed in Sect. 5.3.1, WBR was not an appropriate reference site and, therefore, will no longer be sampled. Sampling sites in EFPC and BF will remain the same except for the possible addition of a site just below the ORWTF. An alternative reference site (possibly Hinds Creek) may also be included in the future, depending on the results of a short-term intensive study that was recently conducted to evaluate the appropriateness of BF as a single reference site.

The sampling frequency for the bioindicator task of the BMAP reduced to one intensive seasonal study. Except for the winter when biological responses are depressed

and atypical of other seasonal patterns, the fall and spring integrated bioindicator responses were similar (Fig. 5-18 to 5-20). Moreover, there is some evidence that multiseasonal sampling may place heavy collection pressure on the redbreast sunfish population, especially females, in EFPC. The new bioindicator initiatives, such as the manipulative caging and reproductive success studies, will compliment the seasonal bioindicator sampling. The major advantage of one intensive seasonal sampling is an increase in sample size, which will improve statistical reliability and the interpretation of results.

6. INSTREAM ECOLOGICAL MONITORING

6.1 BENTHIC MACROINVERTEBRATES

6.1.1 Introduction

Freshwater benthic macroinvertebrates are those organisms large enough to be seen without the aid of magnification that live on or in the substrate. Their limited mobility and relatively long life spans (a few weeks to more than a year) make them ideal for use in evaluating the ecological effects of effluents on streams (Platts et al. 1983). Thus, the composition and structure of the benthic community is a reflection of 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 in the initial year of this task were to (1) spatially and temporally characterize the benthic community of East Fork Poplar Creek (EFPC) and (2) document any effects on the benthic community that may have resulted from the operation of new wastewater treatment facilities at the Y-12 Plant. This information will be used as a baseline to monitor changes in the benthic communities in EFPC over time and to provide direction for future studies.

6.1.2 Materials and Methods

Benthic macroinvertebrates were sampled at approximately monthly intervals from June 1985 through May 1986, at six sites on EFPC (Fig. 2-1). Because a nonaffected reference site was not available on EFPC, a single site on Brushy Fork (BF), a similar-sized tributary of Poplar Creek located north of Oak Ridge (Fig. 2-2), served as a reference stream and was sampled concurrently with EFPC from January through May 1986. Use of a reference stream unaffected by industrial discharges provided an estimate of the structure and variability that might be expected to occur in the benthic community of EFPC.

Benthic macroinvertebrates were collected with a Hess stream bottom sampler (0.1 m^2) fitted with a 363-μm-mesh collection net. Five randomly selected samples were collected from designated riffle areas at each site. To obtain a more complete estimate of species richness within each site, qualitative samples were also taken from riffle and

nonriffle habitats (e.g., pools, leaf packs, detritus, snags, etc.) of both EFPC and BF in April 1986, with a D-frame aquatic dip net (mesh of 800 × 900 μm) and washed in the field in a small handnet (368 μm) and white enamel pan to concentrate the organisms. Both quantitative and qualitative samples were placed in pre-labeled plastic jars and preserved in 80% ethanol; the ethanol was replaced with fresh ethanol within one week.

Various supplemental information was also recorded at the time of sampling. Water temperature and specific conductance were measured at each site with a Coleman-Parmer Model R-1491-20 LCD temperature/ conductivity meter. Water depth, location within the riffle area (distance from permanent headstakes on the stream bank), relative stream velocity (very slow, slow, moderate, or fast), and substrate type using a modified Wentworth particle-size scale (Loar 1985) were recorded for each sample. All samples were washed in the laboratory 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 70% ethanol. Organisms were identified to the lowest practical taxonomic level using a stereoscopic dissecting microscope. A blotted wet weight of all individuals in each taxon was determined to the nearest 0.01 mg with a Mettler analytical balance.

Chironomid larvae were identified from permanent slide mounts. Larvae were initially sorted into groups based on morphological similarities (i.e., body size, head capsule shape and coloration, abdominal setae, etc.), and then the head capsule and body of one or two larvae in each group were mounted on a microscope slide 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 a compound binocular microscope. Slides of mounted larvae were stored in slide boxes and retained for reference purposes.

Individual taxa from a given site and sampling date were preserved in separate vials in 80% ethanol. A reference collection with identifications verified by taxonomic experts will be maintained at ORNL.

All statistical manipulations and analyses were done on transformed data [$\log_{10}$ (X+1)] (Elliott 1977) using the Statistical Analysis System (SAS 1985a,b). The Shannon-Wiener index (H') was used to calculate the taxonomic diversity of benthic macroinvertebrates at each site (Pielou 1977):

$$H' = - \sum P_j \log_2 P_j \ , \qquad (6\text{-}1)$$

where P_j is the proportion of the total number of benthic invertebrates in the sample made up by species j. Density, biomass, species richness, and diversity were analyzed separately with one-way analysis of variance on month-specific data with site as the main effect. Significant differences ($\alpha = 0.05$) were then separated with Duncan's new multiple-range test.

6.1.3 Results

6.1.3.1 Taxonomic composition

Table 6-1 is a checklist of the benthic macroinvertebrates collected in EFPC from June 1985 through May 1986 and in BF from January through May 1986 in both quantitative and qualitative samples. During the year over 111 taxa were collected from EFPC in quantitative samples and an additional 13 taxa were collected in qualitative samples. The number of taxa collected at each of the six sites on EFPC in both quantitative and qualitative samples, respectively, were as follows: EFPC kilometer (EFK) 24.4, 17 and 1; EFK 23.4, 32 and 6; EFK 18.2, 45 and 8; EFK 13.8, 64 and 9; EFK 10.6, 54 and 6; EFK 6.3, 63 and 8. Most major groups of aquatic macroinvertebrates were represented in EFPC, including Planariidae (flatworms), Nematoda (roundworms), Oligochaeta (aquatic earthworms), Isopoda (aquatic sow bugs), Amphipoda (sideswimmers), Decapoda (crayfish), Insecta (insects), and Mollusca (snails and mussels). The majority of the invertebrates collected were insects, comprising 94 of the taxa collected from EFPC. Within this group, the chironomids (Chironomidae) were represented by the greatest number of taxa (52), followed by dragonflies/damselflies (Odonata) (18), nonchironomid dipterans (Diptera) (11), mayflies (Ephemeroptera) (7), caddisflies (Trichoptera) (6), beetles (Coleoptera) and stoneflies (Plecoptera) (5 each), dobsonflies (Megaloptera) (2), and springtails (Collembola) and butterflies/moths (Lepidoptera) (1 each). Chironomid and nonchironomid dipterans, caddisflies, dragonflies/damselflies, and mayflies were collected from all sampling sites on EFPC. At EFK 24.4, however, only a single caddisfly was collected; it had probably adhered to the sampling net at a previously sampled site. Beetles were collected at all sites except EFK 24.4, stoneflies were collected at EFK 6.3 and EFK 13.8 only, and dobsonflies were

Table 6-1. Checklist[a] of benthic macroinvertebrate taxa collected in East Fork Poplar Creek (EFK) from June 1985–May 1986 and Brushy Fork (BFK), a reference stream, from January–May 1986

Taxon	EFK 6.3	EFK 10.6	EFK 13.8	EFK 18.2	EFK 23.4	EFK 24.4	BFK 7.6
Turbellaria							
Planariidae	X						
Nematoda		X	X	X	X	X	X
Oligochaeta	X	X	X	X	X	X	X
Naididae	X	X		X			
Tubificidae				X		X	
Crustacea							
Isopoda							
Asellus	X						
Lirceus	X		X	X		X	X
Amphipoda							
Crangonyx	X	X	X	X	Q		
Decapoda							
Cambarus	X	X	X	X	X		X
Hydracarina							
Parsitengona			X				X
Insecta							
Collembola							
Entomobryidae			X				
Ephemeroptera							
Baetidae							
Baetis	X	X	X	X	X	X	X
Pseudocloeon							X
Caenidae							
Caenis		X					
Ephmerellidae							
Ephemerella					X		X
Eurylophella	Q	X	Q	Q			X
Ephemeridae							
Hexagenia							Q

Table 6-1 (continued)

Taxon	EFK 6.3	EFK 10.6	EFK 13.8	EFK 18.2	EFK 23.4	EFK 24.4	BFK 7.6
Heptageniidae							
Stenacron	Q		X				X
Stenonema	X		X	Q			X
Leptophelebiidae							
Paraleptophlebia							Q
Oligoneuriidae							
Isonychia							X
Tricorythidae							
Tricorythodes			X				
Odonata							
Anisoptera							
Aeshnidae							
Boyeria	X	Q	Q	X			
Boyeria vinosa			X	X			Q
Cordulegastridae							
Cordulegaster maculata			Q				
Gomphidae							
Dromogomphus spinosus	Q						
Gomphus	Q			Q			Q
Gomphus lividus		Q					
Hagenius brevistylus							Q
Omphiogomphus			X				
Stylogomphus							X
Stylogomphus albistylus			X				X
Libellulidae							
Erythemis simplicicollis				Q			
Plathemis lydia					Q		
Macromiidae		Q					
Didymops transversa	Q						
Macromia			Q	Q			
Zygoptera							
Calopterygidae							
Calopteryx maculata	X	Q	Q	Q			Q
Coenagrionidae			X		X		
Argia	X	X	Q	X	Q	Q	
Enallagma	Q	X	Q	Q	Q		Q
Plecoptera							
Capniidae							
Allocapnia			X				X

Table 6-1 (continued)

Taxon	EFK 6.3	EFK 10.6	EFK 13.8	EFK 18.2	EFK 23.4	EFK 24.4	BFK 7.6
Leuctridae							
Leuctra							X
Nemouridae							
Amphinemura	X		X				
Amphinemura delosa	X						X
Perlidae							
Eccoptura xanthenes			Q				
Taeniopterygidae							
Taeniopteryx	X						X
Megaloptera							
Corydalidae							
Corydalus cornutus		Q	X				
Nigronia serricornis	X						Q
Trichoptera							
Glossosomatidae							
Glossosoma	X		X				X
Hydropsychidae							
Cheumatopsyche	X	X	X	X	X		X
Hydropsyche							X
Hydropsyche depravata ?	X	X	X	X	X	X	
Hydroptilidae			X				
Hydroptila			X				
Ochrotrichia							X
Leptoceridae							
Triaenodes							Q
Limnephilidae							
Goera							X
Hydatophylax							Q
Neophylax							X
Pycnopsyche				Q			Q
Philopotamidae							
Chimarra							X
Polycentropodidae							
Phylocentropus							Q
Polycentropus							Q

Table 6-1 (continued)

Taxon	EFK 6.3	EFK 10.6	EFK 13.8	EFK 18.2	EFK 23.4	EFK 24.4	BFK 7.6
Psychomiidae							
Psychomyia							X
Lepidoptera						X	
Coleoptera							Q
Elmidae							
Dubiraphia	X	X	X		X		X
Optioservus	X	X	X	X			X
Stenelmis	X	X	X	X	X		X
Hydrophilidae	Q	X	Q				
Psephenidae							
Psephenus herricki	X	X	X				X
Diptera							
Ceratopogoniidae	X	X	X	X			X
Chironomidae							
Chironomini							
Chironomus	X	X	X	X	X	X	X
Cryptochironomus	X	X	X	X			X
Dicrotendipes		X	X	X			X
Endochironomus ?			X				
Goeldichironomus						X	
Goeldichironomus holoprasinus					X		
Harnischia	X						
Microtendipes				X			X
Paracladopelma							Q
Paraluterborniella							Q
Paratendipes							X
Phaenopsectra	X	X	X	X			X
Polypedilum	X	X	X	X			X
Polypedilum fallax		X		X			X
Stenochironomus				X			
Stictochironomus				X			
Stictochironomus ?		X					
Tanytarsini	X				X		
Cladotanytarsus							X
Paratanytarsus				Q			
Rheotanytarsus	X	X	X	X	X		X
Tanytarsus	X	X	X	X	X		X

Table 6-1 (continued)

Taxon	EFK 6.3	EFK 10.6	EFK 13.8	EFK 18.2	EFK 23.4	EFK 24.4	BFK 7.6
Diamesinae							
Diamesa			X				
Orthocladiinae		X	X				X
Brillia	X	X					
Cardiocladius	X	X	X		X	X	X
Corynoneura		X					
Cricotopus bicinctus gp[b]	X	X	X	X	X	X	
Cricotopus sylvestris gp[b]	X	X			X		
Cricotopus tremulus gp[b]	X	X	X	X	X	X	
Cricotopus trifascia gp[b]	X		X	X	X		
Cricotopus/ Orthocladius[b]	X	X	X	X	X	X	X
Eukiefferiella							X
Hydrobaenus	X	Q		X		X	X
Nanocladius	X	X	X	X			X
Orthocladius		X					
Orthocladius ?			X				
Paracricotopus							Q
Parakiefferiella	X	X	X				
Parakiefferiella ?		X					
Parametriocnemus	X		X				
Psectrocladius	Q						
Rheocricotopus	X	X	X	X	X		X
Synorthocladius			X				X
Thienemanniella	X	X	X				X
Thienemanniella ?				X			
Tvetenia	X						X
Xylotopus par		X					
Tanypodinae			X				
Ablabesmyia	X	X	X	X			X
Coelotanypus	X						
Conchapelopia ?	X						
Labrundinia							Q
Natarsia punctata	X	X	X	X	X	X	X
Nilotanypus		X					
Paramerina							Q
Pentaneura	X						
Procladius	X	X		X	X		

Table 6-1 (continued)

Taxon	EFK 6.3	EFK 10.6	EFK 13.8	EFK 18.2	EFK 23.4	EFK 24.4	BFK 7.6
Thienemannimyia gp	X	X	X	X	X	X	X
Zavrelimyia				X			
Empididae							
Hemerodromia	X	X	X	X	X	X	X
Muscidae						X	
Psychodidae							
Pericoma						X	
Simuliidae	X	X	X	X	X		X
Stratomyiidae		X					
Tabanidae		X	X	X	Q		X
Tipulidae			X				
Antocha			X				X
Tipula	X	X			X		X
Tipula abdominalis					X		X
Mollusca							
Gastropoda							
Ancylidae							
Ferrissia	X	X	X	X	X		X
Bithyniidae							X
Lymnaeidae	X						
Fossaria			X		X		
Physidae							
Physella	X	X	X	X	X		
Planorbidae	X						
Gyraulus	X		X				
Pleuroceridae							
Elimia							X
Pleurocera							X
Pelecypoda							
Sphaeriidae							
Pisidium							X
Sphaerium	X		X				X
Unionidae							
Villosa							X

[a]An "X" indicates that the taxon was collected at least once, a "Q" denotes that the taxon was collected in qualitative samples only, and a "?" indicates that the identification is questionable.

[b]Because of the difficulty in reliably separating the species groups within the genus of *Cricotopus* from each other and from species of *Orthocladius*, they have been lumped into the *Cricotopus/Orthocladius* group for all data analyses except in the discussion of taxonomic composition.

collected at EFK 6.3, EFK 10.6, and EFK 13.8 only. A single individual in the butterfly/moth order was collected at EFK 24.4, but whether its origin was terrestrial or aquatic is not known. Of the noninsect taxa, crustaceans and oligochaetes were collected at all sites on EFPC, mollusks and nematodes were collected at all sites except EFK 24.4, and a planarian was collected at EFK 6.3.

In BF, a total of 88 taxa were collected in the quantitative samples from January through May 1986, of which 69 and 16 were collected in quantitative and qualitative samples, respectively (see Table 6-1). As is EFPC, a majority of the macroinvertebrates collected in BF were insects, totaling 76 taxa. As in EFPC, chironomids had the greatest number of taxa (30). Of the remaining insect taxa, the caddisflies were represented by 13 taxa, the mayflies by 9, the nonchironomid dipterans and dragonflies/damselflies by 7 each, the beetles by 5, stoneflies by 4, and the dobsonflies by 1. Oligochaetes, nematodes, crustaceans, and mollusks were also collected from BF.

6.1.3.2 Density and biomass

The mean density (number of individuals/0.1 m^2) and biomass (mg/0.1 m^2) of benthic macroinvertebrates in EFPC for June 1985 through May 1986, are presented in Fig. 6-1.[1] Density and biomass generally increased in a downstream direction from the Y-12 Plant to EFK 13.8; abundance declined at the lower two sites on EFPC to levels more similar to those found at EFK 23.4 and EFK 18.2. The highest mean density was 254.5 individuals per 0.1 m^2 at EFK 18.2. The lowest mean density, 93.0 individuals per 0.1 m^2, was observed above New Hope Pond (NHP) at EFK 24.4, whereas the mean density of the other sites ranged from 105.7 to 251.1 individuals per 0.1 m^2.

Mean biomass (excluding Mollusca and Decapoda) ranged from a maximum of 1365.1 mg/0.1 m^2 at EFK 13.8 to a minimum of 86.2 mg/0.1 m^2 at EFK 24.4; the means of the other sites ranged from 139.5 to 286.9 mg/0.1 m^2. The mean biomass (mg/0.1 m^2) of

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[1]The biomass of the mollusks and crayfish was not included in discussions of temporal and spatial trends of community biomass because these taxa rarely contributed much in terms of numbers to the community, but because of their large size and/or heavy shells. Their high biomasses often masked the seasonal changes occurring in the more abundant taxa. For example, three crayfish collected at EFK 23.4 in October represented 96.4% of the total community biomass for that month.

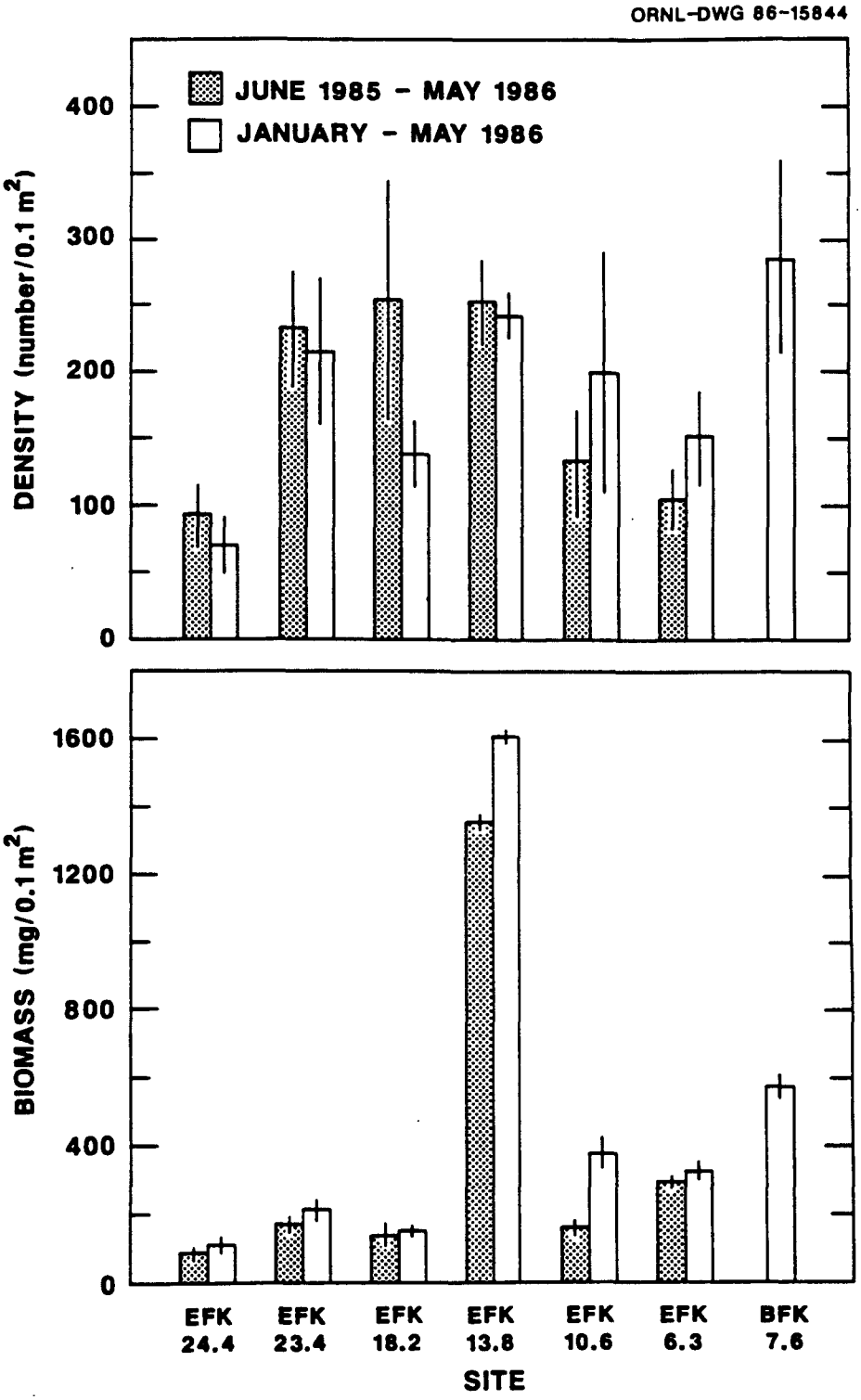

Fig. 6-1. Mean benthic macroinvertebrate density and biomass (excluding Mollusca and Decapoda) in East Fork Poplar Creek (EFK) and Brushy Fork (BFK), a reference stream, June 1985–May 1986. Vertical line represents the standard error of the mean.

decapods and mollusks at each site in EFPC was as follows: EFK 24.4, 0.0; EFK 23.4, 264.9; EFK 18.2, 337.1; EFK 13.8, 1124.8; EFK 10.6, 2.8; and EFK 6.3, 96.4.

The large difference in mean annual biomass of EFK 13.8 relative to the other EFPC sites resulted from differences in the dominant taxa (Fig. 6-2). The benthic community at EFK 13.8 was made up of relatively large numbers of trichopterans (primarily *Hydropsyche*) and chironomids, whereas the other EFPC sites were either dominated by small chironomids or had much lower overall densities of large taxa relative to EFK 13.8.

When the data were examined only for the period in which BF was also sampled (January through May 1986), the trend in mean density and biomass in EFPC was similar to that observed over the entire year (Fig. 6-1). In general, there was a downstream increase in both parameters to EFK 13.8, then a return at the lowest two sites to density and biomass values similar to the three upstream sites. Although slightly higher, the mean density of invertebrates in BF (286.6 individuals per 0.1 m^2) during this period was comparable to the means observed at some sites in EFPC (Fig. 6-1). Mean biomass (excluding Mollusca and Decapoda) in BF (583.2 mg/0.1 m^2) during this period was somewhat greater than that observed at all EFPC sites except EFK 13.8. This was the result of a lack of dominance by any particular taxonomic group and a more-even distribution of the biomass among the various taxa. The mean biomass of mollusks and decapods in BF during this period was 5878.21 mg/0.1 m^2.

Statistical comparison of monthly community density and biomass data (Figs. 6-3 and 6-4) confirmed the general trend of increasing abundance downstream to EFK 13.8; below this site, abundance decreased to levels comparable to those observed at the sites above EFK 13.8 (Table 6-2). Mean monthly densities varied during the year from a low of 5.4 individuals per 0.1 m^2 at EFK 24.4 in June to a high of 1047.4 individuals per 0.1 m^2 at EFK 18.2 in September (Fig. 6-3). Mean monthly biomass (excluding Mollusca and Decapoda) ranged from a low of 3.2 mg/0.1 m^2 at EFK 24.4 in June to a high of 3353.1 mg/0.1 m^2 at EFK 13.8 in April (Fig. 6-4). Mean monthly biomass of the mollusks and decapods ranged from 0 at EFK 24.4, where neither taxon was collected, to 3308.1 mg/0.1 m^2 at EFK 13.8 in April.

Over the 5-month period when the two streams were sampled concurrently, densities in BF and EFPC at EFK 13.8 and EFK 23.4 were not significantly different

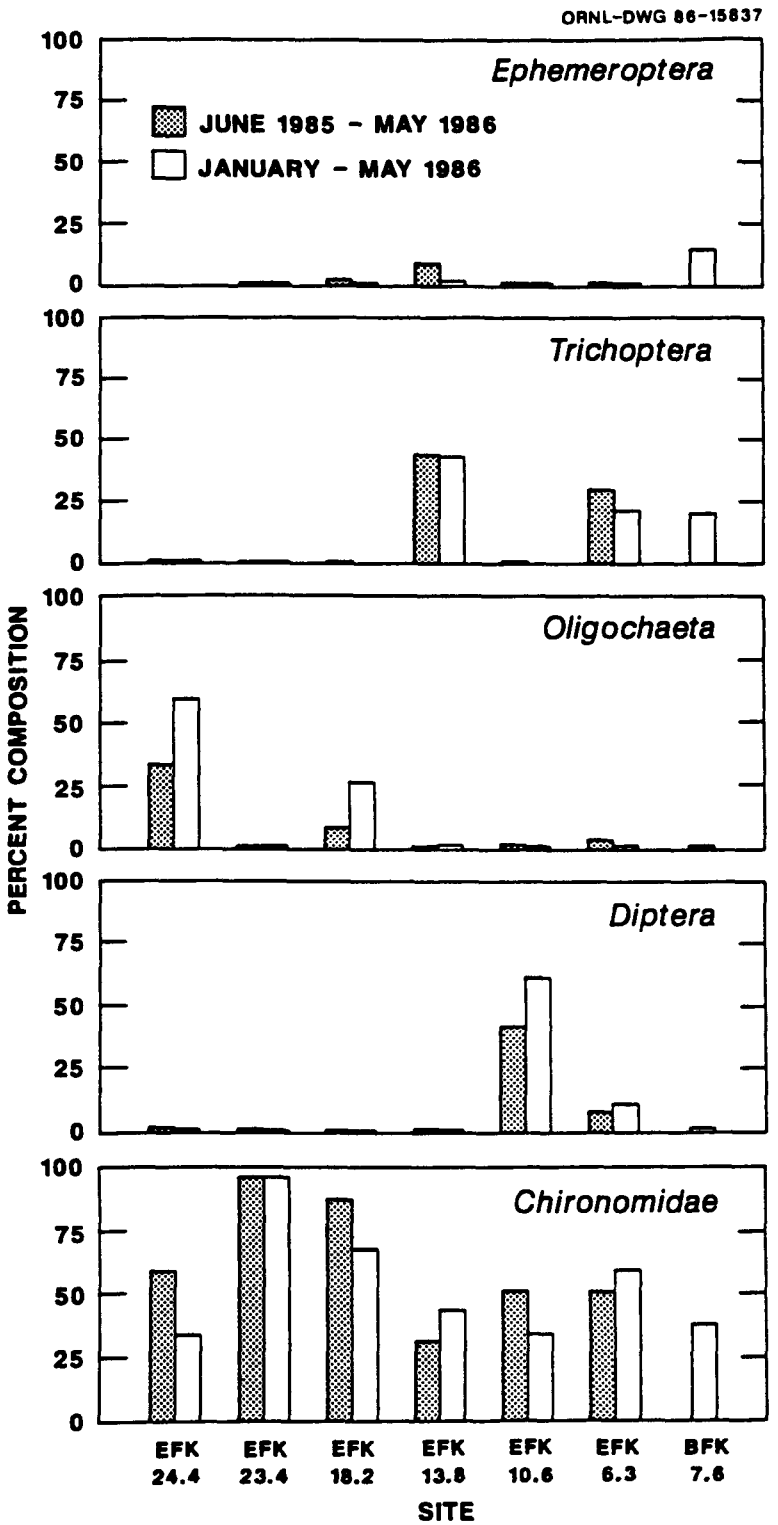

Fig. 6-2. Annual percentage composition of selected benthic macroinvertebrate taxa in East Fork Poplar Creek and Brushy Fork, a reference stream, June 1985–May 1986. Values are based on mean annual density. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

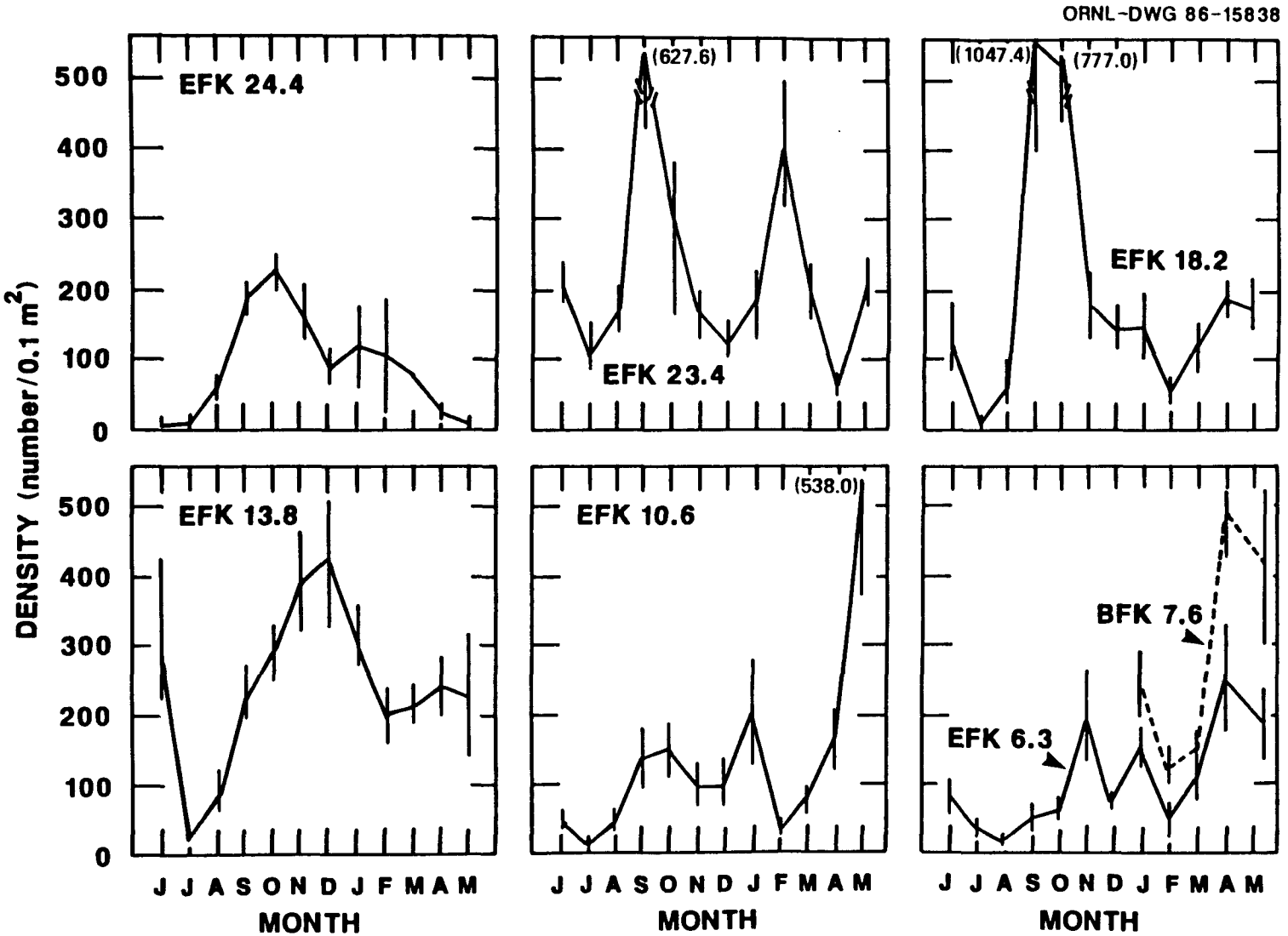

Fig. 6-3. Mean monthly density of benthic macroinvertebrates in East Fork Poplar Creek and Brushy Fork, a reference stream, June 1985–May 1986. Vertical line represents the standard error of the mean. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

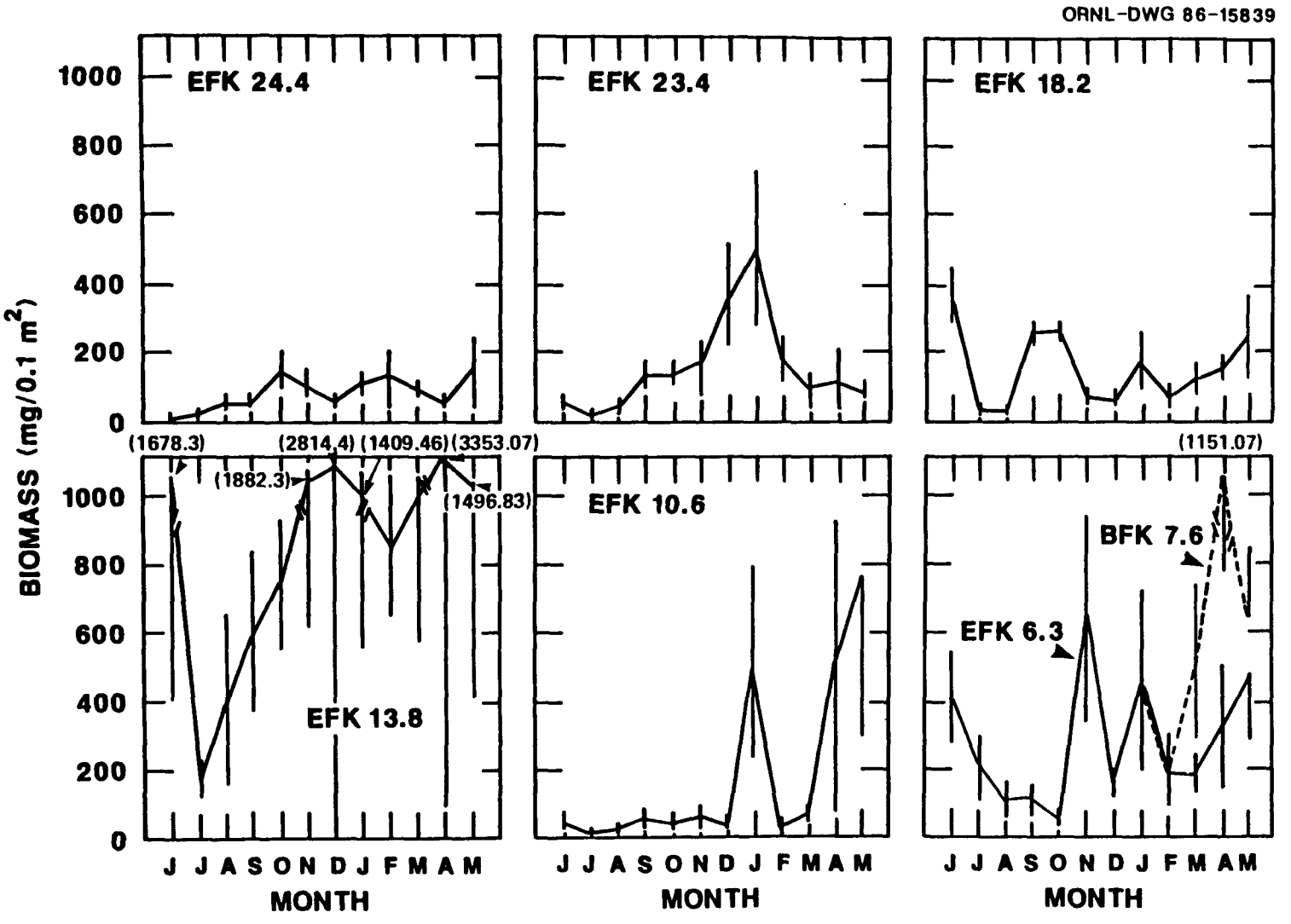

Fig. 6-4. Mean monthly biomass of benthic macroinvertebrates (excluding Mollusca and Decapoda) in East Fork Poplar Creek and Brushy Fork, a reference stream, June 1985-May 1986. Vertical line represents the standard error of the mean. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

Table 6-2. Statistical comparisons of mean values of benthic macroinvertebrate density, biomass (excluding Decapoda and Mollusca), species richness, and diversity (H') in East Fork Poplar Creek,[a] June 1985–May 1986

Ranking					
High					Low
		Density			
EFK 13.8	EFK 23.4	EFK 18.2	EFK 10.6	EFK 6.3	EFK 24.4
		Biomass			
EFK 13.8	EFK 6.3	EFK 23.4	EFK 18.2	EFK 10.6	EFK 24.4
		Richness			
EFK 13.8	EFK 10.6	EFK 18.2	EFK 6.3	EFK 23.4	EFK 24.4
		Diversity			
EFK 10.6	EFK 13.8	EFK 18.2	EFK 6.3	EFK 24.4	EFK 23.4

[a]Those sites not connected by the same line are significantly different ($\alpha = 0.05$).

(Table 6-3). Biomass at EFK 13.8 was significantly greater than that in BF, but biomass in BF was significantly greater than all other sites on EFPC. The greater biomass at EFK 13.8 was the result of the relatively large numbers of caddisflies at this site. Mean monthly biomass in BF (excluding Mollusca and Decapoda) ranged from 193.8 to 1151.1 mg/0.1 m^2, and the mean molluscan and decapod biomass during this period ranged from 2349.3 to 10,408.6 mg/0.1 m^2.

With the exception of EFK 13.8 and EFK 24.4, all sites in EFPC showed a similar seasonal pattern in benthic invertebrate density and biomass (Figs. 6-3 and 6-4, respectively). After a summer low in both density and biomass, a peak occurred in early fall to midfall, followed by a decline in late fall and a second peak in early winter to midwinter; a third peak usually occurred in midspring to late spring. At EFK 13.8, only two peaks occurred in density and biomass, one in late fall/early winter and the other in

Table 6-3. Statistical comparisons of mean values of benthic macroinvertebrate density, biomass (excluding Decapoda and Mollusca), species richness, and diversity (H') in East Fork Poplar Creek (EFK) and Brushy Fork (BFK), January–May 1986. Those sites not connected by the same line are significantly different ($\alpha = 0.05$).

Ranking						
High						Low
			Density			
BFK 7.6	EFK 13.8	EFK 23.4	EFK 10.6	EFK 18.2	EFK 6.3	EFK 24.4
			Biomass			
EFK 13.8	BFK 7.6	EFK 6.3	EFK 10.6	EFK 23.4	EFK 18.2	EFK 24.4
			Richness			
BFK 7.6	EFK 13.8	EFK 10.6	EFK 18.2	EFK 6.3	EFK 23.4	EFK 24.4
			Diversity			
BFK 7.6	EFK 18.2	EFK 10.6	EFK 13.8	EFK 6.3	EFK 24.4	EFK 23.4

midspring. Two peaks in density and biomass also occurred at EFK 24.4, with the largest peak occurring in late fall and another occurring in early winter. Although limited, the temporal data available for density and biomass of the benthic community in BF showed temporal trends similar to those observed in EFPC (Figs. 6-3 and 6-4). In BF, an apparent peak occurred in early winter and was followed by a decline through midwinter before increasing to a peak in midpring. Additional data are needed to more fully characterize temporal patterns at this site.

The seasonal changes in abundance that occurred in EFPC largely reflected changes in one or a few dominant taxa. The relative abundance of chironomids and other key taxa in EFPC is shown in Fig. 6-5. Chironomids generally comprised 50% or more of the total density each month at EFK 24.4, EFK 23.4, EFK 18.2, and EFK 10.6; chironomids were relatively less dominant at EFK 13.8 and EFK 6.3. The *Cricotopus/Orthocladius* group alone accounted for most of the seasonal variability at

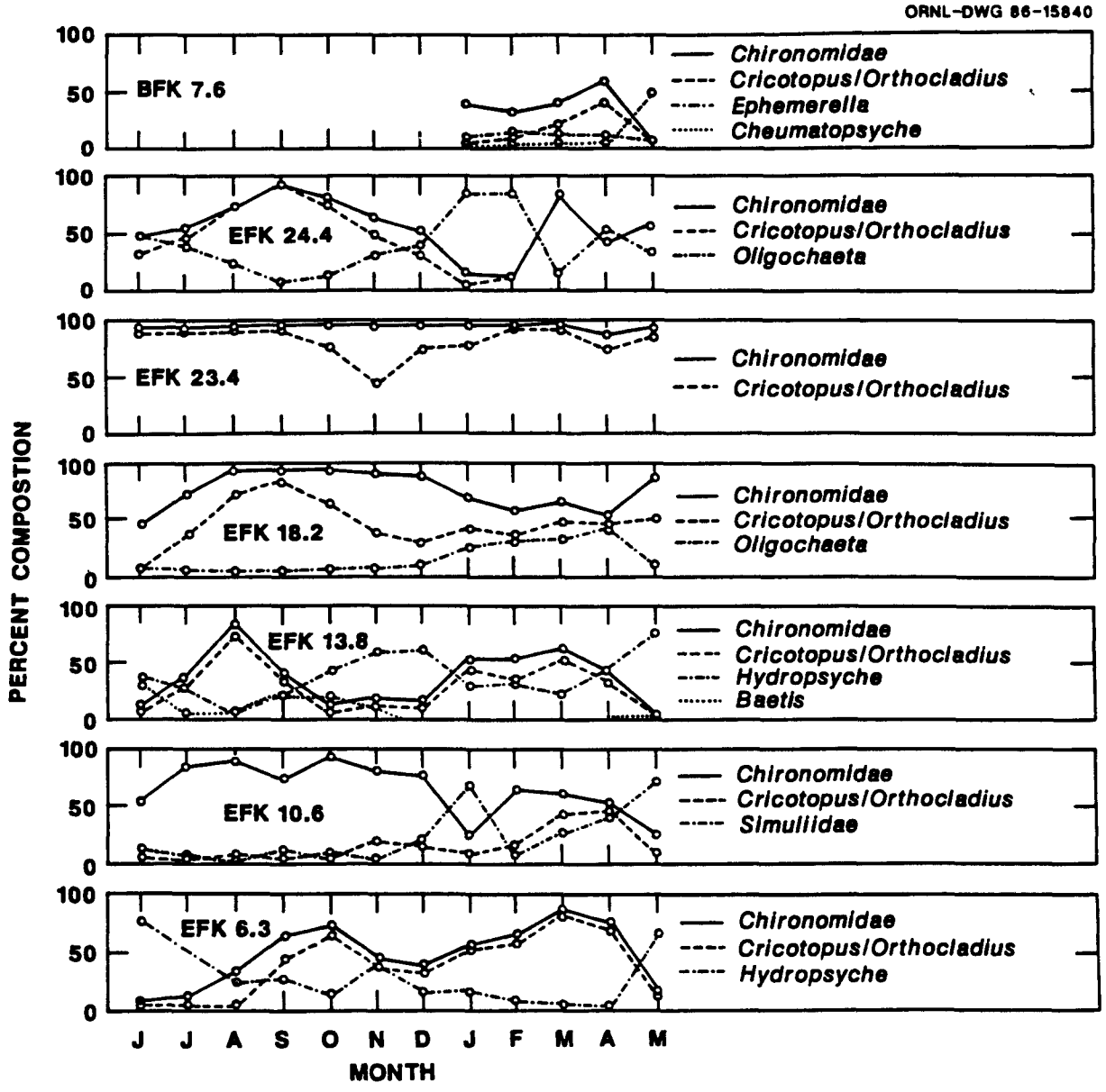

Fig. 6-5. Percentage composition, by month, of selected benthic macroinvertebrate taxa in East Fork Poplar Creek and Brushy Fork, a reference stream, June 1985-May 1986. Values are based on monthly densities. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

EFK 23.4 and much of the variability at EFK 24.4, although oligochaetes were a significant component of the seasonal patterns observed at the latter site.

The structure of the benthic community became increasingly complex from EFK 18.2 downstream to EFK 6.3. The seasonal patterns observed at EFK 18.2 largely reflected changes in the abundance of chironomids, although oligochaetes and the mayfly, *Baetis*, also contributed to the seasonal patterns during certain periods of the year. Within the chironomids, the *Cricotopus/Orthocladius* group was still the dominant group, but *Chironomus*, *Natarsia punctata*, *Procladius*, *Tanytarsini*, and the *Thienemannimyia* group also occurred in relatively high numbers during some months. At EFK 13.8, the observed seasonal variability in abundance primarily reflected the population dynamics of the trichopteran, *Hydropsyche*, but chironomids (primarily the *Cricotopus/Orthocladius* group) and the mayfly, *Baetis*, also contributed substantially to this variability. At EFK 10.6, a major change in key taxa occurred. Although chironomids contributed significantly to the seasonal abundance patterns, simuliids were dominant in some months and were the major contributor to the peaks in benthic density that occurred in winter and late spring. A major shift in the dominant chironomid taxa was also observed. Although the *Cricotopus/Orthocladius* group was periodically dominant, the chironomids *Polypedilum*, *Rheotanytarsus*, *Tanytarsus*, and the *Thienemannimyia* group was often very abundant. At EFK 6.3, *Hydropsyche* and the *Cricotopus/Orthocladius* group, again, were the dominant taxa. Seasonal fluctuations in benthic invertebrate density and biomass largely reflected changes in the abundance of these two taxa.

6.1.3.3 Community structure

Species richness

The total number of taxa present (richness) is a useful parameter for qualitatively assessing the health of a stream (Platts et al. 1983). Under environmental stress, taxonomic richness would be expected to decrease. The total number of taxa collected from the six sites on EFPC from June 1985 through May 1986 is shown in Fig. 6-6.[2] The

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[2]Because of the difficulty in reliably separating the species groups of the genus *Cricotopus* from some species of the genus *Orthocladius* and from each other, the two groups were combined into a single *Cricotopus/Orthocladius* group for analyzing changes in species richness.

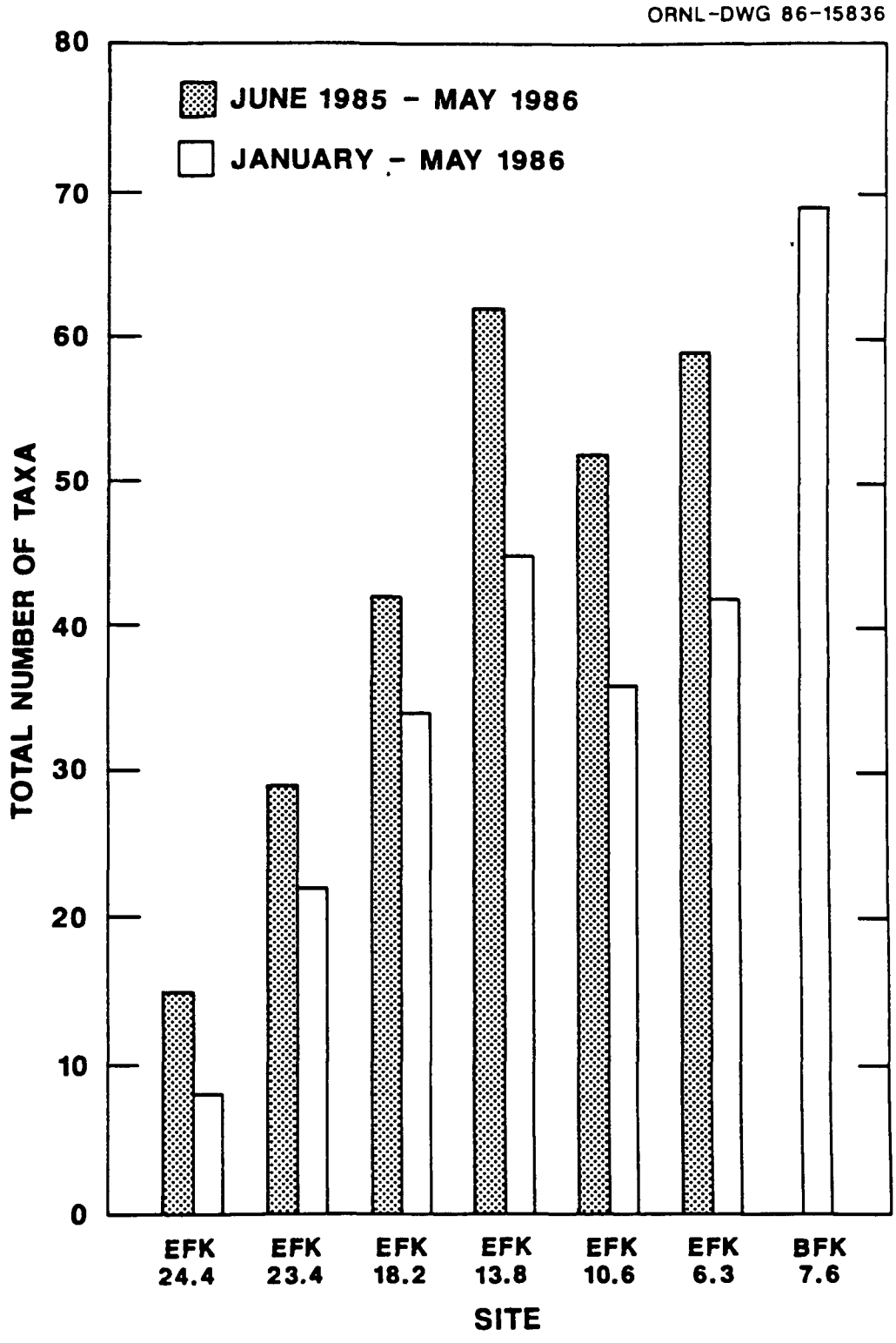

Fig. 6-6. Total number of benthic macroinvertebrate taxa collected in East Fork Poplar Creek and Brushy Fork), a reference stream, June 1985-May 1986. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

number of taxa increased from EFK 24.4 downstream to EFK 13.8, then decreased at EFK 10.6 before increasing slightly at EFK 6.3. The same trend was evident when the data set included the total number of taxa collected during only the 5-month period that BF was also sampled (Fig. 6-6). The most striking feature of these latter data was that more taxa were collected from BF during this 5-month period than were collected from any EFPC site during the entire year. Also, more taxa were collected at EFK 13.8 during the five-month period than were collected at EFK 18.2, EFK 23.4, or EFK 24.4 during the entire year.

This same general trend was exhibited when the data on total and mean number of taxa collected per month were plotted (Figs 6-7 and 6-8, respectively). Both the total and mean number of taxa per month increased steadily from EFK 24.4 to EFK 13.8 before decreasing again at the lower two sites. This trend was confirmed by a statistical comparison of the total number of taxa collected each month (Table 6-2). A similar trend was observed for the 5-month period when BF and EFPC were sampled concurrently (Figs. 6-7 and 6-8). Again, statistical comparison of the mean number of taxa collected during this period confirmed this trend and showed that species richness was significantly greater in BF than at any site on EFPC (Table 6-3). The EFPC site most similar in total and mean number of taxa to BF was EFK 13.8; however, the difference between these sites was still considerable. In general, the increase in the total number of taxa in a downstream direction paralleled an increase in seasonal variability of the mean number of taxa. Such a trend reflects the increasing complexity of the structure of these communities with distance from NHP.

Species diversity

Because species diversity indices combine information on both species abundance and richness into a single dimensionless value, they sometimes provide a useful tool for interpreting changes in the structure of benthic invertebrate communities (Weber 1973; Platts et al. 1983). Platts et al. (1983) indicated that H' values of 3 or greater are generally found in areas of clean water, whereas values of 1 to <3 occur in areas of moderate pollution, and values of <1 occur in areas of heavy pollution. In general, maximum diversity at all sites on EFPC usually occurred between midfall and early winter, whereas minimum diversity typically occurred in midsummer to late summer and/or late

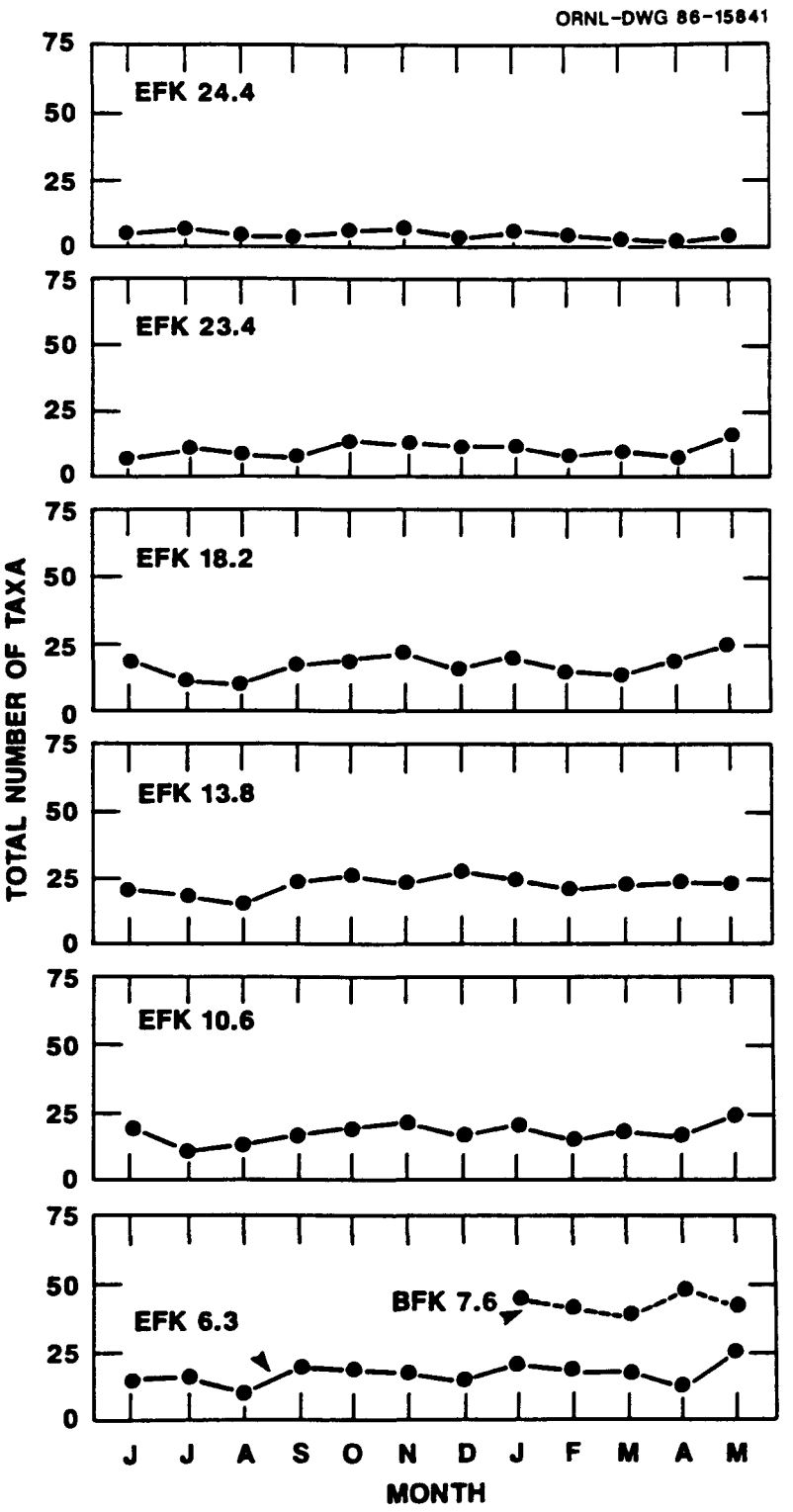

Fig. 6-7. Total number of benthic macroinvertebrate taxa collected each month in East Fork Poplar Creek and Brushy Fork, a reference stream, June 1985–May 1986. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

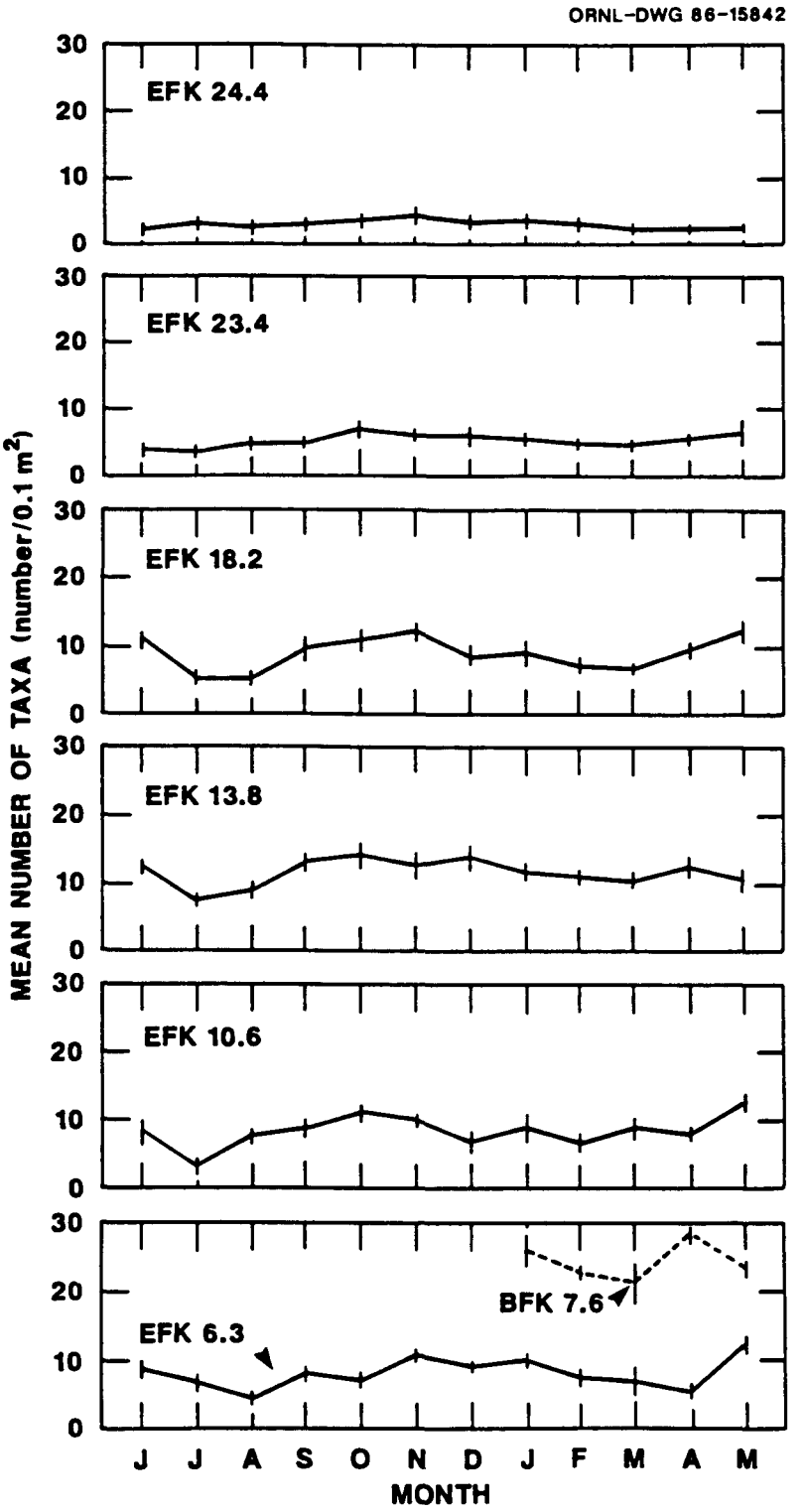

Fig. 6-8. Mean number of benthic macroinvertebrate taxa collected each month in East Fork Poplar Creek and Brushy Fork, a reference stream, June 1985–May 1986. Vertical line represents the standard error of the mean. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

winter to midspring (Fig. 6-9). This is the expected pattern for a community dominated by insects because of the natural seasonal changes that occur in such a community as a result of recruitment, emergence, and death.

Values of H' for the six EFPC sites fell into two distinct groups. The first group consisted of sites EFK 24.4 and EFK 23.4, where H' remained consistently at or below 1 throughout the year. The second group consisted of sites EFK 18.2, EFK 13.8, EFK 10.6, and EFK 6.3, where, with very few exceptions, H' usually varied between 1.5 and 2.5. Statistical analyses of the data confirmed these groupings (Table 6-2). Values of H' for BF were significantly greater than for any EFPC site, and fell below 3 in only one month. Also, the maximum mean H' for any site on EFPC never exceeded the minimum H' observed in BF. These data suggest that, although the two upper sites on EFPC are severely affected, some recovery is occurring in the benthos at the lower four sites.

6.1.4 Discussion

The dominance of insects in both EFPC and BF is not surprising because they are usually the dominant macroinvertebrates in streams (Hynes 1970a). Likewise, the large number of chironomid taxa in both streams is consistent with their dominance in most aquatic ecosystems (Coffman and Ferrington 1984). However, the absence and/or low diversity of pollution-sensitive taxa, such as mayflies and stoneflies (Hubbard and Peters 1978; Surdick and Gaufin 1978), in EFPC relative to BF, suggests that the benthos in EFPC are under varying degrees of stress. This is further illustrated by the significantly greater species richness and diversity at BF relative to all sites on EFPC. The most highly affected site was EFK 24.4 above NHP, but conditions gradually improved downstream to EFK 13.8. Changes in the community below EFK 13.8 indicate the existence of an additional source of perturbation [the Oak Ridge Wastewater Treatment Facility (ORWTF)], as evidenced by the significant reductions in density, biomass, and species richness that occurred at EFK 10.6, ~2.8 km below the ORWTF.

Compared with the results of the limited sampling effort conducted in 1974/1975 (see Loar et al. 1989), little change appears to have occurred in the benthos just below NHP; chironomids continued to dominate the benthic community in 1985/1986, with only minor contributions from a few additional taxa. However, the recent data do suggest that some improvement has occurred in the benthic community farther downstream, at least

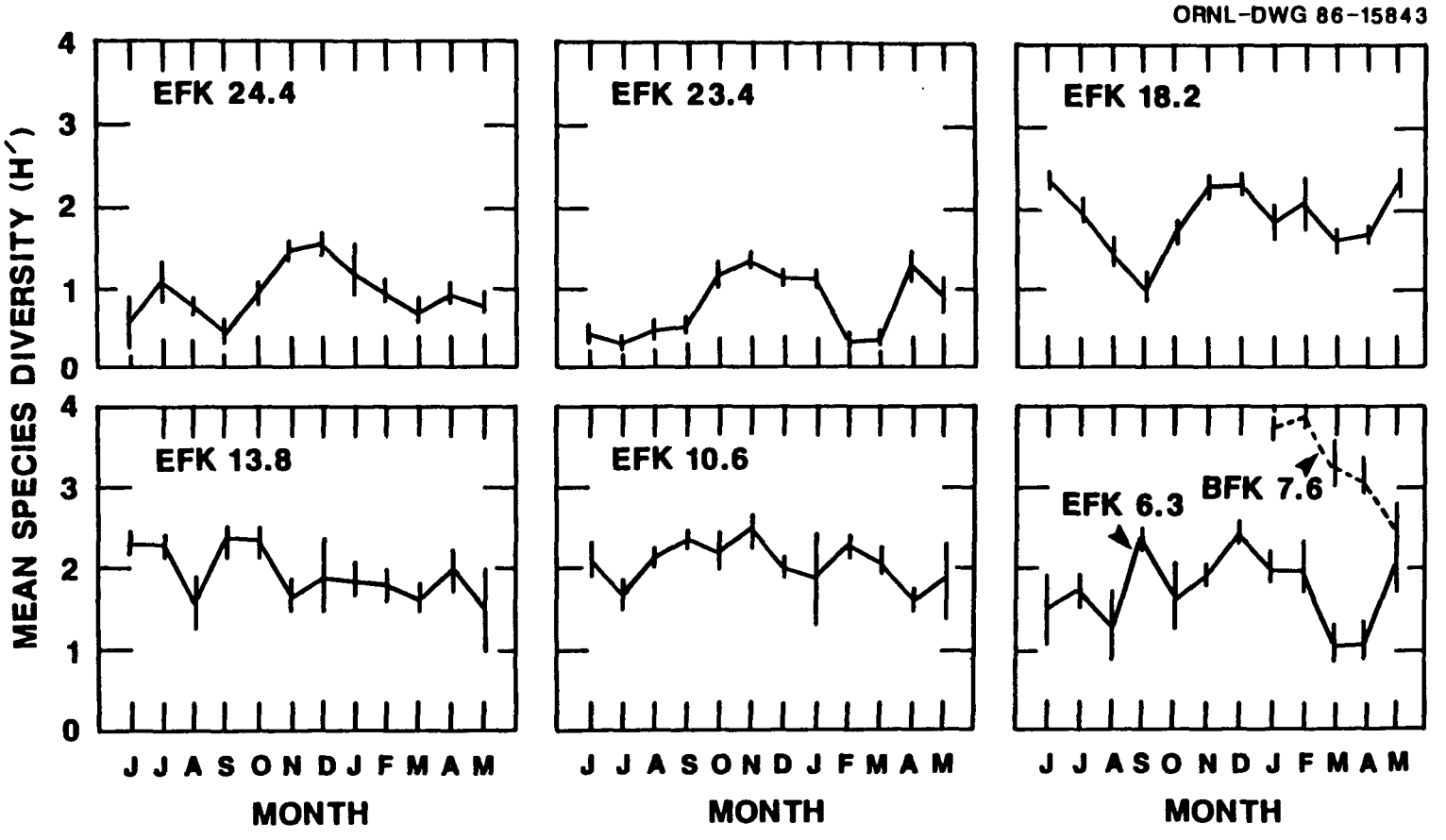

Fig. 6-9. Mean monthly diversity (H') of benthic macroinvertebrates in East Fork Poplar Creek and Brushy Fork, a reference stream, June 1985–May 1986. Vertical line represents the standard error of the mean. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

above the ORWTF, as evidenced by the appearance of caddisflies and mayflies. No clear evidence exists that the benthos downstream of this facility has changed significantly over the past 10 years. Although data from both the current and earlier study showed the community to be dominated by chironomids, results of the current study indicate that the community has become more complex. Additional taxa were dominant during certain times of the year (i.e., Simuliidae) and taxa not reported in the 1974/1975 study (e.g., mayflies and caddisflies) were found in the current study. However, the absence of these taxa in the previous study may have been the result of the limited sampling program. Because many taxa exhibit dramatic seasonal changes in abundance in EFPC, particularly mayflies and simuliids, their presence might not be detected in a limited sampling program.

Although some modifications in the discharge of effluents from the Y-12 Plant occurred as a result of the operation of new wastewater treatment facilities (see Table 1-1), there was no clear evidence from the benthos data that such changes had any effects on the benthic communities in EFPC. Additional data are needed to determine whether the changes observed in the benthos represent natural seasonal changes or recovery.

The obvious source of impact on the benthic community at EFK 24.4 and EFK 23.4 is the Y-12 Plant because these sites are located in the headwaters of EFPC and no other industrial or municipal discharges occur in this reach. Downstream of EFK 23.4, the contribution of the Y-12 Plant to the impact on the benthos is unclear. At EFK 18.2, for example, there is evidence of extensive siltation that could be the result of high sediment loading from operation of a water treatment plant near EFK 22.0 before 1978 (Loar et al. 1989) or to more recent construction activities in the floodplain within the city of Oak Ridge. Between sites EFK 13.8 and EFK 10.6, effluent from the ORWTF enters EFPC (outfall located at EFK 13.4). Additionally, urban and agricultural runoff occur along the creek downstream of the Oak Ridge Reservation (ORR) boundary.

Specific causes of the impact on the benthic community in upper EFPC are not clear. Possible causes include metals, organics, chlorine, nutrient enrichment, and alterations in flow and temperature.

Depending on the type of stress, densities of benthic invertebrates can either increase or decrease (e.g., Wiederholm 1984). Under the stress of a toxicant, the benthic community is usually characterized by low species richness and low numbers (Hynes 1960,

Wiederholm 1984). Although species richness was very low at EFK 24.4 and EFK 23.4, the density of invertebrates at these sites was usually within the normal range found for unpolluted streams within the Oak Ridge area [i.e., 30 to 500 individuals per 0.1 m^2 (G. F. Cada, ORNL, unpublished data; Loar et al. 1981a; Loar et al. 1991)].

Data on concentrations of several metals in EFPC at the outfall of NHP were examined to identify potential toxicants to invertebrate biota. Even though concentrations occasionally exceeded National Pollutant Discharge Elimination System (NPDES) permit limits, the mean concentration at the outfall of NHP between May 1985 and June 1986 was relatively low for most parameters (see Table 2-2). Concentrations of selected metals in EFPC at the outfall of NHP were similar to those found in other studies in which little impact on the benthos was noted (Table 6-4). However, mayflies, a group of insects included in these studies that is very sensitive to elevated metal concentrations (Wiederholm 1984), were severely affected in EFPC. This analysis suggests that these metals are probably not responsible for the absence of mayflies in upper EFPC.

Other compounds, such as ammonia, perchloroethylene, and residual chlorine, occurred in relatively high concentrations at times (see Figs. 2-5 and 2-6), but no obvious changes were observed in the benthic community in those months in which these high concentrations were observed (e.g., Figs. 6-3, 6-4, 6-7, 6-8). Brinkhurst and Cook (1974) indicated that oligochaetes are less tolerant than arthropods of heavy metals and of any chemicals that reduce bacterial activity; above NHP, relatively large numbers of oligochaetes were sometimes found. On the other hand, published data on the toxicity of chlorine to aquatic biota (Mattice and Zittel 1976) suggest that the high levels of total residual chlorine (TRC) in EFPC above NHP (see Fig. 2-6) may account, in part, for the depauperate benthic community at EFK 24.4. Although significant impacts were also evident at EFK 23.4 below NHP, TRC levels at the outfall of the pond are very low (A. J. Stewart, unpublished data). Although episodic increases in chlorine levels undoubtedly affect the benthos at EFK 24.4, they cannot account for the low abundance and diversity of the benthos at EFK 23.4.

Table 6-4. Average metal concentrations (range in parentheses) and the percentage of mayflies (Ephemeroptera) in the total benthos population in East Fork Poplar Creek below New Hope Pond (EFK 23.4) and upstream control sites on three streams where studies were conducted of benthic invertebrate responses to metal contamination

Site	Average total metal concentration in water (mg/L)							% Mayflies	Reference
	N[a]	Cu	Cd	Cr	Zn	Pb	N		
Shayler Run, Ohio (control)	29	9.0 (5–24)	NS[b]	NS	NS	NS	14	42.0	Winner et al. (1980)
(downstream recovery zone)	8	23 (13–41)	NS	NS	NS	NS	14	36.0	
Clinch River, Va.	23	30.1	4.0	12	39	14	33	31.3	Van Hassel and Gaulke (1986)
Adair Run, Va.	35	7 (1–20)	4 (1–80)	13 (1–80)	19 (1–79)	NS	51	~50	Specht et al. (1984)
East Fork Poplar Creek, Tenn.	56[c]	14 (4–190)	2 (2–14)	10 (10–20)	40 (20–70)	10 (10–10)	60	0.1	This study

[a]N = total number of samples collected.
[b]NS = not sampled.
[c]June 1985–June 1986.

Flow and temperature are considered to be two of the most important factors controlling benthic invertebrates in flowing waters (Hynes 1970b; Ward and Stanford 1979; Sweeney 1984; Wiederholm 1984). Numerous studies have demonstrated that modification of these physical factors can adversely affect the benthos in a variety of ways (e.g., Ward and Stanford 1979; Wiederholm 1984). Not only can extreme alteration of stream flow and temperature affect the benthos, but more subtle changes, such as reductions in natural daily and seasonal fluctuations, can also have adverse effects. Like many pollutants, alterations in flow and temperature typically result in a reduction in benthic species richness, while total numbers and biomass may either increase or decrease. Although seasonal fluctuations in stream flow occurred in EFPC at the outfall of NHP in 1985/1986, a minimum discharge was usually maintained (Fig. 2-3; Sect. 2.1). Wide fluctuations occurred in the mean weekly discharge of EFPC at EFK 5.3 and BF near BFK 10.1 in 1985/1986 (see Fig. 2-4); however, the variance of the mean for BF is almost twice as great as that for EFPC. Similarly, temperatures were higher and considerably less variable in EFPC than in BF, especially below NHP (see Fig. 2-7). These data suggest that both the natural flow and temperature regimes in EFPC have been altered, which could be affecting the benthos.

A benthic community under enriched conditions is typically characterized by low species richness and sometimes high numbers of a few taxa (Hynes 1960; Wiederholm 1984). The impact on the community is usually attributed to a reduction in dissolved oxygen. However, Hynes (1960) described a situation where nutrient enrichment does not result in deoxygenation because turbulent flow allows good gas exchange. Under these conditions, certain groups are still able to thrive while many taxa typically associated with eroding substrata, such as mayflies, are absent or sparse. Further downstream from the source of enrichment, species richness increases and filter-feeders, such as *Hydropsyche* and Simuliidae, may occur in great numbers (Wiederholm 1984).

This scenario is similar to the conditions that exist in EFPC below the outfall of NHP, where species richness was low and density was high, particularly within the *Cricotopus/Orthocladius* group. Further downstream at EFK 13.8, species richness increased and large numbers of *Hydropsyche* were found. Data on nutrient levels in upper EFPC are limited (see Table 2-2) but show occasionally high levels of nitrogen at the outfall of NHP (see Fig. 2-5). However, the structure of the benthic community strongly

suggests that nutrient enrichment may be one of the primary sources of perturbation in EFPC. Additional data are currently being collected on nutrient levels and dynamics in EFPC.

The rate of recovery of an affected benthic macroinvertebrate community is dependent on the duration, severity, and type of perturbation; dilution; and the physical and chemical characteristics of the receiving body of water (Hynes 1960; Wiederholm 1984). A source of macroinvertebrates to recolonize affected areas is also important. The major source of recolonization is usually drift, although aerial and upstream migration may also occur (Williams and Hynes 1976; Gore 1985). The long-term nature of the stress to the benthos in EFPC may result in a very slow rate of recovery because any pollution-sensitive organisms would have been eliminated during the early years of operation of the Y-12 Plant. In the upper reaches of EFPC, the rate of recovery will be slowed even further because no major tributaries exist to provide sources of organisms for recolonization. Therefore, the rate of recolonization will greatly depend on aerial migration.

6.1.5 Summary

Quantitative benthos samples were collected at monthly intervals from six sites on EFPC for 1 year. Additionally, quantitative samples were taken at monthly intervals for 5 months from a single reference site in BF.

Results of the first year showed that the benthic macroinvertebrate community in the upper reaches of EFPC is severely affected, while gradually recovering downstream, at least to the middle reaches of the stream; an additional impact related to discharges from the ORWTF is suggested. The general pattern of recovery is indicated by increases in density, biomass, and species richness. Although the source of the impact in upper EFPC is clearly related to operations at the Y-12 Plant, the contribution of the plant to the impact on the benthic community downstream of Bear Creek Road is not clear because of agricultural practices and increases in urbanization along the course of the stream.

Unlike the source of impact, the cause of impact is not clear, but a pattern of upstream nutrient enrichment is evident. Elevated levels of a wide variety of compounds occasionally occurred, and stream flow and temperature regimes have been modified, thus adding to the impact on the benthos.

The rate at which the benthos in EFPC will recover will depend on the correct identification of the causative factors. Because the Y-12 Plant is located in the headwaters of EFPC, recovery relative to streams having unaffected upper reaches may be considerably slower because of the loss of an upstream source of organisms for recolonization.

6.1.6 Future Studies

Sampling of the benthos during the second year will be maintained on a monthly schedule. Because a good baseline data set will be obtained from the initial two years of monthly sampling, the sampling frequency will be reduced to quarterly intervals in the third year (1987/1988). Although the use of a single reference stream provides some information on what might be expected in terms of community structure and composition if EFPC were unaffected, use of an additional reference site would provide a better basis for comparison. Therefore, quarterly sampling of Hinds Creek, a stream similar in size to EFPC, has been initiated. This stream is a tributary of the Clinch River and is located ~8 km southeast of Norris, Tennessee.

Additional studies will be initiated in the second year to help identify the causes of impact to EFPC, possibly including in situ toxicity testing using a relatively sensitive invertebrate species; additional nutrient measurements; and analyses of food availability (e.g., concentrations of particulate organic matter in the seston) and quality (e.g., carbon to nitrogen ratios of periphyton and seston). Adult stages of aquatic organisms will be collected to determine if attempts are being made at recolonization. The latter study could provide insight on recolonization rates and on potential toxic effects as evidenced by the absence of aquatic stages from sites at which their adult (terrestrial) stage occurs. Efforts will also be made to identify more-sensitive methods of data analysis such as similarity indices, functional feeding group analyses, and/or estimates of community secondary production.

6.2 FISHES

6.2.1 Introduction

Use of fishes to assess the ecological effects of water quality and/or habitat degradation offers several advantages over other indicators of environmental quality (see

review by Karr et al. 1986), and several of these advantges are especially relevant to assessment of the biotic integrity of EFPC. Fish communities, for example, are made up of species that represent several trophic levels, and species that comprise the potential sport fishery in EFPC (e.g., bluegill, redbreast sunfish, rock bass) are at or near the ends of food chains. Consequently, they integrate the direct effects of water quality and habitat degradation on lower trophic levels [i.e., 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), an especially important consideration in EFPC where fishing is currently prohibited primarily because of elevated levels of mercury in several species (Sect. 4.1.3.1).

The initial objectives of the in-stream fish monitoring task were to (1) characterize spatial and temporal patterns in the distribution and abundance of fishes in EFPC and (2) document any effects on fish community structure and function resulting from implementation of the Water Pollution Control Program at the Y-12 Plant (Sect. 1).

6.2.2 Methods

6.2.2.1 Population surveys

Quantitative sampling of the fish populations at six sites on EFPC (see Fig. 2-1) and Brushy Fork (see Fig. 2.2) was conducted by electroshocking to estimate population size (densities in numbers and biomass per unit area). Sampling reaches ranged from 100 to 124 m in mean length at all sites except EFK 24.4, where a much longer section (225 m) was needed to adequately determine that no fish were present. Fish sampling sites either overlapped or were within 100 m of the benthic invertebrate sites with one exception: the stream near benthos site EFK 10.6 was too deep to sample by electroshocking, so the fish study site was moved ~600 m downstream. Lengths of the sampling reaches were determined by (1) the presence of at least two riffle-pool sequences, if possible, and (2) location of suitable places for anchoring upstream and downstream block nets (seines).

Sampling was conducted four times during the first year: May/June 1985, October/November 1985, March 1986, and May/June 1986. Additional sampling was conducted in January 1986 at EFK 23.4 and EFK 18.2 because of the substantial changes in species composition and abundance that were observed between May/June and October 1985 at these sites. Qualitative sampling was conducted in May/June 1985 and January 1986 at EFK 23.4, EFK 18.2, and EFK 6.3; in October/November 1985 and January 1986 at EFK 13.8 and BFK 7.6; and in June 1985 at EFK 10.0. NHP was also sampled qualitatively in July 1985, March 1986, and September 1986.

Field sampling procedures

All stream sampling was conducted using two Smith-Root Model 15A backpack electrofishers. This unit utilizes a self-contained, gasoline-powered generator capable of delivering up to 1200 V of pulsed direct current. A pulse frequency of 120 Hz was used at all times, 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. A Smith-Root Type IV electrofisher, which delivers a pulsed direct current through a single boom, was used to sample NHP in March 1986. On the two other sampling dates, the Model 15A electrofisher was used from a johnboat.

After 0.64-cm-mesh seines were stretched across the upper and lower boundaries of the reach to restrict fish movement, a four- or five-person sampling team made three consecutive passes through the study reach in an upstream direction. Fish were processed after each pass to allow sufficient time for the water to clear before another pass was initiated, unless turbidity remained low and passes could be made consecutively. Stunned fish were netted and transferred to open-top, 0.64-cm-mesh holding cages distributed throughout the reach. Smaller fish were placed in plastic buckets containing several rows of very small holes that allowed water circulation yet prevented escape.

Fish were anesthetized with MS-222 (trican methane sulfonate), measured to the nearest 0.1 cm (total length), and weighed on a Pesola spring balance (Bleitz Wildlife Foundation). Fish weighing <50 g were weighed to the nearest 0.1 g, and fish weighing >50 g were weighed to the nearest gram. Representative individuals of those species that

could not be positively identified in the field were preserved in 10% formalin and returned to the laboratory for identification. All other fishes were returned alive to the stream. Supplemental information collected at the time of fish sampling included cloud cover, specific conductance, water temperature, dissolved oxygen, turbidity, and the time at the start and end of each of the three collection passes. The number of seconds that current was being delivered to the water was also recorded at the beginning and end of each pass. Finally, measurements were made of the total length of the reach, the width across transects located at 5-m intervals, and the depth at three, approximately equal intervals along each transect.

Data analyses

Population estimates were calculated following Carle and Strub (1978). Numerical estimates were converted to biomass by multiplying the population estimate by the mean weight per individual. Weights of unweighed individuals were estimated from $\log_{10}$ weight vs $\log_{10}$ length regressions, using the PROC GLM (general linear models) procedure (SAS 1985b) to obtain individual regression parameters (slope and y-intercept) for each site/date combination.

Condition factors (K) were calculated for each weighed and measured fish by site, date, and species as

$$K = 100 \text{ (weight/length}^3\text{)} \ , \qquad (6\text{-}2)$$

where weight is in grams and total length is in centimeters. Fish with a low K because they had been tagged or had lost a tag were not included in the computation of mean K. Very small fish (<2.5 cm) were also excluded because of the error in weighing. Comparisons of condition factors between sites were made using PROC GLM on untransformed data because the condition factors exhibited homogeneity of variance as estimated with the UNIVARIATE procedure (SAS 1985b). If the GLM procedure indicated significant differences in mean condition between groups, the Tukey option was performed to identify those sites that were significantly different.

6.2.2.2 Age determination

Scales were taken for age determination from the target species, redbreast sunfish (*Lepomis auritus*), bluegill (*L. macrochirus*), and rock bass (*Ambloplites rupestris*) collected during routine population surveys. However, because of the low densities of bluegill and rock bass at most sites in EFPC (Sect. 6.2.3), only data on redbreast sunfish are included in this report.

Scales were taken from an area above the lateral line and slightly anterior to the insertion of the dorsal fin. Impressions of the scales were made using a Wildco scale press and acetate slides; those that produced poor impressions (because of attached mucous or skin) were mounted between two glass microscope slides that were taped together. Attempts to improve the impressions by treatment with KOH were not successful, so slides were used without cleaning. Enlarged images of the scales were projected on a screen, using an Eberbach 2700 slide projector with a 16-mm lens. Where possible, at least 10 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 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; for future analyses, all age determinations will be independently cross-checked.

For each scale read, the following data were recorded: 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. Each unit of measurement represented 0.12 mm of actual object length.

6.2.2.3 Fish movement/growth studies

Studies of the pattern of movement of fishes in EFPC were conducted to aid in the interpretation of data collected in Subtask 2a (Accumulation of Contaminants by Biota in EFPC, Sect. 4.1) and Subtask 4b (Instream Monitoring of Fish Populations in EFPC,

Sect. 6.2) of the BMAP (Loar et al. 1989). Between-site comparisons of contaminant levels in fish are meaningful only if movement (and thus exposure) is limited to the vicinity of the site of collection. Knowledge of fish movement is also useful in the interpretation of seasonal changes in fish population abundance.

Four sampling reaches on EFPC were selected for study based on target species abundance, suitability for sampling by electroshocking, and accessibility. With the stream kilometer designation indicating the downstream boundary of the reach, these sites included: EFK 22.7, upstream 950 m to NHP; EFK 17.9, upstream 300 m to the bridge at stoplight 13 on the Oak Ridge Turnpike (Route 95) and continuing an additional 700 m above the bridge; EFK 13.4, upstream 700 m from the outfall of the ORWTF; and EFK 4.7, upstream for 600 m to the U.S. Geological Survey (USGS) gaging station and continuing for an additional 1000 m above the gage. Sampling in EFPC was conducted quarterly from July 1985 through April 1986. Sampling was also conducted in July 1986, but only previously marked fish were collected.

Movement studies in BF were initiated in October 1986 and sampling was conducted over an ~1400-m reach having an upper limit that coincided with the downstream boundary of the fish population reach (BFK 7.6). The reach consisted of an 800-m lower section and a 600-m upper section; the sections were separated by ~500 m of private property with no access. Data collected on fish movements in BF during the first year were qualitative. In October 1986, the reach was surveyed to identify specific stream locations and obtain quantitative data on movements. Sampling in BF was conducted on the same schedule as that in EFPC.

Movement patterns were evaluated by recapturing individually tagged fishes released at known locations in EFPC. Individual fishes were identified by color-coded and numbered Floy fingerling tags sewn through the epaxial musculature at the anterior base of the dorsal fin; all species of centrarchids (sunfishes and basses) and carp were tagged. Fishes were collected using backpack electrofishers, anesthetized with MS 222, weighed, measured, tagged (if not already bearing a tag) or the tag number recorded, and released within 25 m of the location of capture. Stakes were located every 100 m and flagging every 50 m to indicate the stream kilometers so that precise stream locations could be recorded.

The measurements of length and weight obtained for marked and recaptured fish in the movement studies were also used to estimate growth of individual fishes over discrete time intervals. Growth was calculated from changes in weight of fish recaptured at 3-month intervals. Preliminary analyses were based on one species (redbreast sunfish). Initial sample sizes were too small (i.e., recapture rates were too low) for statistical comparisons of mean growth rates between sites.

6.2.3 Results

6.2.3.1 Species composition and richness

A list of the fish species collected from each of the five sites on EFPC and from BFK 7.6 is given in Appendix D. Thirty species were identified from EFPC during the first year of study, and only two of these species were not collected at the lowermost site on EFPC (EFK 6.3). The two species were the Tennessee (*Phoxinus tennesseensis*) and the stripetail darter (*Etheostoma kennicotti*), both of which were found in very low densities at EFK 10.0 only in March 1986. These species are abundant in the middle and lower reaches, respectively, of Bear Creek and also inhabit Mill Branch, both tributaries of EFPC (Loar et al. 1985). Five species were collected only from EFK 6.3 and two of these, the channel catfish (*Ictalurus punctatus*) and smallmouth buffalo (*Ictiobus bubalus*), are primarily reservoir fishes; their occurrence at EFK 6.3 is indicative of the close proximity of the site to the backwater area of Watts Bar Reservoir. No species was collected from the three upper sites (EFK 23.4, EFK 18.2, and EFK 13.8) that was not also encountered at either EFK 10.0 or EFK 6.3.

Much less intensive sampling was conducted in BF (3 vs 22 site/date combinations in BF and EFPC, respectively), yet 32 species were identified from this reference site. Nine of these species have not been collected to date from EFPC. Of the five species collected at EFK 6.3 and not BF, three are reservoir species and, consequently, unique to the site because of its closer proximity to Watts Bar Reservoir than BF [almost 21 km separates the confluences of EFPC and BF with Poplar Creek (Table 2-1 and Fig. 2.2)]. Finally, a greater proportion of the fish community at EFK 6.3 consisted of species not routinely collected in the quantitative population surveys; qualitative sampling alone accounted for 29% of the species collected at EFK 6.3 compared with only 10% at BFK 7.6.

A downstream gradient of increasing species richness was observed in EFPC (Appendix D). Insignificant changes in species composition and richness occurred over the first 5.2 km of EFPC below NHP. Over the next 4.4 km, richness increased from 12 to 16 species at EFK 13.8 and to 23 species at EFK 10.0. The increase in the number of species between EFK 13.8 and EFK 10.0 (44%) was greater than that between any other adjacent sites, even though a significant municipal discharge is located between the sites (Sect. 2.1).

The richness and abundance of species that are intolerant of water quality/habitat degradation (Karr 1981) also increased with increasing distance below NHP (Table 6-5). No intolerant species were collected in EFPC immediately below NHP (EFK 23.4) or at EFK 18.2, but the number of intolerant species increased substantially in the middle reaches of the stream. Intolerant species richness increased from three species at EFK 13.8 (banded sculpin, rock bass, and spotted sucker) to eight species at EFK 10.0 (same three plus northern hog sucker, black redhorse, Tennessee dace, and two darters); no additional species were encountered at EFK 6.3.

6.2.3.2 Density and biomass

Total densities and biomass of the fish communities in EFPC and BF for each of five sampling periods between May 1985 and June 1986 are presented in Table 6-6. Data on species densities and biomass, by site and sampling period, are given in Appendix E, Tables E-1 through E-6. No fish were collected at EFK 24.4 (Table E-1), and no fish were found in NHP in July 1985 and March 1986. However, several large creek chubs (*Semotilus atromaculatus*) were collected near the lower end of the pond in September 1986.

Spatial distribution and abundance in EFPC

A consistent pattern in the longitudinal distribution of fish densities from below NHP to the lower reaches of EFPC was observed. Total density increased from EFK 23.4 to a maximum at EFK 18.2; densities declined from EFK 18.2 to EFK 13.8 but increased again at EFK 10.0, although the peak was generally smaller than that observed at EFK 18.2. Abundance at the lower site on EFPC (EFK 6.3) was consistently low. With one significant exception, this pattern in the distribution of fish abundance in EFPC was

Table 6-5. Mean densities (No./10 m^2) and biomass (g/10 m^2 in parentheses) of fish species in East Fork Poplar Creek and Brushy Fork that are classified by Karr et al. (1986) as intolerant of water quality and/or habitat degradation

Species	Sites[a]			
	EFK 13.8	EFK 10.0	EFK 6.3	BFK 7.6
Black redhorse (*Moxostoma duquesnei*)		<0.01(2.37)	0.01(9.65)	0.14(38.32)
Northern hog sucker (*Hypentelium nigricans*)		0.04(3.31)	<0.01(0.71)	0.14(9.34)
Spotted sucker (*Minytrema melanops*)	<0.01(0.11)	0.01(0.14)	0.04(3.32)	0.06(15.73)
Tennessee dace[b] (*Phoxinus tennessensis*)		<0.01(<0.01)		
Rock bass (*Ambloplites rupestris*)	0.03(1.15)	0.06(4.22)	0.02(0.40)	0.21(17.32)
Banded sculpin[c] (*Cottus carolinae*)	<0.01(0.03)	0.98(1.81)	0.04(0.16)	1.99(8.23)
Greenside darter[d] (*Etheostoma blennioides*)				<0.01(0.13)
Blueside darter[d] (*E. jessiae*)				0.07(0.09)
Stripetail darter[d] (*E. kennicotti*)		<0.01(<0.01)		
Snubnose darter[d,e] (*E. simoterum*)		0.01(0.02)	0.02(0.02)	1.11(1.05)
Total	0.04(1.29)	1.12(11.88)	0.13(14.26)	3.72(90.21)
Percentage intolerant	2.2(2.7)	28.0(26.1)	13.0(47.8)	56.4(57.9)

[a]No intolerant species were collected at EFK 23.4 and EFK 18.2.

[b]Southern redbelly dace (*Phoxinus erythrogaster*) listed by Karr et al. (1986).

[c]Mottled scuplin (*Cottus bairdi*) listed by Karr et al. (1986).

[d]Although not listed, darters (subfamily Etheostomatinae) are included because of their general sensitivity to environmental degradation (Karr et al. 1986).

[e]The black darter (*Etheostoma duryi*) was also identified in a preserved sample from Brushy Fork (M. G. Ryon, ORNL-ESD personnel communication, 1986). The two darters were not separated in the field, so the tabular value represents the combined abundance of the two species.

Table 6-6. Density (D), biomass (B), and species richness (S) of the fish communities in East Fork Poplar Creek and Brushy Fork

Sampling period	Sampling sites					
	BFK 7.6	EFK 23.4	EFK 18.2	EFK 13.8	EFK 10.0	EFK 6.3
May/June 1985						
D[a]	*b*	0.10	0.80	0.22	0.45	0.04
B[c]	*b*	9.30[d]	1.75	28.58[c]	5.91	1.60
S[e]	*b*	5	11	10	9	6
Oct/Nov 1985						
D	0.91	0.78	0.58	0.16	0.51	0.20
B	17.83	10.13	0.64	2.23	3.83	8.05[d]
S	21	6	5	7	11	16
Jan 1986						
D	*b*	0.17	0.50	*b*	*b*	*b*
B	*b*	2.89	0.29	*b*	*b*	*b*
S	*b*	5	4	*b*	*b*	*b*
March 1986	0.51	0.50	0.28	0.12	0.21	0.06
D	0.24	8.23	0.21	6.84[d]	3.13	4.58
B	16	8	4	8	15	10
S						
May/June 1986						
D	0.55	0.17	1.26	0.22	0.41	0.08
B	19.63	4.67	1.73	5.12	5.33	5.86[d]
S	23	6	8	10	14	11
Mean						
D	0.66	0.34	0.68	0.18	0.40	0.10
B (all species	15.57	7.04	0.92	10.69	4.55	5.02
included)	15.57	6.18	0.92	4.70	4.55	2.98
B (excluding carp)	20	6	6	9	12	9
S						

[a]D expressed in No./m^2.

[b]Not samples.

[c]B expressed in g/m^2.

[d]Carp were a major component of the biomass, contributing 4.33 g/m^2 (47%) at EFK 23.4; 20.62 (72%) and 3.37 g/m^2 (49%) at EFK 13.8 in May 1985 and March 1986, respectively; and 4.58 (57%) and 3.61 g/m^2 (62%) at EFK 6.3 in October 1985 and June 1986, respectively.

[e]S expressed as No. species collected.

repeated on each of the four major sampling dates (late spring and fall 1985; late winter and spring 1986). The highest densities in EFPC were not observed at EFK 18.2 in the fall and late winter because of a major fish kill that occurred in July 1985 near this region of the creek (see Sect. 6.2.3.5). By the following June, however, densities of the dominant species at this site (stoneroller, white sucker, and blacknose dace) had recovered and indeed exceeded the pre-kill densities of June 1985 (Table E-2).

The longitudinal distribution of mean total fish biomass (excluding carp)[3] in EFPC exhibited a pattern that was just the opposite of that found for density (Table 6-6). Sites where densities were highest (EFK 18.2 and EFK 10.0) had the lowest biomass per unit area; the communities at these sites consisted of species populations with high densities of relatively small individuals (Tables E-2 and E-4, respectively). At EFK 18.2, the stoneroller, white sucker, blacknose dace, and creek chub populations consisted of very small individuals (<5 cm total length) in late spring, suggesting that this site may be a major spawning and rearing area in EFPC. The high mean total biomass at EFK 23.4, on the other hand, was the result of relatively high densities of redbreast and bluegill sunfish populations made up mostly of older age classes (Table E-1 and Sect. 6.2.3.3). Finally, the absence of any consistent pattern in total fish biomass between EFK 13.8, EFK 10.0, and EFK 6.3 may have been caused by the sporadic occurrence of large catostomids and freshwater drum at these sites (Tables E-3 through E-5). The absence of these species and the low densities of other species, as well as differences in capture efficiency among sites, could explain why biomass at EFK 6.3 was generally the lowest of all the EFPC sites in late spring.

Although densities varied considerably among sites, several species (e.g., redbreast sunfish, stoneroller, striped shiner) occurred throughout EFPC, whereas others had a much more restricted distribution. For example, mean densities (No. of individuals/10 m^2) of the banded sculpin were <0.01, 0.98, and 0.04 at EFK 13.8, EFK 10.0, and EFK 6.3, respectively. A similar distribution pattern was evident for the northern hog sucker (0, 0.04, and <0.01) and, to a lesser extent, the rock bass (0.03, 0.06, and 0.02) at the three

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[3]Carp were numerically insignificant in EFPC (total of ten collected but nine of these ococurred at just two sites: EFK 13.8 and EFK 6.3) yet dominated the estimates of biomass per unit area because all the individuals were large (range 1149 to 3982 g; see Tables E-3 and E-5).

sites, respectively. In addition to the occurrence of maximum densities at the intermediate site (EFK 10.0), densities of two of the three species were markedly lower at both of the adjacent sites.

Comparisons between EFPC and BF

The fish community in BF was diverse and abundant; densities and biomass were consistently higher in BF than at any site in EFPC (Table 6-6). Both mean total density and mean biomass in BF were almost twice that observed at EFK 23.4. The most abundant species in BF included the banded sculpin, snubnose/black darter, stoneroller, and striped shiner (Table E-6). Redbreast sunfish densities were generally similar to those in EFPC except below NHP, where the density of both redbreast and bluegill sunfish was almost three times that in BF. Mean densities of rock bass and bluegill, however, were 3 to 10 times higher in BF than in EFPC below EFK 18.2. The consistently high biomass in BF was primarily the result of relatively high densities of large catostomids and rock bass; such high densities were never approached in EFPC. Another significant attribute of the BF fish community was a high richness, which ranged from 16 to 23 species during the three sampling periods (no sampling was conducted in late spring 1985). Mean richness was generally 2 to 3 times higher in BF than EFPC; only site EFK 10.0 in late winter and late spring 1986 was similar in richness to BF (Table 6-6).

Temporal patterns

With the exception of site EFK 18.2, which was affected in July 1985 by the release of an unknown toxicant downstream of the ORR boundary at EFK 22.7, maximum fish densities in EFPC and BF generally occurred in the fall. This peak in abundance was primarily caused by (1) recruitment of age 0 fish, especially stonerollers and striped shiners, and (2) movement of large numbers of creek chubs and probably striped shiners into the area below NHP (Table E-1). No distinctive peak in fish density was observed at EFK 13.8; the recruitment of age 0 fish was balanced by a sharp decrease in the abundance of adult redbreast sunfish (Table E-3).

Maximum biomass, on the other hand, occurred in May/June at sites EFK 18.2, EFK 13.8, EFK 10.0, and in BF. At sites EFK 23.4 and EFK 6.3, however, late spring was the period of lowest biomass. The contrasting seasonal biomass patterns at these two

groups of sites may be the result of much greater fish movement at the lower and upper sites on EFPC. At EFK 6.3, for example, several large species (e.g., gizzard shad, carp, freshwater drum) accounted for the high biomass in late October (Table E-5). These species are common inhabitants of reservoirs and their movement into lower EFPC reflects the proximity of EFK 6.3 to Watts Bar Reservoir backwaters. The black redhorse may be another migrant; it was only collected at this site in March, yet accounted for 84% of the total fish biomass.

The most significant seasonal shifts in species composition were observed in upper EFPC at EFK 23.4 (Fig. 6-10). Although sunfish populations in the 116-m study reach remained relatively stable over time, the populations of noncentrarchids, especially the creek chub, striped shiner, and blacknose dace, fluctuated by as much as two orders of magnitude. Decreases in abundance were similar in magnitude to increases. Striped shiner numbers, for example, dropped from a peak of 181 individuals in October to zero in January. None of the other sites exhibited such extreme fluctuations in population numbers.

6.2.3.3 Growth and condition

Interpretation of scale annuli was used to estimate fish age and provided the basis for graphical representation of the population growth rate of redbreast sunfish. The population growth rate, or apparent growth rate, is a comparison of the mean size of surviving fish at successive ages (Ricker 1975). When there is size-selective mortality within a year-class (e.g., the larger fish in a year-class often have a greater mortality rate then the smaller individuals), the population growth rate will be different (usually lower) from the true mean growth rate of the fish (Ricker 1975). The growth rate comparisons that follow are based on the assumption of similar biases among sites.

The population growth rates of redbreast sunfish sampled in the fall 1985 and spring 1986 are shown in Figs 6-11 and 6-12, respectively. Although the curves indicate similar rates of growth between the population in Brushy Fork and those in EFPC, such a conclusion must be considered preliminary because of small sample sizes for many age-classes. The mean weights of a given age-class usually differed by a factor of 2 or less between sites; the highest variability was associated with the 1982 year class in fall 1985 (Age 3+ fish; Fig. 6-11) and spring 1986 (Age 4+ fish; Fig. 6-12). In general, the mean

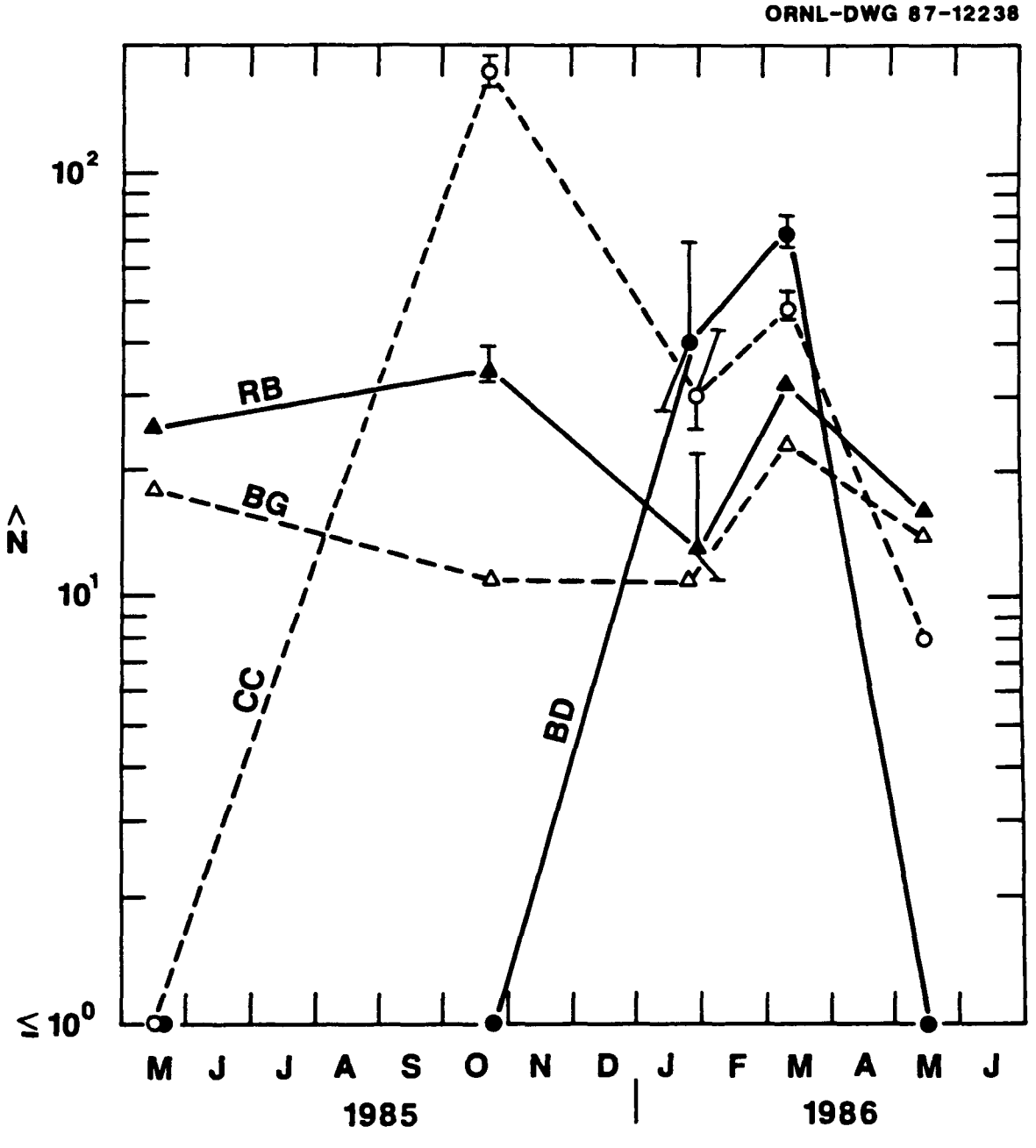

Fig. 6-10. Population size ($\hat{N}$) of four fish species in a 116-m reach of East Fork Poplar Creek (site EFK 23.4) ~150 m below the outfall of New Hope Pond, May 1985–May 1986. Vertical bars represent the 95% confidence interval of N. RB = redbreast sunfish; BG = bluegill; CC = creek chub; BD = blacknose dace.

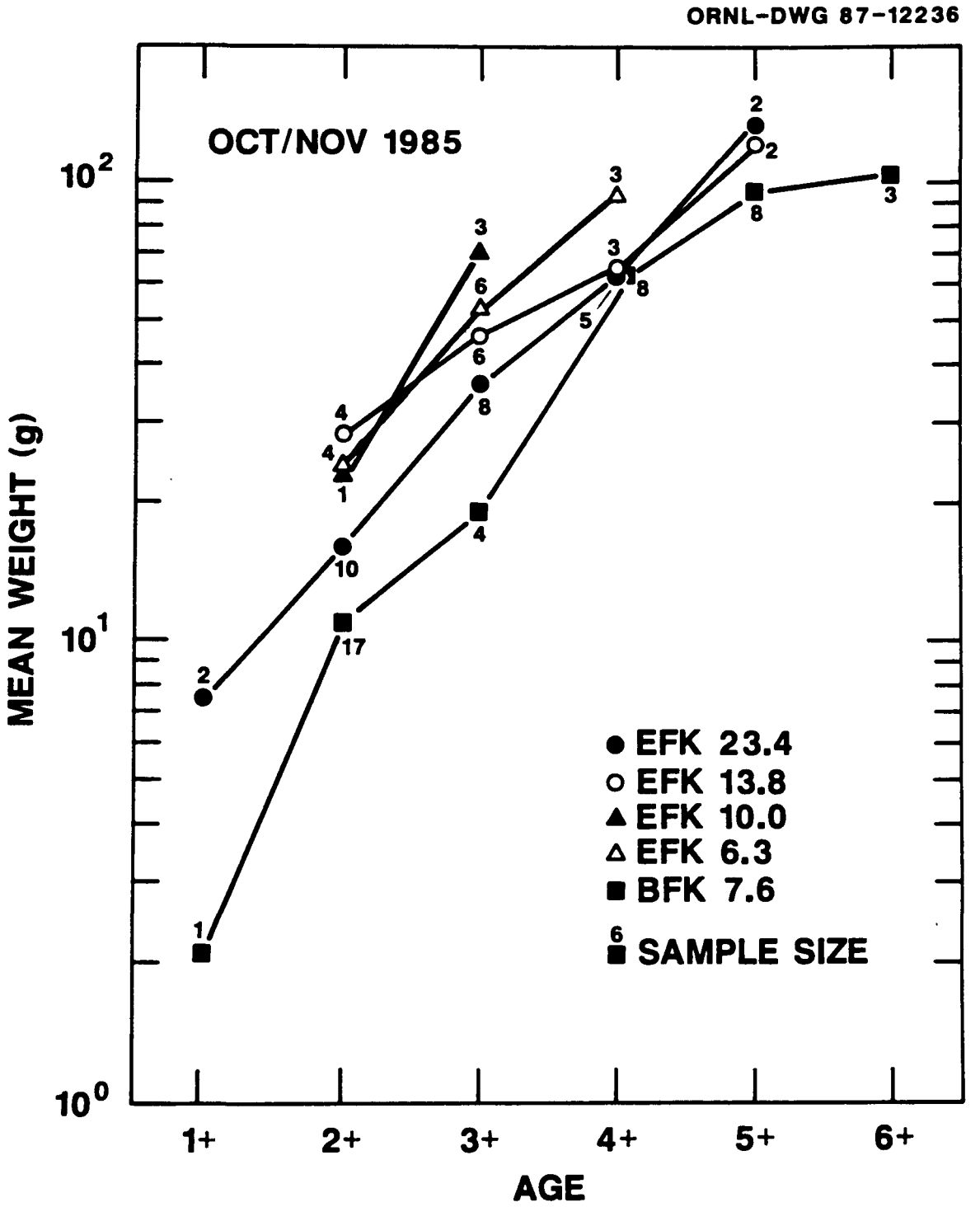

Fig. 6-11. Mean weight at age of redbreast sunfish from four sites on East Fork Poplar Creek and Brushy Fork, the reference stream, fall 1985. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

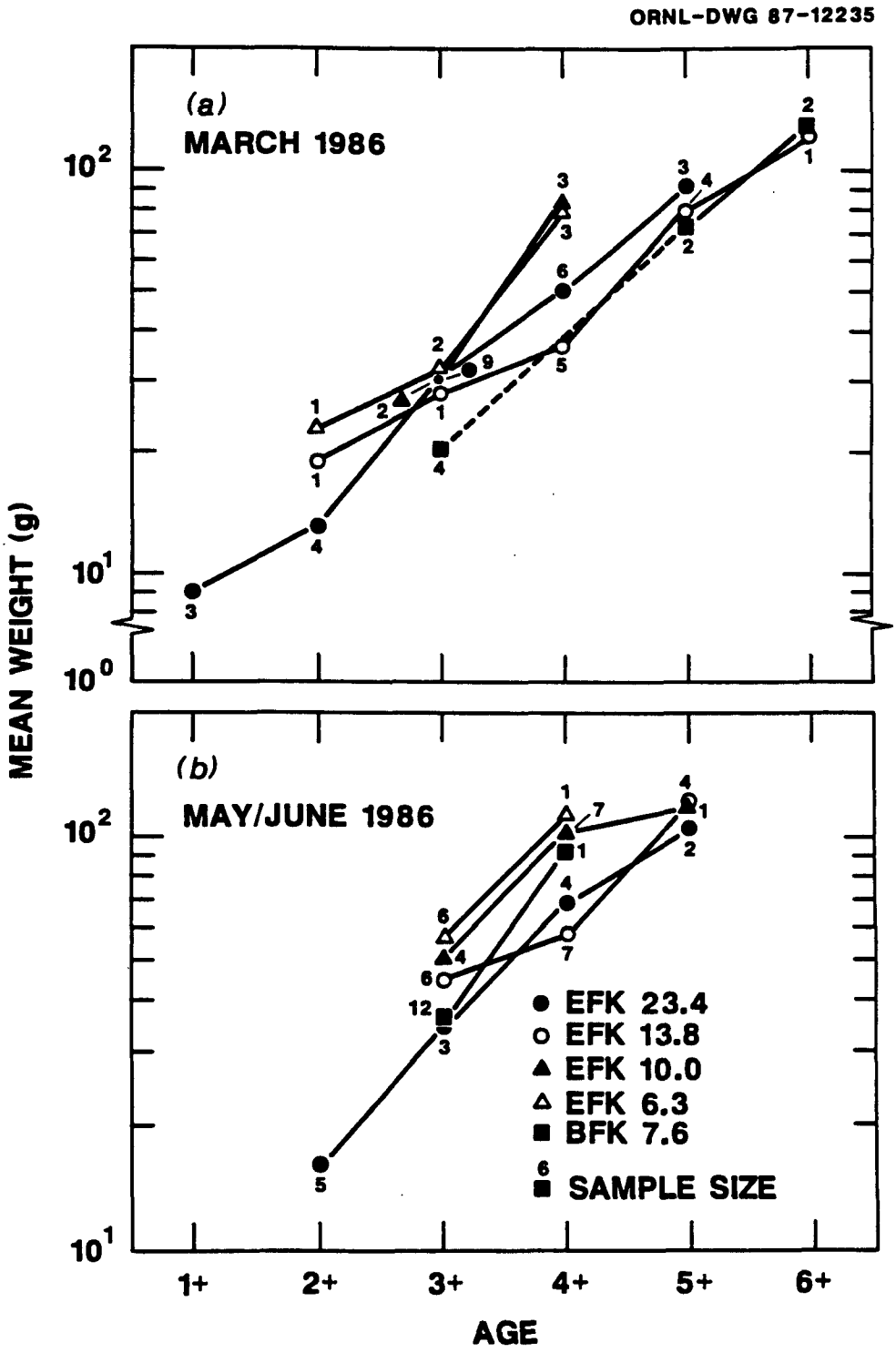

Fig. 6-12. Mean weight at age of redbreast sunfish from four sites in East Fork Poplar Creek and Brushy Fork, the reference stream, winter and spring 1986. EFK = East Fork kilometer; BFK = Brushy Fork kilometer.

weight at age of redbreast sunfish was highest at the downstream sites in EFPC (EFK 10.0 and EFK 6.3) and lowest in BF, possibly because of the effects of the lower water temperatures (Appendix A) and a later spawning period in BF. Such trends in mean weight across sites paralleled the trend in condition factors, as discussed in the following paragraphs.

Because heavier fish of a given length are hypothesized to be in better condition (Bagenal and Tesch 1978), the mean condition factor (K) was used to compare the relative well-being of fish populations at different sites and seasons. Mean condition factors of fish collected from various sites in EFPC and BF in May/June were not significantly different at $\alpha = 0.05$ (Table 6-7), although values of K tended to be higher at downstream sites. Similar results were obtained in January and March 1986 for all three species, but in October/November 1985, the mean K of both redbreast and bluegill sunfish was significantly higher at EFK 23.4 compared with BF and some downstream sites (Table 6-8).

Several other species also had a significantly higher mean K in the fall 1985 at EFK 23.4 compared with other EFPC sites and BF (Table 6-9). Very high population densities of the creek chub and striped shiner were observed at EFK 23.4 at this same time (Table E-1), suggesting that substantial immigration to the upper reaches of EFPC had occurred between May and October 1985. Consequently, the high K of these species may reflect habitat conditions in another region of EFPC prior to emigration.

Comparisons were also made of the mean K of centrarchids between sampling periods at each site, and several significant differences were observed (Table 6-10). At most sites, condition was generally highest in late spring (May/June) and coincided with enlargement of the gonads prior to spawning and with increased growth at that time. Condition was lowest in mid/late winter (January/early March) when growth was presumably low. The condition of other species showed similar seasonal patterns, several of which were statistically significant. For example, the mean K of the striped shiner was highest in late spring and generally lowest in the fall, whereas the mean K of the stoneroller at all sites was highest in early March (but was significantly higher at only three of the sites), coinciding with the earlier maturation of the gonads prior to spawning in March and April.

Table 6-7. Comparisons between sites of mean condition factors (K) for three centrarchid species collected in May and June of 1982, 1985, and 1986

Date	Mean condition factor[a]					
	Redbreast sunfish					
May/June 1985	EFK 6.3 n = 17 (2.030)	EFK 13.8 n = 46 (2.016)	EFK 23.4 n = 23 (2.006)	EFK 10.0 n = 25 (2.002)	EFK 18.2 n = 9 (1.985)	
May/June 1986	EFK 10.0 n = 13 (2.048)	EFK 13.8 n = 17 (1.998)	EFK 6.3 n = 7 (1.933)	EFK 23.4 n = 13 (1.920)	BFK 7.6 n = 18 (1.888)	EFK 18.2 n = 18 (1.865)
	Bluegill					
May 1982[b]	EFK 13.8 n = 11 (2.215)	EFK 1.9 n = 11 (2.052)	EFK 23.6 n = 11 (1.850)	EFK 23.7 n = 7 (1.775)		
May/June 1985	EFK 23.4 n = 17 (2.097)	EFK 10.0 n = 3 (1.965)	EFK 18.2 n = 2 (1.937)	EFK 13.8 n = 4 (1.900)		
May/June 1986	BFK 7.6 n = 18 (1.850)	EFK 23.4 n = 13 (1.797)				
	Rock bass					
May/June 1986	EFK 13.8 n = 2 (1.892)	EFK 10.0 n = 5 (1.845)	BFK 7.6 n = 16 (1.809)			

[a]n = number of fish measured and weighed; sites with less than two individuals were excluded from the analysis. Values connected by the same line are not significantly different (α = 0.05), based on Tukey's studentized range (HSD) test.

[b]K was calculated from data in Van Winkle et al. (1984, Table A-1). The length data in Table A-1 were based on measurements of total length and not standard length as indicated in that table and Sect. 2.2.2 of the report (Elwood 1987).

Table 6-8. Comparisons between sites of mean condition factors (K) for three centrarchid species, October 1985–March 1986

Date	Mean condition factor[a]					
			Redbreast sunfish			
October/ November 1985	EFK 13.8 14 (2.078)	EFK 23.4 33 (2.003)	EFK 6.3 12 (1.823)	EFK 18.2 23 (1.767)	EFK 10.0 5 (1.718)	BFK 7.6 20 (1.670)
January 1986	EFK 23.4 10 (1.671)	EFK 18.2 8 (1.643)				
March 1986	EFK 6.3 6 (1.801)	EFK 13.8 12 (1.788)	EFK 10.0 5 (1.744)	EFK 23.4 31 (1.735)	EFK 18.2 11 (1.713)	BFK 7.6 11 (1.613)
			Bluegill			
October/ November 1985	EFK 23.4	EFK 10.0 8 (2.279)	BFK 7.6 2 (1.912)	4 (1.486)		
			Rock bass			
October/ November 1985	BFK 7.6 24 (1.763)	EFK 6.3 3 (1.644)	EFK 10.0 6 (1.642)			
March 1986	BFK 7.6 13 (1.769)	EFK 10.0 3 (1.732)				

[a]n = number of fish measured and weighed; sites with less than two individuals were excluded from the analyses. Values connected by the same line are not significantly different (α = 0.05), based on Tukey's standardized range (HSD) test.

Table 6-9. Ranking of mean condition factors (K) for seven fish species collected at EFK 23.4[a]

Species	May/June 1985	Oct/Nov 1985	March 1986	May/June 1986
Bluegill	1/4	1/3 (1)[b]	*c*	2/2
Redbreast sunfish	3/5	2/6 (3)	4/6	4/6
Blacknose dace	*d*	NC[e]	1/4	*d*
Creek chub	NC	1/3 (1)	*c*	2/5
Stoneroller	NC	1/5 (3)	2/5	NC
Striped shiner	1/4	1/6 (5)	1/6 (2)	3/6
White sucker	NC	1/2	2/3 (1)	2/5

[a]Numerator is the rank at EFK 23.4 (1 = highest) and denominator is the total number of sites included in the comparison. Sites with less than 2 individuals were excluded from the analysis.

[b]The number of sites with a significantly lower ($\alpha = 0.05$) mean K than EFK 23.4 is given in parentheses.

[c]$n < 2$ at all other sites.

[d]$n = 1$ at EFK 23.4.

[e]NC = species not collected at EFK 23.4

The finding of more statistically significant differences in mean K between sampling periods than between sites (Tables 6-10 and 6-7, respectively) is consistent withthe results of similar comparisons for the fish populations in White Oak Creek watershed (Loar et al. 1987).

6.2.3.4 Fish movement/growth studies

Results of movement studies

From July 1985 through April 1986, 3800 fishes were tagged and released in EFPC and BF (Table 6-11). Although most of the individuals tagged were redbreast (*L. auritus*) and bluegill sunfish (*L. macrochirus*), other centrarchids, including rock bass (*Ambloplites rupestris*), largemouth bass (*Micropterus salmoides*), warmouth (*L. gulosus*), green sunfish (*L. cyanellus*), and several carp (*Cyprinus carpio*) were marked also.

More than 250 tagged sunfish were recaptured over 3-month intervals during the first year of study. Seventy additional fish were recaptured over longer time intervals, but

Table 6-10. Comparisons between sampling periods of mean condition factors (K) for three centrarchid species

Site	Mean condition factors[a]				
			Redbreast sunfish		
BFK 7.6	Jun 1986 n = 18 (1.888)	Nov 1985 n = 20 (1.670)	Mar 1986 n = 11 (1.613)		
EFK 23.4	May 1985 n = 23 (2.006)	Oct 1985 n = 33 (2.003)	May 1986 n = 13 (1.920)	Mar 1986 n = 31 (1.735)	Jan 1986 n = 10 (1.671)
EFK 18.2	Jun 1985 n = 9 (1.985)	Jun 1986 n = 18 (1.865)	Oct 1985 n = 23 (1.767)	Mar 1986 n = 11 (1.713)	Jan 1986 n = 8 (1.643)
EFK 13.8	Oct 1985 n = 14 (2.078)	May 1985 n = 46 (2.016)	Jun 1986 n = 17 (1.998)	Mar 1986 n = 12 (1.788)	
EFK 10.0	Jun 1986 n = 13 (2.048)	Jun 1985 n = 25 (2.002)	Mar 1986 n = 5 (1.744)	Nov 1985 n = 5 (1.718)	
EFK 6.3	Jun 1985 n = 17 (2.030)	Jun 1986 n = 7 (1.933)	Oct 1985 n = 12 (1.823)	Mar 1986 n = 6 (1.801)	
			Bluegill		
BFK 7.6	Jun 1986 n = 18 (1.850)	Nov 1985 n = 4 (1.486)			
EFK 23.4	Oct 1985 n = 8 (2.278)	May 1985 n = 17 (2.097)	May 1986 n = 13 (1.797)	Jan 1986 n = 8 (1.749)	Mar 1986 n = 22 (1.667)
			Rock bass		
BFK 7.6	Jun 1986 n = 16 (1.809)	Mar 1986 n = 13 (1.769)	Nov 1985 n = 24 (1.763)		
EFK 10.0	Jun 1986 n = 5 (1.845)	Mar 1986 n = 3 (1.732)	Nov 1985 n = 6 (1.642)		

[a]n = number of fish measured and weighed. Values connected by the same line are not significantly different ($\alpha = 0.05$), based on Tukey's studentized range (HSD) test.

Table 6-11. Number of fishes tagged and released at four sampling sites in East Fork Poplar Creek and Brushy Fork, a reference stream, July 1985–April 1986[a]

Site	No. of fish				
	July 1985	Oct 1985	Jan 1986	Apr 1986	Total
EFK 22.7	771	463	179	191	1604
EFK 17.9	147	NS[b]	52	80	279
EFK 13.4	347	184	56	87	674
EFK 4.7	282	281	104	94	761
BFK 5.7	NS	261	90	131	482

[a]Sampling was conducted in July 1986, but no fish were tagged.
[b]NS = not sampled.

only the movements that occurred over the shorter intervals (3 months) are emphasized in this report. These data are considered preliminary because increased sample sizes resulting from additional studies during the second year may lead to different interpretations of fish movements in the two streams.

No significant differences ($\alpha = 0.05$) in patterns of movement were observed between any of the sunfish species, so data for all sunfishes were pooled for analysis. Of the sunfishes that were recaptured three months after tagging, 57% had moved less than 100 m; that is, they were recaptured either within the same 50-m reach in which they had been tagged 3 months earlier or in one of the two 50-m reaches immediately adjacent to the tagging section. An additional 16% of the recaptures over 3-month intervals were fishes that had moved between 100 and 200 m; thus, 73% of all recaptured sunfishes had remained within 200 m of the site of their previous capture for at least 3 months.

The pattern of movement varied seasonally, however (Table 6-12). From fall to winter, fishes seemed to be exceptionally sedentary. The proportion of tagged fish exhibiting no movement between captures was highest during this period, and no individuals moved more than 500 m. The greatest number of fishes moving over 500 m occurred from spring to summer. This seasonal component to the movement pattern was even more evident when data on those few individuals that had moved long distances (i.e., >500 m) were examined. For these sunfishes, 85% (17 of 20) moved over the spring

Table 6-12. Patterns of movement of sunfishes, by season, in East Fork Poplar Creek, 1985–1986

Season	Distance moved[a] (m)					
	0	50	100–200	250–500	>500	Total
Summer-fall	17 (18.6)	18 (19.1)	18 (13.1)	10 (10.0)	3 (5.2)	66
Fall-winter	26 (16.1)	16 (17.1)	8 (11.7)	9 (8.9)	0 (4.7)	59
Winter-spring	11 (13.0)	11 (13.3)	10 (9.1)	11 (6.9)	3 (3.7)	46
Spring-summer	17 (22.8)	28 (23.5)	14 (16.1)	8 (12.2)	14 (6.4)	81
Total	71	73	50	38	20	252

[a]The expected values for the null hypothesis of no differences in distance moved among seasons are given in parentheses below the observed values. The null hypothesis was rejected: $\chi^2 = 30.49$, df = 12, $P < 0.005$.

season (i.e., winter to spring or spring to summer) but only 51% of the total recaptures involved this time span. The pattern of seasonality associated with long-distance movements was highly significant ($\chi^2 = 11.99$, df = 1, $P < 0.001$). One redbreast sunfish moved 4450 m from the upper end of site EFK 13.4 to the middle of site EFK 17.9 between 18 March 1986 and 3 April 1986. The greater movement in the spring was probably associated with increased spawning activity at this time of year (nest sites guarded by male redbreast sunfish were observed at EFK 17.9 in early May 1985; J. M. Loar, ESD/ORNL, personal observation).

Significant differences in patterns of movement also occurred among sites (Table 6-13). Site EFK 17.9 was excluded from this analysis because of the very low number of recaptures, a consequence of either the July 1985 fish kill (Sect. 6.2.3.5) or the limited use of this site by sunfishes during only the spawning season. Movements of 250 to 1000 m over a 3-month period occurred more frequently (24% of recaptures) at EFK 22.7 below NHP than at either EFK 4.7 near the USGS gaging station or EFK 13.4 above the ORWTF (8% and 9% of recaptures, respectively). Movements greater than 1000 m

Table 6-13. Patterns of movement of sunfishes, by site, in East Fork Poplar Creek, 1985–1986

Site	Distance moved[a] (m)			
	0–50	100–200	>250	
EFK 22.7	68 (76.3)	27 (26.9)	30 (21.8)	125
EFK 13.4	32 (26.8)	8 (9.5)	4 (7.7)	44
EFK 4.7	33 (29.9)	12 (10.6)	4 (8.5)	49
Total	133	47	38	218

[a]Expected values for the null hypothesis of no differences in movement between sites are given in parentheses below the observed values. The null hypothesis was rejected: $\chi^2 = 9.90$, df = 4, $P < 0.05$.

were excluded from this analysis because they involved movements between sites.

Only 7% of all sunfish recaptures (24 of 322) consisted of fishes that moved more than 1000 m, and the number of individuals moving upstream and downstream was equal. Moreover, most of the movements (71%) involved fishes moving either to or from EFK 22.7, although all sampling sites were involved in at least some movements that exceeded 1000 m (Table 6-14).

To date, only four carp have been recaptured over time intervals of 3 months or more. Three were recaptured at EFK 4.7, and the other at EFK 13.4; none moved more than 200 m between captures. One traveled 50 m downstream from summer to winter, one moved 200 m upstream from spring to summer, and the other two moved 100 and 200 m upstream from one summer to the next. These limited returns indicate that carp are probably very sedentary in EFPC.

Results of growth studies

Small sample sizes within a size class and species greatly restricted the analyses of growth data collected during the first year of study. Data for the largest size category of redbreast sunfish (>15 cm total length) are presented in Table 6-15. The predominant

Table 6-14. Movements of sunfishes (14 redbreast sunfish, 9 bluegill sunfish, and 1 largemouth bass) between sites on East Fork Poplar Creek, 1985/1986

Site of first capture	EFK 4.7	EFK 13.4	EFK 17.9	EFK 22.7
EFK 4.7	X[a]	O	O	2
EFK 13.4	O	X	2	2
EFK 17.9	O	O	X	6
EFK 22.7	O	1	11[b]	X

[a]Entries above and below the diagonal formed by the Xs denote upstream and downstream movement, respectively.

[b]Includes one individual that had only moved approximately one-half the distance downstream to EFK 17.9.

feature of these data is their heterogeneity. Some fish thrived during a given time interval at a given site, whereas other fish lost weight. If the heterogeneity is ignored, then mean growth at EFK 22.7 below NHP was similar to that expected based on the seasonality of water temperatures and food availability: good growth in most fish from summer to fall and from spring to summer, poor growth from fall to winter (although exceptions exist), and weight loss in most fish during the winter (early January to mid-March). Results at EFK 13.4 were even more heterogeneous so that average changes in weight in the two seasons for which data exist were quite small. Heterogeneity among data from EFK 4.7 is probably the result of low recapture rates. Similarly, very few redbreast sunfish were recaptured in BF, so no comparison of growth in the two streams is possible yet.

6.2.4 Discussion

6.2.4.1 Species composition and richness

A longitudinal zonation of the fish assemblage was observed in EFPC during the first year of study. Species richness more than doubled over a 17-km reach between NHP and lower EFPC (EFK 6.3). A smaller Table 6-15. Weight gain or loss (g), over approximately three-month intervals in 1985-1986, of individual redbreast sunfish greater than 15 cm total length increase (from 11 to 18 species) and similar composition was observed at EFK 22.2 and EFK 6.4 by the Tennessee Valley Authority (TVA) in May 1984 (TVA 1985e, Table 11). An increase in the number of species with stream size,

Table 6-15. Weight gain or loss (g), over ~3-month intervals in 1985/1986, of individual redbreast sunfish greater than 15 cm total length

Site	Summer to fall	Fall to winter	Winter to spring	Spring to summer
EFK 22.7	44	30	10	42
	44	27	2	33
	36	16	–1	33
	16	–1	–5	30
	4	–1	–12	27
	–14	–10	–16	18
		–17	–17	17
				10
				–4
				–9
				–10
Mean	+21.7(±9.7)	+6.3(±6.9)	–5.6(±3.8)	+17.0(±5.5)
EFK 13.4	57	ND[a]	ND	43
	–1			10
	–4			0
	–6			–3
	–7			–13
	–7			
Mean	+5.3(±10.4)			+7.4(±8.8)
EFK 4.7	23	–6	22	ND
	14		10	
	12			
Mean	+16.3(±3.4)	–6	+16.0(±6.0)	
BFK 5.7	ND	ND	9	12
Mean			9	12

[a]ND = no data available.

expressed on the basis of either stream order or watershed area, is a distributional pattern that has been well-documented for stream fishes (e.g., Sheldon 1968; Horowitz 1978; Evans and Noble 1979; Fausch et al. 1984). The pattern is generally attributed to the increase in habitat heterogeneity from headwaters to mouth (Gorman and Karr 1978; Schlosser 1982).

Although the longitudinal increase in richness observed in EFPC is typical of most streams, further evaluation of the composition of the fish community indicates some degree of water quality degradation in the upstream reaches. The evaluation is based on the presence of species considered by Karr et al. (1986) to be intolerant of environmental degradation (see Sect. 6.2.3.1 and Table 6-5). No species classified as pollution-intolerant were collected from sites EFK 23.4 and EFK 18.2. A comparison of the longitudinal profiles for intolerant and total species in EFPC showed that the slope of the two curves (the rate of addition of new species) was similar in an 8-km reach below EFK 18.2 (Fig. 6-13). Eight of the 12 species added to the community in this reach were pollution-intolerant species. In addition, the absence of intolerant species at the first two sites below NHP contrasts with the observation of one to four species in much smaller (with respect to discharge) reference streams (Fig. 6-13). Finally, decreased discharge variability is associated with higher species richness (Horowitz 1978). Although discharge is much less variable in upper EFPC at NHP than in BF, both intolerant and total species richness are substantially lower in EFPC just below NHP [C.V. = 12.4 and 76.9% in EFPC at NHP and BF, respectively, based on mean monthly discharge for the period January 1985 through June 1986; mean discharge over the same period was 411 L/s (14.5 cfs) and 374 L/s (13.2 cfs) in the two streams, respectively].

Although the number of intolerant species in lower EFPC was similar to the number in the reference stream (six species at EFK 6.3 compared with eight at BFK 7.6), the fish assemblage in BF reflected a less degraded environment. For example, the darter group, a particularly sensitive indicator of environmental perturbation, was more diverse in BF than in EFPC (Table 6-5). The density and biomass of this group was almost two orders of magnitude greater in BF, an ecologically significant difference. Moreover, the list of intolerant species for BF, as shown in Table 6-5, could be expanded to include additional species not identified by Karr et al. (1986) and not found in EFPC. Both the bigeye chub (*Notropis amblops*) and rosefin shiner (*Lythrurus ardens*) share some of the

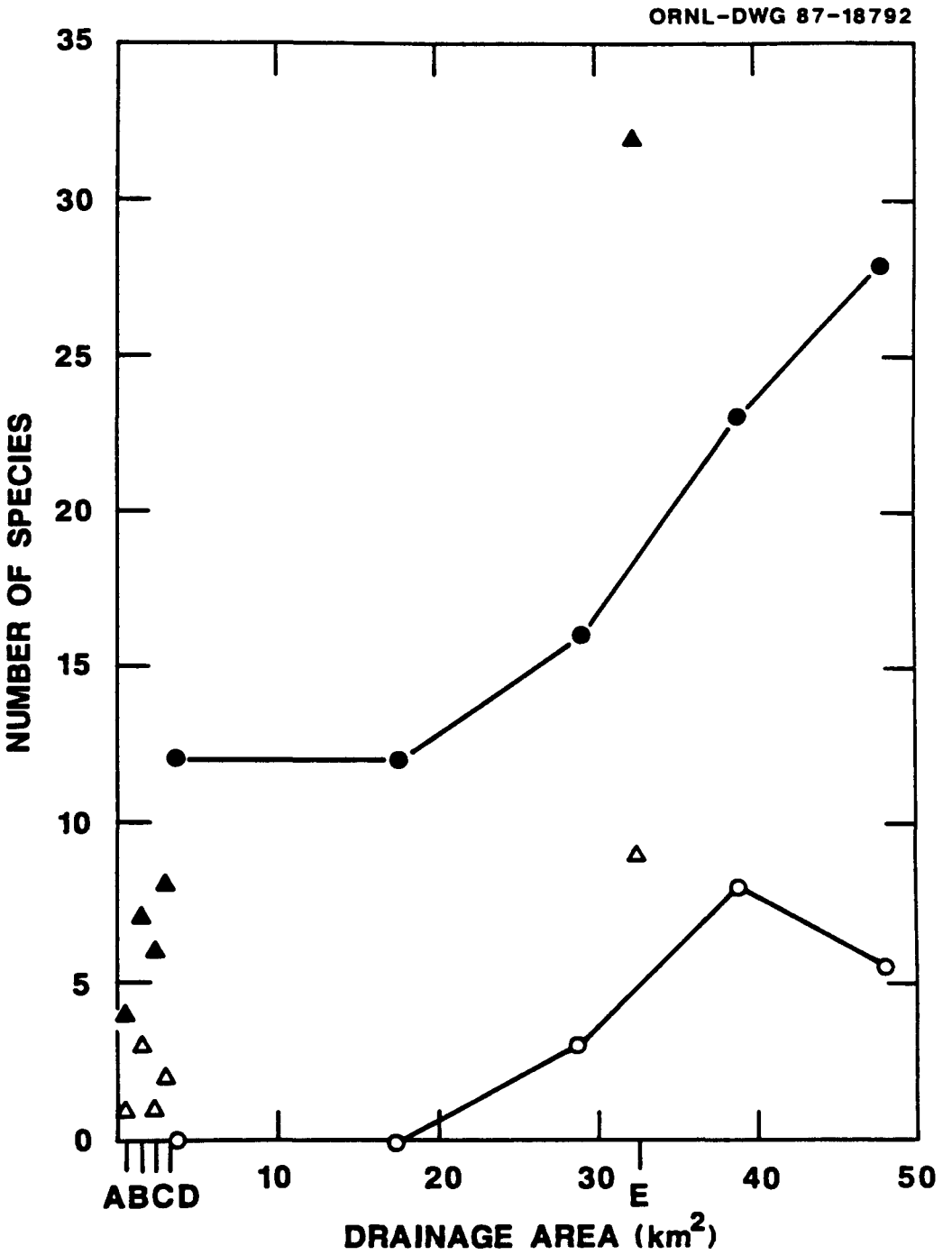

Fig. 6-13. Total number of fish species (solid symbols) and pollution-intolerant species (open symbols) as a function of increasing watershed area in East Fork Poplar Creek below New Hope Pond (circles) and reference streams (triangles), including upper White Oak Creek (A), Mill Branch (B), upper and lower Grassy Creek (C and D, respectively), and Brushy Fork (E). Pollution-intolerant species are identified in Table 6-5. Watershed areas at the EFPC sites were estimated by extrapolation from areas given elsewhere (TVA 1985d, Table 1).

same key attributes (e.g., intolerance of siltation) as the listed species (Trautman 1957; Smith 1979; Lee et al. 1980) but were not on the Karr et al. (1986) list because their current distribution does not include the midwestern United States. The American brook lamprey (*Lampetra appendix*) is also relatively intolerant of water quality degradation (Becker 1983); both this species and the chestnut lamprey (*Ichthyomyzon castaneus*) occur in BF but not EFPC (see Table D-1). No additional intolerant species were identified in EFPC.

The general tolerance rankings outlined by Karr et al. (1986) are subject to error because species tolerance is a function of the type of perturbation. The central stoneroller (*Campostoma anomalum*), for example, is sensitive to turbidity and siltation but may be tolerant of other types of water quality degradation (Leonard and Orth 1986; Berkman and Rabeni 1987). It is classified as a tolerant species by Karr et al. (1986) but as a moderately tolerant species by Leonard and Orth (1986). Similarly, the northern hog sucker (*Hypentelium nigricans*) and rock-bass, both of which inhabit the mid to lower reaches of EFPC, are considered intolerant by Karr et al. (1986) and in our evaluation, yet both species can occur in degraded environments if oxygen levels are high (Leonard and Orth 1986).

Although tolerance rankings are subject to some degree of error, they can be used in qualitative assessments of impacts on fish distributional patterns. For example, the loss of the more sensitive, pollution-intolerant species is a characteristic response of fish communities exposed to environmental degradation (e.g., municipal discharges, mining pollution, and urban development) (Leonard and Orth 1986; Scott et al. 1986). The absence of such species in EFPC below NHP and their appearance in the mid to lower reaches (below EFK 18.2) is indicative of degraded conditions in the upper reaches and downstream recovery. The source of the perturbation includes both the Y-12 Plant and off-site urban development. Both have modified the physical and chemical properties of upper EFPC via stream channel alteration and point and nonpoint source discharges, thus adversely affecting habitat quality.

6.2.4.2 Species abundance

No fish were collected in EFPC above NHP. The high levels of residual chlorine measured at the inlet to the pond (see Fig. 2-6) could account for both the absence of

fishes and the depauperate benthic community in this reach of EFPC. For example, exposure for 60 d to a residual chlorine concentration of 0.9 mg/L (mean concentration for October and November 1985) would have produced significant toxicity to both invertebrates and fishes (Mattice and Zittel 1976). Even if a constant exposure to 0.9 mg/L is considered unrealistic because discharges are actually intermittent, the measured concentrations of residual chlorine were high enough to significantly affect a fish community (Karr et al. 1985).

Immediately below NHP, fish populations exhibited dramatic seasonal shifts in abundance. Densities fluctuated by as much as two orders of magnitude, indicating substantial movement into and out of this region of EFPC. Results of the tagging studies showed that movements of sunfishes occurred more frequently between sites EFK 23.4 and EFK 18.2 than between sites located farther downstream (Table 6-14). The seasonal changes in abundance of cyprinids may also reflect immigration and emigration in the reach of EFPC just below NHP. Recruitment from within this reach cannot account for the seasonality in abundance patterns because (1) no young individuals were ever present in this population (minimum size always exceeded 6 cm), although they were found in the populations at other sampling sites in EFPC (e.g., EFK 18.2) and BF and (2) the densities of all cyprinid species at EFK 23.4 declined almost as dramatically as they increased.

For at least one species, the blacknose dace, the strong seasonality in population abundance below NHP was probably related to water temperature (Fig. 6-14). Hart (1952) reported an ultimate upper lethal temperature of 29.3°C for this species, and Terpin et al. (1976) reported incipient upper lethal temperatures of 28.8 and 29.8°C for short and long day lengths, respectively. Because maximum water temperatures below NHP exceed these thresholds during the summer months (Table A-1), the blacknose dace is not present during this period. It is found year round at EFK 18.2 and EFK 13.8 where maximum temperatures are ~3°C lower (Tables E-2 and E-3).

6.2.4.3 Fish movements

Only 11% of the recaptured sunfishes moved more than 500 m, and 73% moved 200 m or less. These initial results corroborated the work of other investigators. In general, sunfishes are sedentary species, the majority of individuals remain within a highly restricted home range (Gerking 1950, 1953, 1959). For bluegill in particular, Gunning and

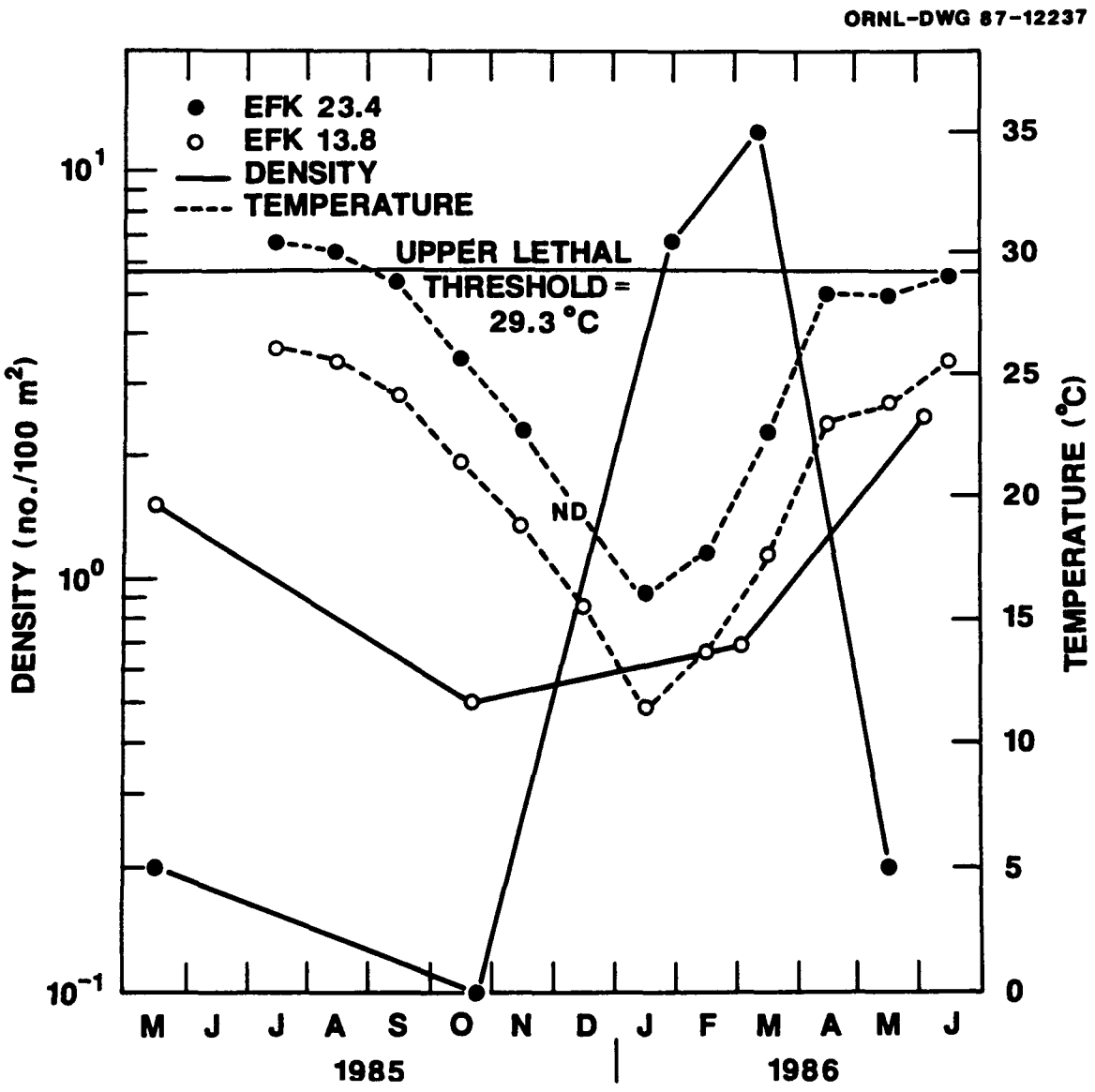

Fig. 6-14. Density of blacknose dace and maximum monthly water temperatures at East Fork kilometer (EFK) 23.4 and EFK 13.8, May 1985–June 1986.

Shoop (1963) reported a home range of less than 38 m. The home range of redbreast sunfish has not been investigated previously. Although too few carp were tagged to adequately assess movement patterns, Funk (1955) classified carp as a species of intermediate mobility in which sedentary and mobile portions of the population are about equally abundant. Most of the studies of home range in fishes have limited application to this study because they were designed to determine the proportion of individuals in a population that are sedentary rather than to quantify the distances moved by the less sedentary members of the population.

The greatest movement occurred at the site (EFK 22.7) just downstream from NHP, which receives effluent discharges from the Y-12 Plant. Such movements may be in response to (1) discharges from the plant, (2) the location of the site in the headwaters of EFPC, (3) limited spawning habitat in this region of the stream, which results in greater movement in late spring in search of suitable spawning sites, or (4) the mid-July 1985 fish kill that occurred near EFK 21.8 and resulted in an essentially fishless reach for several kilometers downstream of that point. The absence of competitors could have increased the amount of movement by fishes in the adjacent, unaffected section, especially the reach immediately below NHP.

The observed movement patterns can easily be attributed to fishes inhabiting a favorable reach of stream in winter and then migrating to suitable breeding habitat between March and July. Such a pattern fits the observations that EFPC near EFK 17.9 is suitable for spawning and that movement to and from this site is high (Table 6-14). Further study is planned to determine whether the patterns of movement between sites observed during the first year are common seasonal occurrences.

High tag loss was observed during the initial studies. Based on the collections in July 1986, for example, the number of sunfishes that had lost tags was approximately equal to the number that had retained them. Although the proportion of fishes tagged and subsequently recaptured has increased throughout the first year of study, the recapture rate declines rapidly after the first 3-month interval following tagging (Table 6-16). This trend will be followed closely in the future, because it is difficult to determine how much of the difference in recapture rate over time is the result of improvement in the tagging technique and how much is the result of lower temperatures and better water quality during the course of the study. The large external lesions that were noted on many fishes

Table 6-16. Percentage of sunfishes tagged on a given date that were recaptured on the subsequent date indicated

Date of recapture	Percentage recaptured			
	Summer 1985	Fall 1985	Winter 1986	Spring 1986
Fall 1985	3.5			
Winter 1986	1.1	5.9		
Spring 1986	0.9	3.6	7.3	
Summer 1986	0.3	2.2	0.4	13.7

at EFK 22.7 in July 1985 were not observed a year later. If this change reflects improved water quality, then one might expect lower infection rate and better tag retention during the course of the study, as has been observed. Further support for the hypothesis of improved water quality below NHP is based on the recapture rates at EFK 22.7, which were the lowest for all sites between July and October 1985 but were the highest for all sites in the three subsequent sampling intervals. Site-specific recapture rates will continue to be monitored closely in the future.

6.2.5 Future Studies

Fish population studies will continue with emphasis during the second year on (1) quantitative population estimates, (2) qualitative sampling to provide a more comprehensive list of species in EFPC watershed, (3) age/growth assessments of target species, (4) evaluation of food habits of target species, and (5) movement/growth studies. Quantitative sampling will be used to (1) determine population size, (2) characterize community structure, and (3) eventually provide an estimate of annual production. Sampling frequency will be reduced from quarterly to twice annually (March and October), but the number of sampling sites will remain unchanged [six sites on EFPC and one site on BF (Fig. 2-1)]. Elimination of the late spring (May/June) sample will reduce inadvertent sampling mortality, which is higher when water temperatures are elevated and fish are more vulnerable to stress. Qualitative sampling will be conducted in tributaries of

EFPC to assess species composition and evaluate the distribution of particular species (e.g., Tennessee dace).

Age/growth of redbreast sunfish will be evaluated in the fall. To provide larger sample sizes, scales will be collected during the tagging studies to supplement those obtained during the routine quantitative population surveys.

Finally, the growth and movement studies will be continued. During the second year, emphasis will be placed on obtaining sufficient data for evaluating growth rates. If tag loss can be reduced and recapture rates are satisfactory, then a reduction in sampling effort from four to three times per year (e.g., elimination of the winter sample because of small sample sizes) will be considered.

6.3 INTERPRETATION OF BIOTIC CHANGES

The ultimate question that must be answered by the Y-12 Plant Biological Monitoring and Abatement Program (BMAP) is whether the various remedial actions that have been taken on EFPC have produced biotic changes that make the stream indistinguishable from unaffected streams of a similar size and type. Several problems must be addressed before this question can be answered. First, are there any unaffected streams of a similar size and type, and if so, what is their biota? Second, if there are impacts to EFPC that are beyond the scope of the remedial action program (e.g., municipal discharges and urban and agricultural runoff), can their impacts on the biota be separated from the impacts resulting from Y-12 Plant discharges? Third, because the final status of the EFPC ecosystem will not be known until the BMAP is completed, can the biotic changes observed during the BMAP be interpreted as favorable or unfavorable to the long-term goal of ecosystem recovery?

Reference streams for EFPC are those without industrial impacts. None of these streams, however, is totally free of past or present disturbances of various types that might influence the biota present. Furthermore, even in the absence of unnatural disturbance, there is great variation in the biota of streams of similar size. This variation can result from bedrock geology, forest vegetation type, chance events of dispersal, and natural disturbances, such as floods and droughts.

To address the first problem, a study was initiated to quantify the variability in the biota of natural streams similar to EFPC by analyzing an extensive unpublished data set on

water conditions, fish, and insects collected by TVA from a great variety of streams within the Tennessee River drainage. This analysis will help determine where EFPC fits in a broader regional perspective, as well as to bracket the range of biotic conditions that should be expected when the current impacts on EFPC have been successfully mitigated. A critical factor is the type and frequency of natural disturbances; this analysis will allow us to evaluate EFPC in the context of other streams with similar patterns of natural disturbance.

The second problem can also be addressed using this large data set in conjunction with our reference streams. Many of the streams in the TVA survey have some impacts similar to those on EFPC beyond the ORR boundary (e.g., urban and agricultural runoff and municipal discharges). Comparison of the biota of these streams to the biota of truly unaffected streams will help establish reasonable criteria to evaluate the biota of EFPC in the presence of impacts that are beyond the scope of existing remedial action program.

The third problem is that the biotic changes observed during the course of implementing the various remedial actions are the primary indication of whether the remedial actions are causing biotic changes in the desired direction. What is the desired direction? Can it be evaluated prior to completion of the BMAP?

Two critical properties of the biota that are expected to change as a result of remedial actions are the diversity (or number of species present) and the productivity (or total biomass) of the biotic community. A low number of species can be an indication of pollution stress, and a high number of species is often considered to be a sign of a healthy stream. Likewise, very low productivity or biomass can be an indication of pollution, whereas high productivity (particularly in terms of fish production) is found in healthy streams having high recreational potential. However, very high productivity is also a sign of certain kinds of pollution, such as nutrient enrichment from agricultural runoff or municipal discharges.

The interpretation of biotic change is complicated by the relationship between productivity and diversity. Beginning at very low levels of productivity, diversity is known to increase with increasing productivity. However, at intermediate levels of productivity, the diversity of many types of organisms has been shown to decrease with further increases in productivity. Thus, depending on where a particular stream or portion of a stream is on the productivity scale, a change in productivity could result in either an

increase or decrease in diversity. It is not possible to maximize both productivity and diversity.

The biotic changes observed in EFPC will be evaluated carefully in the context of the productivity and natural disturbance conditions of the stream. Fortunately, there is a good theoretical understanding of the relation between diversity, production, and disturbance, and analysis of the TVA data set will allow an evaluation of biotic changes in EFPC in an appropriate context.

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Appendix A

MEAN MONTHLY TEMPERATURES IN EAST FORK POPLAR CREEK AND BRUSHY FORK, JULY 1985–AUGUST 1986

Table A-1. Mean monthly stream temperatures in °C (standard deviation in parentheses) at study sites in East Fork Poplar Creek and Brushy Fork; the range (absolute maximum and minimum) and the number of days of record (in parentheses) are given

	BFK 7.6	EFK 23.4	EFK 18.2	EFK 13.8	EFK 10.0	EFK 6.3
			1985			
July	ND[a] ND	25.3 (2.1) 22.0-30.4 (20)[b]	ND ND	23.4 (1.3) 21.0-26.0 (21)[b]	23.0 (1.0) 21.2-25.2 (20)[b]	22.7 (0.9) 21.0-24.6 (19)[b]
August	ND ND	24.1 (2.1) 20.8-30.0 (31)	ND ND	22.1 (1.6) 19.4-25.4 (31)	22.0 (1.3) 20.0-24.8 (31)	21.6 (1.4) 19.4-25.0 (31)
September	ND ND	22.8 (2.4) 18.2-28.8 (29)	ND ND	19.9 (2.3) 15.4-24.0 (29)	20.3 (1.8) 16.4-23.4 (30)	19.3 (1.9) 15.2-22.6 (29)
October	16.8 (1.2) 13.8-19.0 (17)[b]	22.0 (1.9) 17.2-25.6 (30)	19.7 (1.9) 14.4-23.0 (31)	18.3 (2.0) 13.2-21.2 (31)	18.9 (1.5) 15.0-21.2 (34)[c]	17.7 (1.7) 13.2-20.4 (31)
November	13.6 (2.0) 8.0-17.2 (30)	19.7 (1.8) 16.0-22.6 (18)[d]	16.9 (2.0) 12.0-20.8 (30)	15.4 (2.1) 10.2-18.8 (30)	16.5 (1.6) 12.6-19.2 (30)	15.1 (1.9) 10.4-18.0 (30)
December	6.7 (2.9) 2.6-14.8 (31)	ND ND	10.5 (2.7) 5.0-16.8 (29)	8.6 (2.8) 4.2-15.6 (24)[d]	10.7 (2.2) 6.6-16.0 (21)[d]	9.0 (2.6) 4.6-15.2 (23)[d]
			1986			
January	5.1 (1.0) 3.2-7.0 (31)	12.1 (1.9) 7.0-16.0 (19)[b]	9.0 (2.0) 5.0-13.2 (19)[b]	6.8 (2.1) 1.0-11.4 (19)[b]	8.3 (1.8) 3.2-11.8 (19)[b]	7.7 (1.8) 3.0-11.4 (19)[b]
February	ND ND	13.8 (1.7) 9.2-17.8 (28)	11.2 (2.1) 5.8-15.4 (28)	9.7 (2.2) 4.2-13.8 (28)	10.7 (1.9) 6.6-13.8 (28)	10.2 (1.9) 5.6-13.8 (28)
March	11.7 (2.2) 6.8-16.2 (13)[d]	15.7 (2.4) 10.2-22.6 (30)	13.2 (2.6) 7.2-19.8 (30)	12.1 (2.8) 6.0-17.6 (30)	12.6 (2.3) 8.0-17.8 (30)	12.4 (2.6) 7.2-17.6 (30
April	14.3 (2.2) 9.6-20.0 (30)	19.3 (3.0) 13.6-28. (30)	16.9 (2.7) 11.0-24.0 (30)	16.2 (2.5) 10.8-24.0 (30)	16.4 (2.0) 10.8-23.0 (30)	16.4 (2.1) 12.0-21.4 (30)
May	16.9 (1.8) 11.2-20.8 (29)	22.0 (2.7) 15.2-28.2 (30)	20.1 (2.0) 13.6-24.0 (29)	19.7 (2.1) 13.0-23.8 (30)	19.4 (1.7) 14.0-22.4 (30)	19.2 (1.7) 14.2-22.2 (30)
June	19.7 (1.2) 17.0-22.6 (30)	24.5 (2.3) 20.2-29.0 (30)	23.2 (1.5) 19.6-26.2 (30)	23.2 (1.2) 19.6-25.8 (30)	22.4 (1.1) 19.6-24.6 (30)	22.1 (0.9) 19.8-24.0 (30)
July	21.6 (1.3) 18.0-24.4 (31)	25.7 (2.1) 21.0-31.0 (31)	24.8 (1.4) 20.8-28.0 (31)	25.0 (1.4) 21.0-28.0 (25)[d]	24.3 (1.1) 20.8-27.0 (31)	23.9 (1.0) 20.8-25.6 (31)
August	20.3 (1.0) 18.0-22.4 (10)[d]	25.3 (1.7) 22.2-29.4 (13)[d]	23.3 (0.8) 21.6-25.0 (11)[d]	ND ND	23.2 (0.6) 21.6-24.4 (9)[d]	22.8 (0.5) 21.6-23.6 (11)[d]

[a]ND = no data.
[b]Data are from days at the end of month.
[c]Overlap of two temperature recorders resulted in 34 sample days.
[d]Data are from days at beginning of month.

Appendix B

RESULTS OF QA/QC ANALYSES OF MERCURY AND PCBs IN FISH SAMPLES

Appendix B

RESULTS OF QA/QC ANALYSES OF MERCURY AND PCBs IN FISH SAMPLES

Duplicate Mercury Analyses

Blind duplicate analyses of total mercury in fish showed good agreement. The mean and standard deviation (SD) for the two populations of all duplicate samples were 0.67 ± 0.42 and 0.63 ± 0.40 ppm. A paired comparison t-test was used to evaluate the precision of these analyses. The mean difference between duplicate determinations was 0.04 ± 0.13 ppm (N = 33), and the mean coefficient of variation (CV) was 11.8%. There was no significant difference between the paired comparisons (α = 0.05).

When blind duplicate samples were evaluated for each sampling period, a significant (α > 0.05) difference was noted between sets of duplicate samples analyzed at different times in December 1985. Mean mercury levels of the two sets were 0.77 ± 0.47 and 0.66 ± 0.47 ppm, and the standard deviation of the mean difference (0.11 ppm) was 0.12 ppm. The mean CV was 15.4%. Blind duplicates submitted at different times in May 1985 were not dissimilar, averaging 0.67 ± 0.37 and 0.70 ± 0.30 ppm. The mean difference between duplicate determinations was 0.03 ± 0.17 (N = 8), and the mean CV was 13.4%. In May 1986, blind duplicates were submitted simultaneously with other samples. Very little difference was observed between these two sets, which averaged 0.59 ± 0.43 and 0.58 ± 0.42 ppm (N = 14). The mean difference of 0.01 ± 0.08 ppm was not significant (α = 0.05), and the mean CV was 8.0%.

These results indicate that systematic differences may sometimes occur between groups of samples analyzed at different times, but the magnitude of the only significant difference observed was small (~0.1 ppm Hg).

EPA/ORNL Split Samples for Mercury Analysis

Homogenized fish samples were split and analyzed for total mercury by the U.S. Environmental Protection Agency (EPA) Environmental Services Laboratory in Athens, Georgia. The results from the EPA laboratory were slightly higher than those of the Oak Ridge National Laboratory (ORNL) Analytical Chemistry Division [0.53 ± 0.26 vs 0.40 ± 0.20 ppm (N = 5)]. The paired comparison t-test showed that the mean difference of 0.13 ± 0.18 ppm was not statistically significant (α = 0.05).

Subsamples of the fish used in the EPA/ORNL split-sample comparisons were also analyzed as blind duplicates along with routine fish samples. Results of these analyses more closely approximated those of the EPA laboratory, averaging 0.51 ± 0.24 ppm total mercury. The means were not significantly different, and the variability between sample pairs (SD = 0.18 ppm) was similar to that observed between the duplicate samples analyzed by ORNL and EPA.

Analysis of Mercury in Archived Samples from TVA in 1984

Samples of fish that were collected and analyzed for total mercury by the Tennessee Valley Authority (TVA) in 1984 (TVA 1985e) were obtained from TVA and analyzed by ORNL. The TVA results were significantly (α = 0.05) higher than the ORNL results, averaging 0.97 ± 0.23 ppm vs 0.73 ± 0.20 ppm (N = 5). The mean difference between paired analyses was 0.24 ± 0.20 ppm.

Results of this comparison must be interpreted with caution. The TVA samples were homogenized and stored frozen, but during storage, a refrigeration failure resulted in thawing and separation of fluids from solid tissue (G. Hickman, TVA Division of Water Resources, Fisheries and Aquatic Branch, Norris, Tennessee, personal communication). Such separation of fluids can deplete the mercury in the remaining solid-phase mercury, resulting in low measures of total mercury in the sample if fluid is not fully reconstituted with solids (W. McDaniel, EPA Environmental Services Laboratory-Athens, personal communication). Thus, it may not be surprising that the ORNL results would be lower than those found by TVA a year earlier.

Mercury in Archived ORNL Fish

Because the TVA fish were stored frozen for more than a year prior to analysis by ORNL and because sample archiving is an important element of the Biological Monitoring and Abatement Program (BMAP), fish collected in May 1985 were resampled from archived tissue and analyzed again in May 1986. The average mercury level in the fish archived for one year was slightly lower than that observed initially [0.64 ± 0.28 vs 0.77 ± 0.34 ppm (N = 5)]. The mean difference between archived and fresh fish was 0.14 ± 0.13, and the mean of the groups was not significantly different in the paired comparison t-test (α = 0.05).

EPA Reference Fish Standards for Mercury

Freeze-dried mercury-contaminated reference fish standards were routinely analyzed along with the fish samples from EFPC. The reference standard was provided by the EPA Environmental Monitoring and Support Laboratory—Cincinnati. Results of ORNL analyses of this material agreed well with the mean value obtained through exhaustive analyses by EPA referee laboratories. The mean value observed by ORNL [was 2.41 $\pm$ 0.20 ppm total mercury (N = 23)], compared with the published mean value of 2.52 ppm. The CV of these analyses (8.3%) corresponded well with that observed for the blind duplicates.

Comparison of ORNL and Y-12 Mercury Results

One group of fish samples was split; one set of samples was submitted for analysis to the ORNL Analytical Chemistry Division and the other set of samples from the same fish was submitted to the Y-12 Analytical Laboratory, which analyzed samples collected in the 1982 study (Van Winkle et al. 1984). The results obtained by the Y-12 were substantially higher than those obtained by ORNL, averaging 0.88 $\pm$ 0.68 ppm vs 0.51 $\pm$ 0.39 (N = 11). The large difference, (0.37 $\pm$ 0.33 ppm) was statistically significant (α = 0.05). The Y-12 Laboratory also reported higher results on analyses of the EPA reference fish standard [3.2 $\pm$ 0.30 ppm (N = 3) vs the EPA-reported mean of 2.52 ppm total mercury]. Finally, analyses conducted by the Y-12 Laboratory on fish from Brushy Fork, a reference stream, averaged 0.32 $\pm$ 0.12 ppm, whereas ORNL analyses of the same fish averaged 0.14 $\pm$ 0.08 ppm. Based on these results, it was concluded that the Y-12 Laboratory systematically overestimated total mercury in fish on this occasion.

Duplicate PCB Analyses

Results of PCB analyses of blind duplicate samples were much more variable than were the results of similar mercury analyses. The first set of blind duplicate samples involved a comparison of two different methods—the alcoholic/KOH digestion [procedure EC-440 (Martin Marietta Energy Systems 1983)] and the methylene chloride extraction/column cleanup method [procedure EPA-600/4-81-055 (EPA 1980)]. The EC-440 procedure provided direct comparison with the 1982 data (Van Winkle et al. 1984) and was initially used for the May 1985 samples. After those results indicated no PCB-1254 and analyses of EPA reference fish standards suggested low recovery of

PCB-1254 by this procedure, analyses were conducted using the EPA-600/4-81-055 procedure. The comparison of blind duplicates analyzed by each procedure indicate a systematic problem with recovery of PCB-1254. The mean total PCBs reported for the May 1985 duplicates analyzed by EC-440 was 1.43 ± 1.77 ppm (N = 12), whereas the duplicates analyzed by EPA-600/4-81-055 had a mean total PCB concentration of 2.29 ± 3.05 ppm. Despite the large difference (0.86 ppm), the means were not significantly different (α = 0.05) because of the high variability between paired replicates (SD = 1.81 ppm and CV = 45%). However, only PCB-1260 was reported using EC-440, whereas EPA-600/4-81-055 reported roughly equal amounts of PCB-1254 and PCB-1260. The amounts of PCB-1260 reported for the paired replicates were similar, 1.43 ± 1.77 ppm vs 1.21 ± 2.15 ppm for EC-440 and EPA-600/4-81-055, respectively.

The December 1985 set of blind duplicates also indicated intermittent problems with the recovery/quantitation of PCB-1254. The means and standard deviations (N = 10) of the two groups of paired samples were 0.56 ± 0.31 and 0.35 ± 0.32 ppm. Again, the difference between the two means (0.21 ppm) was not significantly different from zero (α = 0.05) because of the high variability of the difference between duplicate determinations (SD = 0.44 ppm and CV = 49%). Both these results and those from the EPA/ORNL split sample analyses prompted further study to refine the procedures for quantifying PCB isomer mixtures before the May 1986 analyses were initiated.

Results of the May 1986 blind duplicate analyses were much improved. Total PCBs for the groups of paired samples averaged 0.33 ± 0.43 and 0.37 ± 0.45 ppm (N = 9). The difference between these means was not statistically significant (α = 0.05), and the variability of the mean difference was much lower than that observed in the previous comparisons (SD = 0.18 ppm and CV = 32%). Although these analyses did not include carp samples (which were not yet completed), the results indicated a considerable increase in precision.

Recovery of PCBs Added to Uncontaminated Fish

Measured amounts of PCB standards were added to homogenized samples of fish from the reference site, and samples were then carried through extraction, cleanup, and gas chromatographic analysis. Each 5 to 10-g sample was spiked with 1 to 10 μg each of PCB-1254 and the PCB-1260, and percentage recovery of each PCB mixture was reported. The mean recovery of PCB-1260 was 91% ± 38% and 100% ± 29%, respectively

(N = 11). Although mean recoveries were good, variability was relatively high. Minimum and maximum reported recoveries of spiked PCBs were 28% and 187%. (The very low recovery was observed in a very low concentration spike.)

EPA/ORNL Split Samples for PCB Analysis

Homogenized carp samples were also split and analyzed for PCBs (as for mercury) by ORNL and by the EPA Environmental Services Laboratory in Athens, Georgia. The fish samples had also been analyzed previously for PCBs by ORNL; thus each fish was run as blind duplicate at ORNL and once by EPA. Results of these comparisons showed high variability and a likelihood of underestimating PCB-1254.

The mean total PCB content of the group of fish analyzed by EPA was 4.3 ± 4.7 ppm (which includes one value of <2.4 ppm included as 2.4 ppm), compared with 1.2 ± 1.2 ppm in the initial ORNL analysis (ORNL-1) and 3.6 ± 3.5 ppm in the second ORNL analysis (ORNL-2). The low result (1.2 ppm) was the result of very low reported levels of PCB-1254 in this set of analyses (0.4 ppm vs 2.8 and 2.5 ppm in the EPA and ORNL-2 analyses, respectively). A high degree of variability was observed between paired samples; SD of mean differences between EPA and the two ORNL samples was 3.6 and 4.5 ppm, respectively, and the mean CV was 67% and 32% for EPA vs ORNL-1 and EPA vs ORNL-2, respectively. Similar variability was observed between the two ORNL groups (SD = 3.1 ppm and CV = 60%). Because of the high variability among the differences between paired comparisons, the means of all three data sets did not differ significantly ($\alpha = 0.05$).

Cesium-137

Three fish samples that were counted by the ORNL Analytical Chemistry Division and found to contain ^{137}Cs were repacked in 5-mL plastic vials and counted by L. Larsen of the ORNL Environmental Sciences Division using a well-configured germanium-lithium detector gamma spectrometer. Good agreement was noted between the two sets of samples. The ORNL Analytical Chemistry Division reported a mean ^{137}Cs value of 6.6 ± 3.3 Bq/kg per sample, and the ORNL Environmental Sciences Division reported a value of 5.8 ± 3.6 Bq/kg. These mean values were not significantly different ($\alpha = 0.05$). The results help clarify an uncertainty in the ^{137}Cs data reported by TVA for East Fork Poplar Creek (EFPC) fish in 1984 (TVA 1985e). In the quality assurance/quality control

(QA/QC) split sample exchange carried out in that study, results reported by the EPA-Las Vegas Laboratory exceeded those reported by TVA by a factor of 6 (TVA 1985e). However, the ^{137}Cs levels reported by TVA appear to be more typical of fish collected from EFPC in this study.

Appendix C

MERCURY, PCBs, AND ^{137}Cs IN FISH FROM EAST FORK POPLAR CREEK AND REFERENCE SITES, MAY 1985–MAY 1986

Table C-1. Mercury, PCBs, and ^{137}Cs in fish from East Fork Poplar Creek and reference sites, May–July 1985

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs			^{137}CS
									Total	1254	1260	
EFK23.4	23.4	05/14/85	Blugil	M	6700	99	16.3	0.80	0.23		0.23	
EFK23.4	23.4	05/14/85	Blugil	M	6706	61	14.1	0.91	0.13		0.13	
EFK23.4	23.4	05/14/85	Blugil	M	6707	93	16.6	0.44				
EFK23.4	23.4	05/14/85	Blugil	M	6711	85	16.2	0.72	0.32		0.32	
EFK23.4	23.4	05/14/85	Blugil	M	6713	102	16.6	0.62				
EFK23.4	23.4	05/14/85	Blugil		6714	79	16.6	0.96	0.39		0.39	
EFK23.4	23.4	05/14/85	Blugil	M	8314	129	17.7	1.10				4.9
EFK23.4	23.4	05/14/85	Blugil	M	8326	130	18.2	0.84				<1.9
EFK23.4	23.4	05/14/85	Blugil	M	8356	145	18.5	0.68	0.22		0.22	5.9
EFK23.4	23.4	05/14/85	Blugil	M	8360	105	17.0	1.10	0.40		0.40	
EFK23.4	23.4	05/14/85	Blugil	M	8375	130	18.2	0.84	0.17		0.17	
EFK23.4	23.4	05/14/85	Blugil	M	8379	128	17.8	0.92	0.21		0.21	4.3
EFK13.8	13.8	05/27/85	Blugil	M	8305	62	14.6	0.37				
EFK6.3	6.3	06/05/85	Blugil	M	6816	128	17.5	0.60	0.04		0.04	
EFK6.3	6.3	06/05/85	Blugil	F	6812	79	16.7	0.70	0.35		0.35	
EFK2.1	2.1	05/20/85	Blugil	M	8306	107	16.2	0.44	0.61		0.61	
EFK2.1	2.1	05/20/85	Blugil	F	8334	157	18.0	0.52	0.22		0.22	<0.49
EFK2.1	2.1	05/20/85	Blugil	F	8347	71	15.5	0.52	0.21		0.21	
EFK2.1	2.1	05/20/85	Blugil	M	8341	137	17.2	0.37	0.32		0.32	3.0
EFK2.1	2.1	05/20/85	Blugil	M	8363	115	17.0	0.29	0.14		0.14	
EFK2.1	2.1	05/20/85	Blugil	F	8366	113	15.9	0.64	0.28		0.28	
EFK2.1	2.1	05/20/85	Blugil	M	8382	95	18.1	0.54	0.25		0.25	
EFK2.1	2.1	05/20/85	Blugil	M	8392	137	17.3	0.54	0.41		0.41	
EFK23.4	23.4	07/02/85	Redbre	M	8130	84	16.0	0.58	0.53		0.53	
EFK23.4	23.4	07/02/85	Redbre	M	8131	111	18.9	0.80	0.40		0.40	7.5
EFK23.4	23.4	07/02/85	Redbre	F	8139	54	13.8	1.1	0.17		0.17	
EFK23.4	23.4	07/02/85	Redbre	F	8158	48	13.9	0.63	0.96		0.96	
EFK23.4	23.4	07/02/85	Redbre	M	8180	94	18.2	0.79	1.05		1.05	
EFK23.4	23.4	07/02/85	Redbre	M	8196	79	15.6	0.41	0.22		0.22	
EFK23.4	23.4	05/27/85	Redbre	M	8312	61	14.2	0.25	0.36		0.36	
EFK23.4	23.4	05/27/85	Redbre	M	8318	61	14.3	0.92	0.56		0.56	
EFK23.4	23.4	05/27/85	Redbre	M	8361	95	16.0	0.39	0.17		0.17	
EFK23.4	23.4	05/27/85	Redbre	M	8365	100	15.8	0.28	0.29		0.29	
EFK23.4	23.4	05/27/85	Redbre	M	8396	73	14.5	0.56	0.82		0.82	
EFK23.4	23.4	07/02/85	Redbre	F	8198	48	13.8	0.68	0.26		0.26	
EFK18.2	18.2	06/07/85	Redbre	F	6822	50	13.6	1.30	0.19		0.19	
EFK18.2	18.2	06/07/85	Redbre	M	6826	84	15.6	0.50	0.51		0.51	
EFK18.2	18.2	06/07/85	Redbre	M	8127	112	17.1	0.75	0.38		0.38	
EFK18.2	18.2	06/07/85	Redbre	M	8135	128	18.6	0.62	0.43		0.43	
EFK18.2	18.2	06/07/85	Redbre	M	8307	115	18.1	0.60	0.40		0.40	
EFK18.2	18.2	06/05/85	Redbre	F	8343	38	12.5	0.82	0.28		0.28	
EFK18.2	18.2	06/07/85	Redbre	F	8370	50	14.0	0.94	0.21		0.21	
EFK18.2	18.2	06/07/85	Redbre	M	8397	102	16.6	0.60	0.28		0.28	

Table C-1 (continued)

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs			^{137}CS
									Total	1254	1260	
EFK13.8	13.8	05/15/85	Redbre	M	8322	80	15.8	0.48	0.49		0.49	
EFK13.8	13.8	05/15/85	Redbre	F	8335	98	16.1	0.47	0.23		0.23	
EFK13.8	13.8	05/15/85	Redbre	M	8345	94	16.9	0.77	0.13		0.13	
EFK13.8	13.8	05/15/85	Redbre	M	8351	125	18.8	0.76	0.42		0.42	
EFK13.8	13.8	05/15/85	Redbre	F	8357	103	17.6	0.82	0.41		0.41	
EFK13.8	13.8	05/15/85	Redbre	F	8359	140	19.5	0.90	0.90		0.90	
EFK13.8	13.8	05/15/85	Redbre	F	8369	80	15.3	0.36	0.10		0.10	
EFK13.8	13.8	05/15/85	Redbre	M	8378	117	18.6	0.66	0.41		0.41	
EFK6.3	6.3	06/05/85	Redbre	M	6814	65	14.1	0.39	0.13		0.13	
EFK6.3	6.3	06/05/85	Redbre	F	6815	46	13.0	0.34	0.21		0.21	
EFK6.3	6.3	06/05/85	Redbre	M	6817	103	17.7	0.40	0.23		0.23	
EFK6.3	6.3	06/05/85	Redbre	M	8300	38	12.5	0.41	0.25		0.25	
EFK6.3	6.3	06/05/85	Redbre	M	8327	47	13.5	0.29	0.31		0.31	
EFK6.3	6.3	06/05/85	Redbre	M	8368	66	13.9	0.46	0.28		0.28	
EFK6.3	6.3	06/05/85	Redbre	F	8380	42	12.7	0.46	0.38		0.38	
EFK6.3	6.3	06/05/85	Redbre	M	8385	51	13.5	0.32	0.24		0.24	
EFK2.1	2.1	05/20/85	Redbre	M	8138	173	19.6	0.95	0.16		0.16	<0.87
EFK2.1	2.1	05/20/85	Redbre	F	8117	110	16.2	0.83	0.18		0.18	<4.5
EFK2.1	2.1	05/20/85	Redbre	M	8301	70	13.9	0.23	0.23		0.23	
EFK2.1	2.1	05/20/85	Redbre	F	8194	63	13.1	0.27	0.23		0.23	
EFK2.1	2.1	05/20/85	Redbre	F	8151	64	13.9	0.42	0.16		0.16	
EFK2.1	2.1	05/20/85	Redbre	F	8142	50	12.2	0.25	0.15		0.15	
EFK2.1	2.1	05/20/85	Redbre	M	8174	49	12.1	0.20	0.24		0.24	
BEAVR		06/10/85	Redbre	M	6831	90	16.6	0.05	0.03		0.03	
BEAVR		06/10/85	Redbre	F	6836	95	16.1	0.10	0.02		0.02	
BEAVR		06/10/85	Redbre	M	6837	90	16.7	0.07	0.03		0.03	
BEAVR		06/10/85	Redbre	M	8394	47	13.2	0.04	0.02		0.02	
BEAVR		06/10/85	Redbre	M	8395	51	13.5	0.04	0.06		0.06	
Hindscr		06/04/85	Redbre	M	6799	96	16.6	0.08	0.01		0.01	
Hindscr		06/04/85	Redbre	F	6804	73	15.8	0.10	0.01		0.01	
Hindscr		06/04/85	Redbre	F	6807	53	13.9	0.10	0.01		0.01	
Hindscr		06/04/85	Redbre	M	6810	127	18.2	0.07	0.01		0.01	
Hindscr		06/04/85	Redbre	M	8350	127	19.0	0.08				
Hindscr		06/04/85	Blugil	M	8371	54	14.1	0.04	0.01		0.01	
Hindscr		06/04/85	Blugil	M	8387	55	13.8	0.05	0.01		0.01	
EFK23.4	23.4	05/14/85	Cocarp		8302	2751	66.4	0.19	0.44		0.44	
EFK23.4	23.4	05/14/85	Cocarp		8398	1149	42.3	0.38	4.6		4.6	
EFK13.8	13.8	05/15/85	Cocarp	M	6739	3034	58.3	0.38	2.6		2.6	
EFK13.8	13.8	05/15/85	Cocarp	F	8310	3982	64.0	1.1	6.1		6.1	
EFK13.8	13.8	05/15/85	Cocarp	F	8336	2647	53.9	1.1	2.4		2.4	

Table C-1 (continued)

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs			^{137}CS
									Total	1254	1260	
EFK13.8	13.8	05/15/85	Cocarp	F	8346	2367	55.8	2.0	1.5		1.5	
EFK13.8	13.8	05/15/85	Cocarp	M	8364	3251	59.9	1.0	2.1		2.1	
EFK6.3	6.3	05/21/85	Cocarp	M	8311	1804	50.9	0.46	1.6		1.6	
EFK6.3	6.3	05/21/85	Cocarp	F	8313	1002	39.1	0.11	0.37		0.37	
EFK6.3	6.3	05/21/85	Cocarp	M	8323	1697	46.9	0.79	1.4		1.4	
EFK6.3	6.3	05/21/85	Cocarp	F	8328	2112	52.2	0.57	1.7		1.7	
EFK6.3	6.3	05/21/85	Cocarp	F	8337	2495	57.0	1.2	1.8		1.8	
EFK6.3	6.3	05/21/85	Cocarp	M	8367	1290	43.9	0.95	1.2		1.2	
EFK6.3	6.3	05/21/85	Cocarp	M	8383	2000	54.0	1.1	1.1		1.1	
EFK6.3	6.3	05/21/85	Cocarp	M	8386	2715	58.8	1.2	4.4		4.4	
EFK2.1	2.1	05/20/85	Cocarp	M	8332	2120	52.6	0.59	5.3		5.3	
EFK2.1	2.1	05/20/85	Cocarp	M	8391	1533	50.4	0.62	1.2		1.2	
Hindscr		06/04/85	Cocarp	F	8389	909	38.1	0.09	0.08		0.08	
Brushy		07/18/85	Cocarp	M	8105	3005	61.0	0.10	9.4	8.3	1.1	
Brushy		07/18/85	Cocarp	M	8133	2626	58.0	0.26	1.05	0.88	0.17	
Brushy		07/18/85	Cocarp	M	8143	1656	51.5	0.21	0.63	0.49	0.24	
Brushy		07/18/85	Cocarp	F	8160	4374	63.9	0.16	1.61	0.96	0.65	
Brushy		07/18/85	Cocarp	M	8167	1027	40.2	0.16	1.5	1.2	0.30	
Brushy		07/18/85	Cocarp	F	8176	2161	50.3	0.23	0.22	0.15	0.07	
Brushy		07/18/85	Cocarp	F	8197	590	37.0	0.07	0.54	0.33	0.21	

Table C-2. Mercury, PCBs, and ^{137}Cs in fish from East Fork Poplar Creek and reference sites, December 1985–January 1985

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs			^{137}CS
									Total	1254	1260	
EFK23.4	23.4	12/03/85	Redbre	M	8106	68.3	15.8	1.2	0.51	0.36	0.15	
EFK23.4	23.4	12/03/85	Redbre	M	8115	76.4	16.0	1.0	0.31	0.21	0.10	9.4
EFK23.4	23.4	12/03/85	Redbre	F	8118	40.8	13.4	1.5	0.21	0.11	0.10	
EFK23.4	23.4	12/03/85	Redbre	F	8154	92.	16.1	1.2	1.03	0.62	0.41	6.2
EFK23.4	23.4	12/03/85	Redbre	F	8157	106.4	16.5	1.6	0.69	0.48	0.21	7.5
EFK23.4	23.4	12/03/85	Redbre	M	8183	40.9	13.6	0.7	0.32	0.21	0.11	
EFK23.4	23.4	12/03/85	Redbre	F	8185	78.7	15.4	1.7	0.40	0.25	0.15	
EFK23.4	23.4	12/03/85	Redbre	M	8393	75.3	15.5	1.2	0.44	0.21	0.23	<2.8
EFK18.2	18.2	12/03/85	Redbre	M	8315	93.9	16.8	0.61	0.46	0.13	0.33	
EFK18.2	18.2	12/03/85	Redbre	M	8320	45.2	14.8	1.1	0.21	0.11	0.10	
EFK18.2	18.2	12/03/85	Redbre	F	8331	62.3	14.6	0.54	0.20	<0.1	0.10	
EFK18.2	18.2	12/03/85	Redbre	M	8338	84.9	16.7	1.3	0.72	0.50	0.22	
EFK18.2	18.2	12/03/85	Redbre	F	8340	35.7	12.2	0.81	0.20	0.10	0.10	
EFK18.2	18.2	12/03/85	Redbre	F	8344	28.2	10.5	0.90	0.20	<0.1	0.10	
EFK18.2	18.2	12/03/85	Redbre	M	8354	35.2	12.5	0.57	0.39	0.13	0.16	
EFK18.2	18.2	12/03/85	Redbre	F	8381	37.2	13.0	0.79	<0.2	<0.1	<0.1	
EFK13.8	13.8	12/03/85	Redbre	M	8303	60.3	15.6	0.70	0.20	0.10	0.10	
EFK13.8	13.8	12/03/85	Redbre	M	8304	49.1	13.2	0.48	0.20	<0.1	0.10	
EFK13.8	13.8	12/03/85	Redbre	M	8309	43.9	13.9	0.91	0.20	0.10	0.10	
EFK13.8	13.8	12/03/85	Redbre	M	8325	88.1	17.1	0.57	0.20	<0.1	0.10	
EFK13.8	13.8	12/03/85	Redbre	M	8348	39.0	13.1	0.64	0.28	0.12	0.16	
EFK13.8	13.8	12/03/85	Redbre	M	8358	63.3	16.5	0.87	0.23	0.10	0.13	
EFK13.8	13.8	12/03/85	Redbre	M	8372	106.5	19.3	0.70	0.32	0.10	0.22	
EFK13.8	13.8	12/03/85	Redbre	M	8374	45.1	14.5	0.95	0.20	<0.1	0.10	
EFK6.3	6.3	12/09/85	Redbre	M	8101	48.8	14.2	0.42	0.25	0.13	0.12	
EFK6.3	6.3	12/09/85	Redbre	F	8137	44.5	13.4	0.58	<0.2	0.1	<0.1	
EFK6.3	6.3	12/09/85	Redbre	F	8146	55.7	14.4	0.73	<0.2	<0.1	<0.1	
EFK6.3	6.3	12/09/85	Redbre	F	8147	56.2	14.0	0.68	<0.2	<0.1	<0.1	
EFK6.3	6.3	12/09/85	Redbre	F	8152	41.2	13.4	0.47	0.20	0.1	0.10	
EFK6.3	6.3	12/09/85	Redbre	M	8156	43.0	13.4	0.75	<0.2	0.1	<0.1	
EFK6.3	6.3	12/09/85	Redbre	M	8189	52.6	15.0	0.42	0.20	0.10	0.10	
EFK6.3	6.3	12/09/85	Redbre	M	8190	52.8	14.0	0.44	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/09/85	Redbre	M	8102	114.6	18.1	0.29	0.20	0.10	0.10	
EFK2.1	2.1	12/09/85	Redbre	F	8112	48.8	14.0	0.52	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/09/85	Redbre	F	8132	58.0	14.5	0.53	0.47	0.21	0.26	<8.1
EFK2.1	2.1	12/09/85	Redbre	M	8144	140.8	20.2	0.62	0.52	0.42	<0.1	
EFK2.1	2.1	12/09/85	Redbre	M	8150	102.5	17.8	0.18	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/09/85	Redbre	F	8162	67.0	15.8	0.24	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/09/85	Redbre	M	8193	83.6	16.5	0.27	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/09/85	Redbre	F	8199	84.1	16.1	0.26	0.20	<0.1	0.10	

Table C-2 (continued)

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs Total	PCBs 1254	PCBs 1260	^{137}CS
Brushy		12/10/85	Redbre	M	8126	122.4	18.7	0.06	1.03	0.10	1.03	
Brushy		12/10/85	Redbre	F	8136	60.0	15.3	0.08	0.46	0.36	0.1	
Brushy		12/10/85	Redbre	M	8161	83.5	17.3	0.24	1.9	1.8	0.1	
Brushy		12/10/85	Redbre	M	8170	70.6	15.6	0.09	0.24	0.14	0.1	
Brushy		12/10/85	Redbre	M	8172	47.9	14.3	0.04	0.65	0.55	0.1	
Brushy		12/10/85	Redbre	F	8179	53.4	14.6	0.07	0.76	0.66	<0.1	
Brushy		12/10/85	Redbre	F	8187	45.0	14.1	0.09	0.50	0.40	0.1	
Brushy		12/10/85	Redbre	M	8188	113.2	17.8	0.09	0.10	<0.1	0.10	
EFK23.4	23.4	12/03/85	Blugil	M	8103	49.0	14.1	0.77	0.36	0.26	0.10	
EFK23.4	23.4	12/03/85	Blugil	M	8119	105.1	17.4	0.90	0.72	0.52	0.20	6.5
EFK23.4	23.4	12/03/85	Blugil	F	8122	56.8	14.5	0.87	0.34	0.16	0.18	
EFK23.4	23.4	12/03/85	Blugil	M	8145	89.9	17.4	0.67	0.53	0.40	0.13	3.2
EFK23.4	23.4	12/03/85	Blugil	M	8169	83.5	16.1	1.4	0.23	0.13	0.10	
EFK23.4	23.4	12/03/85	Blugil	M	8171	88.5	16.7	0.87	0.26	0.16	0.10	6.4
EFK23.4	23.4	12/03/85	Blugil	M	8177	89.0	16.6	0.78	0.23	0.10	0.33	6.2
EFK23.4	23.4	12/03/85	Blugil	M	8181	47.3	14.4	0.64	0.23	0.13	0.10	
EFK18.2	18.2	12/03/85	Blugil	F	8329	74.0	15.6	0.96	0.75	0.26	0.49	
EFK6.3	6.3	12/03/85	Blugil	M	8124	56.3	14.6	0.15	0.28	0.18	0.1	
EFK6.3	6.3	12/03/85	Blugil	M	8104	111.2	18.6	0.65	0.20	<0.1	0.10	
EFK2.1	2.1	12/03/85	Blugil	M	8100	111.1	17.6	0.16	0.20	0.10	0.10	9.4
EFK2.1	2.1	12/03/85	Blugil	M	8113	29.4	11.9	0.41	0.20	0.10	0.10	
EFK2.1	2.1	12/03/85	Blugil	M	8116	37.0	12.8	0.25	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/03/85	Blugil	M	8123	40.4	13.1	0.11	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/03/85	Blugil	M	8129	26.9	11.6	0.92	<0.2	<0.1	<0.1	
EFK2.1	2.1	12/03/85	Blugil	M	8164	38.3	13.3	0.08	0.20	0.10	<0.1	
EFK2.1	2.1	12/03/85	Blugil	M	8166	110.4	18.2	0.47	0.20	0.10	<0.1	<3.6
EFK2.1	2.1	12/03/85	Blugil	M	8191	113.8	18.4	0.61	0.21	0.10	0.11	<5.5
Brushy		12/10/85	Blugil	F	8110	49.9	13.9		1.6	1.5	<0.1	
Brushy		12/10/85	Blugil	M	8114	27.5	11.0		0.45	0.35	0.10	
Brushy		12/10/85	Blugil	M	8120	58.4	14.0	0.05	0.68	0.58	<0.1	
Brushy		12/10/85	Blugil	M	8155	64.4	14.5	0.06	0.20	0.10	0.10	
EFK13.8	13.8	01/08/86	Cocarp	M	8107	3198	66.2	0.97	2.0	0.38	1.6	
EFK13.8	13.8	01/08/86	Cocarp	M	8111		61.8	0.72	2.6	0.10	2.5	
EFK13.8	13.8	01/08/86	Cocarp	M	8148	2213	56.0	0.89	3.2	0.59	2.6	
EFK13.8	13.8	01/08/86	Cocarp	M	8149	1929	52.9	0.51	3.4	0.89	2.5	
EFK13.8	13.8	01/08/86	Cocarp	M	8159	3329	65.1	0.55	1.03	0.1	0.93	
EFK13.8	13.8	01/08/86	Cocarp	M	8165	2046	50.1	0.59	2.0	0.51	1.5	
EFK13.8	13.8	01/08/86	Cocarp	M	8168	2415	54.7	0.43	0.79	0.25	0.54	
EFK13.8	13.8	01/08/86	Cocarp	M	8186	2439	55.3	0.65	0.64	0.22	0.42	

Table C-2 (continued)

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs			^{137}CS
									Total	1254	1260	
EFK6.3	6.3	01/07/86	Cocarp	M	8006	2664	56.3	0.96	1.1	0.38	0.74	
EFK6.3	6.3	01/07/86	Cocarp	M	8007	2640	57.3	1.1	2.0	1.2	0.82	
EFK6.3	6.3	01/07/86	Cocarp	M	8014	2310	55.0	0.66	1.1	0.51	0.56	
EFK6.3	6.3	01/07/86	Cocarp	M	8032	1787	49.5	0.79	1.0	0.34	0.70	
EFK6.3	6.3	01/07/86	Cocarp	M	8036	2692	57.9	0.63	2.3	1.0	1.3	
EFK6.3	6.3	01/07/86	Cocarp	M	8051	2212	56.0	1.2	0.85	0.28	0.57	
EFK6.3	6.3	01/07/86	Cocarp	M	8075	1322	43.5	0.58	1.1	0.53	0.56	
EFK6.3	6.3	01/07/86	Cocarp	M	8097	2394	56.4	1.2	0.89	0.20	0.69	
EFK23.4	23.4	05/15/86	Redbre	F	8000	56.0	15.1	0.86	0.33	0.21	0.12	

Table C-3. Mercury, PCBs, and ^{137}Cs in fish from East Fork Poplar Creek and reference sites, May 1986

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs			^{137}CS
									Total	1254	1260	
EFK23.4	23.4	5/15/86	Redbre	M	8018	52.1	14.8	1.70	1.46	1.20	0.26	
EFK23.4	23.4	5/15/86	Redbre	M	8025	39.6	13.1	0.45	0.22	0.17	0.05	
EFK23.4	23.4	5/15/86	Redbre	M	8041	91.0	17.1	1.53	0.68	0.52	0.16	
EFK23.4	23.4	5/15/86	Redbre	M	8050	80.5	15.9		0.52	0.40	0.12	
EFK23.4	23.4	5/15/86	Redbre	M	8055	120.6	18.8	1.34	0.80	0.71	0.09	11
EFK23.4	23.4	5/15/86	Redbre	M	8058	103.9	17.8	1.56	1.33	1.10	0.23	<6.2
EFK23.4	23.4	5/15/86	Redbre	M	8078	112.0	17.7	0.75	0.34	0.15	0.19	
EFK23.4	23.4	5/15/86	Redbre	F	8086	37.8	13.1	1.31	0.55	0.34	0.21	
EFK18.2	18.2	5/20/86	Redbre	M	8002	144.6	18.5	0.61	0.13	0.08	0.05	
EFK18.2	18.2	5/20/86	Redbre	M	8008	157.8	18.5	0.51	0.36	0.17	0.19	
EFK18.2	18.2	5/20/86	Redbre	M	8017	101.4	17.2	0.71	0.27	0.12	0.15	
EFK18.2	18.2	5/20/86	Redbre	F	8039	89.3	16.4	0.94	0.07	0.03	0.04	
EFK18.2	18.2	5/20/86	Redbre	F	8057	69.7	14.7					<2.4
EFK18.2	18.2	5/20/86	Redbre	F	8066	65.5	14.8	1.08	0.08	0.06	0.02	
EFK18.2	18.2	5/20/86	Redbre	F	8077	74.0	16.1	1.29	0.24	0.13	0.11	
EFK18.2	18.2	5/20/86	Redbre	F	8098	89.6	15.6	0.86	0.24	0.10	0.14	
EFK18.2	18.2	5/20/86	Redbre	M	8099	115.2	17.5	0.94	0.28	0.15	0.13	7.8
EFK13.8	13.8	5/26/86	Redbre	F	0002	69.5	15.1	0.65	0.20	0.05	0.15	
EFK13.8	13.8	5/26/86	Redbre	M	0014	146.8	18.7	0.55	0.59	0.36	0.23	
EFK13.8	13.8	5/26/86	Redbre	M	0035	115.2	18.3	0.51	0.13	0.08	0.05	
EFK13.8	13.8	5/26/86	Redbre	M	0050	125.3	18.4	0.64	0.12	0.06	0.06	
EFK13.8	13.8	5/26/86	Redbre	F	0069	72.6	15.1	0.84	0.35	0.12	0.23	
EFK13.8	13.8	5/26/86	Redbre	F	0094	78.8	15.1	0.61	0.21	0.11	0.10	
EFK13.8	13.8	5/26/86	Redbre	M	7030	106.6	17.1	0.57	0.20	0.11	0.09	
EFK13.8	13.8	5/26/86	Redbre	M	8010	109.5	17.7	0.40	0.25	0.11	0.14	
EFK13.8	13.8	5/26/86	Redbre	F	8072	95.2	16.4	0.57				34
EFK6.3	6.3	5/19/86	Redbre	M	8005	104.8	17.5	0.29				38
EFK6.3	6.3	5/19/86	Redbre	M	8009	96.0	17.3	0.45	0.18	0.13	0.05	
EFK6.3	6.3	5/19/86	Redbre	M	8013	122.9	17.9	0.34	0.18	0.08	0.10	
EFK6.3	6.3	5/19/86	Redbre	M	8027	118.2	17.9	0.35	0.06	0.03	0.03	
EFK6.3	6.3	5/19/86	Redbre	M	8028	85.8	15.8		0.07	0.04	0.03	
EFK6.3	6.3	5/19/86	Redbre	M	8030	100.2	17.6	0.56	0.07	0.04	0.03	38
EFK6.3	6.3	5/19/86	Redbre	F	8043	63.8	14.7	0.53	0.15	0.09	0.06	
EFK6.3	6.3	5/19/86	Redbre	F	8045	56.5	14.4	0.40	0.41	0.16	0.25	43
EFK6.3	6.3	5/19/86	Redbre	M	8046	89.0	16.0	0.33	0.18	0.12	0.06	
EFK2.1	2.1	5/12/86	Redbre		7999	28.0	11.7	0.27	0.63	0.49	0.14	
EFK2.1	2.1	5/12/86	Redbre	F	8001	72.6	14.7	0.28	0.28	0.20	0.08	
EFK2.1	2.1	5/12/86	Redbre	M	8031	103.6	17.5	0.38	0.10	0.07	0.03	13
EFK2.1	2.1	5/12/86	Redbre	M	8042	154.5	19.7	0.40	0.10	0.04	0.06	
EFK2.1	2.1	5/12/86	Redbre	M	8071	91.6	16.5	0.30	0.14	0.07	0.07	
EFK2.1	2.1	5/12/86	Redbre	F	8080	82.3	15.2	0.35	0.20	0.14	0.06	
EFK2.1	2.1	5/12/86	Redbre	M	8084	91.9	16.1	0.28	0.15	0.12	0.03	
EFK2.1	2.1	5/12/86	Redbre	F	8089	76.9	15.4	0.47	0.17	0.08	0.09	

Table C-3 (continued)

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs Total	PCBs 1254	PCBs 1260	^{137}CS
Hindscr		5/13/86	Redbre	M	8012	34.4	11.9	0.13	0.09	0.05	0.04	
Hindscr		5/13/86	Redbre	F	8037	51.9	14.1	0.07	0.04	0.03	0.01	
Hindscr		5/13/86	Redbre	M	8047	64.1	14.9	0.03	0.02	0.02	0.00	
Hindscr		5/13/86	Redbre	M	8052	158.1	19.1	0.04	0.07	0.06	0.01	
Hindscr		5/13/86	Redbre	F	8062	35.7	11.9	0.14	0.03	0.02	0.01	<8.8
Hindscr		5/13/86	Redbre	F	8074	51.0	13.3	0.07	0.15	0.10	0.05	
EFK23.4	23.4	5/15/86	Blugil	M	8021	67.3	16.2	1.52				3.8
EFK23.4	23.4	5/15/86	Blugil	M	8026	91.7	17.3	0.38	0.14	0.09	0.05	
EFK23.4	23.4	5/15/86	Blugil	M	8049	66.1	15.1	0.22	0.14	0.08	0.02	
EFK23.4	23.4	5/15/86	Blugil	M	8064	86.1	16.4	0.74	0.55	0.40	0.15	
EFK23.4	23.4	5/15/86	Blugil	M	8073	68.4	15.9	0.66	0.68	0.52	0.16	
EFK23.4	23.4	5/15/86	Blugil	M	8081	94.6	17.2	0.60	0.11	0.06	0.05	
EFK23.4	23.4	5/15/86	Blugil	M	8093	76.2	16.1	0.84	0.09	0.06	0.03	
EFK23.4	23.4	5/15/86	Blugil	M	8094	66.1	16.0	0.81	0.74	0.51	0.23	<8.5
EFK23.4	23.4	5/15/86	Blugil	M	8096	60.7	14.6	0.88	0.28	0.22	0.06	
EFK18.2	18.2	5/20/86	Blugil	M	8076	99.5	17.4	0.40	0.07	0.05	0.02	
EFK13.8	13.8	5/26/86	Blugil	M	0030	120.0	18.0	0.47	0.16	0.06	0.10	
EFK2.1	2.1	5/12/86	Blugil	M	8015	91.2	16.3	0.06	0.08	0.05	0.03	
EFK2.1	2.1	5/12/86	Blugil	M	8048	79.8	14.9	0.12	0.06	0.04	0.02	
EFK2.1	2.1	5/12/86	Blugil	F	8054	57.5	13.6	0.42	0.22	0.16	0.06	
EFK2.1	2.1	5/12/86	Blugil	M	8059	81.6	15.8	0.27	0.11	0.08	0.03	
EFK2.1	2.1	5/12/86	Blugil	M	8060	48.8	13.1	0.38	0.14	0.10	0.04	
EFK2.1	2.1	5/12/86	Blugil	M	8063	85.9	17.1	0.47	0.23	0.09	0.14	0.21
EFK2.1	2.1	5/12/86	Blugil	M	8065	53.0	13.0	0.17	0.14	0.07	0.07	
EFK2.1	2.1	5/12/86	Blugil	F	8090	42.5	12.5	0.43	0.43	0.28	0.15	
Hindscr		5/13/86	Blugil	F	8004	72.4	14.6	0.06	0.09	0.08	0.01	
Hindscr		5/13/86	Blugil	M	8029	101.2	16.8	0.05	0.02	0.02	0.00	
Hindscr		5/13/86	Blugil	M	8033	74.3	15.5	0.06	0.11	0.10	0.01	
Hindscr		5/13/86	Blugil	M	8061	76.1	16.2	0.06	0.04	0.04	0.00	
Hindscr		5/13/86	Blugil	F	8067	47.2	13.0	0.06	0.05	0.03	0.02	
Hindscr		5/13/86	Blugil	F	8070	88.4	16.0	0.07	0.13	0.12	0.01	
Hindscr		5/13/86	Blugil	F	8082	37.9	12.5	0.06	0.10	0.08	0.02	
Hindscr		5/13/86	Blugil	M	8091	49.1	13.1	0.07	0.04	0.03	0.01	<6.5
EFK18.2	18.2	5/20/86	Cocarp	M	7022	3206	59.4	0.36	0.86	0.28	0.58	
EFK18.2	18.2	5/20/86	Cocarp	M	7026	2336	51.6	0.22	0.78	0.16	0.62	
EFK18.2	18.2	5/20/86	Cocarp	M	7047	3572	61.5	0.45	0.85	0.32	0.53	
EFK13.8	13.8	5/26/86	Cocarp	M	0024	2584	56.4	0.21	0.94	0.59	0.35	
EFK13.8	13.8	5/26/86	Cocarp	M	0042	2164	55.3	0.62	0.73	0.14	0.59	
EFK13.8	13.8	5/26/86	Cocarp	F	0045	2922	54.1	0.62	1.01	0.27	0.74	
EFK13.8	13.8	5/26/86	Cocarp	M	0053	2144	53.4	0.61	0.19	0.04	0.15	
EFK13.8	13.8	5/26/86	Cocarp	F	0070	3397	58.6	0.81	1.15	0.23	0.92	

Table C-3 (continued)

Site	Dist.	Date	Spp.	Sex	No.	Wt	Lgth	Hg	PCBs			^{137}CS
									Total	1254	1260	
EFK13.8	13.8	5/26/86	Cocarp	F	0097	3546	61.1	0.42	0.29	0.07	0.22	
EFK13.8	13.8	5/26/86	Cocarp	F	0116	2508	55.8	0.47	0.93	0.21	0.72	
EFK6.3	6.3	5/19/86	Cocarp	M	8016	1966	51.9	0.98	0.45	0.05	0.40	
EFK6.3	6.3	5/19/86	Cocarp	M	8020	1920	51.3	0.54	0.31	0.03	0.28	
EFK6.3	6.3	5/19/86	Cocarp	F	8024	2447	56.2	0.97	0.28	0.11	0.17	
EFK6.3	6.3	5/19/86	Cocarp	M	8034	2934	55.6	0.80	0.40	0.15	0.25	
EFK6.3	6.3	5/19/86	Cocarp	F	8069	1780	51.7	0.90	0.48	0.20	0.28	
EFK6.3	6.3	5/19/86	Cocarp	F	8079	2326	57.2	1.1	0.30	0.05	0.25	
EFK6.3	6.3	5/19/86	Cocarp	F	8088	2430	59.0	0.25	0.44	0.20	0.24	
EFK6.3	6.3	5/19/86	Cocarp	F	8092	3640	64.9	0.86	0.83	0.34	0.49	
EFK2.1	2.1	5/12/86	Cocarp	F	8011	1726	50.2	0.64	0.22	0.03	0.19	
EFK2.1	2.1	5/12/86	Cocarp	F	8022	1860	51.9	0.58	0.56	0.08	0.48	
EFK2.1	2.1	5/12/86	Cocarp	M	8023	2628	57.8	0.24	0.76	0.24	0.52	
EFK2.1	2.1	5/12/86	Cocarp	F	8040	2210	52.5	0.63	0.41	0.20	0.21	
EFK2.1	2.1	5/12/86	Cocarp	M	8053	1877	50.6	0.50	0.38	0.11	0.27	
EFK2.1	2.1	5/12/86	Cocarp	M	8056	1310	45.6	0.71	0.59	0.33	0.26	
EFK2.1	2.1	5/12/86	Cocarp	F	8068	1396	45.2	0.27	0.27	0.14	0.13	
EFK2.1	2.1	5/12/86	Cocarp	F	8085	2632	51.0	0.76	0.35	0.13	0.22	
Hindscr		5/13/86	Cocarp	F	8003	1430	46.2	0.12	0.78	0.04	0.74	
Hindscr		5/13/86	Cocarp	M	8019	788	38.7	0.13	3.2	<0.01	3.2	
Hindscr		5/13/86	Cocarp	F	8035	584	36.2	0.24	0.05	0.02	0.03	
Hindscr		5/13/86	Cocarp	F	8038	1430	46.2	0.16	0.06	0.01	0.05	
Hindscr		5/13/86	Cocarp	M	8044	985	42.4	0.14	0.06	0.04	0.02	
Hindscr		5/13/86	Cocarp	F	8083	1764	50.8	0.15	0.08	0.03	0.05	
Hindscr		5/13/86	Cocarp	F	8087	2508	56.2	0.11	0.07	<0.01	0.06	
Hindscr		5/13/86	Cocarp	M	8095	796	39.1	0.07	0.05	0.01	0.04	

Table C-4. Metals[a] (other than Hg) in sunfish (ppm, wet wt) from East Fork Poplar Creek and Hinds Creek, a reference site

Site	Spp.	Tag	Date	Sex	Wt	Lgth	Cd	Cr	Cu	Pb	Se	Zn
EFK23.4	Redbre	7158	06/17/86	M	77.4	16.4	0.013	<0.1	0.33	<0.02	0.83	3.5
EFK23.4	Redbre	7176	06/17/86	F	49.2	14.3	0.016	<0.1	0.52	0.02	0.62	11
EFK23.4	Redbre	7179	06/17/86	M	99.6	16.7	0.019	<0.1	1.2	0.11	0.70	8.1
EFK23.4	Redbre	7196	06/17/86	M	65.4	15.5	0.011	<0.1	3.5	0.06	0.72	6.5
EFK23.4	Redbre	7276	06/17/86	M	97.1	16.7	0.040	<0.1	1.4	<0.02	0.84	5.7
EFK23.4	Blugil	7137	06/17/86	M	53.9	14.6	0.006	0.15	0.51	<0.02	1.0	9.9
EFK23.4	Blugil	7174	06/17/86	M	56.9	15.0	0.015	0.15	0.58	<0.02	0.78	4.8
EFK23.4	Blugil	7190	06/17/86	M	59.1	14.9	0.020	<0.1	12.3	0.12	0.78	8.1
EFK23.4	Blugil	7199	06/17/86	F	37.0	12.9	0.011	0.12	4.3	<0.02	0.84	8.5
EFK23.4	Blugil	7241	06/17/86	M	81.8	16.6	0.007	<0.1	1.1	<0.02	0.67	9.5
Hindscr	Redbre	8037	05/13/86	F	51.9	14.1	0.035	<0.1	1.3	<0.02	0.73	7.4
Hindscr	Blugil	8070	05/13/86	F	88.4	16.0	0.006	<0.1	0.22	<0.02	0.50	4.9
Hindscr	Blugil	6930	01/21/87	M	83.5	16.9	0.004	0.17	0.10	<0.02	0.50	5.8
Hindscr	Blugil	6995	01/21/87	M	85.2	16.5	0.013	0.16	0.17	<0.02	0.46	6.5

[a]Other metals below detection limit in all samples [detection limit as ppm (μg/g) wet wt in paretheses]: Ag (<0.1), As (<0.05), Be (<0.05), Li (0.5), Ni (<1), Sb (<0.3), and Tl (<0.2).

Table C-5. GC/MS and HPLC/fluorescence analyses of organics in fish (ppm, wet wt) from East Fork Poplar Creek and Hinds Creek, a reference site

Site	Spp.	Tag	Date	Sex	Wt	Lgth	Compounds Detected	ppm wet wt
EFK 23.4	Redbre	7179	06/17/86	M	99.6	16.7	None	
EFK 23.4	Redbre	7136	06/17/86	M	77.4	16.4	None	
EFK 23.4	Redbre	7177	06/17/86	M	118.8	18.8	None	
EFK 23.4	Redbre	7194	06/17/86	M	82.9	16.7	Benzo(*k*)fluoranthene	0.014
EFK 23.4	Redbre	7198	06/17/86	M	63.6	14.8	None	
EFK 23.4	Blugil	7241	06/17/86	M	81.8	16.6	None	
EFK 23.4	Blugil	7118	06/17/86	M	69.9	15.4	Benzo(*a*)pyrene	0.026
							Benzo(*ghi*)perylene	0.077
EFK 23.4	Blugil	7156	06/17/86	M	92.7	16.7	None	
EFK 23.4	Blugil	7178	06/17/86	M	79.5	16.4	None	
EFK 23.4	Blugil	7238	06/17/86	M	77.9	16.5	None	
Duplicate		7177					None	
Duplicate		7178					Benzo(*a*)anthracene	0.023
Hindscr	Blugil	8061	05/13/86	M	76.1	16.2	None	
Hindscr	Redbre	8047	05/13/86	M	64.1	14.9	None	

Table C-6. Detection limits of compounds screened for by GC/MS and HPLC fluorescence in East Fork Poplar Creek and Hinds Creek, a reference site, samples [ppm (μg/g) wet wt]

	GC/MS	HPLC/fluorescence
Phenol	<0.4	
Bis(2-chloroethyl)ether	<0.4	
2-Chlorophenol	<0.4	
1,3-Dichlorobenzene	<0.4	
1,4-Dichlorobenzene	<0.4	
Benzylalcohol	<0.4	
1,2-Dichlorobenzene	<0.4	
2-Methylphenol	<0.4	
Bis(2-chlorodisopropyl)Ether	<0.4	
4-Methylphenol	<0.4	
N-nitroso-di-N-propylamine	<0.4	
Hexachloroethane	<0.4	
Nitrobenzene	<0.4	
Isophorone	<0.4	
2-Nitrophenol	<0.4	
2,4-Dimethylphenol	<0.4	
Benzoic acid	<2	
Bis(2-chloroethoxy)methane	<0.4	
2,4-Dichlorophenol	<0.4	
1,2,4-Trichlorobenzene	<0.4	
Naphthalene	<0.4	<1
4-Chloroaniline	<0.4	
Hexachlorobutadiene	<0.4	
4-Chloro-3-methylphenol	<0.4	
2-Methylnaphthalene	<0.4	
Hexachlorocyclopentadiene	<0.4	
2,4,6-Trichlorophenol	<0.4	
2,4,5-Trichlorophenol	<2	
2-Chloronaphthalene	<0.4	
2-Nitroaniline	<2	
Dimethylphthalate	<0.4	
Acenaphthalene	<0.4	
3-Nitroaniline	<2	
Acenaphthene	<0.4	
2,4-Dinitrophenol	<2	
Nitrophenol	<2	
Dibenzofuran	<0.4	
2,4-Dinitrotoluene	<0.4	
2,6-Dinitrotoluene	<0.4	
Diethylphthalate	<0.4	
4-Chlorophenyl-phenylether	<0.4	
Fluorene	<0.4	
4-Nitroaniline	<2	
4,6-Dinitro-2-methylphenol	<2	
N-Nitrosodiphenylamine	<0.4	
4-Brompphenyl-phenylether	<0.4	
Hexachlorobenzene	<0.4	

Table C-6 (continued)

	GC/MS	HPLC/fluorescence
Pentachlorophenol	<2	
Phenanthrene	<0.4	<0.3
Anthracene	<0.4	<1.0
Di-N-butylphthalate	<0.4	
Fluoranthene	<0.4	<0.1
Pyrene	<0.4	<0.2
Butylbenzylphthalate	<0.4	
3,3-Dichlorobenzidene	<2	
Benzo(*a*)anthracene	<0.4	<0.02
Bis(2-ethylhexyl)phthalate	<0.4	
Chrysene	<0.4	<0.1
Di-N-oxtylphthalate	<0.4	
Benzo(*b*)fluoranthene	<0.4	<0.03
Benzo(*k*)fluoranthene	<0.4	<0.01
Benzo(*a*)pyrene	<0.4	<0.01
Indeno(1,2,3-*cd*)pyrene	<0.4	<0.2
Dibenz(*a,h*)anthracene	<0.4	<0.04
Benzo(*g,h,i*)perylene	<0.4	<0.04

Table C-7. List of descriptors used to identify entries in Tables C-1 through C-3

Descriptor	Definition
Site	EFK - East Fork Poplar Creek Beavr - Beaver Creek Hinscr - Hinds Creek Brushy - Brushy Fork
Dist.	Distance above confluence with Poplar Creek
Spp.	Species: blugil-bluegill sunfish, *Lepomis macrochirus*, redbre-redbreast sunfish, *Lepomis auritus*, cocarp-carp, *Cyprinus carpio*
No.	Tag number of individual fish
Sex	M-Male, F-Female
Wt	Fish weight, grams
Lgth.	Fish total length, centimeters
Total	Total PCBs (sum of PCB-1254 and 1260) in fish axial muscle, micrograms per gram (ppm), wet weight
1254	PCB-1254 (arochlor 1254) in fish axial muscle, micrograms per gram (ppm), wet weight
1260	PCB-1260 (arochlor 1260) in fish axial muscle, microgram per gram (ppm), wet weight
^{137}Cs	Cesium-137 in fish axial muscle, bequerels per kilogram, wet weight
Ag	Silver in fish axial muscle, micrograms per gram (ppm), wet weight
As	Arsenic in fish axial muscle, micrograms per gram (ppm), wet weight
Be	Beryllium in fish axial muscle, micrograms per gram (ppm), wet weight
Cd	Cadmium in fish axial muscle, micrograms per gram (ppm), wet weight
Cr	Chromium in fish axial muscle, micrograms per gram (ppm), wet weight
Cu	Copper in fish axial muscle, micrograms per gram (ppm), wet weight
Hg	Mercury in fish axial muscle, micrograms per gram (ppm), wet weight
Li	Lithium in fish axial muscle, micrograms per gram (ppm), wet weight
Ni	Nickel in fish axial muscle, micrograms per gram (ppm), wet weight
Pb	Lead in fish axial muscle, micrograms per gram (ppm), wet weight

Table C-7 (continued)

Descriptor	Definition
Sb	Antimony in fish axial muscle, micrograms per gram (ppm), wet weight
Se	Selenium in fish axial muscle, micrograms per gram (ppm), wet weight
Tl	Thallium in fish axial muscle, micrograms per gram (ppm), wet weight
Zn	Zinc in fish axial muscle, micrograms per gram (ppm), wet weight

Appendix D

CHECKLIST OF FISH SPECIES COLLECTED FROM EAST FORK POPLAR CREEK AND Brushy FORK, MAY 1985–JUNE 1986

Table D-1. Checklist of fish species collected from East Fork Poplar Creek and Brushy Fork, May 1985–June 1986

Scientific name[a]	Common name	Site[b]					
		East Fork Poplar Creek (km)					Brushy Fork (km)
		23.4	18.2	13.8	10.0	6.3	7.6
Catostomidae							
Catostomus commersoni	White sucker	X	X	X	X	X[c]	X
Hypentelium nigricans	Northern hog sucker				X	X	X
Ictiobus bubalus	Smallmouth buffalo					*c*	
Minytrema melanops	Spotted sucker			X	X	X	X
Moxostoma duquesnei	Black redhorse				X	X	X
M. erythrurum	Golden redhorse				X[c]	X	
Moxostoma sp.	Unidentified *Moxostoma*		X		X		
Centrarchidae							
Ambloplites rupestris	Rock bass			X	X	X	X
Lepomis auritus	Redbreast sunfish	X	X	X	X	X	X
L. cyanellus	Green sunfish	X[c]		X[c]		X	X
L. gulosus	Warmouth			X	X	X	X
L. macrochirus	Bluegill	X	X	X	X	X	X
Lepomis sp.	Hybrid sunfish	X[c]	X[c]			X[c]	
Micropterus punctulatus	Spotted bass						X[c]
M. salmoides	Largemouth bass	X[c]	X		X	X[c]	X
Clupeidae							
Dorosoma cepedianum	Gizzard shad	X[c]	X	X		X	X
Cottidae							
Cottus carolinae	Banded sculpin			X	X	X	X
Cyprinidae							
Campostoma anomalum	Stoneroller	X	X	X	X	X	X
Cyprinella spiloptera	Spotfin shiner						X
Cyprinus carpio	Carp	X		X	X[c]	X	X[c]
Luxilus chrysocephalus	Striped shiner	X	X	X	X	X	X
Lythrurus ardens	Rosefin shiner						X
N. atherinoides	Emerald shiner				X[c]	X	X
Notropic amblops	Bigeye chub						X
Phoxinus tennesseensis	Tennessee dace				X		
Pimephales notatus	Bluntnose minnow	X	X	X	X	X[c]	
Rhinichthys atratulus	Blacknose dace	X	X	X	X	X	X
Semotilus atromaculatus	Creek chub	X	X	X	X	X[c]	X

Table D-1 (continued)

Scientific name[a]	Common name	Site[b] East Fork Poplar Creek (km) 23.4	18.2	13.8	10.0	6.3	Brushy Fork (km) 7.6
Ictaluridae							
Ameiurus natalis	Yellow bullhead					X	
Ictalurus punctatus	Channel catfish					X[c]	
Percidae							
Etheostoma blennioides	Greenside darter						X
E. jessiae	Blueside darter						X
E. kennicotti	Stripetail darter				X		
E. simoterum	Snubnose darter				X	X	X
E. duryi[d]	Black darter					X	
Perca flavescens	Yellow perch				X[c]	X[c]	
Percina caprodes	Logperch					X	X
Petromyzontidae							
Ichthyomyzon castaneus	Chestnut lamprey						X
Lampetra appendix	American brook lamprey						X
Poeciliidae							
Gambusia affinis	Mosquitofish	X	X		X	X	
Sciaenidae							
Aplodinotus grunniens	Freshwater drum			X	X	X	X
Number of species collected[e]		12	12	16	23	28	32

[a]Species names are based on Bailey et al (1970).

[b]Sites are identified by the distance (km) above the confluence of the stream with Poplar Creek.

[c]Collected only in qualitative sampling conducted in May/June 1985 and January 1986.

[d]Identification confirmed by D. A. Etnier, Department of Zoology, University of Tennessee (M. G. Ryon, ORNL-ESD, personal communication, 1986).

[e]Excluding *Moxostoma* sp. at EFK 6.3 and all *Lepomis* sp.

Appendix E

DENSITY AND BIOMASS OF FISH IN EAST FORK POPLAR CREEK AND Brushy FORK, MAY 1985–JUNE 1986

Table E-1. Density (number of individuals/10 m^2) and biomass (g/10 m^2, in parentheses) of fishes collected by quantitative electroshocking at three sites on upper East Fork Poplar Creek, May 1985–July 1986

Site/species	Sampling date				
EFK 24.4[a,b]	6-12-85	11-6-85	3-10-86	7-16-86	
	----------No fish collected----------				
Stream sampling section					
Length, m	221	227	225	226	
Mean width, m	4.4	4.2	4.4	4.2	
Mean depth, cm	25	23	23	23	
EFK 23.4[a]	5-14-85	10-22-85	1-29-86	3-11-86	5-15-86
White sucker		0.13 (3.68)	0.14 (4.78)	0.21 (8.98)	0.09 (4.99)
Bluegill	0.34 (27.94)	0.17 (16.37)	0.15 (7.35)	0.41 (19.17)	0.27 (17.19)
Redbreast sunfish	0.49 (18.90)	0.67 (22.10)	0.22 (7.56)	0.56 (19.44)	0.30 (14.31)
Blacknose dace	0.02 (0.12)		0.67 (2.66)	1.27 (6.22)	0.02 (0.10)
Carp	0.04 (43.33)				
Creek chub		3.32 (33.44)	0.50 (0.66)	0.84 (9.82)	0.15 (2.20)
Stoneroller		0.11 (1.82)		0.19 (4.01)	
Striped shiner	0.13 (2.71)	3.44 (23.86)		1.50 (14.66)	0.84 (7.88)
Mosquitofish				0.02 (< 0.01)	
Stream sampling section					
Length, m	113	117	117	116	117
Mean width, m	4.7	4.5	5.1	4.9	4.5
Mean depth, cm	26	26	29	27	27

[a]Sites EFK 24.4 and EFK 23.4 are above and below New Hope Pond, respectively.
[b]Not sampled in January 1986.

Table E-2. Density (number of individuals/10 m^2) and biomass (g/10 m^2, in parentheses) of fishes collected by quantitative electroshocking at EFK 18.2, June 1985–June 1986

Species	Sampling date				
	6-17-85	10-29-85	1-30-86	3-17-86	6-4-86
White sucker	0.29 (0.25)				1.31 (1.09)
Unident. *Moxostoma*	0.03 (0.02)				
Bluegill	0.03 (1.78)				
Largemouth bass	0.02 (2.84)				
Redbreast sunfish	0.11 (8.48)	0.25 (0.30)	0.08 (0.10)	0.11 (0.11)	0.21 (3.93)
Gizzard shad					0.02 (1.84)
Blacknose dace	1.83 (0.69)	0.59 (1.45)	0.57 (1.25)	0.14 (0.21)	2.40 (2.71)
Bluntnose minnow	0.06 (0.02)				
Creek chub	0.58 (0.22)	0.07 (0.16)			0.51 (0.23)
Stoneroller	2.34 (1.76)				5.35 (2.35)
Striped shiner	0.73 (0.18)	1.10 (3.66)	0.21 (0.41)	0.14 (0.74)	0.70 (3.22)
Mosquitofish	2.02 (1.27)	3.74 (0.82)	4.13 (1.16)	2.36 (1.01)	2.07 (1.93)
Stream sampling section					
Length, m	116	122	127	128	128
Mean width, m	8.5	7.9	8.3	8.6	8.3
Mean depth, cm	20	20	20	22	19

Table E-3. Density (number of individuals/10 m^2) and biomass (g/10 m^2, in parentheses) of fishes collected by quantitative electroshocking at EFK 13.8, May 1985–June 1986

Species	Sampling date			
	5-15-85	10-23-85	3-3-86	6-5-86
Spotted sucker				0.01 (0.46)
White sucker				0.14 (0.11)
Bluegill	0.05 (2.63)			
Redbreast sunfish	1.02 (55.66)	0.19 (10.30)	0.18 (10.17)	0.24 (17.37)
Rock bass	0.03 (0.97)	0.01 (0.52)	0.03 (0.93)	0.04 (2.17)
Warmouth	0.05 (4.11)		0.01 (0.71)	0.02 (2.32)
Gizzard shad	0.05 (4.55)			
Banded sculpin			0.03 (0.11)	
Blacknose dace	0.15 (0.50)	0.05 (0.19)	0.07 (0.29)	0.25 (0.77)
Bluntnose minnow		0.01 (0.05)		
Carp	0.07 (206.15)		0.01 (33.67)	
Creek chub		0.03 (0.12)		0.02 (0.31)
Stoneroller	0.42 (4.26)	0.79 (6.91)	0.41 (1.44)	1.19 (7.07)
Striped shiner	0.38 (3.84)	0.54 (4.18)	0.41 (8.16)	0.30 (2.92)
Freshwater drum	0.01 (4.03)			0.02 (17.74)
Stream sampling section				
Length, m	94	101	99	104
Mean width, m	7.9	8.0	7.8	7.7
Mean depth, cm	41	45	45	39

Table E-4. Density (number of individuals/10 m^2) and biomass (g/10 m^2, in parentheses) of fishes collected by quantitative electroshocking at EFK 10.0, June 1985–June 1986

Species	Sampling date			
	6-19-85	11-13-85	3-4-86	6-14-86
Black redhorse			0.02 (9.47)	
Northern hog sucker	0.02 (3.98)	0.04 (3.51)	0.04 (0.68)	0.06 (5.08)
Spotted sucker			0.05 (0.56)	
White sucker			0.07 (0.32)	0.05 (1.19)
Bluegill	0.05 (4.94)	0.04 (4.03)	0.01 (1.99)	
Largemouth bass	0.02 (2.40)	0.01 (0.08)		0.04 (8.57)
Redbreast sunfish	0.91 (43.43)	0.07 (3.60)	0.06 (3.77)	0.21(17.59)
Rock bass		0.10 (7.93)	0.05 (2.87)	0.08 (6.10)
Warmouth				0.03 (0.80
Banded sculpin	2.58 (2.63)	0.67 (1.55)	0.55 (2.25)	0.12 (0.81)
Blacknose dace	0.06 (0.09)	0.43 (0.74)	0.23 (0.75)	0.14 (0.53)
Bluntnose minnow		0.03 (0.05)	0.05 (0.18)	
Creek chub	0.01 (< 0.01)		0.01 (0.02)	0.05 (0.02)
Tennessee dace				0.01 (0.01)
Stoneroller	0.84 (1.38)	1.13 (5.36)	0.72 (6.94)	2.62 (6.10)
Striped shiner	0.02 (0.27)	2.53 (9.18)	0.15 (1.36)	0.07 (4.60)
Stripetail darter			0.02 (0.03)	
Snubnose darter			0.05 (0.07)	
Mosquitofish				0.01 (0.01)
Freshwater drum		0.01 (2.26)		0.01 (1.87)
Stream sampling section				
Length, m	118	116	116	118
Mean width, m	7.2	6.7	7.1	6.6
Mean depth, cm	56	57	60	49

Table E-5. Density (number of individuals/10 m^2) and biomass (g/10 m^2, in parentheses) of fishes collected by quantitative electroshocking at EFK 6.3, June 1985–June 1986

Species	Sampling date			
	6-14-85	10-30-85	3-5-86	6-26-86
Black redhorse			0.04 (38.62)	
Northern hog sucker		0.02 (2.84)		
Spotted sucker			0.12 (1.16)	0.05 (12.11)
Unidentified *Moxostoma*	0.01 (4.35)			
Bluegill	0.01 (0.98)			
Green sunfish	0.01 (0.08)			
Redbreast sunfish	0.27 (9.88)	0.15 (7.96)	0.07 (4.09)	0.08 (5.10)
Rock bass		0.04 (1.31)		0.02 (0.27)
Warmouth		0.02 (0.77)		
Gizzard shad		0.07 (9.43)		0.03 (2.79)
Banded sculpin		0.09 (0.26)	0.06 (0.30)	0.02 (0.10)
Blacknose dace				0.02 (0.08)
Bluntnose minnow		0.11 (0.18)	0.03 (0.06)	
Carp		0.02 (45.85)	0.01 (36.12)	
Emerald shiner		0.04 (0.19)	0.02 (0.06)	0.01 (0.12)
Stoneroller	0.05 (0.32)	1.18 (4.54)	0.09 (0.87)	0.54 (0.68)
Striped shiner	0.01 (0.38)	0.12 (1.22)	0.10 (0.11)	0.03 (0.28)
Yellow bullhead		0.02 (0.30)		0.02 (0.95)
Logperch		0.01 (0.17)	0.03 (0.44)	
Snubnose darter		0.03 (0.03)	0.03 (0.04)	
Mosquitofish		0.01 (< 0.01)		
Freshwater drum		0.03 (5.44)		
Stream sampling section				
Length, m	100	109	101	98
Mean width, m	9.3	9.0	9.6	9.3
Mean depth, cm	56	54	60	50

Table E-6. Density (number of individuals/10 m^2) and biomass (g/10 m^2, in parentheses) of fishes collected by quantitative electroshocking in Brushy Fork (BFK 7.6), November 1985–June 1986

Species	Sampling date		
	11-8-85	3-6-85	6-25-86
Black redhorse	0.15 (33.82)	0.08 (22.41)	0.19 (58.74)
Golden redhorse	0.11 (32.65)		0.06 (15.14)
Northern hog sucker	0.16 (11.29)	0.06 (2.99)	0.19 (13.73)
Spotted sucker	0.05 (16.28)	0.02 (6.70)	0.11 (24.20)
White sucker	0.10 (17.31)	0.07 (10.45)	0.25 (21.21)
Bluegill	0.05 (0.81)		0.21 (8.61)
Green sunfish	0.02 (0.13)		
Largemouth bass			0.05 (0.42)
Redbreast sunfish	0.24 (9.17)	0.16 (7.39)	0.22 (7.13)
Rock bass	0.29 (26.01)	0.14 (13.76)	0.20 (12.18)
Warmouth			0.02 (2.68)
Gizzard shad			0.06 (4.88)
Banded sculpin	1.56 (5.98)	3.44 (14.76)	0.97 (3.96)
Blacknose dace	0.16 (0.13)	0.01 (0.01)	0.21 (0.29)
Bigeye chub	0.32 (0.48)	0.05 (0.06)	0.22 (0.30)
Creek chub	0.03 (0.04)		0.11 (0.51)
Emerald shiner		0.01 (< 0.01)	
Rosefin shiner			0.02 (0.03)
Spotfin shiner			0.13 (0.21)
Stoneroller	0.72 (8.13)	0.15 (4.88)	0.97 (7.30)
Striped shiner	2.25 (11.85)	0.22 (7.17)	1.10 (8.30)
Blueside darter	0.10 (0.12)	0.08 (0.14)	0.02 (0.02)
Greenside darter	0.02 (0.38)		
Logperch	0.02 (0.19)	0.03 (0.39)	
Snubnose darter	2.61 (2.35)	0.55 (0.63)	0.17 (0.18)
American brook lamprey	0.03 (0.30)		0.04 (0.38)
Chestnut lamprey	0.02 (0.20)		
Petromyzontidae	0.08 (0.67)	0.07 (0.71)	
Freshwater drum			0.01 (5.91)
Stream sampling section			
Length, m	116	118	118
Mean width, m	8.3	8.7	7.6
Mean depth, cm	39	49	32

Appendix F

OFF-SITE FISH KILLS

Appendix F

OFF-SITE FISH KILLS

Fish kills occurred in East Fork Poplar Creek (EFPC) in July 1985 and in a tributary of EFPC in December 1985. These two events are described below.

F.1. JULY 1985 FISH KILLS

F.1.1 Field Surveys

F.1.1.1 Friday, July 19, 1985

J. M. Loar of the Environmental Sciences Division (ESD), Oak Ridge National Laboratory (ORNL), was notified of a fish kill on EFPC by R. H. Kingrea of the Department of Environmental Management (DEM), Y-12 Plant, at 1630 h on July 19, 1985. Loar proceeded to EFPC at the Route 95 bridge near Jefferson Avenue and surveyed approximately 200- and 100-m sections of the stream above and below the bridge, respectively. Dead and distressed fish of at least six species and a wide range of sizes (10 to >1000 g) were observed. One large white sucker, one large redhorse, four large redbreast sunfish, and several small stonerollers were collected at the site and preserved on ice. Loar surveyed six additional sites on EFPC upstream of the Route 95 bridge (Table F-1) before returning to ORNL at 1830 h to store the fish samples in a freezer.

C. C. Coutant of the ESD was also notified of the fish kill at 1800 h on July 19, 1985, by T. R. Butz, Head of the Y-12 Plant DEM. Coutant arrived at the stream at ~1900 h and observed numerous dead fish behind K-Mart and the NOAA Atmospheric Turbulence and Diffusion Laboratory (ATDL) near EFK 22.0. A small number of dead fish was found downstream of the office of Dr. S. G. Hammons, but none was observed above a temporary culvert across the creek near EFK 22.5 at the northern (or downstream) end of the adjacent vacant lot where recent road construction activity was evident. A volatile organic smell was apparent in the air above riffle areas behind ATDL. He observed no dead fish at the Bear Creek Road bridge across EFPC or immediately above the bridge in ~15 min of searching.

Table F-1. Observations of J. M. Loar, ORNL Environmental Sciences Division, of East Fork Poplar Creek (EFPC) fish kill between 1700 and 1830 h on July 19, 1985

Location	EFPC site (km)[a]	Length of stream walked (m)[b]	Observation
Route 95 bridge near Jefferson Avenue (1700 h)	18.2	200 U 100 D	Dead and dying fish of several species representing all sizes
Vanderbilt Drive	20.4	None; observations made from bridge	Several dead fish
Tulsa Road bridge[c]	21.4	50 D	Several dead fish (stonerollers)
Behind office of S. G. Hammons, DDS	22.6	200 D (walked stream from dentists office to ~100 m below culvert on EFPC behind adjacent vacant lot	Same as above
Scarboro Road guard station	22.8	50 U	No dead fish; several live fish

[a]Distance above the mouth of the stream.
[b]U = upstream; D = downstream.
[c]Southernmost bridge.

F.1.1.2 Saturday, July 20, and Monday, July 22, 1985

Personnel from the Y-12 Plant DEM (J. D. Gass, R. R. Hodge, and J. L. Murphy) monitored pH, temperature, and dissolved oxygen at numerous sites in EFPC between New Hope Pond (NHP) and the City of Oak Ridge Wastewater Treatment Facility (ORWTF) in the afternoons of July 20, 1985, and July 22, 1985. On both dates, the three parameters were within normal ranges at all sites on the stream.

F.1.1.3 Tuesday, July 23, 1985

At ~0900 h on July 23, ESD staff (C. C. Coutant, D. K. Cox, A. J. Gatz, W. C. Kyker, and J. G. Smith) started electroshocking sites on EFPC where sampling had been conducted between May and July 1985 as part of the Y-12 Plant Biological Monitoring

and Abatement Program (BMAP). The purpose of the survey was to assess the impact of the July 19 fish kill on the BMAP study sites. While conducting this survey, they observed that another fish kill was occurring and contacted C. W. Gehrs (ESD) at ~1100 h with this information. Results of the electroshocking surveys conducted on July 23, 1985, are described below.

East Fork Poplar Creek kilometer (EFK) 18.2 to EFK 18.8

Approximately 300 m of EFPC on either side of Route 95 bridge at Jefferson Avenue was sampled by electroshocking between 0900 and 1100 h Numerous freshly dead and distressed fish were observed. Some live fish were collected in the main channel, but most live fish were found congregated in the mouths of tributaries at abnormally high densities (the fish were packed together so tightly that their sides touched and their backs were out of the water). Far fewer fish were observed than were found in a previous quantitative survey of the same reach on June 17, 1985 (Table E-2). In the judgment of the survey crew, this reach of EFPC was affected by an ongoing fish kill.

EFK 23.3 to EFK 23.6

A 300-m reach of EFPC, extending from 100 m below the railroad crossing upstream to ~100 m below NHP, was electroshocked between 1130 and 1230 h. The survey revealed an abundance of live fish, and no dead or distressed fish were observed in this reach of stream. A high proportion of the fish tagged in a July 1985 study of fish movements in EFPC were recovered. There was no evidence of a fish kill in this reach of stream, and fish abundance appeared similar to that observed in a quantitative survey of this site on May 14, 1985 (Table E-1).

EFK 21.2 to EFK 22.8

A 1.6-km reach of EFPC between the Tuskegee Drive bridge and the Scarboro Road guard station was electroshocked between 1300 and 1600 h. Live and distressed fish were found in backwater areas and tributary mouths, and freshly dead fish were observed in the main channel. An odor of volatile organics was present in riffle areas. A pool at the mouth of a tributary at the southern (upstream) boundary of Dean Stallings Ford (EFK 22.4) contained several live sunfish. No live fish were found between this site and just below a temporary culvert located ~100 m upstream. Live fish were observed in an

eddy immediately below the culvert, and a normal abundance of live, healthy fish was observed in the main channel above the culvert. The change in fish abundance was noticeably abrupt near the southern (upstream) boundary of Oak Ridge Storage located just downstream of the culvert. Herbicide spraying was observed at Oak Ridge Storage during the electroshocking survey, and foliage along a perimeter fence parallel to EFPC was coated with an oily residue.

F.1.1.4 Wednesday, July 24, 1985

Normal abundances of fish were observed by electrofishing a 600-m section of stream immediately above the ORWTF outfall at EFK 13.4. (A 100-m reach of EFPC at EFK 13.8 had been quantitatively sampled in May 1985.) Decomposed carcasses of fish killed on previous dates were observed. These fish probably drifted into this reach from upstream. It appears that the fish kill did not adversely affect EFPC this far downstream.

F.1.2 Investigations of Fish Pathology

F.1.2.1 Histopathology

Histopathological examinations were conducted on fish collected dead by J. M. Loar on July 19 at EFK 18.2 and fish collected alive by S. M. Adams and B. D. Jimenez of the ESD on July 23 at EFK 23.5 (just below NHP and outside the kill area) and EFK 22.4 (behind Dean Stallings Ford and within the kill area). Gills, liver, and heart were removed from three fish at each site except EFK 18.2 (N = 2), preserved in 10% formalin with 0.1 M phosphate buffer, and shipped to Dr. David Hinton, Department of Pathology, University of West Virginia School of Medicine, for analysis. Results of the histopathological examination indicated high levels of parasite infestation in target organs of fish from both the kill area and the unaffected reach of stream. Observations by Adams and Jimenez of fish collected dead on July 19 and alive on July 23 at EFK 22.4 indicated lesions and/or parasites in kidneys, heart, and liver. Post-mortem changes in gills and liver as a result of freezing interfered with the histopathological examination and interpretation. Effects of the possible toxicant could not be distinguished from these post-mortem effects.

F.1.2.2 Blood chemistry

Levels of serum glutamate oxaloacetate transaminase (SGOT), glucose, sodium, cholesterol, triglycerides, and potassium were measured in fish collected alive on July 23 from (1) the kill area (EFK 21.2 to EFK 22.4; N = 10), (2) just below New Hope Pond (EFK 23.5; N = 7), and immediately above the ORWTF (EFK 13.5; N = 5). Only the level of cholesterol was significantly different among fish from the three sites. Although the fish from the kill area had lower cholesterol levels, such a reduction is not indicative of a specific toxin or toxicological effect.

Mixed function oxidase activities were also measured in livers of five redbreast sunfish collected alive from the site of the fish kill (EFK 22.4), but no difference in activities was observed between these fish and those collected in May 1985 as part of the Y-12 Plant BMAP.

F.1.3 Toxicity Tests

Toxicity tests were conducted by J. M. Giddings, J. L. Keller, W. K. Roy, and D. K. Whitmore of the ESD on water samples from several sites on EFPC (Table F-2). Laboratory personnel detected the odor of volatile organics in the water samples on July 23 and July 24 but not on subsequent days.

Survival was unaffected in all water samples, whereas growth (difference between initial and final weight over the 7-d test) was reduced in all samples by 13% to 28% compared with controls (dechlorinated tap water). The effects on growth were small, though statistically significant ($\alpha = 0.05$), and showed no pattern consistent with the observations of live and dead fish in EFPC. The results did not indicate that toxic substances were entering EFPC downstream from NHP.

F.1.4 Residue Analyses and Water Chemistry

Dead fish were collected from EFPC on July 19 and July 23, 1985, and frozen, pending analysis for the presence of suspected toxicants. Suspect agents were to be determined from a gas chromatography/mass spectrometry (GC/MS) analysis of samples of water and of vegetation sprayed with herbicides. The samples were collected on July 23 from the area where the fish kill appeared to originate.

The water samples obtained for toxicity testing (Sect. 1.3) were analyzed for organics by EPA procedure 625 (EPA 1984b). The results of these analyses indicated

Table F-2. Survival and growth of fathead minnow larvae in a 7-d toxicity test of water samples from five sites on East Fork Poplar Creek (EFPC) collected between 1230 and 1500 h on July 23, 1985

Sampling location	EFPC site (km)[a]	Percentage survival		Dry weight (mg)	
		Mean	SD[b]	Mean	SD[b]
Control		97.5	5.0	1.02	0.12
Below New Hope Pond	23.7	97.5	5.0	0.73[c]	0.07
25 m above temporary culvert	22.5	95.0	5.8	0.73[c]	0.05
5 m below temporary culvert	22.5[d]	97.5	5.0	0.89[c]	0.06
Behind NOAA Atmospheric and Turbulence Diffusion Laboratory	22.0	97.5	5.0	0.79[c]	0.04
Tuskegee Drive bridge	21.2	97.5	5.0	0.84[c]	0.03

[a]Distance above the mouth of the stream.
[b]SD = standard deviation.
[c]Significantly different from controls, $\alpha = 0.05$.
[d]Approximate location of the temporary culvert.

trace levels (low ppb concentrations) of an herbicide, 2-methoxy-4,6 bis(isopropylamino)-s-triazine, in all samples. Other substances detected in low concentrations were butyl carbitol, 2-butoxyethylphosphate, and tributylphosphate. (Methyl hexanone was noted in some samples and blanks.) Results of analyses of sprayed foliage did not reveal the presence of specific herbicides because of interference from the oily matrix. Because specific toxicants could not be detected in the vegetation, fish tissues were not analyzed.

F.1.5 Benthic Invertebrate Survey

Benthic invertebrates were sampled by J. G. Smith of ESD on Wednesday, July 24, 1985. Replicate samples were collected with a Surber sampler from riffles in EFPC at each of four sites: sites EFK 23.2 (just above Bear Creek Road) and EFK 22.6 (behind

office of Dr. S. G. Hammons) were upstream of the region where dead and dying fish were observed and sites EFK 22.3 (~100 m downstream of Dean Stallings Ford) and EFK 21.3 (behind K-Mart) were within the region of the observed fish kill.

Benthos densities were substantially lower at EFK 22.3, the site ~200 m below the upper end of the reach in which dead and dying fish were observed, than at either of the adjacent sites upstream and downstream (Fig. F-1). The difference primarily resulted from reduced numbers of chironomids at EFK 22.3 (Table F-3). Although the presence of a depauperate benthic fauna at that site cannot be attributed conclusively to the toxic agent responsible for the fish kill, it is consistent with the presence of a toxic agent in the stream at approximately EFK 22.5 (the site of road construction activity and herbicide spraying).

F.1.6 Conclusions

A significant fish kill occurred in EFPC on Friday, July 19, 1985. A second kill occurred four days later, on Tuesday, July 23, 1985. Field observations on July 19 and an electrofishing survey on July 23 indicated that the reach of stream between the outfall of New Hope Pond (EFK 23.7) and a road construction site 1.1 km downstream was not affected by the kill. Observations of numerous dead fish, the absence of living fish in the mainstream, the presence of fish congregated in apparent refuge areas at the mouth of tributaries, and the reduced numbers of benthic insects all indicated the presence of toxic conditions in the reach of EFPC below the construction site.

A major concern was the adverse impact of the fish kill to ongoing ecological studies associated with the Y-12 Plant BMAP for EFPC. An electroshocking survey conducted on July 23, 1985, indicated that only site EFK 18.2 located downstream of the Route 95 bridge at Jefferson Avenue was directly affected by the fish kill. The next site downstream (EFK 13.8) appeared to have normal fish densities and faunal composition, and EFK 23.4, the next site upstream of EFK 18.2, was above the reach where the fish kill occurred. A quantitative survey of the fish populations at the five BMAP sites below NHP was conducted in the fall 1985, and the results indicated that the July fish kill substantially reduced the populations of several species, especially the stoneroller, at EFK 18.2 (Table E-2). Adult redbreast sunfish were also absent from this study reach in the fall, but densities appeared normal in the reach just upstream of the study site. The upstream reach contains deeper pool habitat that may be preferred by this species in

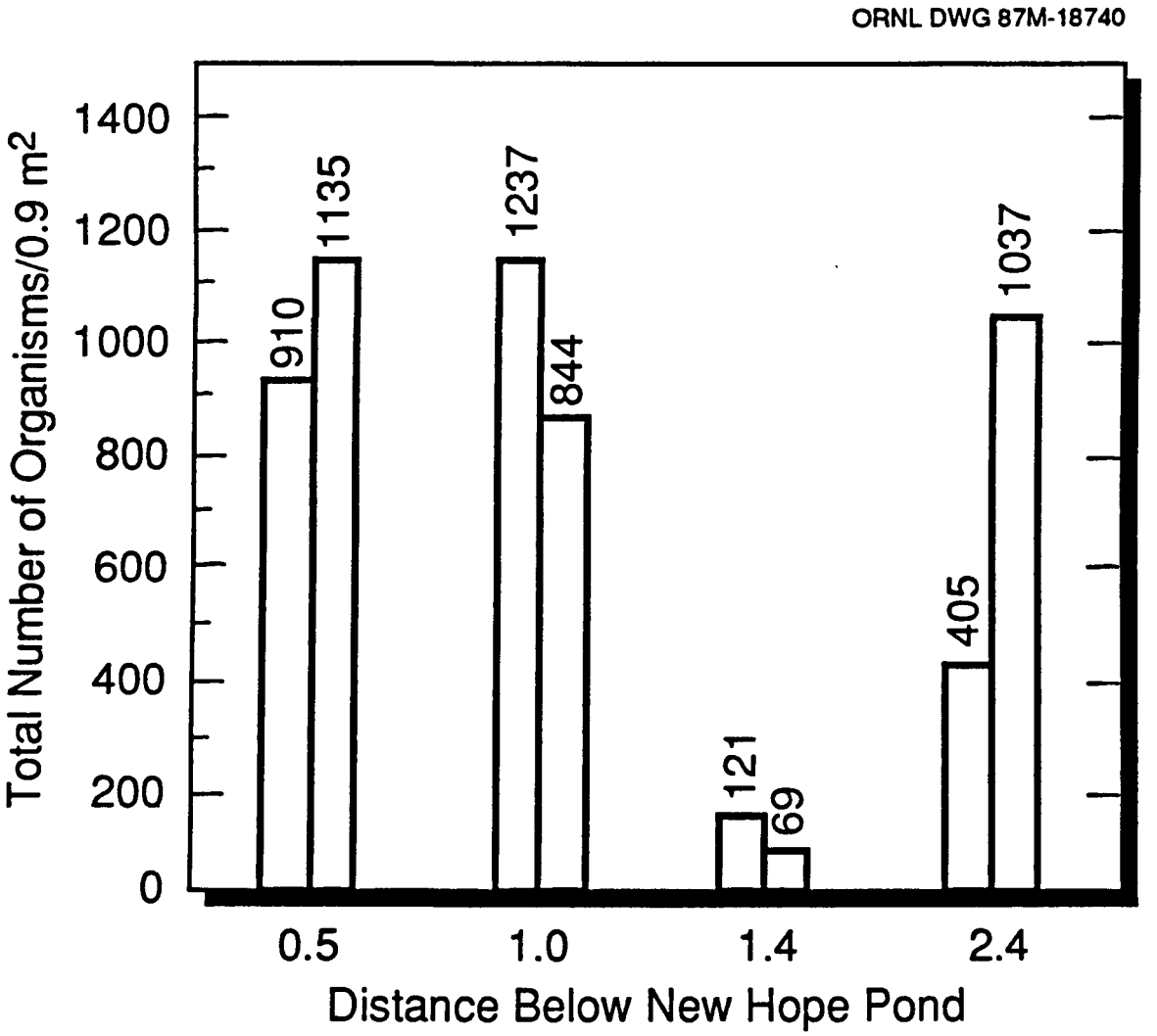

Fig. F-1. Total number of benthic invertebrates in replicate (A, B) Surber samples collected at four sites on East Fork Poplar Creek, July 24, 1985. The outfall of New Hope Pond (NHP) is located at stream kilometer 23.7. The upper limit of the fish kill was ~1.1 km below NHP.

Table F-3. Mean density (number of individuals/0.09 m^2) of taxa comprising the benthic invertebrate community at four sites on East Fork Poplar Creek,[a] July 24, 1985

Taxon	Mean density[b]			
	EFK 23.2 (0.5)	EFK 22.6 (1.0)	EFK 22.3 (1.4)	EFK 21.3 (2.4)
Nematoda	3	2	1	4
Oligochaeta	8	2	3	20
Arthropoda				
Crustacea				
Decapoda				
Astacidae				
Cambarus	0	0	0	2
Insecta				
Odonta				
Zygoptera				
Coenagrionidae	1	0	1	0
Megaloptera				
Cordalidae				
Corydalus	0	1	0	0
Coleoptera				
Elmidae				
Stenelmis	0	1	1	2
Diptera				
Ceratopogonidae	0	0	0	1
Chironomidae				
Chironominae	0	1	0	1
Orthocladinae	959	840	59	576
Tanypodinae				8
Unidentified Chironomidae	21	134	14	78
Empididae	23	18	3	5
Mollusca				
Gastropoda				
Physidae				
Physa	2	2	0	0
Total	1025	1043	97	723

[a]Distance (km) below New Hope Pond (EFK 23.7) in parentheses.
[b]Values are based on two samples collected at each site.

winter, whereas the study reach was more shallow and utilized primarily during the spawning season.

Although one study site was adversely affected by the fish kill, the impact on the BMAP was not significant. Four sites were unaffected, and a relatively abundant and diverse fish community currently inhabits EFK 18.2. The fish assemblage observed at this site 10 months after the kill was similar in species composition and density to the community that existed in June 1985 just one month before the fish kill (Table E-2).

F.2 DECEMBER 1985 FISH KILL

F.2.1 Wednesday, December 4, 1985

C. E. Textor, secretary of the Aquatic Ecology Section of ESD, was notified of a gasoline spill near Midway Road by M. Walker of the City of Oak Ridge at 0950 h on December 4, 1985. The gasoline entered the headwaters of a small tributary that flows through the Woodland area, under Illinois Avenue, and along the southern boundary of Dean Stallings Ford before entering EFPC at EFK 22.4. To contain the spill and prevent it from entering EFPC, the City of Oak Ridge Fire Department placed sorbent paper and straw in the stream.

After returning from other field studies, J. M. Loar received the message; he and S. M. Adams of ESD surveyed the tributary and EFPC between 1530 and 1700 h on December 4. No dead or distressed fish were observed in EFPC near the mouth of the tributary (EFK 22.4), and no dead fish were found in the tributary between EFPC and Illinois Avenue. Live fish and a few distressed individuals (one redbreast sunfish and two striped shiners) were observed in a pool just below the culvert at Illinois Avenue. A survey of the section of stream between Illinois Avenue and the culvert at Purdue Avenue produced one dead creek chub; a petroleum odor was also present. No dead or live fish were observed at other sites upstream. Sorbent paper and straw were observed above the culvert at Illinois Avenue and several sites upstream.

Additional information on the spill was obtained from the Tennessee Department of Health and Environment (TDHE) (Melgaard 1985). The spill, which apparently occurred on Tuesday evening, December 3, 1985, involved ~750 L (200 gal) of gasoline and water that entered a ditch near Midway Road and flowed to the stream via a storm sewer. Several dead fish were observed above the culvert at Purdue Avenue by

D. Melgaard of TDHE on the morning of December 4. He also mentioned that ~1900 to 2300 L (500 to 600 gal) of cutting oil, a combination lubricant and drying agent, was disposed of on December 4, 1985, in the vicinity of Lafayette Drive and Emory Valley Road. Runoff of the cutting oil from the disposal area produced a white color in the stream, but no dead or live fish were observed by TDHE when the stream cleared the next day (Melgaard 1985).

F.2.2 Conclusions

Based on the on-site surveys conducted by the ESD and TDHE and the information provided by the TDHE (Melgaard 1985), the area of impact was confined to a tributary of EFPC. The headwaters of this same stream are located near two small, light-industrial areas just east of Lafayette Drive, where the spills originated. The spills had no adverse effect on EFPC largely because of the actions taken by the City of Oak Ridge Fire Department to contain the gasoline spill within the tributary.

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