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SECOND REPORT ON THE OAK RIDGE K-25 SITE BIOLOGICAL MONITORING AND ABATEMENT PROGRAM FOR MITCHELL BRANCH

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Environmental Sciences Division
Publication No. 3928

March 1994



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ENVIRONMENTAL SCIENCES DIVISION
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J. G. Smith, Editor

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Environmental Sciences Division
Publication No. 3928

Manuscript Completed: December 1993

Date Published: March 1994

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Prepared by the
OAK RIDGE NATIONAL LABORATORY
Oak Ridge, Tennessee 37831-6285
managed by
MARTIN MARIETTA ENERGY SYSTEMS, INC.
for the
U.S. DEPARTMENT OF ENERGY
under contract No. DE-AC05-84OR21400



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ACRONYMS AND INITIALISMS

ACD	Analytical Chemistry Division
ALT	alanine aminotransferase
ANOVA	analysis of variance
BCK	Bear Creek kilometer
BMAP	Biological Monitoring and Abatement Program
BUN	blood urea nitrogen
CDFC	condition factor
CNF	Central Neutralization Facility
CRK	Clinch River kilometer
DEM	Department of Environmental Management [name changed in 1987 to Department of Environmental Monitoring and Compliance (DEMC)]
DO	dissolved oxygen
DOE	U.S. Department of Energy
EDTA	ethylenediaminetetracetic acid
EFK	East Fork Poplar Creek kilometer
EFPC	East Fork Poplar Creek
EPA	U.S. Environmental Protection Agency
EPT	Ephemeroptera, Plecoptera, and Tricoptera
ESD	Environmental Sciences Division
FDA	U.S. Department of Agriculture Food and Drug Administration
GCK	Grassy Creek kilometer
GC/ECD	gas chromatography/electron capture detector
GC/MS	gas chromatography/mass spectrometry
GLM	general linear model
HESA	Department of Health, Safety, and Environmental Affairs
HSRD	Health and Safety Research Division
LFPMA	functional parenchyma in liver
LMACA	macrophage aggregates in liver
LOEC	lowest observed-effect concentration
LPARS	liver parasites
LSI	liver-somatic index
MAF	mean annual flow
MHR	Melton Hill Reservoir

MIK	Mitchell Branch kilometer
NAD(H)	nicotinamide-adenine dinucleotide (reduced form)
NADP(H)	nicotinamide-adenine dinucleotide phosphate (reduced form)
NOEC	no observed-effect concentration
NPDES	National Pollutant Discharge Elimination System
ORGDP	Oak Ridge Gaseous Diffusion Plant
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyl
PCK	Poplar Creek kilometer
PGV	preliminary guidance values
PPM	parts per million
SAS	Statistical Analysis System
SD	storm drain
TDEC	Tennessee Department of Environment and Conservation
TDS	total dissolved solids
TRC	total residual chlorine
TRK	Tennessee River kilometer
TSCA	Toxic Substances Control Act
TSS	total suspended solids
TVA	Tennessee Valley Authority
USGS	U.S. Geological Survey
VSI	visceral-somatic index
WCK	White Oak Creek kilometer
WOC	White Oak Creek

PREFACE

On September 11, 1986, a modified National Pollutant Discharge Elimination System permit was issued for the Oak Ridge Gaseous Diffusion Plant (ORGDP; now referred to as the Oak Ridge K-25 Site), a former uranium-enrichment production facility operated by Martin Marietta Energy Systems, Inc., for the U.S. Department of Energy. As required in Part III (L) of the permit, a plan for the biological monitoring of Mitchell Branch (K-1700 stream) was prepared and submitted for approval to the U.S. Environmental Protection Agency and the Tennessee Department of Environment and Conservation (Loar et al. 1992a). The plan, referred to as the

Oak Ridge K-25 Site Biological Monitoring and Abatement Program (BMAP), described biomonitoring activities that would be conducted over the duration of the permit.

This document is the second in a series of reports presenting the results of the studies that were conducted from August 1987 through June 1990 for the Oak Ridge K-25 Site BMAP. The actual period covered by each task or subtask varied, depending on when the task was initiated and the time needed for sample analysis and data review. These reports also address any significant modifications in the scope of work outlined in Loar et al. (1992a).

ACKNOWLEDGMENTS

We thank D. K. Cox, W. M. Harris, W. C. Kyker, L. M. Stubbs, M. S. Greeley, S. L. Niemela, B. K. Beane, P. W. Braden, L. S. Ewald, G. P. Morris, J. Richmond, L. F. Wicker, and A. J. Stewart for their assistance in the laboratory and/or field. We thank N. M. Ferguson, M. P. Maskarinec, T. B. Schope, and other staff of the Oak Ridge National Laboratory (ORNL) Analytical Chemistry Division for sample analyses. We thank B. F. Clark, W. C. Dickinson, A. W. McWhorter, and J. A. Wojtowicz of JAYCOR for processing the benthic macroinvertebrate samples;

W. C. Dickinson also reviewed an earlier draft of Sect. 6.1. We are grateful to G. F. Cada and A. J. Stewart, who reviewed a draft of this report and provided many helpful suggestions and comments. Finally, thanks to E. B. Bryant for editorial support and L. J. Jennings for electronic publishing of this document. This work was funded by the Department of Health, Safety, and Environmental Affairs of the Oak Ridge K-25 Site. The K-25 Site and ORNL are operated by Martin Marietta Energy Systems, Inc., under contract DE-AC05-84OR21400 with the U.S. Department of Energy.

EXECUTIVE SUMMARY

As a condition of the modified National Pollutant Discharge Elimination System (NPDES) permit issued to the Oak Ridge Gaseous Diffusion Plant (ORGDP; now referred to as the Oak Ridge K-25 Site) on September 11, 1986, a Biological Monitoring and Abatement Program (BMAP) was developed for the receiving stream (Mitchell Branch or K-1700 stream). The objectives of BMAP are to (1) demonstrate that the effluent limitations established for the Oak Ridge K-25 Site protect and maintain the use of Mitchell Branch for growth and propagation of fish and other aquatic life and (2) document the effects on stream biota resulting from operation of major new pollution abatement facilities, including the Central Neutralization Facility (CNF) and the Toxic Substances Control Act (TSCA) incinerator. The BMAP consists of four tasks: (1) ambient toxicity testing; (2) bioaccumulation studies; (3) biological indicator studies; and (4) ecological surveys of stream communities, including benthic macroinvertebrates and fish. This document is the second in a series of reports on the results of the Oak Ridge K-25 Site BMAP; it describes studies that were conducted over various periods of time between August 1987 and June 1990.

Background

Mitchell Branch is a small stream that originates near the northeast boundary of the Oak Ridge K-25 Site; it flows only 1.5 km from its headwaters to its mouth at Poplar Creek kilometer 7.0. The water quality of Mitchell Branch is influenced by the geology of the drainage basin, effluents entering the stream via storm drains and

the K-1407-E/F ponds, leachate from waste disposal sites, and remediation projects. The water quality of lower Mitchell Branch was characterized by (1) moderate levels of dissolved solids and occasionally high turbidity; (2) low concentrations of nitrogen relative to concentrations of phosphorus; (3) elevated levels of many metals and some organics; and (4) elevated temperatures. The geology of Mitchell Branch watershed contributes to periods of no flow in its upper reaches, whereas flow in its lower reaches is augmented by as much as 31% by discharges from the Oak Ridge K-25 Site. Samples for BMAP were routinely collected from eight primary sites in Mitchell Branch. The toxicity monitoring and community studies included at least five of the primary sites; four of the eight sites were common among all tasks. The site farthest upstream served as an undisturbed reference; the remaining sites were selected to coincide with the ambient NPDES monitoring station or to bracket known areas or sources of ecological disturbance. Additional reference sites on nearby streams were used for some tasks, depending on the specific objectives of the task.

Toxicity Testing

The bimonthly evaluation of the toxicity of the discharges from the K-1407-B pond initiated in October 1986 continued through April 1988. Following closure of the K-1407-B pond in October 1988, testing of the effluents from the K-1407-E/F ponds was initiated and continued bimonthly through June 1990. Similarly, water from the K-1407-J basin was tested for toxicity at bimonthly intervals from

December 1988 through June 1990. The effluent from storm drain (SD) 170 was evaluated for toxicity eight times, and the toxicity of effluents from SDs 180 and 190 was evaluated six times each between July 1988 and May 1990. The toxicity of the effluents (grab samples) was determined by 7-d static-renewal tests that measured survival and growth of fathead minnow (*Pimephales promelas*) larvae and survival and reproduction of a microcrustacean (*Ceriodaphnia dubia*). Toxicity patterns were similar for the K-1407-B pond, the K-1407-E/F ponds, and the K-1407-J basin. Effluents from these basins/ponds were either never toxic (K-1407-E/F ponds) or rarely toxic (K-1407-B pond and K-1407-J basin) to fathead minnows but were nearly always toxic to *Ceriodaphnia*. The toxicity of the effluents from the K-1407-E/F ponds appeared to be linked to constituents that cause high hardness and conductivity levels, whereas the toxicity of effluent from the K-1407-J basin appeared to be related, in part, to elevated concentrations of sodium, chloride, and sulfate. The effluent from SD 170 was always toxic to *Ceriodaphnia* and frequently toxic to fathead minnows. Similarly, the effluent from SD 190 was almost always toxic to *Ceriodaphnia* and occasionally toxic to fathead minnows, whereas the effluent from SD 180 was occasionally toxic to both species. Dechlorination did not always remove toxic components from the storm drains, suggesting that toxic constituents other than chlorine were sometimes present.

The ambient (instream) toxicity at six sites in Mitchell Branch was evaluated bimonthly from January 1987 through July 1990 by using the same testing protocols that were used to evaluate the toxicity of point-source discharges. Water from the two sites farthest upstream in Mitchell Branch [Mitchell Branch kilometers (MIKs) 1.43 and 1.0] adversely affected

fathead minnows but not *Ceriodaphnia*. Because there were no obvious sources of toxicants at either of these sites, a bacterial or fungal fish pathogen may have been involved. Water from MIK 0.71 downstream to MIK 0.45 showed strong evidence of acute toxicity to both species through 1988. In 1989 and 1990, the incidence of acute toxicity at these sites declined. Water from MIK 0.12 was never acutely toxic to fathead minnows but was acutely toxic to *Ceriodaphnia* in many tests conducted from 1987 through 1989. Chronic toxicity to fathead minnows was evident only in water from MIK 0.54, whereas chronic toxicity was detected in *Ceriodaphnia* in water from MIKs 0.71, 0.54, and 0.12. Evidence to date suggests that chlorine appears to be the primary toxicant contributing to the acute toxicity of water from some sites. Chronic toxicity may be caused by low levels of total residual chlorine from the storm drains, constituents released from the K-1407-E/F ponds, and/or unidentified area-source contamination.

Bioaccumulation Studies

Results of monitoring polychlorinated biphenyl (PCB) accumulation in aquatic biota in Mitchell Branch and downstream aquatic systems from 1987 through 1990 continued to show that Mitchell Branch was clearly a source of PCB contamination to its biota during this period. PCB monitoring also suggested that Mitchell Branch may be a significant source of PCB contamination to lower Poplar Creek. Some fish in Mitchell Branch contained PCB concentrations in excess of the 2 µg/g U.S. Department of Agriculture Food and Drug Administration (FDA) action limit (FDA 1984a), and accumulation of PCBs in caged Asiatic clams (*Corbicula fluminea*) was also substantial. Significant PCB contamination was also evident in some

species of fish collected from Poplar Creek a short distance downstream of Mitchell Branch. However, the significance of Mitchell Branch as a source of PCBs in Poplar Creek and the Clinch River arm of Watts Bar Reservoir cannot yet be determined. Except for slightly elevated concentrations of polycyclic aromatic hydrocarbons in caged clams, no elevated concentrations of other organic compounds were positively identified in caged clams.

Mercury concentrations in Mitchell Branch fish continue to exceed background levels, but the levels were not higher than the FDA limit (FDA 1984b) and were typical of the degree of contamination found in fish in nearby Poplar Creek. Thus, the significance of Mitchell Branch as a source of mercury to its biota, and to the biota of downstream aquatic systems, was inconclusive. Concentrations of other metals were not elevated in fish from Mitchell Branch.

Population and Community Studies

Studies of fish and benthic invertebrates continued to show that the biota in Mitchell Branch downstream of SD 170 are severely affected by operations at the Oak Ridge K-25 Site. Furthermore, results from benthic invertebrate studies showed evidence of substantial impact at the two monitoring sites located immediately upstream of SD 170 (i.e., MIKs 0.78 and 0.86). Fish were absent from all three sampling sites downstream of SD 170 (i.e., MIKs 0.71, 0.54, and 0.45), and they tended to congregate at MIK 0.78, just upstream of SD 170. The benthic invertebrate community was strongly stressed at MIK 0.71, whereas minor improvements were evident further downstream. Results from ambient toxicity tests, analyses of the structure and composition of the benthic macroinvertebrate community, and the absence of fish in lower Mitchell Branch suggest that the biota near SDs 170, 180,

and 190 were influenced primarily by frequent releases of one or more toxicants such as chlorine. The lack of toxicity to *Ceriodaphnia* in ambient tests of water from MIK 1.10, plus the congregation of fish upstream of all major discharges, suggests that the impact observed in the benthic invertebrate community at MIKs 0.78 and 0.86 may be the result of a nontoxic stress (e.g., siltation from construction activities).

The health of redbreast sunfish (*Lepomis auritus*) in lower Mitchell Branch was assessed by observing a suite of biochemical and physiological parameters. These parameters provided information in five functional categories: (1) detoxification enzyme induction, (2) organ dysfunction, (3) histopathology, (4) overall fish health and condition, and (5) nutritional status. Results from this study indicated that redbreast sunfish appeared to be physiologically stressed. Evidence suggesting exposure to toxicants and/or poor water quality included (1) elevation of a detoxification enzyme, (2) gill dysfunction, and (3) evidence of some liver impairment. There was also evidence of indirect stress related to nutritional status; fish in Mitchell Branch generally had less food in their stomachs and intestines, and the level of serum triglycerides was low relative to fish from reference sites.

Future Studies

Results of studies conducted since 1986 in association with the Oak Ridge K-25 Site BMAP will be used to guide future monitoring efforts. Sampling sites and frequencies will remain the same for the effluent and ambient toxicity studies and benthic macroinvertebrate and fish studies, although the frequency of benthic invertebrate sample collection was reduced to quarterly after the second year of sampling (i.e., quarterly beginning August 1988). Bioaccumulation studies will key on

identifying the source(s) of PCBs and the significance of Mitchell Branch as a source of mercury to its biota. Bioindicator studies will continue largely as before but will also include the evaluation of reproductive competence of the target species.

Although routine monitoring of Mitchell Branch will continue, increasing emphasis will be placed on development and testing of hypotheses on the factors

and mechanisms contributing to the adverse conditions that have been observed to date. Ultimately, the rate of recovery of the biotic communities, the elimination of toxicity in the middle reaches of Mitchell Branch, and the reduction of contaminant residues in biota of Mitchell Branch will all depend on accurate identification of the causal factor(s).

1. INTRODUCTION

J. G. Smith

On September 11, 1986, a modified National Pollutant Discharge Elimination System (NPDES) permit was issued for the Oak Ridge Gaseous Diffusion Plant (ORGDP; now referred to as the Oak Ridge K-25 Site), a former enriched-uranium production facility currently operated by Martin Marietta Energy Systems, Inc., for the U.S. Department of Energy (DOE). As specified in Part III (L) of the permit, a plan for the biological monitoring of the receiving stream (K-1700 stream or Mitchell Branch) was submitted for approval to the U.S. Environmental Protection Agency (EPA) and the Tennessee Department of Environment and Conservation (TDEC) in December 1986. Because it was anticipated that the chemical composition of several effluents would be altered shortly after the permit modifications were issued, the Oak Ridge K-25 Site Biological Monitoring and Abatement Program (BMAP) was implemented in August 1986, before the plan had been formally approved by the regulatory agencies.

The BMAP for Mitchell Branch was developed to meet two major objectives. First, studies were designed to provide sufficient data to determine whether the interim effluent limits established for the K-25 Site protect and maintain the use of Mitchell Branch for growth and propagation of fish and other aquatic life. A second major objective was to document the effects on stream biota resulting from

construction and operation of major new pollution abatement facilities, including the Central Neutralization Facility (CNF) and the Toxic Substances Control Act (TSCA) incinerator. The ecological effects of remedial actions (e.g., closure of the K-1407-B and K-1407-C holding ponds) can also be evaluated by this monitoring program.

The effluents discharged to Mitchell Branch are chemically complex and contain various trace metals, organic chemicals, neutral salts, and radionuclides (Sect. 2.2). Moreover, the composition of these effluents will change as various pollution abatement measures are implemented over the next several years. Although contaminant inputs to the stream originate primarily as point-sources from existing plant operations, area or non-point sources such as the classified burial grounds cannot be eliminated as potential sources of contaminants. A multitiered, integrated approach to biological monitoring was developed to address this complexity. The Mitchell Branch BMAP consists of four major tasks: (1) effluent and ambient toxicity testing, (2) bioaccumulation studies, (3) biological indicator studies, and (4) ecological surveys of stream communities (e.g., benthic macroinvertebrates and fish). Because few fish occur in Mitchell Branch, the third task, biological indicator studies, will use hatchery-reared species as needed. These introduced fish will be maintained in enclosures in the stream.

2. DESCRIPTION OF STUDY AREA

R. L. Hinzman and J. M. Loar

Mitchell Branch is a small stream located near the northeast boundary of the Oak Ridge K-25 Site (Figs. 2.1 and 2.2). The stream has a drainage area of 1.78 km² and is similar in size to upper Grassy Creek [2.59 km² at Grassy Creek kilometer (GCK) 2.4], a reference stream located about 2 km southeast of the

Oak Ridge K-25 Site (Fig. 2.3). Mitchell Branch flows only 1.5 km from its headwaters to its mouth, where it drains into Poplar Creek about 150 m downstream of the Blair Road bridge. The confluence of the two streams is about 1.5 km below the mouth of East Fork Poplar Creek (EFPC) and 7 km above

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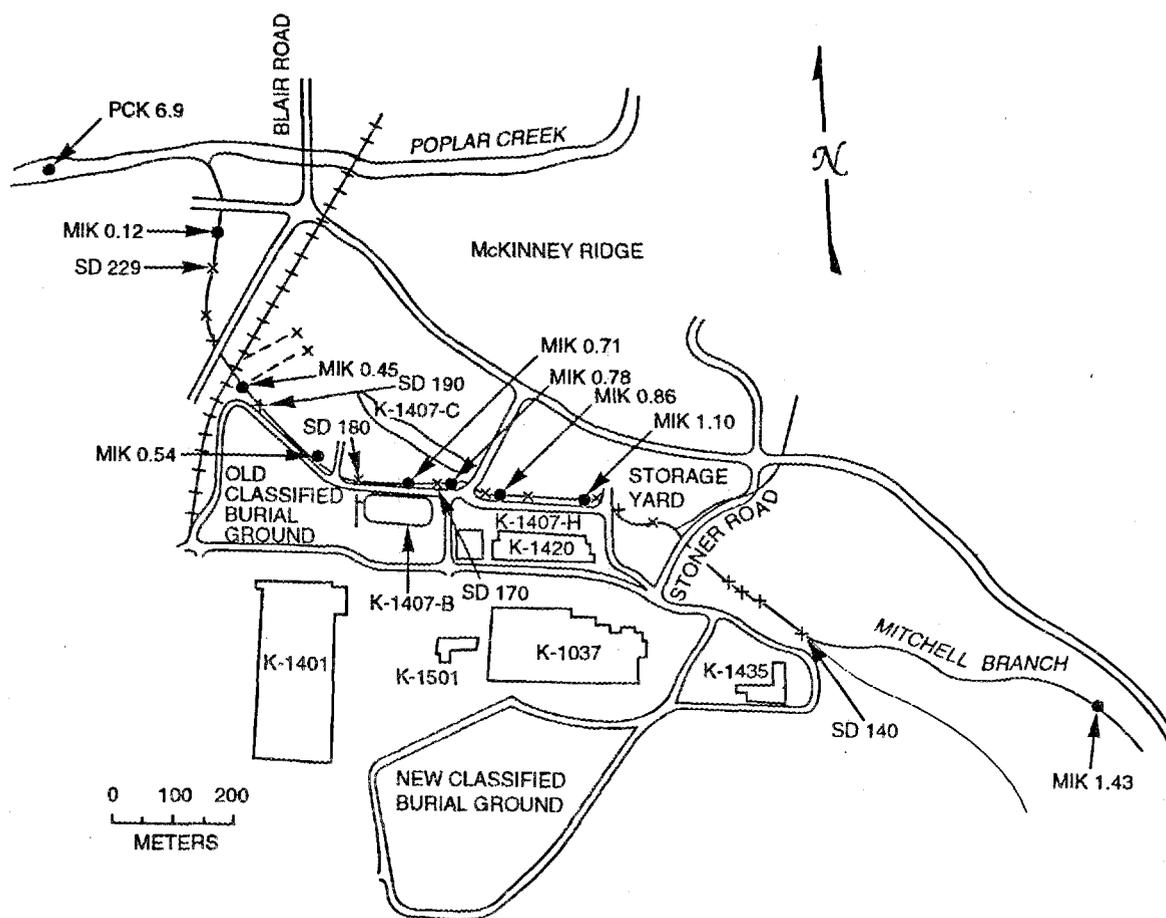


Fig. 2.1. Map of Mitchell Branch and a portion of the Oak Ridge K-25 Site showing the locations of the biological monitoring sites (●) in relation to the storm drains (x) and the K-1407-B pond, which closed after October 31, 1988.

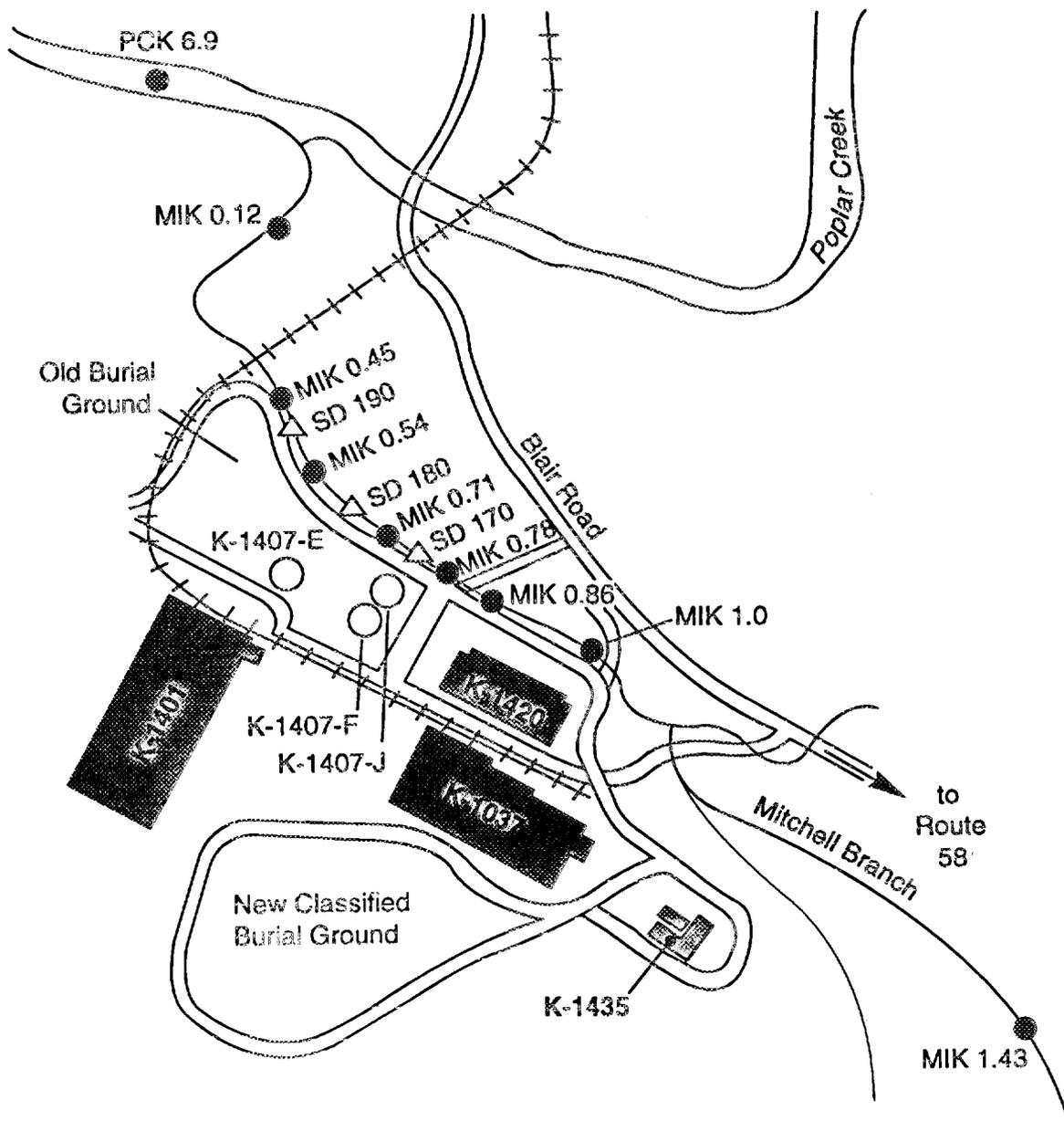


Fig. 2.2. Map of Mitchell Branch and a portion of the Oak Ridge K-25 Site showing the locations of biological monitoring sites (*) in relation to selected storm drains (Δ) and the K-1407-E/F ponds and the K-1407-J basin, which replaced the K-1407-B pond on November 1, 1988.

the confluence of Poplar Creek with the Clinch River (Fig. 2.3).

Poplar Creek, which has a drainage area of 352 km², originates northeast of Oliver Springs, Tennessee, on the

Cumberland Plateau (Figs. 2.1 and 2.3). The general direction of streamflow of Poplar Creek is southeasterly, entering the DOE Oak Ridge Reservation (ORR) north of the K-25 Site, and then flowing

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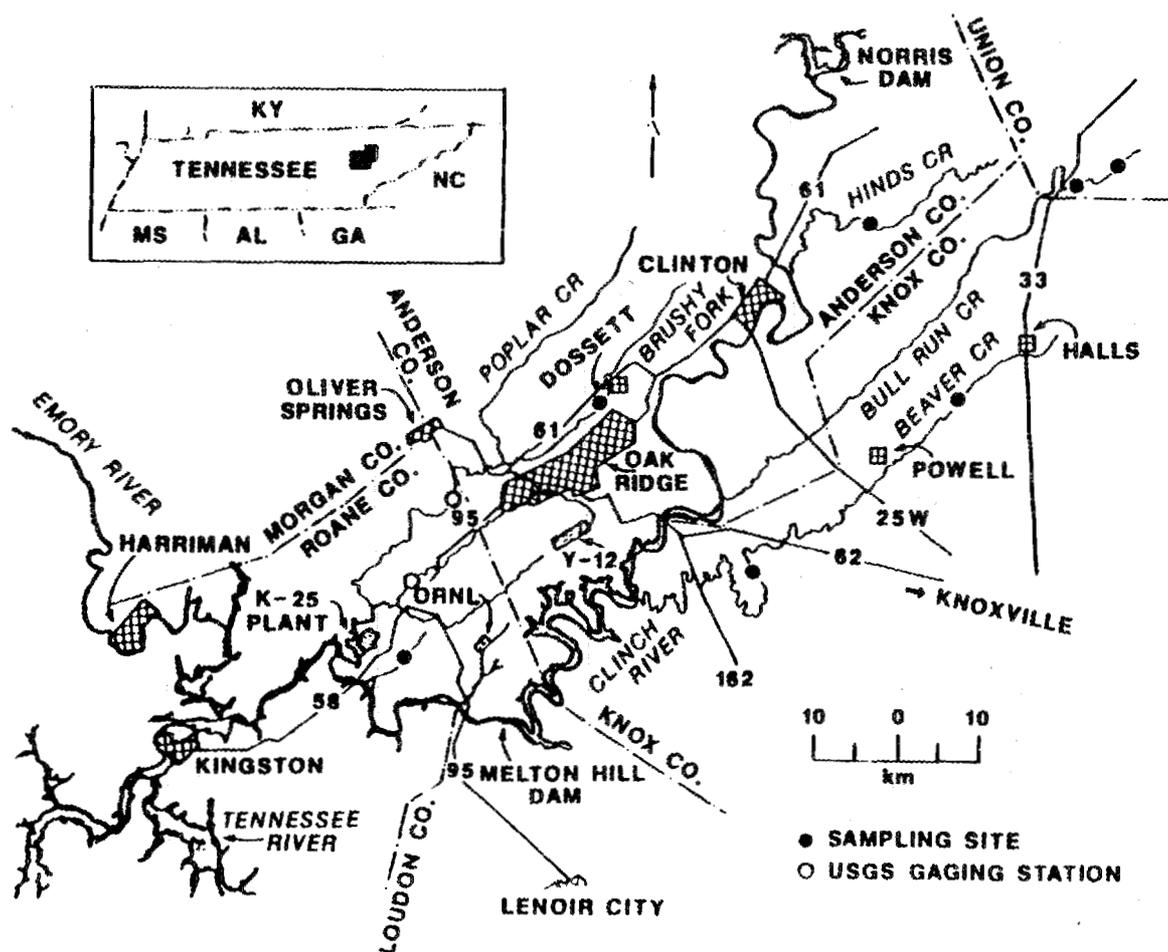


Fig. 2.3. Map of the Oak Ridge area showing locations of the reference (control) sites.

for a distance of 10 km through the plant area before entering the Clinch River near Clinch River kilometer (CRK) 19.3. EFPC, a major tributary of Poplar Creek, originates from springs on the northwest slope of Chestnut Ridge within the boundaries of the Oak Ridge Y-12 Plant. Streamflow is controlled by Lake Reality, a 1.0-ha settling pond located ~1.5 km below the spring. The creek flows for ~24 km before joining the West Fork at Poplar Creek kilometer (PCK) 8.8.

2.1 GEOHYDROLOGY

2.1.1 Mitchell Branch

Mitchell Branch originates near the base of a small knoll southwest of McKinney Ridge. The knoll is underlain by the Conasauga Group, which consists of a calcareous shale interbedded with thinner layers of limestone and siltstone (DOE 1979). Streams such as Mitchell Branch, which are underlain by shale and sandstone, have a smaller low-flow discharge

and greater range in flow than streams underlain by carbonate rocks such as Knox Dolomite (McMaster 1967).

Periods of zero discharge sometimes occur in portions of Mitchell Branch just upstream of the BMAP sampling site at Mitchell Branch kilometer (MIK) 0.86 [J. G. Smith, Environmental Sciences Division (ESD), Oak Ridge National Laboratory (ORNL), personal observation]. Periods of zero discharge are also characteristic of the upper reaches of Melton Branch near ORNL (Loar 1990, Table 2.1), a small stream that is also underlain predominantly by shale and sandstone (McMaster 1967, Table 10).

Discharges from the Oak Ridge K-25 Site augment the flow of Mitchell Branch downstream of MIK 0.78. Once-through cooling water and process water account for about 21% and 10%, respectively, of the streamflow at NPDES monitoring station K-1700 on lower Mitchell Branch at MIK 0.12 (Kasten 1986). Surface runoff probably accounts for most of the remaining flow (69%) (Kasten 1986), although there is a minor contribution from groundwater (Scheib 1987, Table 7). Based on these estimates, ~31% of the flow in lower Mitchell Branch can be attributed to discharges from the Oak Ridge K-25 Site. However, in years of below-normal precipitation and minimal runoff, such as in 1985-87, nearly 100% of the flow in the stream could be plant effluent. The potential benefit to biota derived from increasing the minimum flow in the stream (i.e., reduction in stream bed dewatering and habitat loss) could be offset by the adverse impacts of insufficient dilution of these effluents.

Mitchell Branch has a relatively low temporal variability in discharge volume compared with streams without flow augmentation (Table 2.1). The variability in discharge in Mitchell Branch was higher than in EFPC at East Fork Poplar Creek kilometer (EFK) 5.3, where flow is

augmented by ~50% in dry years. The effect of flow augmentation may be beneficial to some biota. For example, Horowitz (1978) found that greater numbers of fish species occurred in streams where flow was more constant.

The mean annual flow (MAF) in Mitchell Branch decreased by ~25% in 1986 and again in 1987 (Table 2.1). Placing the Oak Ridge K-25 Site on standby status during this period reduced the releases of water from small, once-through cooling systems by almost 50% (Kasten 1986, Fig. 12). Because water from these systems accounts for less than 25% of the flow in Mitchell Branch, they probably contributed less to the reduction in stream discharge than the below-normal precipitation that occurred from 1985-88. Generally, streams with and without flow augmentation decreased in flow from 1985-87 (Table 2.1). Even in 1985, which was used as a baseline for this comparison, the MAFs in Poplar Creek and EFPC were only 62% and 79%, respectively, of the historical MAF (period of record: 1960-85) (Lowery et al. 1987).

Although precipitation was below normal from 1985-88, the Mitchell Branch hydrograph in 1985 differed substantially from those for 1986-89 (Appendix A, Fig. A.1). Flows were generally less variable in 1985 (Table 2.1), and there was no prolonged low-flow period, which was a dominant feature of the 1986-89 hydrographs. During a 5-month period from summer through early fall, mean monthly flows were below 450 and 350 L/min in 1986 and 1987, respectively; for two consecutive months in both years, flows averaged less than 150 L/min. In comparison, mean monthly flows in 1985 never fell below 500 L/min and were never less than 700 L/min for more than 2 consecutive months. Also, the minimum daily flow in 1985 was almost an order of magnitude higher than the minimum flows in 1986 and 1987. Annual precipitation was

Table 2.1. Comparison of mean annual discharge (in liters per second) and percentage change between years for local streams with and without significant flow augmentation

Year	With flow augmentation		Without flow augmentation	
	Mitchell Branch	East Fork Poplar Creek (EFK 5.3)	Upper Melton Branch (MEK 1.93)	Bear Creek (BCK 4.55)
1985	16.7 (50.0)	1114 (33.8)	^a	^a
1986	12.5 (81.6)	1011 (52.1)	10.5 (137.1)	107 (110.6)
1987	9.6 (70.8)	998 (50.6)	10.2 (148.4)	111 (110.6)
1988	23.1 (48.8)	1100 (147.7) ^b	9.1 (469.3)	92 (238.9)
1989	23.7 (25.5)	NA	28.0 (187.1)	325 (145.8)
Percentage change				
1985-1986	-25.2	-9.2	NA	NA
1986-1987	-23.2	-1.3	-2.9	3.7
1987-1988	140.6	9.2	-10.8	-17.2
1988-1989	2.6	NA	207.7	253.3

^aNew USGS station; no record prior to April 1, 1985, for Melton Branch or before March 1, 1985 for Bear Creek.

^bValues based on 182 d of record; station was discontinued after June 30, 1988.

Note: Values in parentheses are the coefficient of variation (CV) for the mean flow values. The mean and CV were computed from mean monthly values taken from monthly NPDES reports (Mitchell Branch at MIK 0.12). EFK = East Fork Poplar Creek kilometer; MIK = Mitchell Branch kilometer; BCK = Bear Creek kilometer.

Sources: Lowery, J. F., et al., 1986, *Water Resources Data for Tennessee, Water Year 1985*, Report No. USGS/WRD/HD-86/216, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F., et al., 1987, *Water Resources Data for Tennessee, Water Year 1986*, Report No. USGS/WRD/HD-87/225, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F., et al., 1988, *Water Resources Data for Tennessee, Water Year 1987*, Report No. USGS/WRD/HD-88/236, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F., et al., 1989, *Water Resources Data for Tennessee, Water Year 1988*, Report No. USGS/WRD/HD-89/258, U.S. Geological Survey, Nashville, Tennessee, and USGS provisional discharge data [L. D. Voorhees, ORNL, Environmental Sciences Division (ESD), unpublished data].

124.3 cm in 1988 and 167.7 cm in 1989 (90% and 120% of normal, respectively, for the 1951-80 period of record), which increased mean monthly flows in both years; during this same period, flows fell below 500 L/min only once (September 1989).

The number of days of zero discharge in upper Melton Branch also provides a relative measure of ecologically meaningful differences in the hydrographs of the past 3 years. For example, no flow was recorded for 104 d in 1986, 172 d in 1987, and 141 days in 1988 (Table 2.2). More

Table 2.2. Number of days of zero discharge (number of consecutive days in parentheses) in upper Melton Branch at USGS gaging station 03537100 near Oak Ridge National Laboratory

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

Sources: Lowery, J. F., et al., 1986, *Water Resources Data for Tennessee, Water Year 1985*, Report No. USGS/WRD/HD-86/216, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F., et al., 1987, *Water Resources Data for Tennessee, Water Year 1986*, Report No. USGS/WRD/HD-87/225, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F., et al., 1988, *Water Resources Data for Tennessee, Water Year 1987*, Report No. USGS/WRD-HD-88/236, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F., et al., 1989, *Water Resources Data for Tennessee, Water Year 1988*, Report No. USGS/WRD-HD-89/258, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F., et al., 1990, *Water Resources Data for Tennessee, Water Year 1989*, Report No. USGS/WRD/TN-89-1, U.S. Geological Survey, Nashville, Tennessee, and USGS provisional discharge data [L. D. Voorhees, ORNL, Environmental Sciences Division (ESD), unpublished data].

important, the number of consecutive days with zero discharge in 1987 (155 d) exceeded the number in 1986 and 1985 by factors of 3 and 25, respectively, and in 1988 (113 d) by factors of 2 and 18 respectively. These data indicate that any adverse ecological effects resulting from reduced streamflow were also greater in 1987 because of below-normal precipitation that year and the previous year. Thus, the first year of biological monitoring was conducted over extreme, and possibly worst case, ambient conditions.

2.1.2 Poplar Creek

Poplar Creek is the largest tributary of the Clinch River between Melton Hill Dam and the northwest boundary of ORR. It has an average annual discharge that is approximately ten times greater than the combined discharges of other tributaries in this 21-km reach of the river (Loar et al. 1981). It flows from Poplar Creek Valley, which is underlain by the Conasauga shale, through a gap in Black Oak Ridge, which is underlain by the Knox Formation, and enters the Clinch River southeast of the

Oak Ridge K-25 Site. With the exception of the silty shale and siltstone-sandstone members of the Rome Foundation adjacent to Route 58, southeast of the plant, most of the K-25 Site is underlain by Chickamauga Limestone (DOE 1979).

Poplar Creek exhibits seasonal fluctuations in discharge that, in general, reflect precipitation and runoff patterns typical of this region of East Tennessee. Maximum precipitation is in the winter (December-February), when ~31% of the annual precipitation occurs, the wettest

months being February and March (NOAA 1990). Maximum runoff is also likely to occur in January, February, or March, when rainfall is normally high and soil moisture and groundwater storage are highest (McMaster 1967). As shown in Fig. 2.4, the discharge in Poplar Creek was highest during the winter months and lowest in late summer, when rainfall is normally low and runoff is minimal. Water levels in Poplar Creek are also influenced by the operation of two Tennessee Valley Authority (TVA) dams: Melton Hill Dam

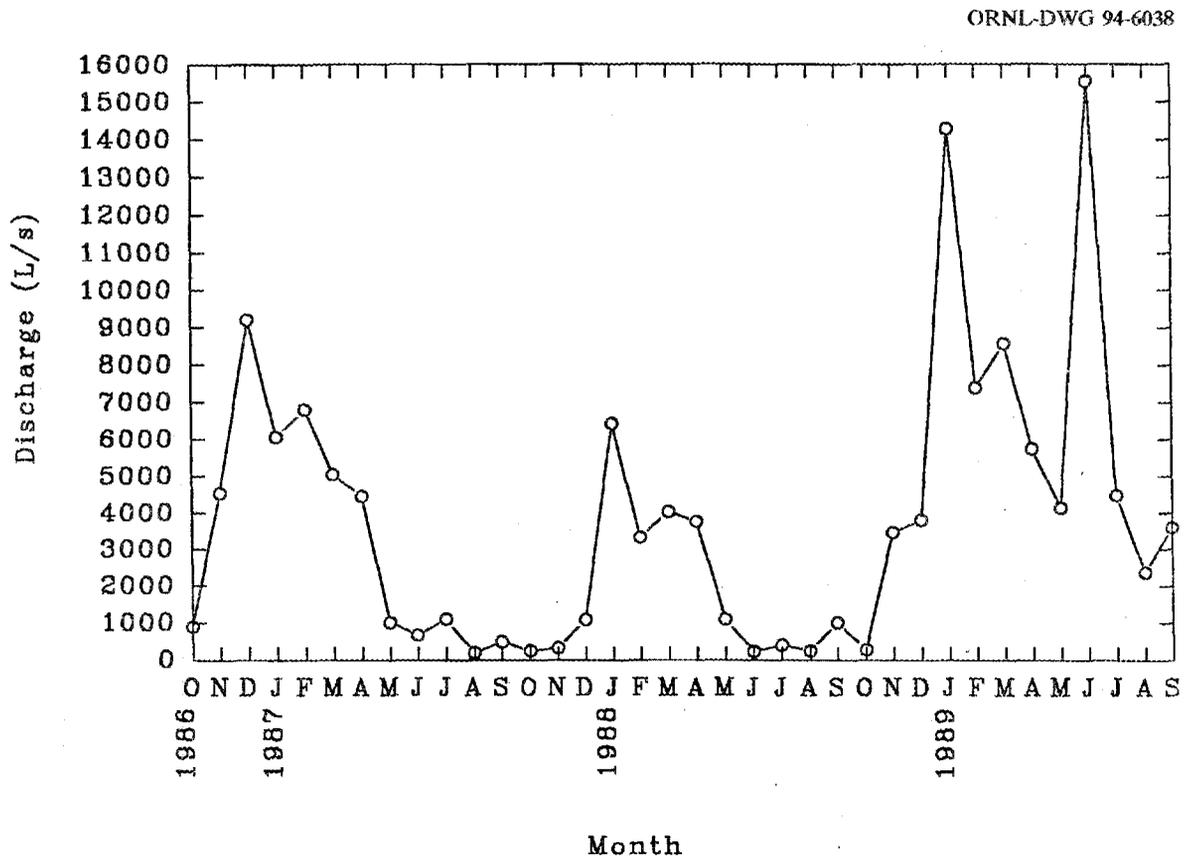


Fig. 2.4. Mean monthly discharge in Poplar Creek. The value for June 1989 is estimated. Source: J. F. Lowery et al., 1987, *Water Resources Data for Tennessee Water Year 1986*. Report No. USGS/WRD/HD-87/225, U.S. Geological Survey, Nashville, Tenn.; J. F. Lowery et al., 1988, *Water Resources Data for Tennessee Water Year 1987*. Report No. USGS/WRD/HD-88/236, U.S. Geological Survey, Nashville, Tenn.; J. F. Lowery et al., 1989, *Water Resources Data for Tennessee Water Year 1988*. Report No. USGS/WRD/HD-89/258, U.S. Geological Survey, Nashville, Tenn.

at CRK 37.2 (completed in 1963) and Watts Bar Dam located at Tennessee River kilometer (TRK) 852 (completed in 1942). Dam operations not only affect the magnitude and frequency of water level fluctuations, but they also influence stream velocities and flow direction in the lower reaches of Poplar Creek.

2.2 WATER QUALITY

Water quality of Mitchell Branch is influenced not only by the geology of the drainage basin (Sect. 2.1) but also by (1) effluents that enter the stream via the K-1407-E/F ponds and storm drains, (2) leachate from waste disposal sites (i.e., area-source discharges), and (3) remedial action projects. The following characterizations of water quality are preceded by general descriptions of the sources of effluents discharged by the Oak Ridge K-25 Site. The characterizations of water quality are based on data from NPDES monitoring station K-1700 (Sect. 2.2.2.1), effluent data from the K-1407-J basin (Sect. 2.2.2.2), and BMAP-related measurements of water temperatures (Sect. 2.2.3). Water quality, characterizations of Mitchell Branch associated with toxicity testing are given in Sect. 3.2.3.1.

2.2.1 Description of the Oak Ridge K-25 Site Discharges

2.2.1.1 Mitchell Branch

Point-source discharges to Mitchell Branch from current Oak Ridge K-25 Site operations generally fall into one of two categories: process water or storm drain effluents. Prior to October 1988, wastes from the uranium recovery facility, the metals cleaning facility, the chemical process development facility, the steam plant, and the coal yard were neutralized

in a 113,500-L vat prior to discharge into the K-1407-B pond. Metal hydroxides were allowed to settle to the bottom of the pond, and the supernatant was conveyed, via a 1.6-km long ditch, to Mitchell Branch (Fig. 2.1).

Closure of the K-1407-B pond began on October 31, 1988, under the Resource Conservation and Recovery Act. The pond now receives rainwater and surface runoff, which is pumped to the CNF, for treatment before being discharged to Poplar Creek. On November 1, 1988, the K-1407-B pond was replaced with the K-1407-E and K-1407-F ponds and the K-1407-J settling basin (Figs. 2.2 and 2.5). Prior to September 1, 1989, the K-1407-E/F ponds and the K-1407-J basin discharged directly into Mitchell Branch through storm drains 170 and 180 (SDs 170 and 180). Currently, effluent from the K-1407-J basin is transferred via pipeline to the 801 area, where it is mixed with Clinch River water, run through a series of baffles, and discharged into Poplar Creek.

Treated effluent from the CNF is pumped to the K-1407-J settling basin. The CNF has two treatment modes, "hazardous" and "nonhazardous". Included in the hazardous treatment mode are effluents from the TSCA incinerator. After treatment, hazardous waste is pumped into the K-1407-J basin and then discharged. Nonhazardous treated effluents, such as coal pile runoff and steam plant effluent, are treated in the CNF and then discharged into the K-1407-E/F ponds (Fig. 2.5).

Most of the process water currently being discharged to Mitchell Branch is via the K-1407-E/F ponds, two holding ponds constructed primarily for settling solids and pH control (K-1407-E, volume \approx 781,000 L; K-1407-F, volume \approx 132,000 L). The K-1407-E/F ponds receive primarily caustic wastes from the steam plant water treatment process and coal yard runoff. The

FINAL SYSTEM

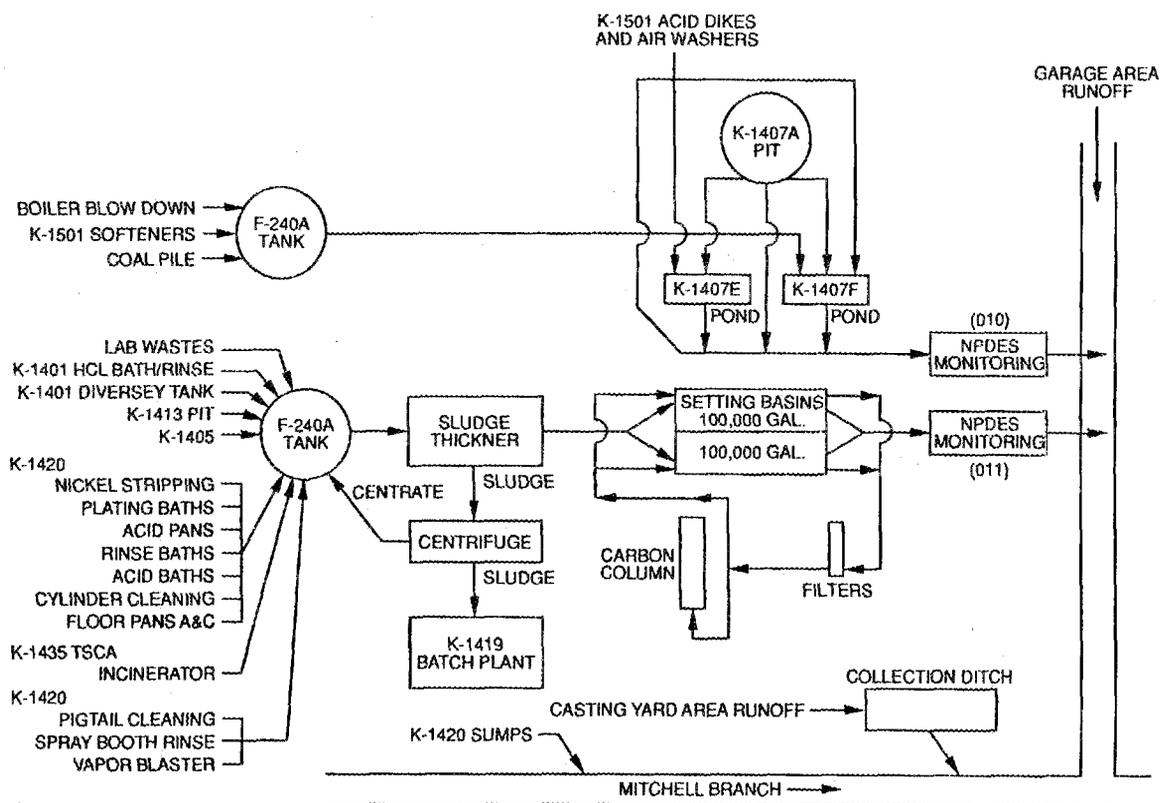


Fig. 2.5. Diagram of effluent input into the Central Neutralization Facility and ponds K-1407-E/F. Source: National Pollutant Discharge Elimination System permit for Central Neutralization Facility.

overflow from the K-1407-F and K-1407-E ponds discharges to SDs 170 and 180 respectively. Before August 1987, some coal yard runoff and boiler blowdown was discharged directly to the stream through SD 170, just upstream of the K-1407-B pond outfall.

Eighteen storm drains enter Mitchell Branch (Fig. 2.1). Although some of these drains contribute only runoff water and suspended particulate matter to the stream during rainfall events, others may convey discharge groundwater, once-through cooling water, or floor drain wastewater in addition to runoff from roofs and parking lots. In the NPDES permit, storm drains

are classified according to their source and potential for contamination. Nine storm drains at the Oak Ridge K-25 Site are classified as category III outfalls (those that may receive untreated process wastewaters), and seven of these nine drains discharge to Mitchell Branch (Smith et al. 1992a).

Leachate from waste disposal sites (i.e., area-source discharges) may also enter the stream. The old classified burial ground, a 1.50-ha site located 120 m west of the K-1407-B pond, was created by filling in a large swampy area that drained into Mitchell Branch (Fig. 2.1). This disposal site contains both radioactive and

nonradioactive wastes. An ephemeral stream drains the site, which is located within the Mitchell Branch watershed. The K-1407-C retention basin has an area of 0.80 ha and is located 120 m northeast of the K-1407-B pond. The basin was constructed in 1973 and received dredged material from other holding ponds, including K-1407-B. Although the basin has no surface effluent, a groundwater plume extending from the pond toward the stream has been detected (Ashwood et al. 1986).

2.2.1.2 Poplar Creek

The CNF (Building K-1407-H) went on-line in October 1987. The facility treats effluents from (1) Building 1420, a decontamination and recovery facility; (2) Building 1401, a metals preparation facility and machine shop; and (3) the TSCA incinerator (Building K-1435). The TSCA incinerator will be used to dispose of polychlorinated biphenyls (PCBs) and other hazardous wastes. Liquid discharges from the incinerator will include (1) scrubber blowdown and (2) fire water and rainwater. These two waste streams are collected in a surge tank, analyzed, and, as appropriate, discharged to CNF; pumped through carbon-bed absorbers; or pumped to the waste feed tanks for burning in the incinerator (EPA 1986).

Hazardous wastes from CNF are pumped to the K-1407-J basin, which consists of two aboveground settling ponds (volume = 378,000 L each) with an average monthly discharge that is slightly less than 5×10^5 L. Effluent from the K-1407-J basin is treated in a batch mode; piped to the 801 area; mixed with water from the Clinch River, which flows over a series of baffles; and then discharged to Poplar Creek.

2.2.2 NPDES Monitoring

2.2.2.1 Mitchell Branch

The following characterization of water quality in Mitchell Branch is based on routine monitoring of parameters at NPDES station K-1700. This site is located on lower Mitchell Branch (MIK 0.12) downstream of all point-source and most area-source discharges (Figs. 2.1 and 2.2).

From 1986 through 1989 water quality in Mitchell Branch at MIK 0.12 was characterized by (1) moderate levels of dissolved solids and occasionally high levels of turbidity, (2) relatively low levels of nutrients, (3) elevated levels of most metals and some organics, and (4) high temperatures (discussed in Sect. 2.2.3).

Total dissolved solids (TDS) averaged 340 mg/L from January through July 1985 and 553 mg/L for the rest of the year (W. J. Scheib, unpublished data from 1985 NPDES monthly reports). Average annual TDS levels also exceeded 500 mg/L in 1986 and 1987 (Appendix A, Tables A.1--A.5) but were less than 400 mg/L in 1989. TDS data were not available for 1988. High levels of TDS were usually associated with high streamflows, although runoff from construction sites adjacent to Mitchell Branch contributed to the periodically high suspended loads in the stream.

Concentrations of nitrogen in Mitchell Branch are low, whereas concentrations of phosphorus are high. Even though concentrations of nitrate nitrogen were higher than background (~ 0.1 mg/L; Boyle et al. 1982, Table 3.16), they averaged less than 1 mg/L and never exceeded 8 mg/L at MIK 0.12 (Appendix A, Tables A.1--A.5). Phosphorus is not monitored in Mitchell Branch, but water from the K-1407-J basin had a median phosphorus concentration of 0.405 mg/L, with a maximum recorded value of 9.3 mg/L in 1989 (Appendix A,

Table A.6). K-1407-B and K-1407-E/F ponds had median phosphorus concentrations of <0.2 mg/L, but the K-1407-B pond had a maximum concentration of 8.4 mg/L in 1988 (Appendix A, Tables A.3 and A.5).

Many of the metals (Appendix A, Tables A.1-A.5) in Mitchell Branch exceeded concentrations that are typical of small, relatively undisturbed streams on the ORR (Boyle et al. 1982, Tables 3.16 and 4.16). Except for aluminum, iron, and zinc, all measured metals had a median concentration that was at or below the detection limit (i.e., via inductively coupled plasma optical emission spectrophotometry) in each of the past 4 years. Of the organics measured in Mitchell Branch, all were above detection limits at least some of the time. Except for the increase in TDS that occurred in July 1985, the water quality of Mitchell Branch has remained relatively constant over the last 4 years (1986-89).

2.2.2.2 Discharges to Poplar Creek

Effluent from the K-1407-J basin was characterized by high concentrations of TDS in 1988 and 1989, with median values of 2619 and 1358 mg/L, respectively (Appendix A, Table A.6).

Concentrations of nitrate, phosphorous, and sulfate were all high in K-1407-J, especially during 1989. Median values for nitrate exceeded background, and the maximum value in 1989 was 1390 mg/L. The median concentration of phosphorous in 1989 was 0.4 mg/L, and the maximum concentration was 9.3 mg/L. The median concentration of sulfate was higher than 600 mg/L in 1988 and 400 mg/L in 1989; maximum values in these 2 years were 1200 and 1500 mg/L respectively.

Concentrations of all metals that were monitored from 1988-89 were elevated at

some time during this period, and most had median values above the detection limits (Appendix A, Table A.6).

Although most of the organics that were monitored had median values at or below detection limits, most were detectable at some time during the sampling period (Appendix A, Table A.7).

2.2.3 Temperature

Temperature monitoring was initiated in Mitchell Branch on April 9, 1987, at MIK 0.50 immediately below the outfall of SD 190. Data were collected with a Ryan Tempentor digital temperature recorder, and values were obtained every 20 min. Because of periodic equipment problems, the 3-year record was incomplete.

Through the period of record, Mitchell Branch temperatures were high relative to unaffected streams on ORR. Mean weekly temperatures in Mitchell Branch sometimes exceeded 25°C, but were generally below 20°C in Grassy Creek (Fig. 2.6), a nearby drainage with a similar geology (Sect. 2.1; McMaster 1967, Table 10). Water temperature averaged ~4 to 8°C higher in Mitchell Branch than Grassy Creek, although maximum temperatures in the two streams in June 1987 differed by as much as 19°C (Table 2.3).

The temperature of Mitchell Branch is influenced strongly by cooling water discharges (via storm drains) and the retention of effluent in open ponds (K-1407-B prior to October 1988 and K-1407-E/F after this period) prior to discharge just 130 m above the monitoring station at MIK 0.50. Furthermore, except for the headwaters, there is little canopy cover above MIK 0.50 to help moderate water temperatures.

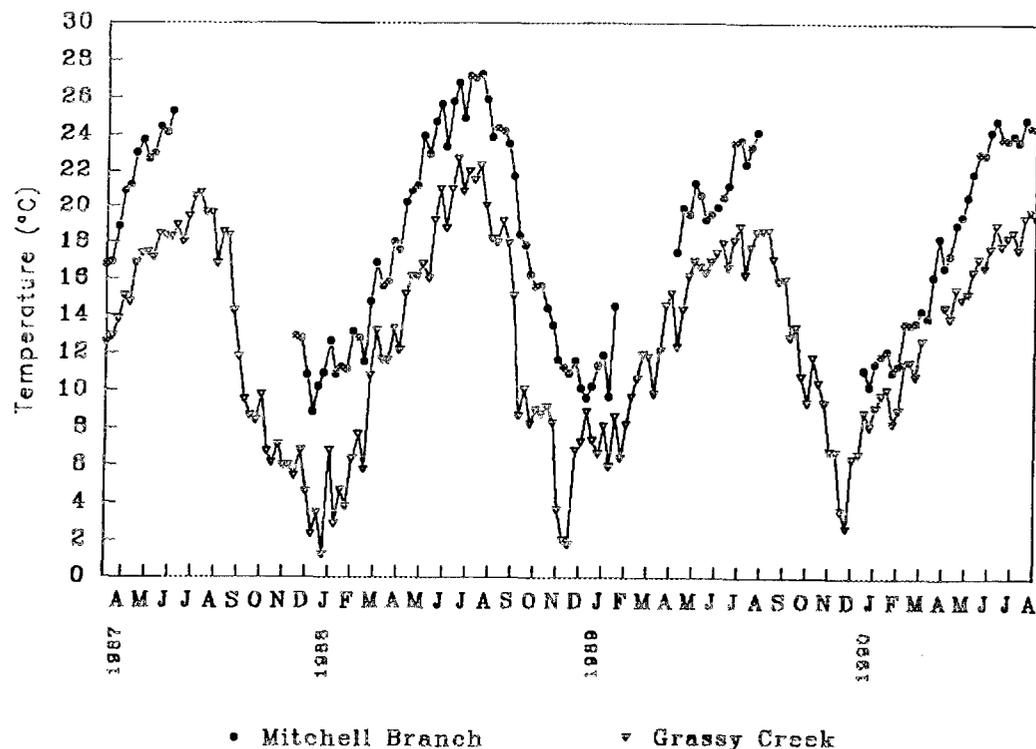


Fig. 2.6. Weekly mean temperatures for Mitchell Branch kilometer 0.50 and Grassy Creek kilometer 2.40 for 1987-90. Temperature record is incomplete as a result of equipment failure.

2.3 BIOLOGICAL MONITORING SITES

Eight sites on Mitchell Branch (Figs. 2.1 and 2.2 and Table 2.4) were routinely sampled to assess ecological conditions in the stream. Although the upper site (MIK 1.43) was the primary reference site, sites on other area streams were also used as reference stations (Fig. 2.3). The lowermost site at MIK 0.12 coincided with the location of the NPDES monitoring station (K-1700). Construction of the weir at MIK 0.12 created a large pool immediately upstream. Mitchell Branch below the weir is also a pool or embayment of Poplar Creek when Watts Bar Reservoir is at full pool (approximately April to October). In this reach of stream,

water levels are controlled by operation of Watts Bar Dam, located on the Tennessee River approximately 61 km downstream of the confluence with the Clinch River. A sampling site for the bioaccumulation task is located at PCK 6.9 downstream of the mouth of Mitchell Branch (Fig. 2.2).

Four of the remaining six sites (MIKs 0.45, 0.54, 0.71, and 0.78) were selected based on the location of the three most significant discharges to Mitchell Branch: SDs 170, 180 (and effluent from the K-1407-E/F holding ponds), and 190 (Figs. 2.1 and 2.2); these monitoring sites are located above and below each of these outfalls. The remaining two sites (MIKs 0.86 and 1.0) were selected to assess the potential for adverse impacts associated with (1) construction activities

Table 2.3. Monthly means (± 1 SD) and range (number of days of record) of water temperature ($^{\circ}\text{C}$) at Mitchell Branch kilometer (MIK) 0.50 just below the outfall of storm drain 190 and at Grassy Creek kilometer (GCK) 2.4, a reference site

Sampling period		MIK 0.50		GCK 2.4		
Year	Month	Mean	Range	Mean	Range	
1987	April	17.7 (2.1)	12.1-24.7 (20)	13.2 (2.4)	6.8-19.5 (20)	
	May	22.2 (2.2)	17.1-29.6 (31)	16.0 (2.0)	9.9-19.7 (31)	
	June	23.5 (3.9)	11.9-41.2 (30)	17.8 (1.4)	13.3-21.7 (30)	
	July	25.3 (6.4)	16.8-41.4 (5.5)	18.9 (1.2)	15.7-31.1 (28)	
	August	NA ^a	NA	20.1 (1.0)	16.8-22.0 (31)	
	September	NA	NA	16.9 (2.0)	12.0-19.9 (30)	
	October	NA	NA	9.5 (2.2)	4.9-15.9 (31)	
	November	NA	NA	7.4 (2.4)	3.1-11.3 (30)	
	December	12.8 (1.2)	10.4-16.4 (12)	6.0 (1.8)	3.4-10.7 (12)	
	1988	January	9.9 (1.4)	5.6-13.4 (25)	3.0 (2.7)	-(6.8)-4.3 (25)
		February	11.4 (1.7)	7.6-16.5 (29)	4.5 (2.4)	0.6-10.1 (29)
		March	13.1 (2.1)	8.3-21.8 (31)	7.9 (3.9)	-(3.6)-18.0 (31)
April		16.6 (1.9)	12.3-21.9 (30)	2.4 (2.7)	6.1-19.7 (30)	
May		20.1 (2.0)	14.4-24.6 (31)	14.9 (2.8)	7.4-20.9 (31)	

Table 2.3 (continued)

Sampling period		MIK 0.50		GCK 2.4	
Year	Month	Mean	Range	Mean	Range
1989	June	24.3 (1.9)	19.1-29.5 (30)	18.3 (3.0)	11.4-25.3 (30)
	July	25.3 (1.9)	19.6-32.2 (26)	20.8 (2.13)	14.7-25.6 (26)
	August	26.7 (1.2)	23.1-29.9 (26)	21.3 (1.4)	16.8-24.1 (26)
	September	23.9 (1.3)	20.4-28.3 (30)	18.3 (2.0)	2.7-23.5 (30)
	October	18.3 (2.3)	13.6-24.9 (31)	10.3 (3.7)	2.6-21.4 (31)
	November	14.7 (1.5)	10.8-19.1 (30)	8.8 (2.0)	4.1-13.1 (30)
	December	11.3 (1.2)	5.8-14.5 (31)	3.7 (2.7)	-(0.2)-10.3 (31)
	January	10.6 (1.3)	5.7-14.2 (20)	7.2 (1.8)	2.5-10.7 (20)
	February	10.9 (2.6)	5.9-16.6 (16)	7.3 (2.2)	1.7-13.8 (28)
	March	NA	NA	10.2 (2.5)	4.9-18.1 (31)
	April	NA	NA	12.1 (2.5)	6.0-18.4
	May	19.5 (1.7)	15.1-24.2 (18)	14.2 (2.4)	9.0-20.9 (26)
June	20.2 (1.3)	17.5-25.4 (30)	16.7 (1.03)	14.1-20.8 (30)	
July	21.4 (1.7)	19.4-27.8 (31)	17.6 (1.1)	14.4-20.1 (31)	

Table 2.3 (continued)

Sampling period		MIK 0.50		GCK 2.4	
Year	Month	Mean	Range	Mean	Range
	August	23.3 (0.9)	21.0-27.3 (28)	17.8 (1.5)	13.1-21.5 (28)
	September	NA	NA	17.5 (1.8)	12.8-20.9 (30)
	October	NA	NA	13.1 (2.8)	7.1-18.5 (31)
	November	NA	NA	10.3 (2.0)	5.3-14.3 (30)
	December	NA	NA	4.9 (2.4)	0.4-9.8 (28)
1990	January	10.5 (1.0)	8.0-12.5 (12)	7.5 (1.6)	2.9-11.1 (31)
	February	11.6 (1.5)	7.7-16.7 (28)	9.4 (1.7)	4.6-12.8 (28)
	March	13.1 (1.8)	9.2-18.1 (31)	10.7 (2.1)	5.9-16.6 (31)
	April	15.7 (2.6)	9.8-24.1 (30)	12.7 (1.8)	10.0-16.6 (2)
	May	18.3 (2.0)	13.2-23.1 (29)	14.7 (1.5)	10.4-19.2 (29)
	June	22.1 (2.0)	17.6-31.1 (30)	16.4 (1.3)	12.2-20.8 (30)
	July	24.1 (1.3)	21.4-2.0 (31)	18.2 (1.1)	15.3-22.0 (31)
	August	24.2 (1.2)	20.8-28.8 (31)	18.9 (1.23)	16.1-23.3 (31)

*ND = no data available.

Note: Data were obtained with a Ryan Tempmentor digital temperature recorder with values recorded every 20 min, April-June 1987 and every 60 min beginning July 1987.

Table 2.4. Location and description of the eight sites on Mitchell Branch that were sampled in various tasks of the Oak Ridge K-25 Site Biological Monitoring and Abatement Program

Site ^a	Location ^b	Task			
		Toxicity monitoring	Bioaccumulation	Community studies	
				Benthos	Fish
MIK 0.12	NPDES monitoring station K-1700	X	X ^c	NS ^d	NS
MIK 0.45	Below storm drain 190 (45 m)	X	NS	X	X
MIK 0.54	Below K-1407-B effluent and storm drain 180 (90 m) ^e	X	NS	X	X
MIK 0.71	Below storm drain 170 (50 m)	X	NS	X	X
MIK 0.78	Above storm drain 170 (20 m)	NS	NS	X	X
MIK 0.86	Above storm drain 170 (100 m) near storage yard	NS	NS	X	NS
MIK 1.0	Above storm drain 170 (240 m) just downstream of Stoner Rd	X	NS	NS	NS
MIK 1.43 ^f	Above storm drain 170 (650 m) and the Oak Ridge K-25 Site	X	^g	X	X

^aMIK = Mitchell Branch kilometer; refers to the distance (in kilometers) above the confluence of Mitchell Branch with Poplar Creek.

^bDistance above/below storm drain is given in parentheses.

^cSampling was also conducted in Poplar Creek at PCK 6.9, ~300 m below the mouth of Mitchell Branch. At MIK 0.12, bioaccumulation began and continued throughout a reach which is actually designated as MIK 0.2.

^dNS = not sampled.

^eEffluent from the K-1407-B holding pond and storm drain 180 join just below the pond to form a single discharge to Mitchell Branch.

^fReference (control) site.

^gOther reference sites were sampled (see Fig. 2.2).

on a storage yard located immediately northeast of Mitchell Branch and (2) minor inputs from several storm drains located further upstream. The sampling sites for

the toxicity monitoring and community studies tasks overlap; these tasks/subtasks share four sites, and each includes at least five of the eight primary sites.

3. TOXICITY MONITORING

L. A. Kszos

The toxicity monitoring task outlined in the BMAP for Mitchell Branch (Loar et al. 1991) included three subtasks. The goals of the task were to (1) monitor ambient water toxicity (subtask 1a), (2) measure the toxicity of selected effluents (subtask 1b), and (3) determine point sources of toxicity (subtask 1c). Results of subtasks 1b and 1c are discussed in Sect. 3.1; results of subtask 1a are discussed in Sect. 3.2.

3.1 EFFLUENT TOXICITY

3.1.1 Introduction

The EPA supports the use of test organisms to determine the chronic toxicity of a test water (Horning and Weber 1985). As required under the modification of the Oak Ridge K-25 Site NPDES permit (EPA 1986), the toxicity of effluents discharging to Mitchell Branch were evaluated by using the 7-d fathead minnow (*Pimephales promelas*) larval survival and growth test and the 7-d *Ceriodaphnia* survival and reproduction test. These two tests are described in detail by Horning and Weber (1985). These tests are static renewal tests, which means that the test solutions are replaced daily for each species.

In October 1988, the K-1407-B pond (and its discharge point) was closed and replaced with the K-1407-E pond, the K-1407-F pond, and the CNF. Because the K-1407-E and K-1407-F ponds are filled and discharged alternately, they

will be discussed as one discharge (K-1407-E/F). Effluent from the CNF (which includes effluent from the TSCA Incinerator) discharges through the K-1407-J basin. Until September 1989, the K-1407-J basin discharged to Mitchell Branch via SD 170. Subsequently, the basin discharged to Poplar Creek. Because the K-1407-E/F pond and the K-1407-J basin are NPDES monitoring points, they were evaluated most extensively.

The three major storm drains that discharge to Mitchell Branch were monitored because (1) they discharge into a reach of Mitchell Branch that is known to be stressed (Smith et al. 1993); (2) toxicity tests conducted during the first year of the BMAP indicated that the effluents were toxic to fathead minnows and *Ceriodaphnia*; and (3) after the K-1407-B pond was closed, SDs 170 and 180 were the conduits for effluent from K-1407-E and K-1407-F ponds respectively. Sources of water in the storm drains are as follows: SD 170, which includes K-1407-E pond effluent, K-1407-J basin effluent (until September 1989), once-through cooling water, cooling tower blowdown, roof drains, area runoff, and groundwater; SD 180, which includes K-1407-F pond effluent, groundwater, garage area runoff, K-1405 sink drains, K-1401 floor drains, roof drains, once-through cooling water, and area runoff; and SD 190, which includes K-1401 floor drains, roof drains, once-through cooling water, effluent from K-1045-A Fire Training Facility, and area runoff (Scheib 1987).

3.1.2 Materials and Methods

Toxicity tests with effluents from the K-1407-B pond, K-1407-E/F pond, K-1407-J basin, SDs 170, 180, and 190 were conducted by using the fathead minnow and *Ceriodaphnia dubia* chronic toxicity tests described in Sect. 3.1.1. The K-1407-B pond effluent was evaluated every other month from October 1986 to April 1988. Results of the tests conducted with effluent from K-1407-B pond during 1986-87 are discussed in Smith et al. (1993). A summary of all the tests conducted to date is provided in Sect. 3.1.2.1. The K-1407-E/F pond effluent was evaluated about every other month from April 1988 to June 1990. The K-1407-J basin effluent was evaluated every other month from December 1988 to June 1990. The number of toxicity tests used to evaluate storm drain effluents during the same period were, for SD 170, eight; for SD 180, six; and for SD 190, six.

For each effluent toxicity test, seven daily grab samples were taken at the point of discharge. All samples were delivered to the Aquatic Toxicology Laboratory at ORNL by personnel of the Oak Ridge K-25 Site via chain-of-custody procedures (Kszos et al. 1989). Time of collection, water temperature, and arrival time in the laboratory were recorded. Upon arrival in the laboratory, the water was warmed or cooled to 25°C and dilutions were made if necessary. Tests with the two species were usually conducted concurrently. On each day of a test, subsamples of each effluent or water sample were routinely analyzed for pH, conductivity, alkalinity, water hardness, total residual chlorine (TRC), and free chlorine (Kszos et al. 1988). Other chemical measurements were made by the Oak Ridge K-25 Site Process Support Department.

Dechlorination of the effluents from SDs 170, 180, and 190 were frequently

used to evaluate the contribution of TRC to toxicity. Dechlorination was accomplished by adding 0.1 N sodium thiosulfate dropwise to the effluent until TRC was 0.0 mg/L.

SAS statistical software [Statistical Analysis System for personal computers (PC-SAS), release 6.02] was used to analyze all data. Survival percentages for fathead minnow larvae and for *Ceriodaphnia* were transformed (arsine square-root; Steel and Torrie 1960) before being analyzed. All fecundity values are for females that survived all 7 d of the test. Significant reductions in *Ceriodaphnia* survival and fecundity (compared with a control) were determined by using Fisher's Exact Test and Dunnett's Procedure, respectively (Horning and Weber 1985). Significant reductions in fathead minnow survival and growth (compared with a control) were determined by using Dunnett's Procedure (Horning and Weber 1985). Dunnett's Procedure yields the least significant difference and the no-observed-effect concentration (NOEC). Unless noted otherwise, statements of significance are based on $p < 0.05$.

3.1.3 Results

3.1.3.1 K-1407-B Pond

A summary of all the toxicity tests and concurrent water quality measurements for tests conducted during 1987-88 are provided in Table 3.1. Results of the tests conducted during 1986-87 are discussed in Smith et al. 1993 and provided here for comparison with those from the K-1407-E/F pond. The effluent's NOECs for the two tests conducted in 1988 were 100% for both species. Water quality during the two tests was similar to that in 1986-87.

Table 3.1. Summary of toxicity test results and mean (± 1 SD) water quality parameters ($n = 7$) for effluent from the K-1407-B pond

Test Period	No-observed-effect concentration (%) ^a				
	Fathead minnow	<i>Ceriodaphnia</i>	Alkalinity ^b	Hardness ^b	Conductivity ^c
Oct 1986	100	<100	143 (25)	180 (47)	441 (74)
Dec 1986	100	<20	66 (8)	725 (124)	2260 (167)
Feb 1987	60	<20	36 (21)	525 (110)	1465 (261)
Mar 1987	NT ^d	<50	78 (12)	599 (235)	2464 (995)
Apr 1987	100	<100	61 (12)	701 (168)	1967 (256)
Jun 1987	<100	100	56 (14)	310 (18)	1353 (184)
Jul 1987 ^e	NT	100	60 (0)	445 (0)	1610 (0)
Aug 1987	<50	50	44 (3)	510 (120)	2295 (1159)
Oct 1987	100	<50	70 (8)	453 (41)	1657 (67)
Dec 1987	100	<100	72 (8)	410 (23)	1572 (120)
Feb 1988	100	100	97 (28)	455 (763)	3061 (427)
Apr 1988	100	100	97 (16)	502 (76)	1805 (294)

^aThe no-observed-effect concentration designates the highest concentration of the effluent tested causing no significant ($p > 0.05$) reduction in survival or growth of fathead minnow larvae, or survival or reproduction of *Ceriodaphnia*.

^bmg/L CaCO₃.

^c μ S/cm.

^dNT = not tested.

^eTest used one grab sample.

3.1.3.2 K-1407-E/F Pond

The results of the toxicity tests and concurrent water quality measurements are summarized in Table 3.2. From April 1988 to June 1990, effluent from K-1407-E/F pond was tested 14 times with fathead minnows and *Ceriodaphnia*; however, 2 of the *Ceriodaphnia* tests were invalid because of low fecundity in the controls. Results of these toxicity tests are summarized in terms of the NOEC of the effluents (Table 3.2). The effluent NOEC for fathead minnows was never less than 100%, whereas the effluent NOEC for *Ceriodaphnia* ranged

from 6 to 100%. The NOEC was 100% in only 4 of the 12 valid *Ceriodaphnia* tests.

Effluent from K-1407-E/F pond was characterized by periods of high conductivity and hardness (Table 3.2). During the April 1988 to June 1990 tests, pH, mean alkalinity, mean hardness, and mean conductivity ranged from 7.53 to 10.17, 37 to 93 mg/L, 343 to 741 mg/L, and 854 to 3695 μ S/cm respectively.

Results of chemical analyses obtained concurrently with some of the toxicity tests by the Oak Ridge K-25 Site Process Support Department are summarized in Table 3.3. Only those constituents that

Table 3.2. Summary of toxicity test results and mean (± 1 SD) water quality parameters ($n = 7$) for effluent from the K-1407-E/F ponds

Test Period	No-observed-effect concentration (%) ^a				
	Fathead minnow	<i>Ceriodaphnia</i>	Alkalinity ^b	Hardness ^b	Conductivity ^c
Apr 1988	100	100	68 (12)	538 (162)	2204 (425)
Jun 1988	100	100	71 (21)	343 (103)	1655 (365)
Aug 1988	100	100	68 (11)	535 (151)	2355 (1856)
Oct 1988	100	I ^d	75 (5)	722 (106)	2838 (331)
Dec 1988	100	I	75 (3)	764 (77)	2761 (131)
Feb 1989	100	50	60 (5)	445 (88)	1610 (335)
Apr 1989	100	<25	89 (23)	475 (141)	1466 (335)
Jun 1989	100	100	68 (9)	297 (68)	854 (195)
Aug 1989	100	25	66 (7)	413 (102)	1864 (619)
Oct 1989	100	12	93 (11)	398 (72)	1686 (286)
Dec 1989	100	<6	69 (8)	703 (155)	2112 (213)
Feb 1990	100	25	76 (2)	434 (93)	2210 (543)
Apr 1990	100	6	59 (8)	741 (132)	2265 (236)
Jun 1990	100	6	37 (10)	814 (141)	3695 (324)

^aThe no-observed-effect concentration designates the highest concentration of the effluent tested causing no significant ($p > 0.05$) reduction in survival or growth of fathead minnow larvae, or survival or reproduction of *Ceriodaphnia*.

^bmg/L CaCO₃.

^c μ S/cm.

^dI = invalid test because of low fecundity.

had concentrations consistently above detection levels or that were of toxicological concern were included; complete data sets are available elsewhere (McGaha 1989a, 1989b, 1989c, 1989d; Shoemaker et al. 1990). High concentrations of Ca, Cl, dissolved solids, Na, and SO₄ are discharged from the K-1407-E/F ponds; the maximum concentrations of these substances measured during the toxicity tests were 250 mg/L, 714 mg/L, 2048 mg/L, 440 mg/L, and 693 mg/L respectively. Temporally, the concentrations of Ca, Cl, dissolved solids, Na, and SO₄ were quite variable and corresponded with patterns

of conductivity and hardness. The concentrations of most other substances varied less from test to test, with a few notable exceptions: (1) aluminum was elevated in the August 1988, February 1989, and June 1989 tests; (2) iron was elevated in the June and October 1989 tests; (3) manganese was elevated in the February 1989 test; and (4) nickel was elevated in the October 1989 test.

The concentrations of chemicals in K-1407-E/F pond effluent were very similar to those found in the K-1407-B pond effluent (Smith et al. 1993). For example, the K-1407-B pond and K-1407-E/F pond

Table 3.3. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in the effluents from the K-1407-E/F ponds obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity tests

Analysis	Aug 1988	Oct 1988	Dec 1988	Feb 1989	Apr 1989	Jun 1989	Oct 1989
Aluminum	0.41 (0.18-0.74)	0.03 (BD ^a -0.06)	0.18 (BD-0.78)	0.61 (BD-1.2)	0.18 (BD-0.43)	0.52 (0.14-1.1)	0.04 (0.14-0.57)
Barium	0.02 (0.007-0.04)	0.01 (0.004-0.011)	0.01 (BD-0.02)	0.03 (BD-0.04)	0.02 (0.01-0.03)	0.03 (0.03-0.04)	0.04 (0.02-0.04)
Boron	0.12 (0.08-0.16)	0.04 (0.03-0.05)	0.16 (0.05-0.35)	0.33 (0.02-0.59)	0.06 (0.05-0.08)	0.08 (0.06-0.12)	0.05 (0.04-0.06)
Calcium	127 (81-210)	190 (170-220)	203 (85-250)	133 (85-180)	157 (76-220)	91 (59-120)	125 (75-170)
Chloride	233 (54-465)	597 (500-714)	514 (456-563)	261 (167-419)	227 (136-282)	91 (32-133)	231 (173-324)
Dissolved solids	1051 (452-1788)	1790 (1570-2048)	1888 (1790-1998)	1041 (788-1416)	1017 (610-1266)	599 (414-838)	1229 (1160-1470)
Fluoride	0.3 (0.1-1.0)	0.2 (0.2-0.3)	0.3 (0.3-0.3)	0.2 (0.2-0.2)	0.2 (0.2-0.4)	0.4 (0.2-0.8)	0.2 (BD-0.2)
Iron	0.3 (0.13-0.58)	0.2 (0.03-0.27)	0.6 (0.2-1.1)	0.7 (BD-1.4)	0.4 (0.02-0.96)	1.0 (0.3-2.0)	1.2 (0.1-2.0)
Magnesium	11.2 (1.7-30.0)	23.4 (20-29)	28.3 (13-34)	22.9 (14-30)	21.1 (14-26)	12.0 (9-17)	18.4 (14-22)
Manganese	0.02 (0.008-0.03)	0.04 (0.03-0.06)	0.10 (0.04-0.14)	0.24 (0.13-0.31)	0.07 (0.02-0.15)	0.07 (0.04-0.10)	0.08 (0.04-0.13)

Table 3.3 (continued)

Analysis	Aug 1988	Oct 1988	Dec 1988	Feb 1989	Apr 1989	Jun 1989	Oct 1989
Nickel	BD	0.01 (BD-0.012)	0.05 (0.02-0.1)	BD	0.06 (BD-0.14)	0.02 (0.01-0.04)	0.32 (0.10-0.88)
Silicon	2.5 (1.3-3.0)	3.3 (2.8-4.1)	4.5 (3.9-7.1)	5.3 (2.9-7.3)	2.2 (1.7-2.7)	3.9 (3.2-4.6)	6.9 (5.9-7.4)
Sodium	150 (48-280)	374 (330-440)	315 (270-440)	192 (130-290)	160 (100-190)	63 (31-98)	197 (140-260)
Sulfate	385 (138-592)	507 (456-581)	651 (610-693)	372 (201-527)	343 (183-479)	249 (151-341)	381 (208-498)
Suspended solids	6.7 (4-14)	3.7 (2-6)	7.9 (4-21)	3.1 (BD-8)	6.6 (1-24)	11.7 (3-36)	10.6 (1-22)
Zinc	0.14 (0.09-0.22)	0.12 (0.03-0.37)	0.15 (0.05-0.29)	0.27 (0.08-0.57)	0.01 (BD-0.03)	0.03 (0.02-0.07)	0.03 (0.01-0.04)

*BD = below detection.

Sources: McGaha, M. A., 1989a, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-288, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989b, *Toxicity Monitoring at ORGDP October-December 1988*, Report No. K/QT-0311, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989c, *Toxicity Monitoring at ORGDP January-March 1989*, Report No. K/QT-312, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989d, *Toxicity Monitoring at ORGDP April-June 1989*, Report No. K/QT-313, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; and Shoemaker, J. L., et al., 1990, *Toxicity Monitoring at ORGDP October-December 1989*, Report No. K/QT-0373, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee

contained similar amounts (average of the mean for each test) of Al (0.21 vs 0.28 mg/L), Ca (173 vs 307 mg/L), Cl (297 vs 307 mg/L), Mg (18.5 vs 19.6 mg/L), and Na (197.5 vs 207 mg/L).

3.1.3.3 K-1407-J Basin

The results of the toxicity tests and concurrent water quality measurements are summarized in Table 3.4. From December 1988 to June 1990, effluent from K-1407-J basin was tested 11 times with fathead minnows and *Ceriodaphnia*. Results of these tests are summarized in terms of the effluent NOEC (Table 3.4). One of the *Ceriodaphnia* tests was invalid because of

low fecundity in the controls. A definitive NOEC could not be determined for two additional *Ceriodaphnia* tests and one fathead minnow test because (1) there was no effect at the highest concentration of effluent tested (60%) or (2) the lowest concentration tested (6%) caused a reduction in survival. The effluent NOEC was 100% for fathead minnows in nine of ten tests; the NOEC was <100% only during the test conducted in April 1990. The effluent NOEC was 100% for *Ceriodaphnia* in only two of eight conclusive tests; NOECs for the remaining tests ranged from 1 to 50%.

The effluent periodically had high alkalinity, hardness, and conductivity (Table 3.4). During the December 1988

Table 3.4. Summary of toxicity test results and mean (± 1 SD) water quality parameters ($n = 7$) for effluent from the K-1407-J basin

Test Period	No-observed-effect concentration (%) ^a				
	Fathead minnow	<i>Ceriodaphnia</i>	Alkalinity ^b	Hardness ^b	Conductivity ^c
Dec 1988	100	I ^d	444 (112)	250 (25)	4215 (850)
Feb 1989	100	12	228 (30)	284 (28)	2744 (504)
Mar 1989	>60	>60	174 (14)	244 (40)	1356 (305)
Apr 1989	100	50	148 (59)	332 (99)	1450 (383)
Jun 1989	100	12	382 (70)	136 (19)	8171 (1499)
Aug 1989	100	12	125 (30)	1431 (423)	4257 (1130)
Oct 1989	100	6	79 (8)	402 (85)	1148 (207)
Dec 1989	100	<6	81 (7)	452 (117)	1148 (226)
Feb 1990	100	1	93 (2)	418 (70)	1285 (223)
Apr 1990	50	100	177 (21)	262 (14)	676 (101)
Jun 1990	100	100	461 (85)	197 (12)	1926 (256)

^aThe no-observed-effect concentration designates the highest concentration of the effluent tested causing no significant ($p > 0.05$) reduction in survival or growth of fathead minnow larvae, or survival or reproduction of *Ceriodaphnia*.

^bmg/L CaCO₃.

^cμS/cm.

^dI = invalid test because of low fecundity in the control water.

to June 1990 tests, pH, mean alkalinity, mean hardness, and mean conductivity ranged from 7.67 to 8.88, 79 to 461 mg/L, 140 to 1431 mg/L, and 630 to 8536 $\mu\text{S}/\text{cm}$ respectively (Table 3.4).

Results of chemical analyses obtained concurrently with the toxicity tests by the Oak Ridge K-25 Site Process Support Department are summarized in Table 3.5. Only those constituents that had concentrations consistently above detection levels or that were of toxicological concern were included; complete data sets are available elsewhere (McGaha 1989a, 1989b, 1989c, 1989d; Shoemaker et al. 1990). High concentrations of Ca, Cl, dissolved solids, Na, and SO_4 were discharged from the pond; releases of these constituents corresponded with patterns of high conductivity and hardness. The highest concentrations of Ca, Cl, dissolved solids, Na, and SO_4 discharged during the toxicity tests were 240, 3470, 6632, 2200, and 970 mg/L respectively. The concentrations of most other substances varied less from test to test, with a few notable exceptions: (1) Al and Zn were elevated in the March 1989 test, (2) Ni was elevated in the April and October 1989 test, and (3) F was elevated in the June 1989 test.

3.1.3.4 Storm drains

Toxicity test results and water quality measurements of the storm drain effluents are summarized in Table 3.6. Daily concentrations of TRC in SD 170 effluent ranged from 0 to 1.78 mg/L; maximum concentrations during each test ranged from 0.22 to 1.78 mg/L. Untreated effluent was toxic (NOEC <100%) to minnows in six of seven tests; dechlorinating the effluent eliminated the toxicity in four of the five tests in which it was attempted. Untreated SD 170 effluent was toxic to *Ceriodaphnia* in all eight tests; dechlorination eliminated the toxicity in

two of the seven tests in which it was attempted.

Daily and maximum concentrations of TRC in SD 180 effluent ranged from 0 to 0.57 mg/L. In four tests (October 1988, July 1989, and January and May 1990), the concentration of TRC was not high enough to cause mortality of fathead minnows or *Ceriodaphnia* in 7 d (A. J. Stewart, Environmental Sciences Division, unpublished data). When the effluent contained high concentrations of TRC (0.36 and 0.57 mg/L), it was toxic to both species. Dechlorination of the effluent during one test (August 1988) removed the toxicity.

Daily concentrations of TRC in SD 190 effluent ranged from 0 to 0.99 mg/L; maximum concentrations during each test ranged from 0.06 to 0.99 mg/L. During all tests but one (May 1990), the maximum concentration of TRC was high enough to be toxic to *Ceriodaphnia* (A. J. Stewart, Environmental Sciences Division, unpublished data). Untreated SD 190 effluent was toxic to fathead minnows in three of six tests; dechlorination eliminated toxicity to the minnows in one of five of the tests in which it was attempted. Untreated SD 190 effluent was toxic to *Ceriodaphnia* in five of six tests; dechlorination eliminated the toxicity in three of five of the tests in which it was attempted.

Results of chemical analyses conducted concurrently with the toxicity tests by the Oak Ridge K-25 Site Process Support Department during July 1988 to July 1989 are summarized in Tables 3.7 through 3.9. Effluent from SD 180 typically contained the highest concentrations (compared with SDs 170 and 190) of Al, Ba, Ca, Cl, F, Fe, Mn, Na, SO_4 , dissolved solids, and suspended solids. The concentrations of these substances were particularly high in SDs 170 and 180 during the July 1989 test. Nickel was elevated in the July 1988 test of effluent from SD 190.

Table 3.5. Mean (range in parentheses) concentrations (in milligrams per liter) of selected parameters for the K-1407-J basin effluent obtained by the Oak Ridge K-25 Site Process Support Department taken in conjunction with some of the toxicity tests

Analysis	Dec 1988	Feb 1989	Mar 1989	Apr 1989	Jun 1989	Oct 1989
Aluminum	0.7 (BD ^a -1.2)	0.7 (BD-1.7)	1.1 (0.75-1.4)	0.2 (0.07-0.41)	0.7 (0.51-1.0)	0.7 (0.19-1.9)
Calcium	94 (41-240)	90 (79-100)	65 (52-86)	94 (70-150)	39 (31-56)	133 (60-180)
Chloride	730 (378-976)	311 (126-395)	129 (106-222)	191 (81-329)	2405 (1520-3470)	129 (73-258)
Dissolved solids	2695 (1640-3348)	1862 (1140-2094)	880 (746-1296)	920 (554-1320)	4900 (3278-6632)	894 (700-1170)
Fluoride	33.3 (15-50)	22.3 (10-28)	11.1 (8-20)	3.9 (2.6-4.6)	40.7 (30-55)	0.9 (0.8-1.1)
Iron	0.8 (0.19-1.2)	0.6 (0.31-1.1)	0.6 (0.34-1.2)	0.4 (0.13-0.77)	0.6 (0.2-1.0)	0.2 (BD-1.2)
Magnesium	14.1 (9-33)	9.7 (8-10)	18.7 (11-27)	17.9 (14-23)	4.5 (4-6)	10.6 (6-13)
Manganese	0.05 (0.04-0.07)	0.15 (0.12-0.20)	0.02 (BD-0.03)	0.04 (0.02-0.07)	0.03 (0.03-0.04)	0.04 (0.01-0.07)
Nickel	0.07 (0.02-0.11)	BD	BD	0.27 (BD-0.05)	0.05 (0.02-0.09)	0.22 (0.06-0.47)
Silicon	6.0 (2.3-8.9)	8.2 (5.5-11.0)	5.1 (4.1-7.0)	1.9 (1.5-2.3)	6.6 (5.3-8.3)	2.9 (1.3-4.8)

Table 3.5 (continued)

Analysis	Dec 1988	Feb 1989	Mar 1989	Apr 1989	Jun 1989	Oct 1989
Sodium	654 (40-970)	554 (290-630)	228 (180-410)	188 (97-260)	1671 (1200-2200)	92 (32-110)
Sulfate	667 (473-970)	679 (488-762)	280 (230-406)	289 (175-429)	464 (315-611)	398 (275-656)
Suspended Solids	18.6 (9-25)	13.4 (4-46)	10.0 (230-406)	6.6 (4-9)	14.4 (8-23)	13.6 (3-32)
Zinc	0.26 (0.02-0.45)	0.37 (0.11-1.1)	0.48 (0.17-0.78)	0.01 (0.01-0.02)	0.05 (0.04-0.06)	0.06 (0.04-0.08)

*BD = below detection.

Sources: McGaha, M. A., 1989a, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-288, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989b, *Toxicity Monitoring at ORGDP October-December 1988*, Report No. K/QT-0311, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989c, *Toxicity Monitoring at ORGDP January-March 1989*, Report No. K/QT-312, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989d, *Toxicity Monitoring at ORGDP April-June 1989*, Report No. K/QT-313, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; and Shoemaker, J. L., et al., 1990, *Toxicity Monitoring at ORGDP October-December 1989*, Report No. K/QT-0373, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee

Table 3.6. Summary of toxicity test results and concentration (in milligrams per liter) of total residual chlorine (TRC) in the storm drain effluents

Storm drain	Test period	No-observed-effect concentration ^a (%)		Range TRC
		Fathead minnow	<i>Ceriodaphnia</i>	
170	Jul 1988	25	<25	0.04-1.65
	Aug 1988	<100	<100	0.00-0.47
	dechlor.	100	<100	
	Oct 1988	<100	<100	0.00-1.50
	dechlor.	<100	<100	
	Dec 1988	<100	<100	0.90-1.78
	dechlor.	100	100	
	Feb 1989	NT ^b	<100	0.11-1.10
	dechlor.	NT	<100	
	Jul 1989	100	<100	0.08-0.22
	dechlor.	100	<100	
	Jan 1990	<100	<100	0.00-0.64
dechlor.	100	<100		
May 1990	<100	<100	0.00-0.69	
dechlor.	100	100		
180	Jul 1988	50	50	0.00-0.36
	Aug 1988	<100	<100	0.00-0.57
	dechlor.	100	100	
	Oct 1988	100	100	0.00-0.00
	Jul 1989	100	100	0.00-0.09
	Jan 1990	100	<100	0.00-0.00
	May 1990	100	100	0.00-0.05
190	Jul 1988	25	25	0.00-0.99
	Aug 1988	<100	<100	0.00-0.87
	dechlor.	100	100	
	Oct 1988	<100	<100	0.00-0.96
	dechlor.	<100	<100	
	Jul 1989	100	<100	0.20-0.30
	dechlor.	NT	100	
	Jan 1990	100	<100	0.00-0.26
	dechlor.	100	100	
May 1990	100	100	0.00-0.06	

^aNo-observed-effect concentration designates the highest concentration of the effluent tested causing no significant ($p > 0.05$) reduction in survival or growth of fathead minnow larvae, or survival or reproduction of *Ceriodaphnia*.

^bNT = not tested.

Table 3.7. Mean (range in parentheses) concentration (in milligrams per liter) of selected parameters in Storm Drain 170 effluent obtained by the Oak Ridge K-25 Site Process Support Department taken in conjunction with some of the toxicity tests

Analysis	Jul 1988 (n = 3) ^a	Aug 1988 (n = 6)	Oct 1988 (n = 1)	Dec 1988 (n = 2)	Feb 1989 (n = 7)	Jul 1989 (n = 7)
Aluminum	0.07 (0.05-0.1)	0.14 (0.02-0.51)	0.065	0.24 (0.23-0.25)	0.50 (BD ^b -0.56)	0.18 (0.06-0.55)
Barium	0.028 (0.027-0.029)	0.024 (0.02-0.027)	0.031	0.026 (0.026-0.026)	0.041 (BD-0.13)	0.036 (0.031-0.045)
Calcium	34 (33-36)	30 (14-37)	33	41 (40-43)	40 (30-51)	54 (32-120)
Chloride	44 (20-86)	20 (BD-28)	13	29 (26-32)	18 (15-27)	48 (2-154)
Chromium	0.16 (BD-0.43)	0.012 (BD-0.02)	0.02	0.020 (0.015-0.024)	BD	0.02 (BD-0.03)
Copper	0.01 (0.01-0.01)	0.02 (BD-0.06)	0.02	0.012 (BD-0.014)	BD	0.04 (BD-0.21)
Dissolved solids	284 (212-396)	213 (96-252)	174	285 (280-290)	205 (164-274)	339 (164-722)
Fluoride	BD	0.12 (BD-0.2)	0.10	0.25 (0.2-0.3)	BD	BD
Iron	0.6 (BD-1.5)	0.2 (0.01-0.44)	0.1	0.4 (0.40-0.41)	BD	0.4 (0.11-0.37)
Magnesium	9.4 (8.9-10)	7.5 (2.2-8.9)	8.3	10.2 (9.3-11)	8.9 (5.9-12)	10.8 (9.1-14)
Manganese	0.02 (BD-0.04)	0.02 (0.02-0.03)	0.02	0.04 (0.03-0.05)	0.04 (BD-0.09)	0.08 (0.04-0.16)
Nickel	0.09 (BD-0.24)	0.02 (BD-0.03)	0.01	0.02 (0.01-0.02)	BD	0.03 (BD-0.086)

Table 3.7 (continued)

Analysis	Jul 1988 (n = 3)	Aug 1988 (n = 6)	Oct 1988 (n = 1)	Dec 1988 (n = 2)	Feb 1989 (n = 7)	Jul 1989 (n = 7)
Phosphorus	0.90 (0.47-1.3)	1.25 (BD-1.6)	0.3	1.22 (0.65-1.8)	NM ^c	NM
Potassium	3.5 (3.3-3.7)	5.6 (3.3-6.4)	NM	NM	37.7 (19-54)	9.3 (1.5-13)
Silicon	1.2 (0.8-1.6)	2.4 (2.2-2.5)	1.9	2.1 (1.9-2.3)	0.8 (BD-1.3)	3.5 (2.9-4.8)
Sodium	17 (15-20)	25 (6-31)	12	35 (32-37)	23 (15-31)	33 (2-85)
Sulfate	90 (47-169)	52 (16-64)	43	81 (79-82)	46 (34-72)	100 (12-235)
Suspended solids	BD	4.2 (BD-9)	2.0	4.5 (3-6)	BD	5.0 (BD-15)
Zinc	0.02 (BD-0.03)	0.10 (0.03-0.16)	0.09	0.08 (0.06-0.1)	0.08 (BD-0.15)	0.03 (0.01-0.06)

^an = sample size.

^bBD = below detection.

^cNM = not measured.

Sources: McGaha, M. A., 1989a, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-288, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989b, *Toxicity Monitoring at ORGDP October-December 1988*, Report No. K/QT-0311, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989c, *Toxicity Monitoring at ORGDP January-March 1989*, Report No. K/QT-312, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989d, *Toxicity Monitoring at ORGDP April-June 1989*, Report No. K/QT-313, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; and Shoemaker, J. L., et al., 1990, *Toxicity Monitoring at ORGDP October-December 1989*, Report No. K/QT-0373, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

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Table 3.8. Mean (range in parentheses) concentrations (in milligrams per liter) of selected parameters in Storm Drain 180 effluent obtained by the Oak Ridge K-25 Site Process Support Department taken in conjunction with some of the toxicity tests

Analysis	Jul 1988 (n = 3) ^a	Aug 1988 (n = 6)	Oct 1988 (n = 7)	Jul 1989 (n = 7)
Aluminum	0.07	3.90	1.70	0.06
Barium	0.044 (0.042-0.045)	0.068 (0.047-0.17)	0.096 (0.06-0.12)	0.057 (0.047-0.068)
Calcium	39 (38-41)	36 (20-41)	56 (30-70)	73 (43-160)
Chloride	20.7 (19-23)	15.3 (BD-26)	31.1 (BD-44)	64.0 (26-166)
Chromium	0.15 (BD-0.37)	0.01 (BD-0.02)	0.01 (BD-0.02)	BD
Dissolved solids	238 (222-254)	198 (150-252)	263 (134-318)	604 (230-1788)
Fluoride	0.10 (0.1-0.1)	0.13 (BD-0.2)	0.34 (BD-0.7)	0.53 (0.1-3.0)
Iron	0.4 (BD-1.2)	3.6 (0.01-21)	1.9 (BD-11)	0.10 (0.05-0.17)
Magnesium	10.0 (9.9-10)	10.0 (5.9-11)	11.9 (5.5-15)	13.1 (10-21)
Manganese	0.05 (0.04-0.07)	0.10 (0.06-0.23)	0.26 (0.21-0.37)	0.38 (0.16-0.55)
Nickel	0.08 (BD-0.19)	BD	0.01 (BD-0.02)	0.02 (BD-0.04)
Potassium	3.0 (2.6-3.7)	7.1 (2.5-7.5)	NM ^c	12.4 (1.9-58)
Silicon	1.3 (0.1-1.6)	7.6 (2-35)	4.6 (2.2-15)	3.2 (2.8-4.6)
Sodium	8.6 (8.1-9.3)	8.1 (1.8-10)	13.5 (5.8-18)	73.2 (7.9-340)
Sulfate	38.0 (38-38)	32.5 (17-36)	40.0 (19-47)	293.3 (31-1090)

Table 3.8 (continued)

Analysis	Jul 1988 (n = 3) ^a	Aug 1988 (n = 6)	Oct 1988 (n = 7)	Jul 1989 (n = 7)
Suspended	BD	83.5	30.9	2.9
Zinc	0.02 (BD-0.03)	0.10 (0.06-0.13)	0.06 (0.03-0.08)	0.02 (0.01-0.03)

^an = sample size.

^bBD = below detection.

^cNM = not measured.

Sources: McGaha, M. A., 1989a, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-288, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989b, *Toxicity Monitoring at ORGDP October-December 1988*, Report No. K/QT-0311, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989c, *Toxicity Monitoring at ORGDP January-March 1989*, Report No. K/QT-312, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989d, *Toxicity Monitoring at ORGDP April-June 1989*, Report No. K/QT-313, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; and Shoemaker, J. L., et al., 1990, *Toxicity Monitoring at ORGDP October-December 1989*, Report No. K/QT-0373, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee

Effluent from SD 190 typically contained the lowest concentrations of the parameters measured.

3.1.4 Discussion

3.1.4.1 K-1407-B Pond

During the two tests completed in 1988, the effluent from the K-1407-B pond was not toxic to fathead minnows or *Ceriodaphnia*. These were the last two tests completed before closure of the pond began.

3.1.4.2 K-1407-E/F Pond

Effluent from K-1407-E/F pond was never toxic to fathead minnows but was nearly always toxic to *Ceriodaphnia*. The

mean effluent NOEC for *Ceriodaphnia* was 52.4% for those tests that were conclusive (range = 6 to 100%). The toxicity of the effluent appeared to be linked to those constituents that cause high hardness and conductivity levels (e.g., calcium and sulfate). For example, the maximum calcium concentration during the February 1989 test was 189 mg/L. Assuming the Ca is balanced with SO₄, the concentration of CaSO₄ would equal 643 mg/L. In a 7-d toxicity test with reagent-grade CaSO₄, 615 mg of CaSO₄ per L reduced fecundity of *Ceriodaphnia* by 62% (L. A. Kszos, Environmental Sciences Division, and M. L. Holtz, University of Kentucky, unpublished data). The measured NOEC (50%) for the February 1989 test with K-1407-E/F pond effluent could therefore be accounted for by the concentration of CaSO₄ in the effluent. Although chemical analyses are not yet published for the tests

Table 3.9. Mean (range in parentheses) concentrations (in milligrams per liter) of selected parameters in Storm Drain 190 effluent obtained by the Oak Ridge K-25 Site Process Support Department taken in conjunction with some of the toxicity tests

Analysis	Jul 1988 (n = 4) ^a	Aug 1988 (n = 3)	Oct 1988 (n = 7)	Jul 1989 (n = 7)
Aluminum	0.71	0.35	0.19	0.07
Barium	0.04 (0.03-0.05)	0.03 (0.01-0.04)	0.03 (0.02-0.05)	0.04 (0.04-0.04)
Calcium	36.3 (19-51)	28.3 (11-40)	31.9 (17-40)	39.0 (37-40)
Chloride	16.8 (11-25)	9.7 (BD ^b -14)	12.3 (BD-20)	16.9 (14-21)
Chromium	0.41 (BD-0.62)	0.01 (BD-0.01)	BD	BD
Copper	0.02 (0.01-0.04)	0.015 (BD-0.025)	0.013 (BD-0.029)	0.008 (BD-0.02)
Dissolved solids	208 (70-286)	148 (94-184)	158 (68-196)	190 (98-212)
Fluoride	0.10 (0.01-0.10)	0.10 (BD-0.10)	0.20 (BD-0.30)	0.10 (0.10-0.10)
Iron	2.0 (1.8-2.3)	0.4 (0.28-0.49)	0.4 (0.12-1.40)	0.20 (0.13-0.44)
Magnesium	8.8 (4.4-11)	7.6 (1.9-11)	8.9 (3.6-12)	9.9 (9.8-10)
Manganese	0.08 (0.06-0.10)	0.04 (0.03-0.05)	0.06 (0.03-0.09)	0.07 (0.06-0.09)
Nickel	0.21 (BD-0.33)	0.01 (BD-0.02)	0.01 (BD-0.03)	BD
Potassium	2.8 (2.1-3.8)	1.4 (BD-2.2)	NM ^c	1.8 (1.0-2.3)
Silicon	1.8 (1.6-2.1)	1.9 (1.5-2.2)	1.6 (1.0-2.3)	2.7 (2.6-3.0)
Sodium	6.2 (0.9-9.3)	5.1 (1.7-7.1)	5.7 (2.5-7.6)	5.9 (2.6-6.5)

Table 3.9 (continued)

Analysis	Jul 1988 (n = 4) ^a	Aug 1988 (n = 3)	Oct 1988 (n = 7)	Jul 1989 (n = 7)
Sulfate	34.8	25.3	29.6	31.6
Suspended solids	23.5 (BD-86)	11.0 (7-16)	5.4 (BD-15)	2.1 (BD-6)
Zinc	0.10 (0.05-0.25)	0.10 (0.09-0.12)	0.10 (0.06-0.14)	0.05 (0.04-0.07)

^an = sample size.

^bBD = below detection.

^cNM = not measured.

Sources: McGaha, M. A., 1989a, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-288, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989b, *Toxicity Monitoring at ORGDP October-December 1988*, Report No. K/QT-0311, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989c, *Toxicity Monitoring at ORGDP January-March 1989*, Report No. K/QT-312, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; McGaha, M. A., 1989d, *Toxicity Monitoring at ORGDP April-June 1989*, Report No. K/QT-313, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee; and Shoemaker, J. L., et al., 1990, *Toxicity Monitoring at ORGDP October-December 1989*, Report No. K/QT-0373, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

conducted during April and June 1990, the low NOECs for *Ceriodaphnia* are probably linked to high Ca, as indicated by the high hardness of the effluents (741 and 814 mg/L respectively). Nickel may also contribute to the toxicity of the effluent. During the October 1989 test, the mean concentration of nickel was 0.32 mg/L; 0.05 mg/L in Bear Creek water is toxic to *Ceriodaphnia* (Kszos et al. 1992).

The toxicity and water quality of K-1407-E/F ponds and the closed K-1407-B pond were similar. Although the toxicity tests of the K-1407-B pond effluent were often inconclusive (adverse affect at the lowest concentration tested), the K-1407-B ponds and the K-1407-E/F pond effluent NOECs were <100% for 8 of 12 *Ceriodaphnia* tests completed. Although

effluent from the K-1407-E/F ponds was never toxic to fathead minnows, effluent NOEC from the K-1407-B pond was <100% in three of ten tests. Water quality of effluents from the K-1407-E/F and K-1407-B ponds were also similar: average values for the mean concentrations of alkalinity, hardness, and conductivity measured for all tests conducted were 69.6 vs 73.3 mg/L, 544 vs 484 mg/L, and 2122 vs 1829 μ S/cm respectively.

The effect of the K-1407-E/F pond effluents on the aquatic biota in Mitchell Branch will obviously depend on the flow characteristics of the stream. During periods when flow in Mitchell Branch is low and the concentration of the effluent in the stream is >50%, the toxicity tests indicate that K-1407-E/F pond

effluent may adversely affect the stream biota. Currently, however, the toxicity of the TRC discharged from SD 170 overwhelms the toxicity that might be caused in Mitchell Branch because of inputs of effluent from the K-1407-E/F ponds (see discussion of storm drains, Sect. 3.1.4.4). During periods when flow in Mitchell Branch is high, it is less likely that the pond effluent will adversely affect the stream biota.

3.1.4.3 K-1407-J Basin

Effluent from the K-1407-J basin was generally not toxic to fathead minnows; an NOEC of <100% was obtained in only one test. *Ceriodaphnia*, on the other hand, were much more sensitive to this effluent, with a mean NOEC of 36.6% for those tests that were definitive (range: 1 to 100%). The toxicity of the K-1407-J basin effluent is partly the result of the high concentrations of Na, Cl, and SO₄ in the effluent. During the June 1989 test, for example, 1671 mg of Na per liter in the effluent balanced ironically with Cl and SO₄, yielding total concentrations of NaCl and Na₂SO₄ in the effluent of ~3942 mg/L and 682 mg/L respectively. In *Ceriodaphnia* toxicity tests with reagent-grade NaCl and Na₂SO₄, 1225 mg of NaCl per liter was acutely toxic and 500 mg of NaSO₄ per liter reduced fecundity (L. A. Kszos and L. F. Wicker, Environmental Sciences Division, unpublished data). Thus, 3942 mg of NaCl per liter in the K-1407-J basin effluent would be toxic to *Ceriodaphnia* at a concentration of 30%; the measured lowest-observed-effect concentration (LOEC) during the June 1989 test was 25% (NOEC = 12%). The toxicity of the effluent, however, may not always be accounted for by the presence of high Na, Cl, and/or SO₄. For example, during the April 1989 test, 313 mg of NaCl per liter and 424 mg of Na₂SO₄ per liter were present (assuming

the Cl and SO₄ were completely balanced with Na). Neither of these concentrations were found to be toxic in the tests with reagent-grade chemicals, yet the measured NOEC was 50%. Thus, it appears that there are one or more unknown toxicant(s) present in the effluent besides Na, Cl, and SO₄.

Because K-1407-J effluent is discharged to Poplar Creek, this wastewater does not affect the biota in Mitchell Branch. The receiving water concentration of the effluent in Poplar Creek is 1% (B. A. Shoemaker, ORGDP Health, Safety and Environment Division, personal communication), thus, it is unlikely that the biota of Poplar Creek would be adversely affected.

3.1.4.4 Storm drains

High TRC is a major source of toxicity in SDs 170 and 190. The lowest maximum concentration of TRC in SDs 170 and 190 was 0.22 mg/L and 0.06 mg/L respectively. In general, TRC concentrations of about 0.20 mg/L are lethal to *Ceriodaphnia* (A. J. Stewart, Environmental Sciences Division, unpublished data). The EPA water quality criterion for Cl is even lower (0.011 mg/L; EPA 1986). Effluent from SD 170 was always toxic to *Ceriodaphnia* and in most cases, to fathead minnows. Dechlorinating the effluent from SD 170 did not always eliminate the toxicity to either species, indicating the presence of one or more toxicants other than chlorine in the effluent.

Effluent from SD 190 was toxic to *Ceriodaphnia* during all tests but one; the maximum concentration of TRC during the one test was 0.06 mg/L. Dechlorination of the effluent always removed the toxicity to fathead minnows, but did not always eliminate toxicity to *Ceriodaphnia*. *Ceriodaphnia* were therefore more sensitive than fish to the toxicant(s) other than chlorine in the effluent.

Effluent from SD 180 typically contained only low levels of TRC and was the least toxic of the storm drain effluents. In the two tests with elevated TRC, the effluent was toxic to both species. Effluent collected during one of the four tests without elevated TRC was toxic to *Ceriodaphnia*. (Metal concentrations in this effluent are not yet available.)

Chemical analyses of metals and other miscellaneous constituents in the effluent from the storm drains (Tables 3.7 through 3.9) provided little insight. Effluents that were dechlorinated but still toxic did not appear to have any constituents that were elevated relative to the dechlorinated effluent that was not toxic.

Effluents from the storm drains, particularly SDs 170 and 190, have a high potential to adversely impact aquatic life in Mitchell Branch. Measurements of TRC taken at the end-of-pipe (as reported above) and the toxicity tests indicate that the TRC in these effluents will adversely affect the biota. In addition, tests with dechlorinated effluents have shown that, on occasion, other constituents in the effluents may contribute to toxicity.

3.2 AMBIENT TOXICITY

3.2.1 Introduction

Ambient toxicity testing was incorporated into the Mitchell Branch BMAP to (1) evaluate area-source contributions to stream toxicity, (2) characterize patterns of toxicity in Mitchell Branch, (3) document changes in water quality attributable to changes in operations at the Oak Ridge K-25 Site, and (4) provide data to demonstrate that the effluent limitations established for the Oak Ridge K-25 Site protect and maintain the use of Mitchell Branch for growth and propagation of fish and aquatic life (Loar et al. 1992a). The sites chosen for testing were selected to bracket

area- and point-source discharges into the stream and to correspond closely to those selected as instream monitoring study sites.

3.2.2 Materials and Methods

Ambient toxicity was evaluated by means of the fathead minnow (*Pimephales promelas*) larval survival and growth test and the *Ceriodaphnia* survival and reproduction test as described by Horning and Weber (1985). These tests are 7-d static-renewal chronic tests, based on the survival and growth of the fathead minnow and survival and fecundity of the microcrustacean *Ceriodaphnia*.

The six sites evaluated were located at MIKs 1.43, 1.0, 0.71, 0.54, 0.45, and 0.12 (Fig. 2.1). MIK 1.43 was selected as a reference site because it is located upstream of the Oak Ridge K-25 Site operations and any known source of perturbation. Twenty-two tests were conducted on a bimonthly basis from January 1987 through July 1990. Water sampling and water chemistry analyses were conducted as described in Sect. 3.1.2. Water collected from MIK 0.12, however, was a daily 24-h composite for the first four tests and a daily grab sample for the remaining tests. The switch to daily grab samples for this site was made when it was discovered that TRC could sometimes be detected at MIK 0.12.

All data analyses were accomplished as described in Sect. 3.1.2, with the following exceptions. Differences in hardness, alkalinity, conductivity, fathead minnow growth, and *Ceriodaphnia* reproduction between sites were evaluated with an analysis of variance (ANOVA) by using the SAS general linear models (GLM) procedure (SAS 1985a, 1985b); significant differences were then separated with Tukey's studentized range test. The Tukey test was selected to test for differences because it accommodates unequal sample

sizes, whereas Dunnett's test does not. The ANOVA proved to be inappropriate for distinguishing differences among sites and years for fathead minnow and *Ceriodaphnia* survival because the values at most sites had bimodal distributions (i.e., indicative of an all-or-none response). Unless otherwise noted, statements of significance are based on $p < 0.05$.

3.2.3 Results

3.2.3.1 Water chemistry

Measurements taken of three of the water quality parameters (conductivity, hardness, and alkalinity) are presented in Fig. 3.1. Six tests per year were completed (7 d of analyses per test) in 1987, 1988, and 1989; four tests were completed in 1990. Each year, conductivity and hardness increased with distance downstream. During 1987 and 1988, three reaches of the stream were distinctly different based on conductivity and hardness: (1) MIKs 1.43 and 1.0; (2) MIKs 1.0 and 0.71; and (3) MIKs 0.54, 0.45 and 0.12. The increase in mean conductivity and hardness from MIKs 1.43 to 0.12 was $\sim 600 \mu\text{S}/\text{cm}$ and $140 \text{ mg}/\text{L}$ respectively. The largest increase in hardness and conductivity occurred downstream of the K-1407-B pond discharge point between MIKs 0.71 and 0.54. In 1989 and 1990, three reaches of the stream were still distinct based on significant differences in conductivity: (1) MIK 1.43; (2) MIK 1.0; and (3) MIKs 0.71, 0.54, 0.45, and 0.12. The sharp increase between MIKs 0.71 and 0.54 no longer existed. In 1989 and 1990, downstream increases in conductivity were less dramatic, with mean conductivity from MIKs 1.43 to 0.12 increasing by only about $300 \mu\text{S}/\text{cm}$. This reduction was the result of a drop in conductivity at MIKs 0.54, 0.45, and 0.12 during 1989 and 1990; there was no significant change in

the conductivity at MIK 0.71 from 1987 to 1990. In 1989 and 1990, mean hardness showed a trend similar to that of conductivity; there was no significant difference in mean hardness from MIKs 0.71 to 0.45 in 1989 or from MIKs 0.71 to 0.12 in 1990. In 1990, the increase in hardness from MIKs 1.43 to 0.12 was $120 \text{ mg}/\text{L}$.

Trends for alkalinity in Mitchell Branch were different than those for conductivity and hardness. In 1987, alkalinity at MIK 1.0 was significantly higher than at MIKs 1.43, 0.71, 0.54, and 0.45; there was no significant difference in alkalinity from MIKs 0.71 to 0.12. During 1988, alkalinity also increased at MIK 1.10, but unlike the trend in 1987, alkalinity remained elevated at all downstream sites. In 1989 and 1990, there was a more gradual increase in alkalinity with distance downstream. In 1989, alkalinity at MIK 0.12 was significantly higher than that at all upstream sites, with no difference from MIKs 0.71 to 0.45. In 1990, there was no difference in alkalinity from MIKs 0.71 to 0.12.

The pH in Mitchell Branch generally increased with distance downstream (Table 3.10). During 1988, water from MIKs 0.54, 0.45, and 0.12 had the largest range in values. During the remaining years, the range in values at each site was similar.

Cumulative frequency distributions of TRC for all sampling dates since 1987 at MIKs 0.12, 0.45, 0.54, and 0.71 are shown in Fig. 3.2. The concentration of TRC in Mitchell Branch was typically highest at MIK 0.71 (below SD 170). At MIK 0.71, $\sim 40\%$ of the measured values were $>0.20 \text{ mg}/\text{L}$ (the concentration that is usually lethal to *Ceriodaphnia*; A. J. Stewart, Environmental Sciences Division, unpublished data). The concentration of TRC diminished with distance downstream: 30% of the TRC values at MIK 0.54 were $>0.20 \text{ mg}/\text{L}$, about 25% of the values at MIK 0.45 were $>0.20 \text{ mg}/\text{L}$,

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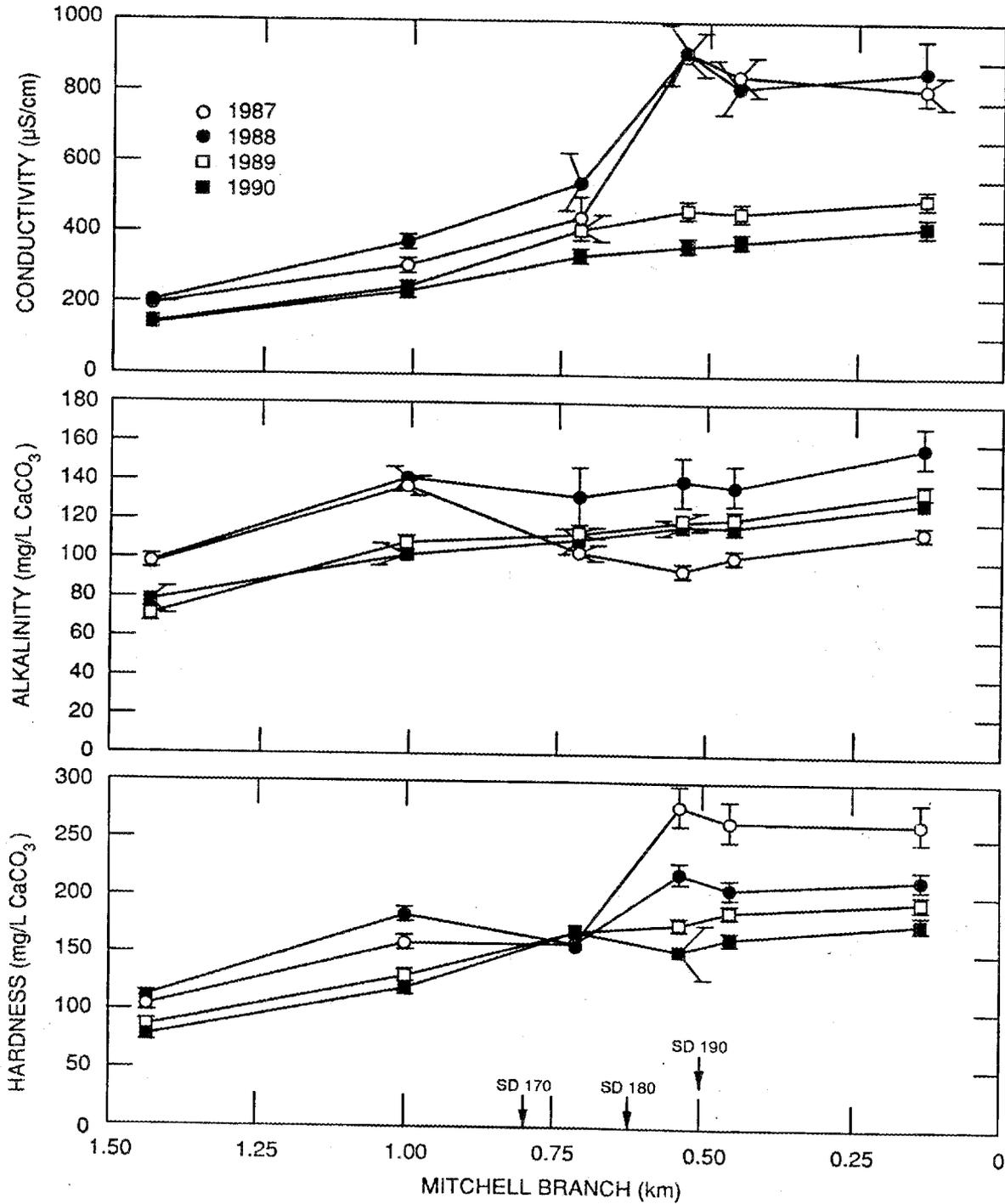


Fig. 3.1. Annual mean conductivity, alkalinity, and hardness for each site in Mitchell Branch, 1987-90. Vertical bars represent ± 1 SE. SD = storm drain.

Table 3.10. Range in pH values at each site in Mitchell Branch for all tests within each year

Site	Range in pH values			
	1987	1988	1989	1990
MIK 0.12	7.72-8.42	7.06-8.90	7.61-8.40	7.74-8.18
MIK 0.45	7.74-8.21	7.01-8.48	7.54-8.12	7.73-8.20
MIK 0.54	7.82-8.21	7.29-8.37	7.52-8.09	7.61-8.21
MIK 0.71	7.79-8.34	7.88-8.32	7.48-8.14	7.62-8.23
MIK 1.0	7.78-8.29	7.73-8.20	7.53-8.12	7.49-8.14
MIK 1.43	7.52-8.18	7.64-8.40	7.30-8.11	7.13-8.53

Note: MIK = Mitchell Branch kilometer.

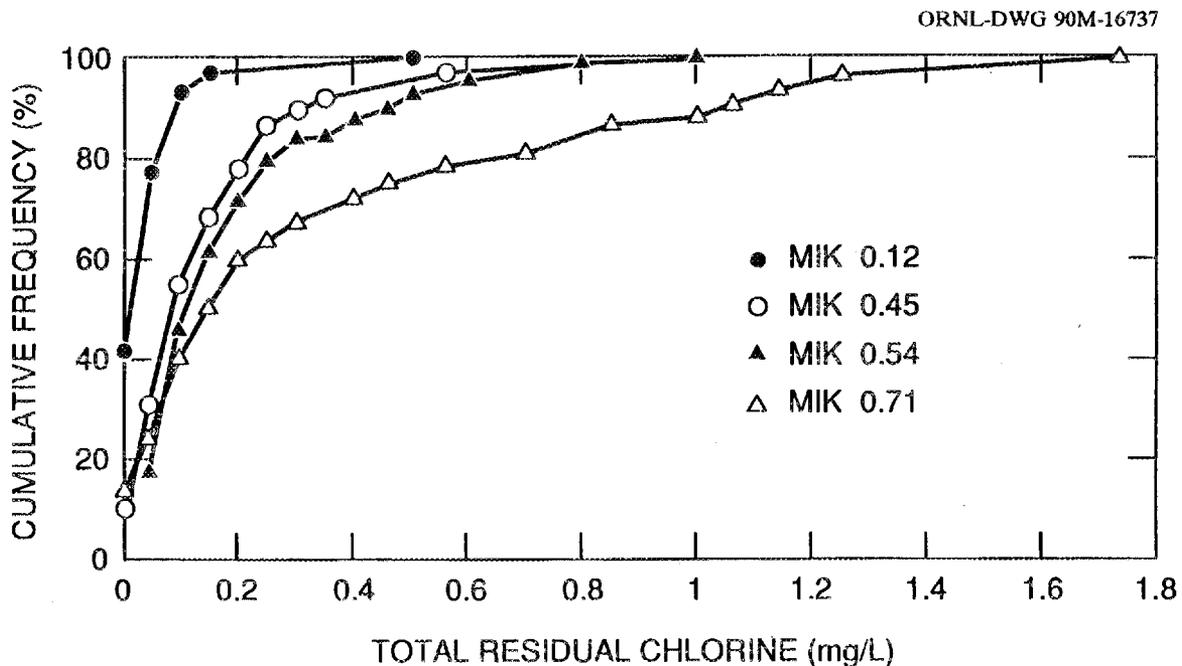


Fig. 3.2. Cumulative frequency (percentage) of total residual chlorine at 4 sites in Mitchell Branch for 22 tests, 1987-90.

and only about 5% of the values at MIK 0.12 were >0.20 mg/L. Cumulative frequency distributions of TRC for each year at MIKs 0.12, 0.45, 0.54, and 0.71 are shown in Figs. 3.3 and 3.4. At all four

sites, TRC concentrations were highest in 1988. During 1987, 1989, and 1990, TRC at MIK 0.12 was never higher than 0.20 mg/L; however, in 1988, TRC was ≥ 0.20 mg/L about 16% of the time.

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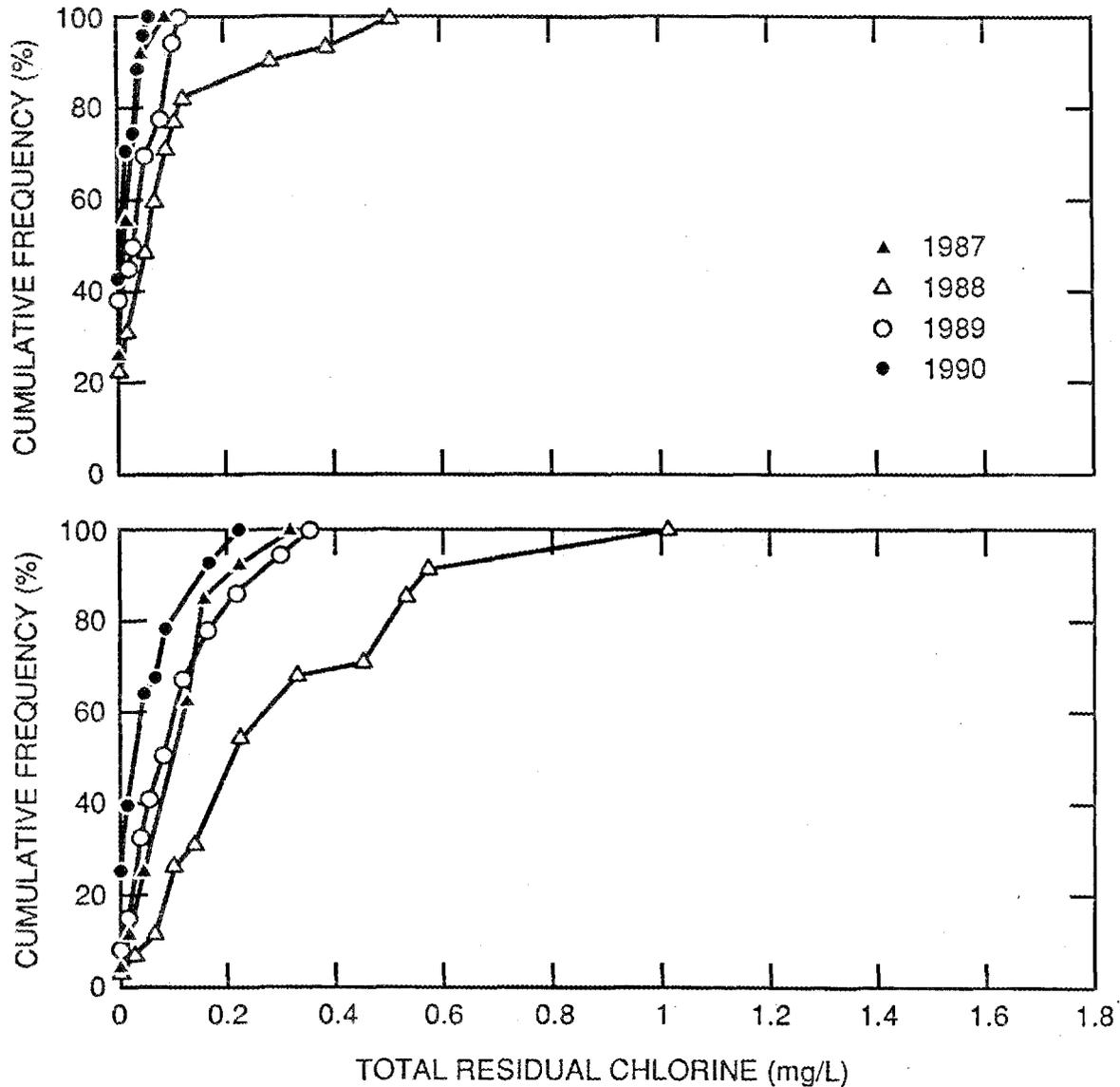


Fig. 3.3. Cumulative frequency (percentage) of total residual chlorine at Mitchell Branch kilometer (MIK) 0.12 (top) and MIK 0.45 (bottom), 1987-90.

During 1987, 1989, and 1990, TRC at MIK 0.45 was >0.20 mg/L 10 to 20% of the time; during 1988, TRC was ≥ 0.20 mg/L about 60% of the time. TRC concentrations at MIK 0.54 and MIK 0.71 were more variable between years. In 1987, TRC at MIK 0.54 was ≥ 0.20 mg/L 10% of the time; during 1988 and 1990, TRC was ≥ 0.20 mg/L about 30% of the

time; and in 1990, TRC at MIK 0.54 was never >0.20 mg/L. In 1987, 1988, 1989, and 1990, TRC at MIK 0.71 has been >0.20 mg/L about 50, 70, 40, and 20% of the time respectively.

Results of chemical analyses obtained concurrently with some of the toxicity tests by the Oak Ridge K-25 Site Process Support Department are summarized in

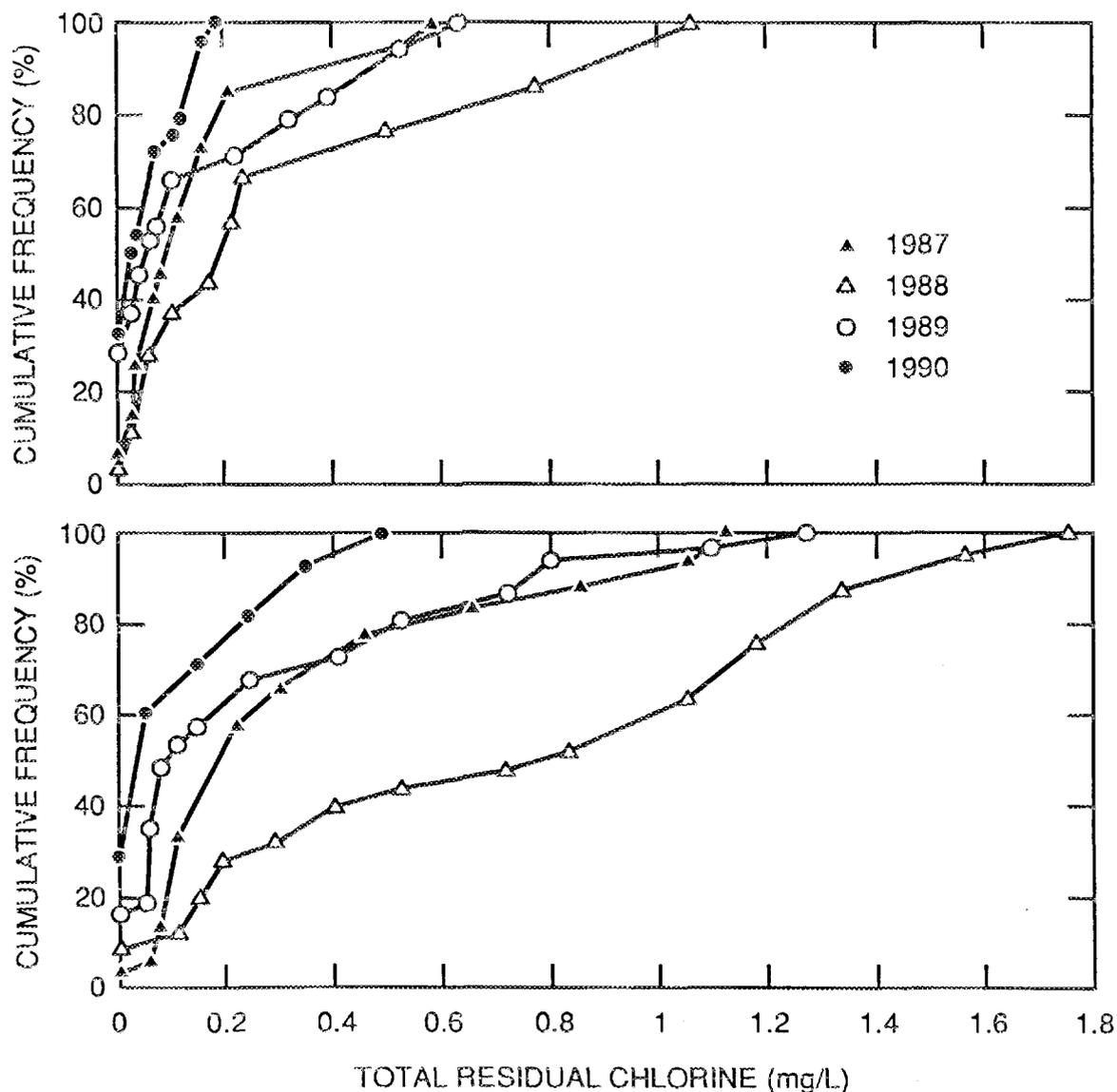


Fig. 3.4. Cumulative frequency (percentage) of total residual chlorine at Mitchell Branch kilometer (MIK) 0.54 (top) and MIK 0.71 (bottom), 1987-90.

Appendix B, Tables B.1 through B.9. Only those constituents that had concentrations consistently above detection levels or that were of toxicological concern were included; complete data sets are available elsewhere (McGaha 1989a, 1989b, 1989c, 1989d, 1989e; Shoemaker et al. 1990). The concentrations of none of the measured constituents at any of the sites were high

enough to correlate with the toxicity observed. However, the concentration of nearly every constituent increased with distance downstream. Chemical data collected before and after the K-1407-B pond closed (October 1988) showed the influence that the pond had on water chemistry in Mitchell Branch. During the September 1988 test, before the pond

closed, concentrations of chloride, dissolved solids, and sulfates were 2.5 to 4.2 times higher at MIK 0.54 (located below the K-1407-B outfall) than at MIK 0.71 [located above the K-1407-B outfall (Table B.2)]. In the November 1988 test, after the pond had closed, there were only minor differences (<1.1 times higher) between these two sites in the same parameters (Table B.3). Of the eight test dates shown, Al, dissolved solids, chloride, fluoride, Mn, Si, Na, sulfate,

and suspended solids were generally highest at all sites during the November 1988 test.

3.2.3.2 Fathead minnow tests

Mean survival and growth of fathead minnows for each site in each year are plotted in Fig. 3.5. Minnow survival in water from the reference site (MIK 1.43) was low (50 to 75%) during each year

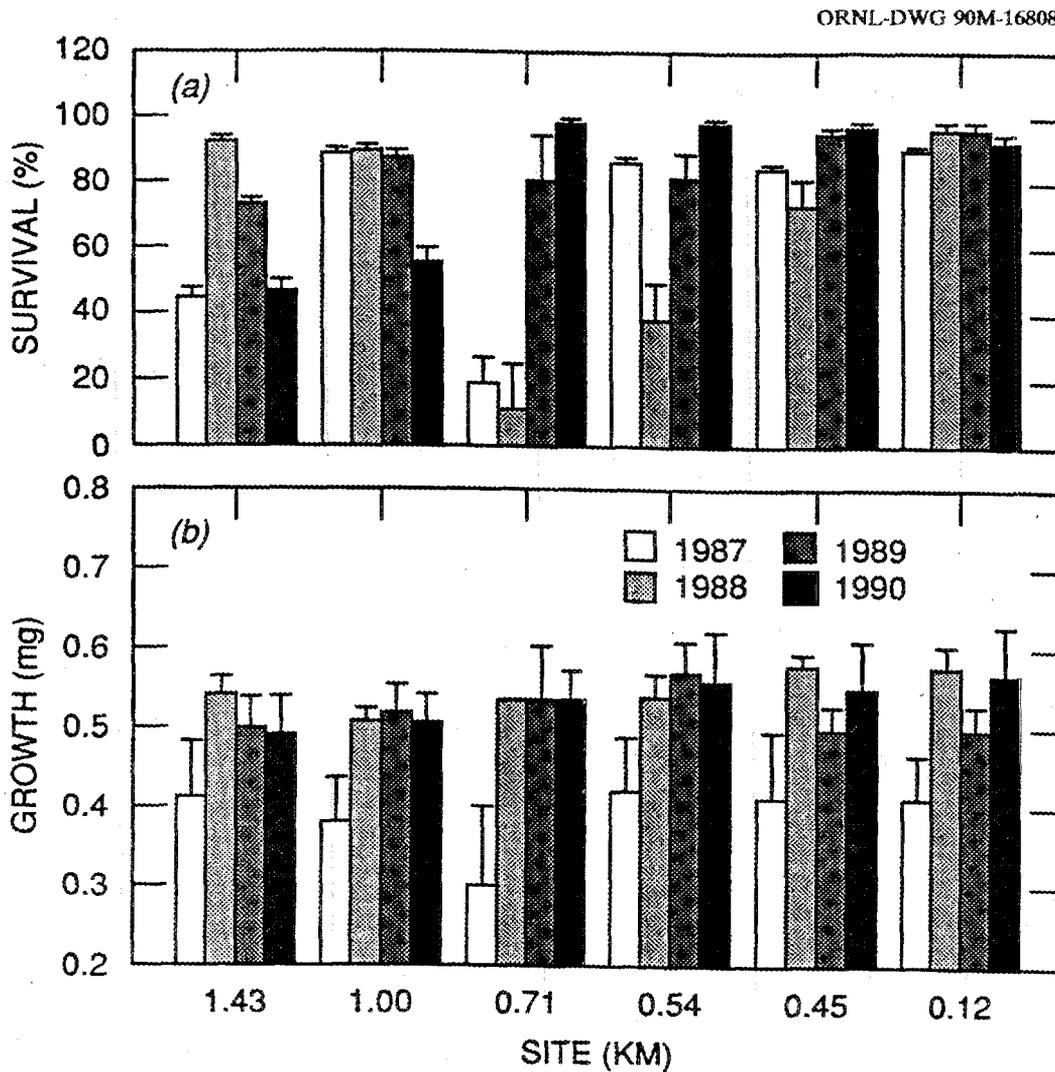


Fig. 3.5. Mean percentage survival (a) and growth of fathead minnows (b) for each site in Mitchell Branch, 1987-90. Vertical bars represent ± 1 SE.

except for 1988. At MIK 1.0, mean survival was about 90% in each year except for 1990, when mean survival was 55%. Survival at MIK 0.71 was much lower than at all other sites in 1987 and all but MIK 0.54 in 1988. Mean survival of the minnows at MIK 0.54 dropped substantially from 1987 to 1988 but rebounded in 1989 and 1990, whereas survival at MIKs 0.45 and 0.12 remained relatively high in all 4 years. In 1989 and 1990, mean survival

at MIKs 0.71, 0.54, 0.45, and 0.12 were similar. Also, survival at these four sites during the same 2-year period was greater than that at MIKs 1.43 and 1.0. Growth of the fish in water from all sites was significantly lower in 1987 compared with growth in 1988, 1989, and 1990. Within each year, however, there were no significant differences in growth among sites.

The frequency distributions of survival at each site for each year (Figs. 3.6

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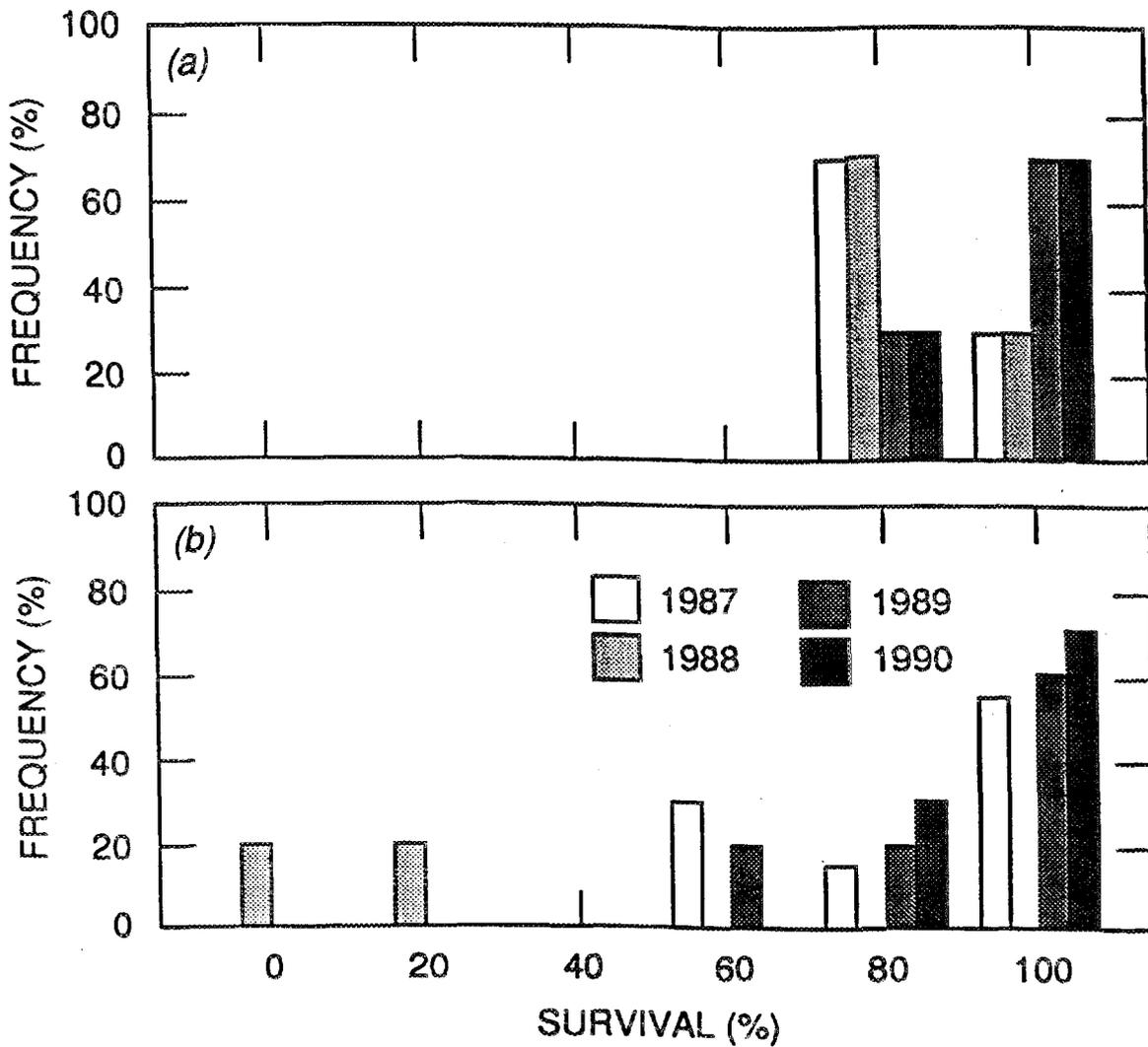


Fig. 3.6. Frequency distribution of fathead minnow survival at Mitchell Branch kilometer (MIK) 0.12 (a) and MIK 0.45 (b), 1987-90.

through 3.8) show that acute toxicity (survival 60%) was detectable at MIKs 0.45, 0.54, and 0.71 but declined after 1988. Mean survival of minnows in tests of water from MIK 0.12 was never <80%. At MIK 0.45, acute toxicity was demonstrated in 40% of the tests in 1988; in 1989 and 1990, no acute toxicity was found. At MIK 0.54, acute toxicity was found in 60% of the tests in 1988 and 17% in 1989; in 1987 and 1990, no

acute toxicity was found. Water from MIK 0.71 was typically very toxic to fat-head minnows. In 1987, 1988, and 1989, acute toxicity was found in 70, 80, and 30% of the tests, respectively, but in 1990, water from MIK 0.71 was not acutely toxic to fish in any test. Low survival was occasionally seen at MIK 1.0 and was frequently observed at MIK 1.43. However, this was probably not caused by a toxicant per se.

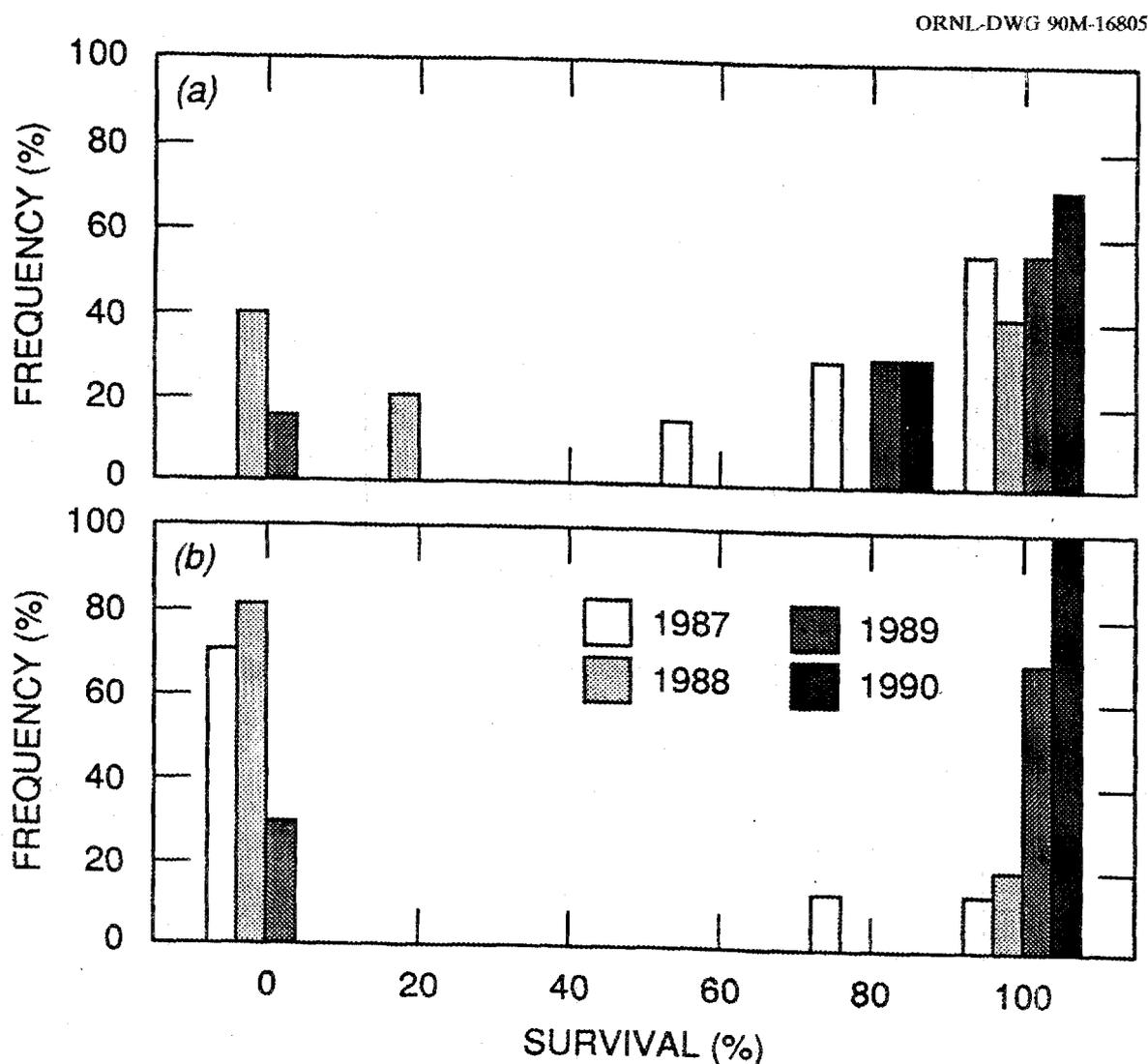


Fig. 3.7. Frequency distribution of fathead survival at Mitchell Branch kilometer (MIK) 0.54 (a) and MIK 0.71 (b), 1987-90.

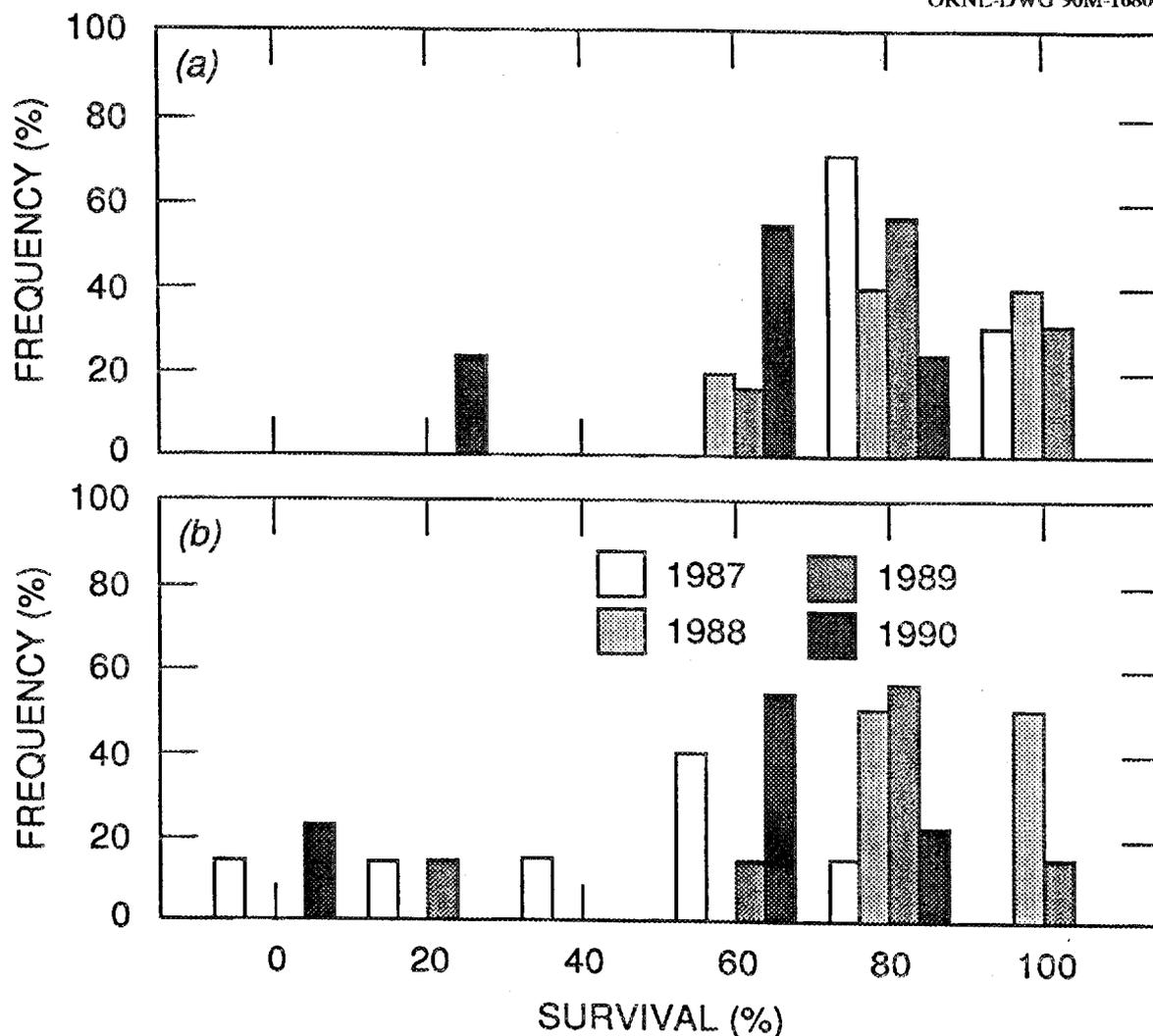


Fig. 3.8. Frequency distribution of fathead survival at Mitchell Branch kilometer (MIK) 1.0 (a) and MIK 1.43 (b), 1987-90.

To evaluate the presence of chronic toxicity (significant reduction in growth compared with the control) at each site, separate ANOVAs were conducted for each test. Only sites where mean survival was $>60\%$ were included. Results of this analysis (Table 3.11) showed that chronic toxicity was evident only at MIK 0.54 but at a very low frequency. In contrast, water from MIKs 0.71, 0.54, 0.45, and 0.12 was more often acutely toxic (survival $<60\%$) or not toxic at all.

3.2.3.3 *Ceriodaphnia* tests

Mean survival and reproduction of *Ceriodaphnia* for each site in each year are plotted in Fig. 3.9. Differences in survival among years and sites could not be effectively determined by means of an ANOVA because most values had a bimodal distribution, indicating an all-or-none type of response. However, survival was reduced every year at MIKs 0.71, 0.54, and 0.45 compared with 1.43. In general, mean

Table 3.11. Percentage of fathead minnow tests at each site where no toxicity, acute toxicity, or chronic toxicity was observed

Site	No toxicity ^a (%)	Acute toxicity ^b (%)	Chronic toxicity ^c (%)
MIK 0.71	50	50	0
MIK 0.54	68	27	5
MIK 0.45	82	8	0
MIK 0.12	100	0	0

^aSurvival was $\geq 60\%$ and no reduction in growth.

^bSurvival was $< 60\%$.

^cSurvival was $\geq 60\%$ and growth was significantly reduced compared with the control.

Note: MIK = Mitchell Branch kilometer.

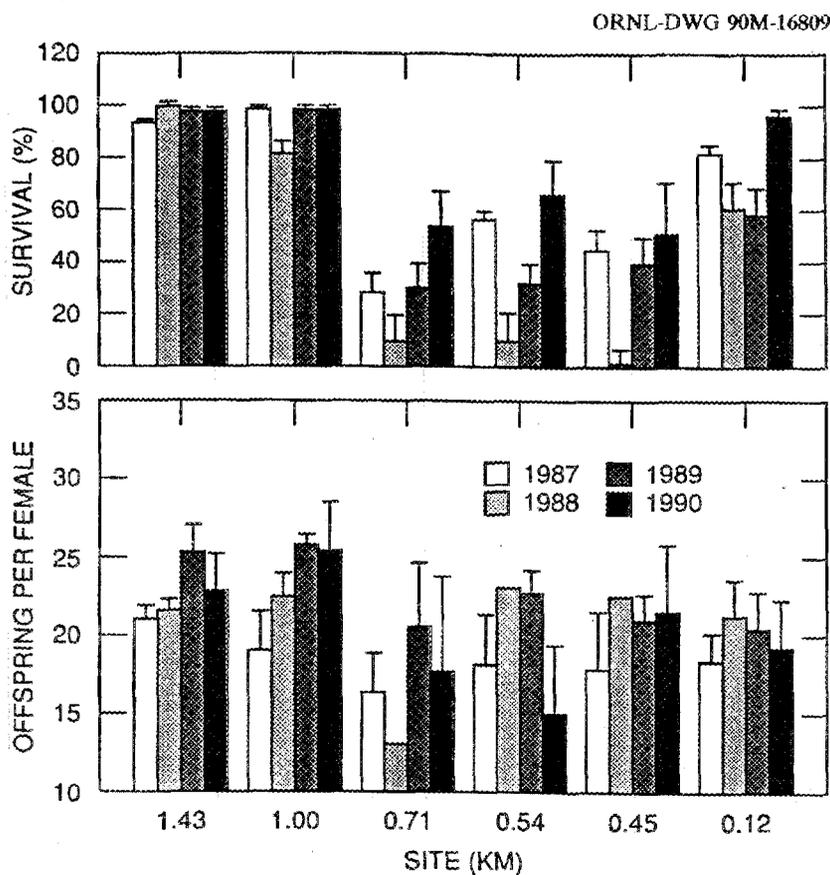


Fig. 3-9. Mean survival and fecundity of *Ceriodaphnia* for each site in Mitchell Branch, 1987-90. Vertical bars represent ± 1 SE.

survival increased with distance downstream from MIK 0.71, reaching a peak at MIK 0.12. In 1988 and 1989, mean fecundity at MIKs 0.71, 0.54, 0.45, and 0.12 was also reduced compared with that at MIK 1.43. In 1990, fecundity at MIKs 0.71 and 0.54 were reduced compared with that at MIK 1.43.

Frequency distributions of survival of *Ceriodaphnia* at each site for each year show that acute toxicity was present at MIKs 0.71, 0.54, 0.45, and 0.12 (Figs. 3.10 through 3.12). As for fat-head minnows, the incidence of acute toxicity (survival <60%) declined after 1988. Water from MIK 0.12 was acutely

toxic to *Ceriodaphnia* during 1988 and 1989 but not in 1990. At MIKs 0.45 and 0.54, acute toxicity was found in 22 to 80% of the tests conducted each year. At MIK 0.71, the incidence of acute toxicity was higher, occurring in 40 to 80% of the tests each year. Except for one test in 1988, survival was always been $\geq 80\%$ at MIKs 1.0 and 1.43.

To evaluate the presence of chronic toxicity (i.e., a significant reduction in reproduction compared with that at MIK 1.43) at each site, separate ANOVAs were conducted for each test. Only those sites where mean survival was >60%

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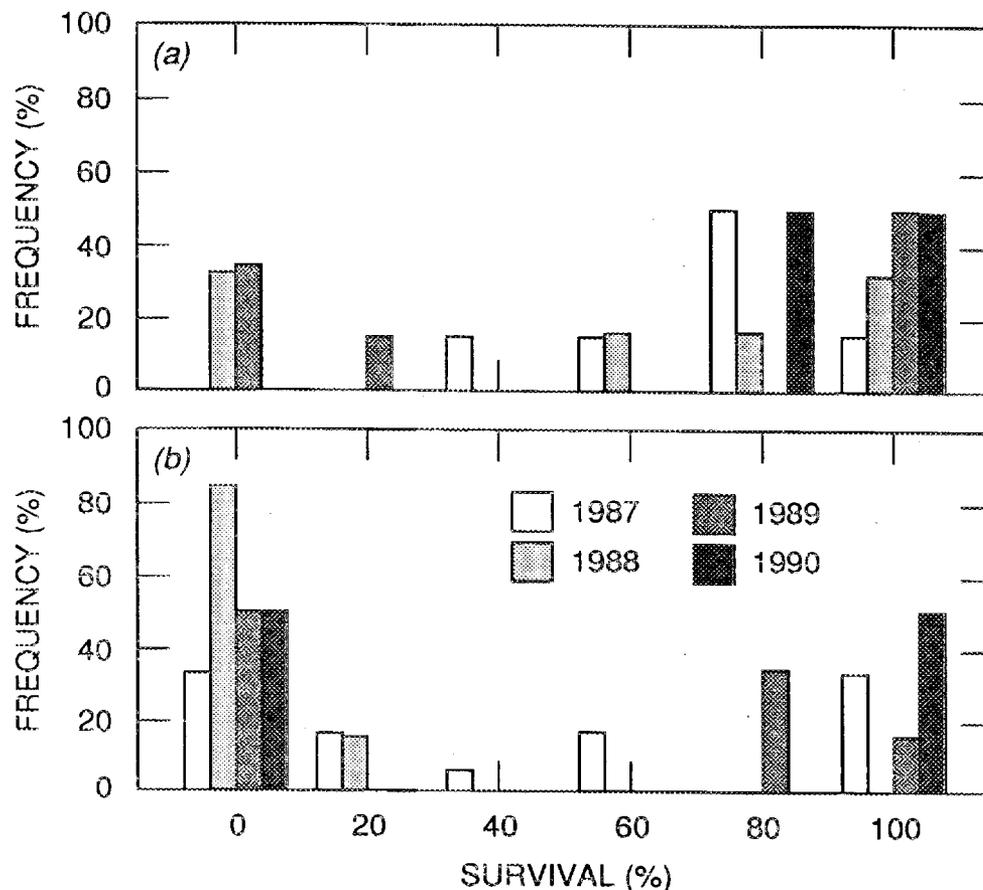


Fig. 3.10. Frequency distribution of *Ceriodaphnia* survival at Mitchell Branch kilometer (MIK) 0.12 (a) and MIK 0.45 (b), 1987-90.

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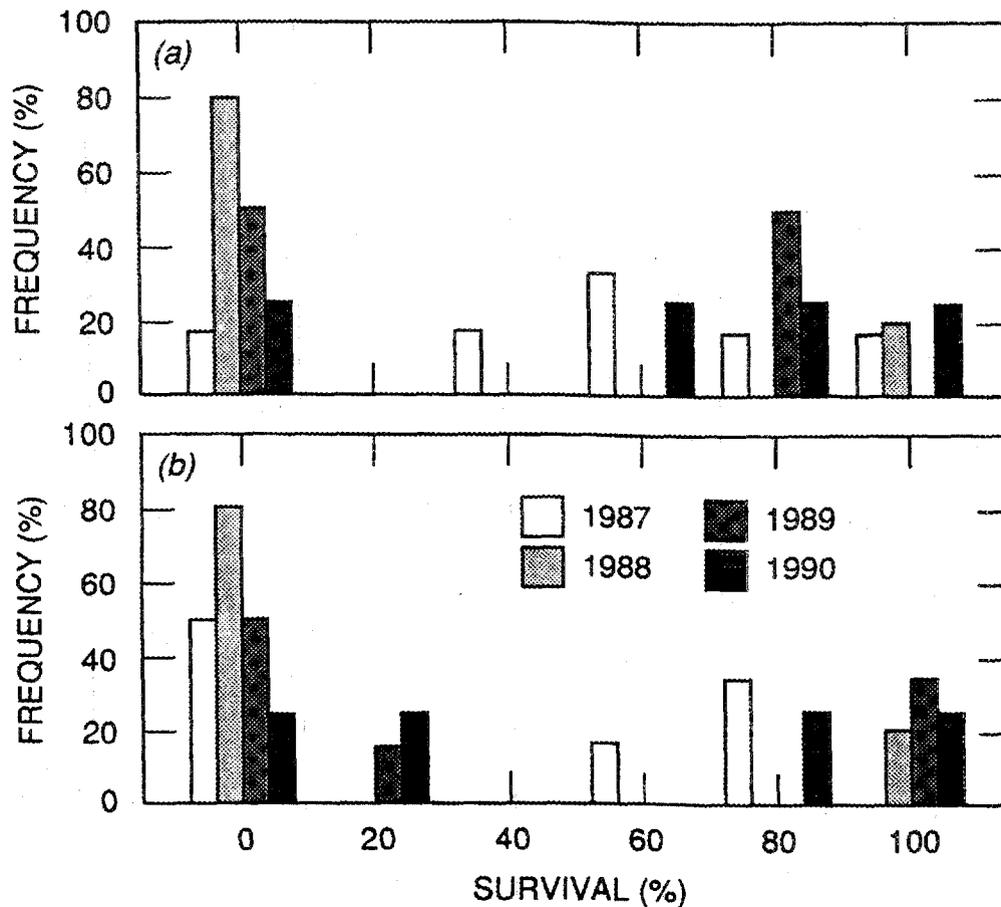


Fig. 3.11. Frequency distribution of *Ceriodaphnia* survival at Mitchell Branch kilometer (MIK) 0.54 (a) and MIK 0.71 (b), 1987-90.

were included. Results of this analysis (Table 3.12) show that chronic toxicity was evident at MIKs 0.71, 0.54, and 0.12. For example, in 23% of the tests conducted with water from MIK 0.12, *Ceriodaphnia* survival was $\geq 60\%$ but fecundity was significantly reduced compared with that at MIK 1.43.

3.2.4 Discussion

A noticeable improvement in the water quality of Mitchell Branch has occurred since the end of 1988. Conductivity and hardness of the water at and

below MIK 0.71 remained elevated relative to the two sites furthest upstream, indicating impact from the K-1407-E/F ponds. However, the magnitude of increases in these parameters with distance downstream has declined since 1988. The toxicity tests also documented that water quality improved during 1989 and 1990. After 1988, fathead minnow survival was no longer adversely affected by water from the midreach section of Mitchell Branch. Survival of *Ceriodaphnia* continued to be low in this section of the stream in 1989 and 1990 because of high concentrations of TRC, but the frequency of acute toxicity was not as high as in 1987 and 1988. This

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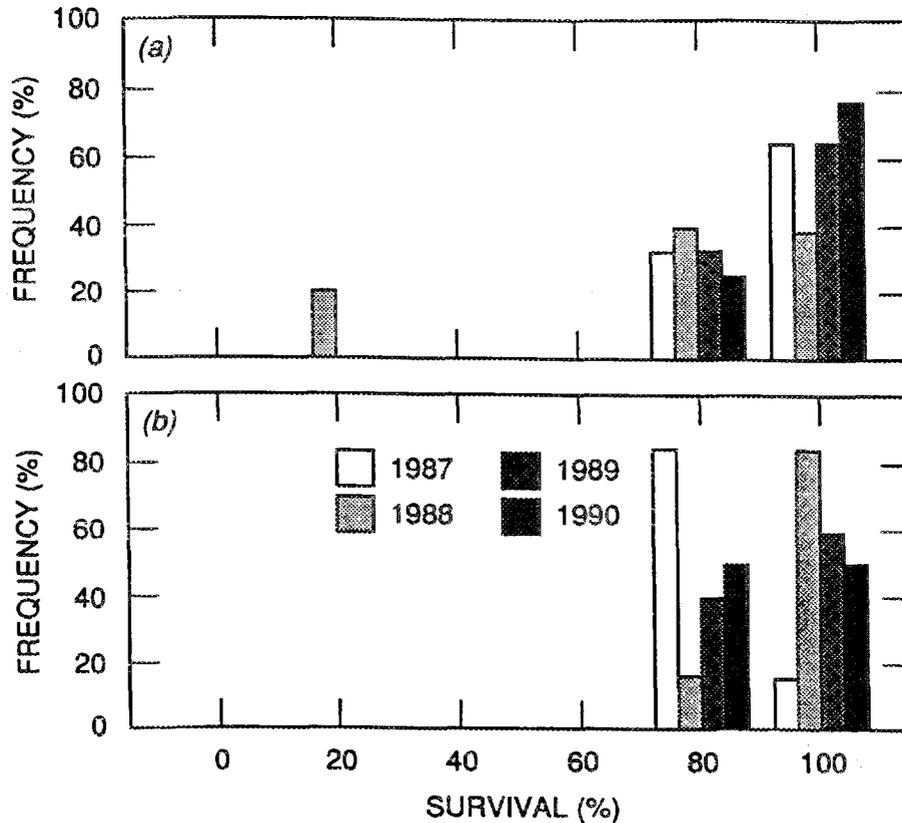


Fig. 3.12. Frequency distribution of *Ceriodaphnia* survival at Mitchell Branch kilometer (MIK) 1.0 (a) and MIK 1.43 (b), 1987-90.

Table 3.12. Percentage of *Ceriodaphnia* tests at each site where no toxicity, acute toxicity, or chronic toxicity was observed

Site	No toxicity ^a (%)	Acute toxicity ^b (%)	Chronic toxicity ^c (%)
MIK 0.71	23	64	13
MIK 0.54	36	55	9
MIK 0.45	41	59	0
MIK 0.12	59	18	23

^a*Ceriodaphnia* survival was $\geq 60\%$ and no reduction in fecundity.

^bSurvival was $< 60\%$.

^cSurvival was $\geq 60\%$ and reproduction was significantly reduced compared with MIK 1.43.

Note: MIK = Mitchell Branch kilometer.

was probably because of fewer excursions in TRC concentrations above 0.20 mg/L (Figs. 3.3 and 3.4). It was difficult to determine if the improvements in water quality were the result of the closure of K-1407-B pond (October 1988) or an increase in streamflow (thus, greater dilution) during 1989 and 1990. (See Sect. 2.1.1 for flow data.) Changes in water quality caused by the closure of K-1407-B pond were evident in terms of lower concentrations of chloride, sulfate, and dissolved solids at MIK 0.54 (Tables B.2 and B.3). Fewer excursions of high TRC at all sites were probably the result of higher streamflows during 1989 and 1990 because no known process changes occurred during this period. Thus, the closure of the K-1407-B pond in 1988 and the end of the drought in late 1988 both contributed to the improved water quality of Mitchell Branch. There continues to be evidence of chronic toxicity in Mitchell Branch: *Ceriodaphnia* fecundity was reduced (compared with that at MIK 1.43) in some tests at MIKs 0.71, 0.54, and 0.12 when survival was $\geq 60\%$. This toxicity may be the result of low levels of TRC from the storm drains, constituents released from the K-1407-E/F ponds, and/or area-source contamination. However, measurements of the major metals, anions, and cations in the stream water have not yet revealed a probable toxicant.

The cumulative frequency distribution of TRC in Mitchell Branch and the occurrence of acute toxicity (0% survival) at each site provide a means to estimate the concentration of TRC that can cause acute toxicity in Mitchell Branch. The frequency of acute toxicity at a particular site should coincide with the concentration of TRC that is acutely toxic. For example, survival of *Ceriodaphnia* was 0% in 48% of the toxicity tests at MIK 0.71 (Fig. 3.11). By using the cumulative frequency of TRC at MIK 0.71 (Fig. 3.2), the TRC

concentration was calculated to be greater than 0.18 mg/L about 48% of the time. Thus, when the TRC at MIK 0.71 is ≥ 0.18 mg/L, the water should be acutely toxic to *Ceriodaphnia*. If the same rationale is used for each site, the concentration of TRC predicted to be acutely toxic to *Ceriodaphnia* ranged from about 0.08 to 0.15 mg/L at MIKs 0.12, 0.45, 0.54, and 0.71. The concentration of TRC predicted to cause acute toxicity to fathead minnows ranged from about 0.2 to 0.5 mg/L at the same sites. It is not possible to predict the chronic toxicity of TRC because of the static-renewal nature of the toxicity test.

Mean survival of fathead minnow larvae in water from the uncontaminated reference site (MIK 1.43) continued to be low. As stated in the First Annual BMAP report (Smith et al. 1988), this is probably caused by a pathogen in the water.

3.3 FUTURE STUDIES

The results of effluent and ambient toxicity tests show the need for continued monitoring of effluents and Mitchell Branch. Monitoring of the storm drains should be continued because they are a major source of TRC to the stream and they may contain other toxicants as well. Discharge from the K-1407-E/F ponds is expected to be discontinued in 1993. To continue to document improvements in water quality as the ponds are closed and as reductions are made in the release of chlorine, monitoring will continue on a bimonthly basis. Because fathead minnows are typically insensitive to effluent from the K-1407-E/F ponds and the K-1407-J basin and to Mitchell Branch water, depending on the approval of the Oak Ridge K-25 Site and the Tennessee Department of Environment and Conservation, these effluents and ambient sites will be evaluated with *Ceriodaphnia* only.

4. BIOACCUMULATION STUDIES

M. J. Peterson and G. R. Southworth

The primary objectives of contaminant monitoring in Mitchell Branch biota are to (1) identify substances that accumulate to levels exceeding those observed in biota from nearby uncontaminated reference streams and (2) evaluate the extent and significance of contamination by those substances in Mitchell Branch and downstream aquatic systems. Secondary objectives are to assist in locating sources of contaminants that accumulate to unacceptable levels and to evaluate the relative importance of current vs past discharges in determining contaminant levels in biota.

4.1 INTRODUCTION

Results presented in the first report for Mitchell Branch (Smith et al. 1993) demonstrated that Hg and PCBs accumulate to levels significantly above background in the biota. Whether or not Mitchell Branch was a significant source of the elevated mercury levels in its fish or in fish from lower Poplar Creek was unclear. PCB monitoring data from clams were more conclusive and clearly documented that Mitchell Branch is a source of PCBs. The data also suggested that the stream may be a significant source of PCB contamination to biota in lower Poplar Creek.

Monitoring of metals and organics in Mitchell Branch biota continued on about a yearly basis between May 1987 and March 1990. To better assess possible contributions of mercury and PCBs from Mitchell Branch to downstream waters, sunfish sampling was expanded to include a number of sites in Poplar Creek and the

Clinch River in 1987 and 1988 as part of a coordinated effort between ORNL, the Oak Ridge Y-12 Plant, and the Oak Ridge K-25 Site biomonitoring programs. Caged clams (*Corbicula fluminea*) were used in Poplar Creek in 1988 to gain additional information on the importance of Mitchell Branch as a source of PCBs to Poplar Creek. In coordination with the ORNL BMAP, channel catfish (*Ictalurus punctatus*) were collected from Poplar Creek and the Clinch River and analyzed for PCBs in 1988 and 1989. Channel catfish probably accumulate the highest concentrations of PCBs attainable in a sport species in Poplar Creek that could be influenced by PCBs from Mitchell Branch. In 1988, gizzard shad (*Dorosoma cepedianum*) were also collected and analyzed for PCBs from Poplar Creek in an effort to evaluate the role of forage fish in PCB dispersal.

4.2 METHODS

Fish were collected by electrofishing from the lower reaches of Mitchell Branch (MIK 0.2) beginning in May 1987 and continuing approximately yearly through 1990 (Fig. 2.1). Approximately 12 red-breast sunfish (*Lepomis auritus*) were collected from Mitchell Branch in each sampling period. Attempts were made to obtain eight fish samples for mercury analysis, eight for PCB analysis, and four for metals other than mercury. Early attempts to collect fish from Mitchell Branch, however, revealed that the stream did not support an adequate population of adult fish to meet the full sampling

requirements of the contaminant monitoring program. Thus, because of the small size of the few redbreast sunfish present in the stream in 1987 and 1988, only analyses of mercury and other metals were conducted in those years. In 1989 and 1990, however, enough adult redbreast sunfish were collected to include an analysis for PCBs. Analysis of metals (other than mercury) was not performed on Mitchell Branch fish in 1989.

Bluegill (*Lepomis macrochirus*) were routinely collected for mercury and PCB analysis from various sites on Poplar Creek and the Clinch River from 1987 to 1990 (Figs. 2.1 and 2.2). Fish from Poplar Creek and the Clinch River were sampled to evaluate the relative importance of mercury and PCB inputs from multiple sources (Bear Creek, East Fork Poplar Creek, Mitchell Branch, and White Oak Creek Watershed) that enter this system. Eight bluegill sunfish were collected by electrofishing from each of four sites on Poplar Creek (PCKs 10.4, 8.2, 6.9, and 2.1, Figs. 2.1 and 2.2) and the Clinch River (CRK 15.0) in November/December of 1987 and 1988. When possible, collections were restricted to individuals of a size likely to be taken by sport fishermen (>50 g). Sites on Poplar Creek immediately above (PCK 8.2) and below (PCK 6.9) Mitchell Branch were resampled for bluegill in March 1990. Reference fish for the Poplar Creek and Mitchell Branch collections were obtained from Hinds Creek (1985-89) and Melton Hill Reservoir (MHR; 1987-88) in Anderson County, Tennessee (Fig. 2.1).

Upon collection, sunfish from all sites were tagged with a unique four-digit tag wired to the lower jaw and placed on ice in a labelled ice chest; the chest was locked for transport to the laboratory. Upon return to the laboratory, each fish was weighed and measured (total length), and scale samples were taken for age determination. The fish were then fileted and

skinned, and a 1- to 2-g portion of the anterior dorsal axial muscle filet was excised for mercury determination, if a 5-g portion of tissue was also available, it was used for analysis of PCBs and/or other metals. Samples were wrapped in heavy-duty aluminum foil, labelled, and stored at -20°C in a locked freezer in Building 1505 at ORNL until delivered to the Analytical Chemistry Division (ACD) laboratory at ORNL.

Channel catfish (*Ictalurus punctatus*) were collected for PCB analysis from PCK 6.9 and CRK 15.0 by means of trotlines in the summers of 1988 and 1989. Upon collection, the channel catfish were handled and processed as sunfish were, except that: (1) the dorsal spine was taken for age determination and (2) fish filets were frozen and ground three times in a hand-powered meat grinder before a 10- to 20-g sample was removed for analysis.

In the summer of 1988, gizzard shad (*Dorosoma cepedianum*) were collected for PCB analysis by electrofishing at the same sites as catfish. Three composite samples of five shad each (22-29 cm total length) were obtained at each site. Each shad composite was wrapped in aluminum foil and placed on ice in the field. Upon return to the laboratory, the digestive tract was removed from each fish and the composite frozen. Later, each frozen composite was ground three times in the meat grinder before a 10- to 20-g sample was removed for analysis.

Caged clams (*Corbicula fluminea*) were placed in Mitchell Branch (MIK 0.2) to monitor for organic contaminants on an approximately annual basis from 1987 to 1990. In 1988, clams were placed at PCKs 8.2 and 6.9 (upstream and downstream of Mitchell Branch, respectively) to monitor for PCBs in Poplar Creek. Clams were obtained in March 1987 from Beaver Creek near Karns in Knox County, Tennessee; in May 1988 from Bull Run Creek in Union County, Tennessee; and

in July 1989 from Paint Rock Creek in Meigs County, Tennessee. After the clams were held for 24 h in the laboratory in dechlorinated process water, they were put in polypropylene cages that were held securely at each site. One set of clams from the reference stream was frozen immediately for analysis as a control. Each cage placed in a stream held ~50 clams; each clam contained 0.5 to 2 g (wet weight) of soft tissue. The cages remained at the sites for 4 weeks, after which the clams were removed, placed on ice in a locked cooler, transported to the laboratory, and deposited in a locked freezer in Building 1505 at ORNL. After the clams were frozen, the shells were removed and the frozen soft tissue was placed in a 20-ml glass vial. Composite samples weighing ~5 and 10 g each were obtained for PCB analysis and gas chromatographic/mass spectrometric analysis respectively. Samples were refrozen prior to delivery to the ORNL ACD laboratory for chemical analysis.

Mercury determinations were carried out by the ACD recording to 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. Samples collected in 1987 and 1988 were analyzed for metals by graphite furnace atomic adsorption spectrophotometry following digestion with concentrated nitric acid (EPA 1980). Samples obtained in 1990 were analyzed for metals by inductively coupled argon plasma emission spectroscopy or by inductively coupled plasma mass spectrometry following digestion with concentrated nitric acid (EPA 1980). PCBs were determined by packed-column gas chromatography following methylene chloride extraction and adsorption column cleanup (EPA 1980). Organic priority pollutants were

analyzed by procedure PPB 12/83 (EPA 1983), in which the homogenized sample is extracted in methylene chloride, cleaned up by using column chromatography, and analyzed by capillary column gas chromatography with mass spectrometric or electron capture detectors.

Statistical evaluations of the data were made by using SAS procedures and software (SAS 1985a, 1985b) for the ANOVA, Duncan's multiple range test, t-test, linear regression analysis, and the calculation of means, standard deviations, standard errors, and coefficients of variation. Quality assurance was maintained by using a combination of (1) blind duplicate analyses; (2) split-sample analyses between the EPA Environmental Services Laboratory, Athens, Georgia, and the ACD Laboratory at ORNL; and (3) the analyses of biological reference standards and uncontaminated fish. Recoveries of PCBs were verified by analysis of uncontaminated fish or clam samples spiked with known amounts of PCBs. Details and results of QA analyses are summarized in Appendix C and results of tissue analyses are tabulated in Appendix D.

4.3 RESULTS AND DISCUSSION

4.3.1 Mercury in Mitchell Branch Fish

Mercury was clearly elevated in fish from Mitchell Branch relative to fish from the reference stream (Table 4.1). However, the difference was not large, and levels were well below the Food and Drug Administration (FDA) action level of 1.0 $\mu\text{g/g}$ (FDA 1984a). No fish from Mitchell Branch exceeded the more conservative Preliminary Guidance Value (PGV) for Hg of 0.42 $\mu\text{g/g}$ (Travis et al. 1986).

The mean mercury levels in Mitchell Branch redbreast varied among years, with the highest mean Hg concentration in fish

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Table 4.1. Mercury concentrations (in micrograms per gram, wet weight) in bluegill (*Lepomis macrochirus*) and redbreast sunfish (*L. auritus*) collected at sites in the Poplar Creek drainage area in the vicinity of the Oak Ridge K-25 Site

Site ^a	Species	Date	Hg mean \pm 1 SE	Range	PGV exceedences ^b
EFK 2.1	Bluegill	December 1987	0.43 \pm 0.06	0.05-0.62	6/8
		November 1988	0.37 \pm 0.05	0.07-0.52	3/8
PCK 10.4	Bluegill	November 1987	0.10 \pm 0.03	0.05-0.16	0/8
		November 1988	0.08 \pm 0.03	0.02-0.24	0/8
PCK 8.2	Bluegill	November 1987	0.43 \pm 0.06	0.21-0.69	4/8
		November 1988	0.41 \pm 0.05	0.23-0.59	3/8
		March 1990	0.36 \pm 0.05	0.14-0.65	3/8
MIK 0.2	Redbreast	May 1987	0.17 \pm 0.01	0.15-0.18	0/7
		March 1988	0.12 \pm 0.01	0.09-0.14	0/5
		March 1989	0.19 \pm 0.01	0.12-0.22	0/8
		January 1990	0.27 \pm 0.03	0.11-0.39	0/8
PCK 6.9	Bluegill	June 1987	0.42 \pm 0.05	0.24-0.65	3/8
		November 1987	0.43 \pm 0.05	0.26-0.66	5/8
		November 1988	0.29 \pm 0.05	0.09-0.54	2/8
		March 1990	0.43 \pm 0.04	0.30-0.68	5/8
PCK 1.6	Bluegill	December 1987	0.16 \pm 0.04	0.06-0.35	0/8
		November 1988	0.17 \pm 0.02	0.06-0.24	0/8
CRK 15.0	Bluegill	November 1987	0.14 \pm 0.03	0.06-0.31	0/8
		November 1988	0.14 \pm 0.03	0.02-0.28	0/8
Reference sites					
MHR	Bluegill	1987-1989	0.04 \pm 0.003	0.02-0.10	0/30
HC	Redbreast	1985-1989	0.08 \pm 0.01	0.03-0.16	0/46

^aMHR = Melton Hill Reservoir; HC = Hinds Creek; EFK = East Fork Poplar Creek kilometer; PCK = Poplar Creek kilometer; MIK = Mitchell Branch kilometer; CRK = Clinch River kilometer.

^bPreliminary Guidance Value. Number of samples exceeding the PGV value for Hg of 0.42 $\mu\text{g/g}$ (C. C. Travis et al. 1986, *Preliminary reviews of TVA fish sampling and analysis report*, Report of Task Group Five to Oak Ridge Task Force, Oak Ridge National Laboratory, Oak Ridge, Tenn., Mimeo, Jan. 1986) divided by the total number of samples. No fish exceeded the FDA action limit of 1.0 $\mu\text{g/g}$ (U.S. Department of Agriculture Food and Drug Administration (FDA), 1984 *Action level for methylmercury in fish*, Fed. Reg. 49(224):45663.

occurring in the most recent (1990) sampling period. However, accumulation of Hg in fish tends to increase with body size. A significant positive relationship was observed between Hg concentration in fish tissue and fish weight (linear regression analysis, $p < 0.05$) in Mitchell Branch. Thus, the larger fish collected in Mitchell Branch in 1989 and 1990 (mean weight of 26 and 41 g, respectively) would be expected to contain higher concentrations of mercury than the extremely small fish collected in 1987 and 1988 (mean weight of 5 g in both years).

Results previously reported for Mitchell Branch showed that sunfish collected from this stream had accumulated elevated levels of Hg (Smith et al. 1993). However, it was not clear from the data whether the source of mercury was Mitchell Branch or whether these fish had accumulated it while in another stream (i.e., Poplar Creek or EFPC) before immigrating into Mitchell Branch. By comparing mercury concentrations of fish from Mitchell Branch with those of fish from Poplar Creek, it may be possible to determine the extent to which Mitchell Branch serves as a source of mercury in fish. Although these comparisons are for two different species of sunfish (redbreast sunfish and bluegill), past studies (Loar 1992b; Loar 1993a) have shown no significant differences in mercury concentrations between the two species where the concentrations in fish are $\sim 0.5 \mu\text{g/g}$ or less. The mean mercury concentration in Mitchell Branch fish ($0.20 \mu\text{g/g}$) was approximately half that in fish ($0.39 \mu\text{g/g}$) at sites on Poplar Creek nearest Mitchell Branch (PCKs 8.2 and 6.9) in all years. However, because there is a positive relationship between mercury concentrations in Mitchell Branch fish and fish weight, adult fish, equal in weight to Poplar Creek fish (approximate mean weight of 70 g), would be expected to

contain much higher levels of mercury, if they were present.

Linear regression of mercury vs weight for MIK 0.2 sunfish yielded the expression $\text{Hg} = 0.13 + 0.00285 (\text{wt})$, $R^2 = 0.69$. When this expression was used to estimate the concentration of mercury in a hypothetical 70-g fish from Mitchell Branch, a value of $0.33 \mu\text{g/g}$ was obtained. This value is very similar to the mean concentration in Poplar Creek fish of $0.39 \mu\text{g/g}$. These results suggest that fish collected from Mitchell Branch may have accumulated their mercury burden from Poplar Creek before immigrating into Mitchell Branch. Conversely, a single creek chub (*Semotilus atromaculatus*) collected from Mitchell Branch, which was unlikely to have come from Poplar Creek, was analyzed for mercury in 1987 (Smith et al. 1993) and found to contain a substantial mercury concentration ($0.29 \mu\text{g/g}$). Elevated mercury levels in Mitchell Branch sediments (Ashwood et al. 1986) and water (Smith et al. 1993) also provide strong evidence that Mitchell Branch is a source of mercury to its biota. Because of this conflicting evidence, the significance of Mitchell Branch as a source of the elevated mercury levels in its fish remains unresolved.

4.3.2 Mercury in Poplar Creek Fish

Mercury concentrations were measured in bluegill from several sites in Poplar Creek and the Clinch River to assess the relative importance of various streams (Bear Creek, EFPC, Mitchell Branch, and White Oak Creek) to the system. Results for the Poplar Creek area are shown in Table 4.1; results for the entire reservation are shown in Figs. 4.1 and 4.2.

The most upstream site on Poplar Creek (PCK 10.4) was upstream of any

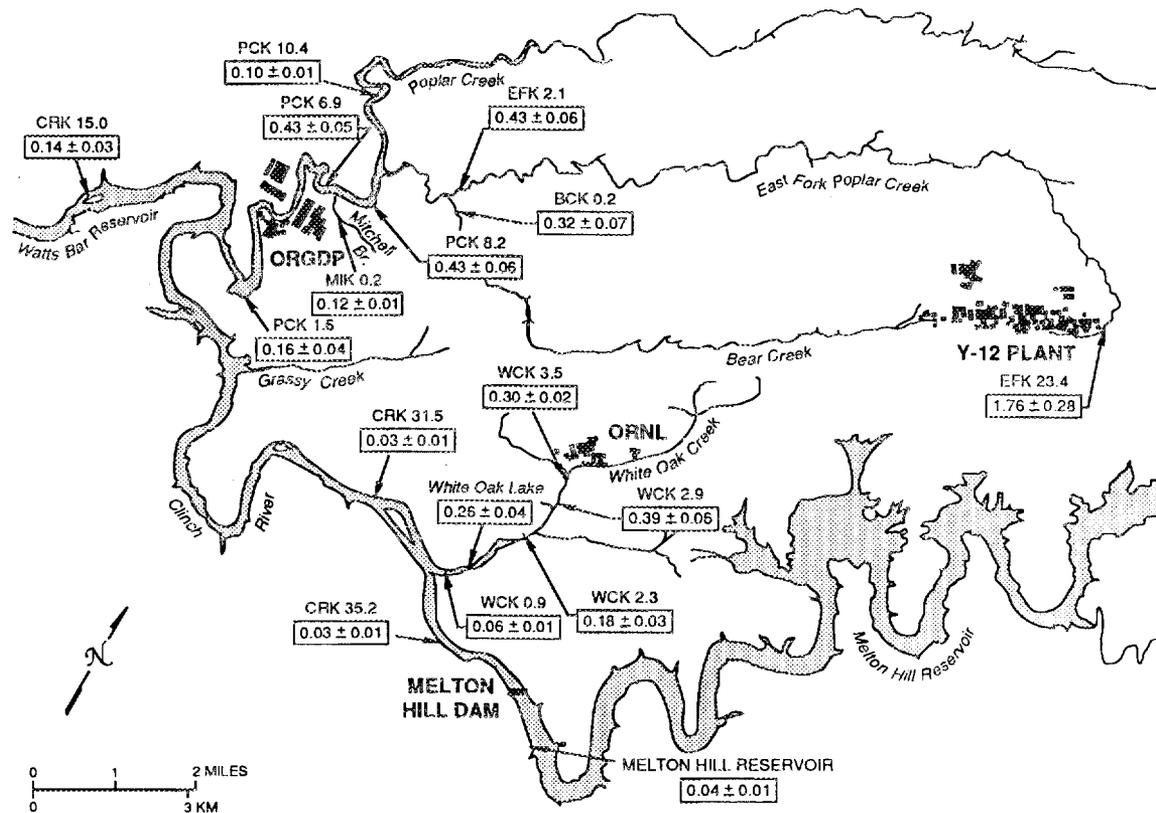


Fig. 4.1. Mean concentrations (± 1 SE) of mercury (in micrograms per gram, wet weight) in bluegill ($n = 8$) collected in fall/winter 1987 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River. Rockbass (*Ambloplites rupestris*) and redbreast sunfish (*Lepomis auritus*) were substituted for bluegill at Bear Creek kilometer 0.2 and Mitchell Branch kilometer 0.2 respectively.

discharges associated with DOE Oak Ridge operations (Table 4.1). Fish from PCK 10.4 contained mercury concentrations typical of fish from reference streams (Table 4.1), indicating that neither upstream sources of mercury in Poplar Creek nor movement of fish between sites were probable explanations for the high mercury levels at PCKs 8.2 and 6.9, located just upstream (PCK 8.2) and downstream (PCK 6.9) of Mitchell Branch. Although no fish exceeded the FDA action level of $1 \mu\text{g/g}$, 25 of 56 fish collected from PCKs 8.2 and 6.9 exceeded the more conservative PGV of $0.42 \mu\text{g/g}$ (Travis et al. 1986). Mean mercury concentrations

in fish from these two sites changed little between 1987 and 1990; mean values were about $0.4 \mu\text{g/g}$ at both sites in most years. Furthermore, no significant difference (ANOVA) in mean mercury concentrations was evident between PCKs 8.2 and 6.9 (0.40 and $0.38 \mu\text{g/g}$, respectively), suggesting that Mitchell Branch was not a significant additional contributor of mercury to fish in Poplar Creek between 1987 and 1990.

The high mercury levels in fish from PCKs 8.2 and 6.9 are not readily explainable by a one-source-hypothesis, such as input from EFPC only. Levels of mercury in sunfish from EFPC decreased in propor-

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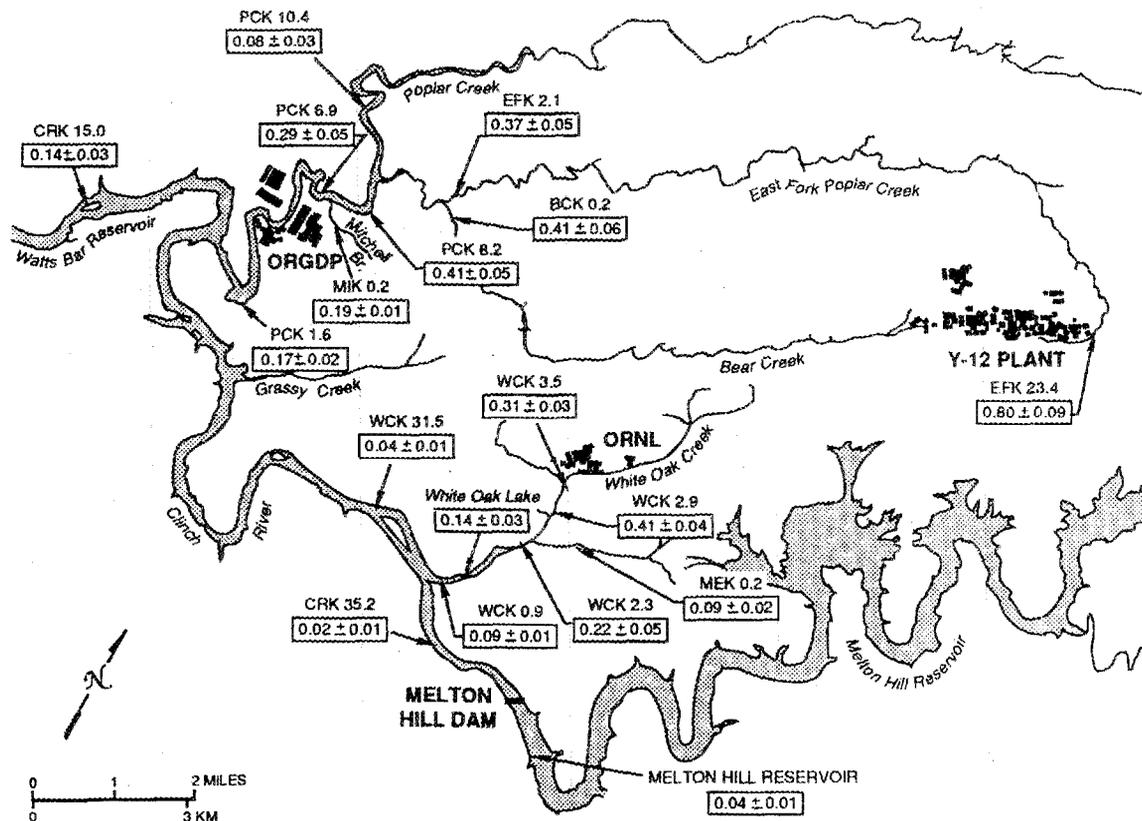


Fig. 4.2. Mean concentrations (± 1 SE) of mercury (in micrograms per gram, wet weight) in bluegill ($n = 8$) collected in fall/winter 1988 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River. Rockbass (*Ambloplites rupestris*) and redbreast sunfish (*Lepomis auritus*) were substituted for bluegill at Bear Creek kilometer 0.2 and Mitchell Branch kilometer 0.2 respectively.

tion to dilution of Oak Ridge Y-12 Plant discharges at sites along the length of EFPC (G. R. Southworth, ESD/ORNL, unpublished data), yet mercury levels in fish from PCKs 8.2 and 6.9 were similar to those found in lower EFPC, despite an almost fourfold dilution of EFPC upon entering Poplar Creek. One potential source of this mercury is Bear Creek. Mercury contamination is evident in fish from lower Bear Creek, but the extent of contamination in fish does not appear to be large enough to explain the high levels seen in Poplar Creek fish (Figs. 4.1 and 4.2). It is possible that either changes occur in the environmental chemistry of

mercury in lower Poplar Creek, enhancing the biological availability of water-borne or sediment-associated mercury, or that additional sources of mercury occur in the reach of Poplar Creek near the Oak Ridge K-25 Site.

Mercury contamination in fish at the most downstream site on Poplar Creek (PCK 1.6) was much lower than at PCKs 6.9 or 8.2. Similarly low concentrations were found in fish in the Clinch River (CRK 15.0) downstream of the mouth of Poplar Creek. Mercury concentrations in fish at CRK 15.0 were slightly elevated over concentrations in fish at sites on the Clinch River upstream of the

mouth of Poplar Creek; however, these levels were very similar to those found in fish from a reference stream (Hinds Creek). Thus, mercury contamination from Oak Ridge Operations appears to be detectable in Clinch River fish downstream of the mouth of Poplar Creek, but the degree of contamination appears to be minimal.

The degree of mercury contamination in bluegill from Poplar Creek sites near the K-25 Site was generally low compared with the degree of contamination in fish from the middle to upper reaches of EFPC in 1987 and 1988 (Loar et al. 1993b). Mean Hg concentrations at PCKs 8.2 and 6.9 ranged from approximately one-fourth to one-half the mercury concentrations in bluegill at EFK 23.4 (Figs. 4.1 and 4.2). In general, the mean Hg concentrations in fish at these two sites were comparable with those from lowermost EFPC (EFK 2.1) and Bear Creek (BCK 0.2) and the worst mercury-contaminated sites on White Oak Creek (WCKs 3.5 and 2.9). Mean mercury concentrations in fish from lowermost Poplar Creek (PCK 1.6) and the Clinch River (CRK 15.0) were comparable with those of fish from downstream sites on White Oak Creek. Fish from Mitchell Branch had Hg concentrations similar to those in fish from PCK 1.6, CRK 15.0, and lower White Oak Creek sites (WCKs 1.0 and 2.3; Loar 1989). However, the Mitchell Branch fish are substantially smaller than those from PCK 1.6, CRK 15.0, and WCK 2.9. As discussed, mean mercury concentrations in Mitchell Branch fish would probably be much higher if more adult fish were present.

4.3.3 Other Metals in Mitchell Branch Fish

Despite abnormally high levels of some metals in Mitchell Branch water (Smith et al. 1993) and sediment (Ashwood

et al. 1986), concentrations of metals (other than mercury) in Mitchell Branch fish in all three sampling years were similar to those found in the reference stream (Table 4.2). Although cadmium and copper concentrations in Mitchell Branch fish in 1987 were slightly higher than in Hinds Creek fish collected in that year (Smith et al. 1993), the difference was not statistically significant. The most recent sampling of Mitchell Branch (1990) revealed that only two metals other than Hg (Se and Zn) were found above the analytical detection limit. The levels of metals (except for mercury) in Mitchell Branch fish in all three sampling years were also quite similar to those observed in Hinds Creek bluegill in 1987 (Smith et al. 1993) and by the Tennessee Valley Authority (TVA) at one of their reference sites (Melton Hill Reservoir) (TVA 1985, 1986). Moreover, the levels of metals observed in fish in the National Contaminant Biomonitoring Program (geometric mean of 112 sites sampled for As, Cd, Cu, Pb, Hg, Se, and Zn; Lowe et al. 1985) were also generally similar to levels observed in Mitchell Branch fish, except for mercury.

A comparison of the concentrations of metals in Mitchell Branch fish with PGVs, derived to screen for levels of contamination that may potentially threaten human health (Travis et al. 1986; Hoffman et al. 1984), indicates that only As, Be, and Hg approach this threshold (Table 4.2). Beryllium and arsenic were also the only two metals in Mitchell Branch fish that exceeded fish tissue screening levels established for protection of human health by the EPA (EPA 1990). Neither arsenic nor beryllium was detected in Mitchell Branch fish; however, PGV and EPA screening levels are set at levels below current detection limits because of the carcinogenicity of these two metals. The PGV screening approach is very conservative and is designed to eliminate

Table 4.2. Metal concentrations (mean \pm 1 SE, in micrograms per gram wet weight) in redbreast sunfish (*Lepomis auritus*) collected from Mitchell Branch and Hinds Creek (reference stream)

Metal	Mitchell Branch			Hinds Creek ^b	PGV ^c
	1987 ^a	1988	1990		
Antimony	<0.2	<0.3	<0.54	<0.44	5.2
Arsenic	<0.10	<0.3	<0.05	<0.3	0.0007
Beryllium	<0.05	<0.3	<0.003	<0.3	0.004
Cadmium	0.05 \pm 0.01	<0.005	<0.22	<0.18	1.0
Chromium	<0.1	<0.5	<0.54	<0.5	1.8
Copper	0.68 \pm 0.14	0.09 \pm 0.03	<0.54	0.32 \pm 0.13	36
Lithium	<0.5	<0.3		<0.3	
Mercury	0.17 \pm 0.01 ^d	0.12 \pm 0.01 ^e	0.27 \pm 0.03	0.08 \pm 0.01 ^f	0.42
Nickel	<1.0	<1.0	<0.54	<1.0	5.2
Lead	0.08 \pm 0.05	<0.1	<0.54	<0.44	1.8
Selenium	0.50 \pm 0.10	0.40 \pm 0.04	0.61 \pm 0.03	0.52 \pm 0.08	12
Silver	<0.05	<0.05	<0.22	<0.18	0.29
Thallium	<0.2	<0.2	<0.02	<0.2	0.66
Zinc	5.5 \pm 0.3	1.8 \pm 0.5	4.7 \pm 0.3	5.0 \pm 0.5	180

^a*n* = 6 except for mercury.

^bHinds Creek reference fish for metals other than mercury were collected in December of 1988 and 1990 (*n* = 2).

^cPreliminary Guidance Values (C. C. Travis et al. 1986, *Preliminary review of TVA fish sampling and analysis report*, Report of Task Group Five to Oak Ridge Task Force, Oak Ridge National Laboratory, Oak Ridge, Tenn., Mimeo, Jan. 1986. F. O. Hoffman et al. 1984, *Preliminary screening of contaminants in sediments*, ORNL/TM-9730, Oak Ridge National Laboratory, Oak Ridge, Tenn.).

^d*n* = 7.

^e*n* = 5.

^fFish collected from 1985 to 1989 (*n* = 46).

Note: Mitchell Branch fish were collected in May 1987, March 1988, and January 1990 (*n* = 4 except where noted). When all values used to calculate the mean were below detection limits, the highest detection limit value was cited.

from concern any substances not exceeding the PGV (Hoffman et al. 1984).

4.3.4 PCBs in Mitchell Branch Fish

Results presented previously for Mitchell Branch (Smith et al. 1993) revealed that Asiatic clams held for 4-week exposures in cages accumulated substantial concentrations of PCBs. It was hypothesized that if adequate size and numbers of resident fish were present they would contain substantial PCB concentrations as well. Results of PCB analysis of redbreast collected in March 1989 and January 1990 confirmed this hypothesis (Table 4.3). Mitchell Branch fish contained significantly higher mean PCB concentrations than reference stream fish in both years ($p < 0.05$, t-test). The level of PCB contamination in fish (mean of 1.18 $\mu\text{g/g}$, both years combined) is high for a relatively short-lived, low-lipid species such as redbreast sunfish; longer-lived older, fattier species such as channel catfish, carp (*Cyprinus carpio*), or large-mouth bass (*Micropterus salmoides*), if present, would probably contain much higher levels. In general, the degree of PCB contamination in redbreast from Mitchell Branch was similar to the level of contamination found at the most highly contaminated sites on East Fork Poplar Creek (EFK 23.4) and White Oak Creek (WCK 2.9) (Figs. 4.3 and 4.4). PCB levels in Mitchell Branch fish could pose concern for human exposure; two of eight redbreast in 1989 contained PCB concentrations above the FDA action level (FDA 1984b).

The significance of Mitchell Branch as a source of PCB contamination is evident when comparing the levels of contamination in fish collected in Poplar Creek upstream and downstream of the mouth of Mitchell Branch. Because no statistically significant relationship was found between PCB concentrations in fish tissue and fish

weight (ANOVA; $p > 0.05$), comparisons were made among sites without adjusting for fish size. Additionally, no adjustments were made when mean PCB concentrations were compared between redbreast sunfish and bluegill from different sites, because past studies on the reservation (Loar 1992b; Loar 1993a) have shown that mean PCB concentrations between redbreast and bluegill collected from the same sites were not significantly different. Mean PCB concentrations in Mitchell Branch redbreast were significantly higher ($p < 0.05$) than mean PCB concentrations in bluegill at PCKs 6.9 and 8.2 in each year (Table 4.3). Much higher concentrations were found in fish from Mitchell Branch than in fish from PCK 6.9, even though PCK 6.9 is only a short distance downstream of Mitchell Branch. These results confirm the conclusions drawn from the earlier clam data (i.e., Mitchell Branch is a substantial source of PCBs to its own biota; fish were not likely to have obtained such high PCB concentrations elsewhere).

4.3.5 PCBs in Poplar Creek Fish

PCB concentrations were measured in bluegill from several sites in Poplar Creek and the Clinch River to assess the relative importance of various streams (Bear Creek, EFPC, Mitchell Branch, and White Oak Creek) to the system. PCB results for the Poplar Creek area are shown in Table 4.3; results for the entire reservation are shown in Figs. 4.3 and 4.4.

Fish from PCK 10.4, upstream of any discharges associated with DOE Oak Ridge operations, contained PCB concentrations typical of reference stream fish (Table 4.3), indicating that neither upstream sources of PCBs in Poplar Creek nor movement of fish between sites was a probable explanation for the high PCB levels at PCKs 8.2 and 6.9. As was observed for mercury, the mean concentrations of PCBs in fish from

Table 4.3. PCB concentrations in bluegill (*Lepomis macrochirus*), redbreast sunfish (*L. auritus*), channel catfish (*Ictalurus punctatus*), and gizzard shad (*Dorosoma cepedianum*) collected at sites in the Poplar Creek drainage area in the vicinity of the Oak Ridge K-25 Site

Site ^a	Species	Date	PCB mean \pm 1 SE ($\mu\text{g/g}$, wet wt)	Range	FDA exceedences ^b
EFK 2.1	Bluegill	December 1987	0.14 \pm 0.04	0.03-0.41	0/8
		November 1988	0.13 \pm 0.03	0.03-0.33	0/8
PCK 10.4	Bluegill	November 1987	0.06 \pm 0.01	0.02-0.11	0/8
		November 1988	0.05 \pm 0.03	0.01-0.26	0/8
PCK 8.2	Bluegill	November 1987	0.20 \pm 0.03	0.11-0.36	0/8
		November 1988	0.31 \pm 0.07	0.08-0.59	0/8
		March 1990	0.07 \pm 0.02	0.02-0.15	0/8
MIK 0.2	Redbreast	March 1989	1.61 \pm 0.32	0.50-3.27	2/8
		January 1990	0.80 \pm 0.14	0.23-1.55	0/8
PCK 6.9	Bluegill	June 1987	0.22 \pm 0.06	0.05-0.53	0/8
		November 1987	0.26 \pm 0.04	0.14-0.47	0/8
		November 1988	0.21 \pm 0.05	0.08-0.40	0/7
		March 1990	0.12 \pm 0.01	0.07-0.23	0/8
	Channel catfish	August 1988	0.71 \pm 0.13	0.28-1.31	0/8
		August 1989	1.07 \pm 0.35	0.25-3.37	1/8
	Gizzard shad ^c	August 1988	2.97 \pm 0.76	1.76-4.37	
PCK 1.6	Bluegill	December 1987	0.17 \pm 0.03	0.10-0.37	0/8
		November 1988	0.19 \pm 0.04	0.06-0.39	0/8
CRK 15.0	Bluegill	November 1987	0.08 \pm 0.01	0.04-0.15	0/8
		November 1988	0.11 \pm 0.03	0.01-0.28	0/8
	Channel catfish	August 1988	0.50 \pm 0.07	0.15-0.90	0/8
		August 1989	0.79 \pm 0.25	0.01-2.10	1/8
	Gizzard shad ^c	August 1988	0.82 \pm 0.12	0.58-0.94	
	Reference sites				
MHR	Bluegill	1987-1988	0.09 \pm 0.02	0.02-0.28	0/16
	Channel catfish	1988-1989	0.41 \pm 0.09	0.07-1.61	0/18
	Gizzard shad ^c	August 1988	0.40 \pm 0.03	0.35-0.44	0/8
HC	Redbreast	1985-1989	0.04 \pm 0.01	0.00-0.18	0/50

^aMHR = Melton Hill Reservoir; HC = Hinds Creek; EFK = East Fork Poplar Creek kilometer; PCK = Poplar Creek kilometer; MIK = Mitchell Branch kilometer; CRK = Clinch River kilometer.

^bNumber of samples exceeding the FDA action level of 2.00 $\mu\text{g/g}$ [U.S. Department of Agriculture Food and Drug Administration 1984, *Polychlorinated biphenyls (PCBs) in fish and shellfish: Reduction of Tolerances*, Final Decision, Fed. Regist. 49:(100):21520] divided by the total number of samples.

^cThe mean and range for gizzard shad represents three composite samples of five shad each.

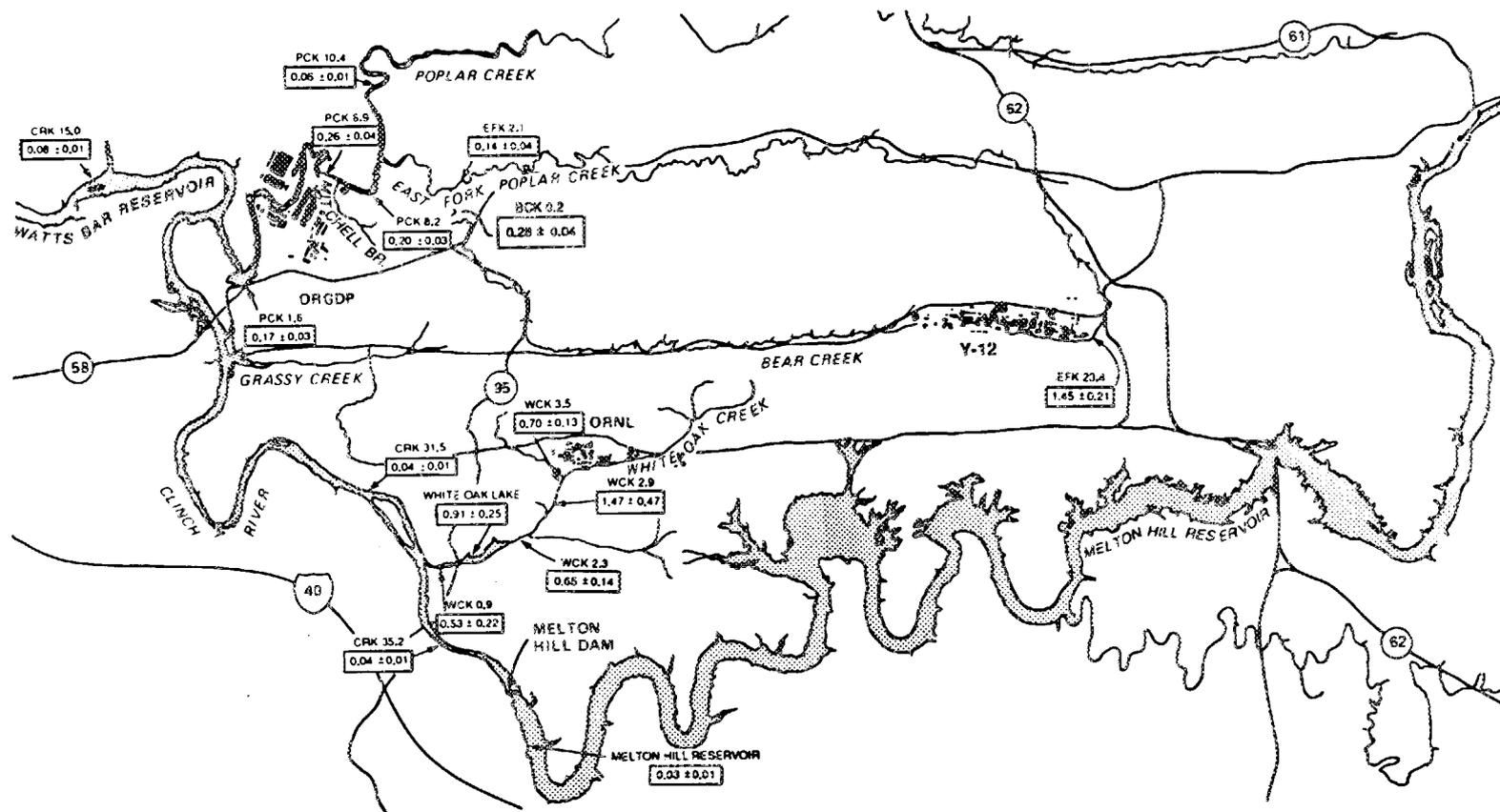


Fig. 4.3. Mean concentrations (± 1 SE) of polychlorinated biphenyl (in micrograms per gram, wet weight) in bluegill ($n = 8$) collected in fall/winter 1987 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River. Rockbass (*Ambloplites rupestris*) and redbreast sunfish (*Lepomis auritus*) were substituted for bluegill at Bear Creek kilometer 0.2.

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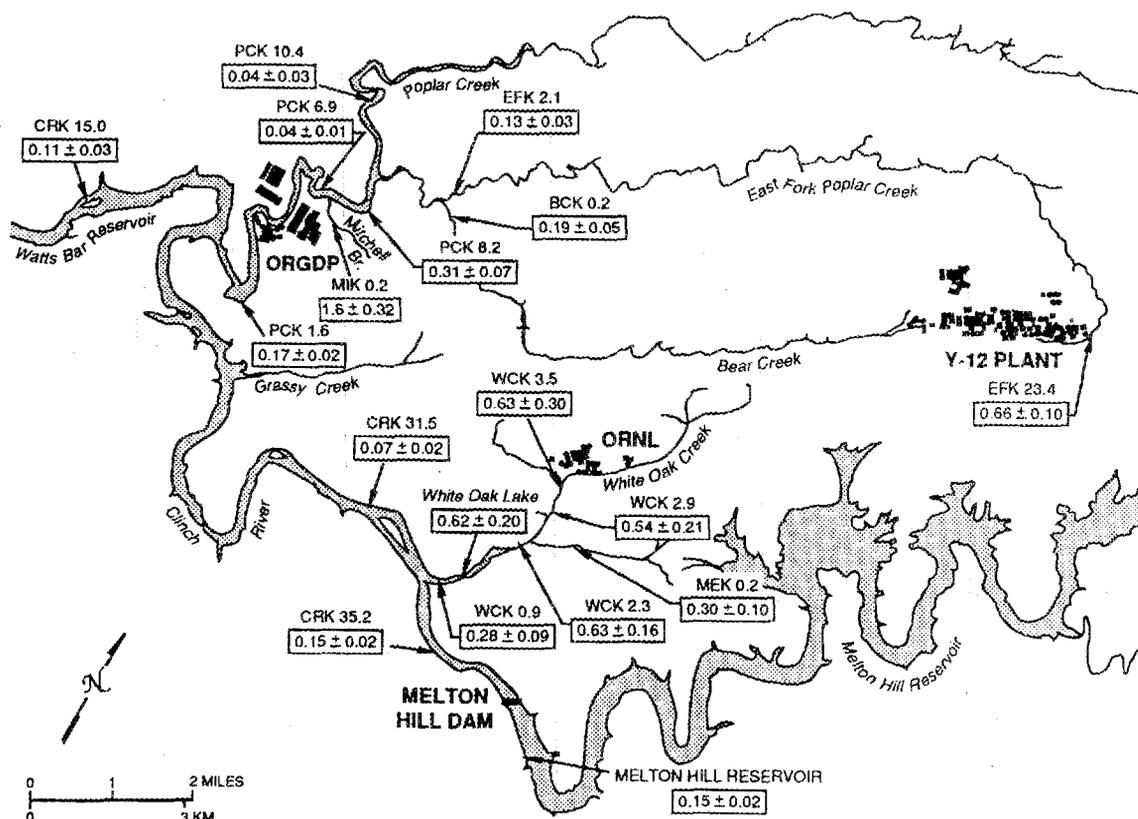


Fig. 4.4. Mean concentrations (± 1 SE) of polychlorinated biphenyl (in micrograms per gram, wet weight) in bluegill ($n = 8$) collected in fall/winter 1988 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River. Rockbass (*Ambloplites rupestris*) and redbreast sunfish (*Lepomis auritus*) were substituted for bluegill at Bear Creek kilometer 0.2 and Mitchell Branch kilometer 0.2 respectively.

lower Poplar Creek (PCKs 8.2, 6.9, and 1.6) did not decline proportionally to the dilution of EFPC in Poplar Creek, but rather remained at levels typical or higher than those of lower EFPC (Figs. 4.3 and 4.4). In 1987 and 1988, the mean PCB concentration in bluegill collected downstream of Mitchell Branch (PCK 6.9) was not significantly different from that for bluegill at the site upstream of Mitchell Branch (PCK 8.2). In 1990, the mean PCB concentration in bluegill at PCK 6.9 was higher than at PCK 8.2, but concentrations at both sites were at or near background. The relatively low levels of PCBs in bluegill at PCK 6.9 (mean of

0.22, all years), coupled with the lack of substantial increases in PCBs at PCK 6.9 compared with PCK 8.2, suggest that PCB inputs from Mitchell Branch have little impact on Poplar Creek. However, the use of sunfish to monitor very dilute inputs of PCBs may be limited because lipid-rich species collected at PCK 6.9 (channel catfish and gizzard shad) contained much higher levels of PCBs (Table 4.3).

The degree of PCB contamination in bluegill from the Poplar Creek sites near the Oak Ridge K-25 Site (PCKs 8.2, 6.9, and 1.6) was relatively low compared with other sites on the reservation (Figs. 4.3 and 4.4). The level of PCB contamination

in Poplar Creek bluegill was most similar to levels in fish from lowermost EFPC, in rockbass from Bear Creek (Loar 1993b), and in fish from the lowermost part of the White Oak Creek drainage [White Oak Lake and White Oak Creek kilometer (WCK) 0.2].

PCB contamination in Poplar Creek appeared to have little or no impact on PCB concentrations in bluegill in the Clinch River (Table 4.3). Mean PCB concentrations in bluegill from the Clinch River downstream of the mouth of Poplar Creek (CRK 15.0) were slightly elevated over upstream sites on the Clinch River [CRKs 31.5 and 35.5 and Melton Hill Reservoir (MHR)] in 1987, but in 1988 concentrations in fish at CRK 15.0 were at or below PCB levels upstream (Figs. 4.3 and 4.4).

The Poplar Creek embayment continues to receive ongoing PCB inputs from at least three sources: EFPC, Bear Creek, and Mitchell Branch. Resolving the significance of each PCB source on the PCB burden of Poplar Creek biota is not possible with the existing data. Further complicating an evaluation of possible sources to Poplar Creek is the fact that Poplar Creek is a depositional backwater of Watts Bar Reservoir and thus is likely to accumulate PCB-contaminated sediments. Some of the observed contamination in fish may be a result of mobilization of sediment associated PCBs rather than ongoing stream discharges.

Channel catfish were collected at PCK 6.9 and CRK 15.0 in 1988 and 1989 as part of an evaluation of the importance of inputs from DOE sources in contributing to PCB contamination in Watts Bar Reservoir. Results of this study are given in detail in Loar (1993a) and are only summarized here. Mean PCB concentrations in channel catfish from PCK 6.9 in both years were elevated over concentrations in catfish at both the downstream site on the Clinch River (CRK 15.0) and

MHR (Table 4.3). However, these differences were not great, considering the discrete nature of EFPC/Poplar Creek as a source of PCB contamination. It is likely that movements of channel catfish and forage fish act to spread PCB contamination away from point sources, which results in a rather uniform distribution of PCB contamination in catfish throughout Watts Bar Reservoir, as was observed in the Oak Ridge Task Force study in 1984 (TVA 1985). One in eight fish collected in 1989 from both CRK 15.0 and PCK 6.9 exceeded the 2-ppm action level. The mean PCB concentrations in channel catfish in 1988 and 1989 at PCK 6.9 support the precautionary fish consumption advisory issued by the Tennessee Department of Health and Environment.

Gizzard shad were collected concurrently with catfish in 1988 to assess the significance of PCB accumulation in forage fish. The mean PCB concentration of three composite samples (five gizzard shad each) collected at PCK 6.9 was significantly higher than for shad collected from CRK 15.0 and MHR ($p < 0.05$) (Loar 1993a). The PCB concentration of nearly 3 ppm in shad from PCK 6.9 clearly indicates that Poplar Creek embayment, like White Oak Creek embayment (Loar 1993a), is a relative "hot spot" for PCB contamination. Highly contaminated gizzard shad from the area are likely to be important vectors for PCB dispersal to larger game fish that feed on shad (e.g., largemouth bass, catfish, and rockfish).

The mean PCB concentration in shad from CRK 15.0, as the case with bluegill and channel catfish, was not significantly different from the concentrations in shad from upstream sites (CRK 32.2 and MHR) (Loar 1993a). Evaluating the effect of various PCB inputs from DOE facilities on fish in the Clinch River proper is complicated by two factors: (1) substantial PCB contamination is evident in fish immediately upstream of any DOE impacts

(MHR) and (2) the species that tend to accumulate the highest levels of PCBs (catfish and shad) appear to range widely, obscuring relationships between source location and PCB levels in fish.

4.3.6 PCBs in Caged Clams

Caged clams held in Mitchell Branch for four weeks accumulated substantial concentrations of PCBs in all exposure periods from 1988 to 1990 (Table 4.4). Mean concentrations ranged from 0.34 $\mu\text{g/g}$ in April 1988 to 0.98 $\mu\text{g/g}$ in July 1989. Previous clam exposures in Mitchell Branch also resulted in substantial PCB accumulation (Smith et al. 1993). The degree of PCB accumulation noted in clams was consistent with the relatively high concentrations of PCBs found in redbreast and bluegill sunfish from Mitchell Branch in 1989 and 1990 (Table 4.3).

PCB concentrations in clams in 1988-90 were lower than those observed in 1987, when clams were exposed soon after a PCB discharge in the Mitchell Branch drainage (Smith et al. 1993). However, there does not appear to have been a continuing decrease in biotic PCB contamination over the 1988-90 period. In fact, the highest PCB concentrations observed in 1988-90 occurred in July 1989 and April 1990 (Table 4.4). Interpretation of temporal trends in the degree of PCB contamination must be made with caution because adverse environmental conditions, such as lack of food or the presence of toxicants, may result in atypically low PCB accumulation by clams despite the presence of PCB contamination. Thus, it may be possible under improving ecological conditions in a highly affected stream, such as Mitchell Branch, for PCB accumulation in caged clams to increase over time without a concurrent increase in PCB inputs to the system. The data collected to date show that Mitchell Branch contains significant

PCB contamination, which is not rapidly increasing or decreasing.

In 1988, clams placed downstream from the mouth of Mitchell Branch accumulated slightly higher concentrations of PCBs than clams placed upstream from Mitchell Branch (Table 4.4). These data suggest that the impact of PCB discharges from Mitchell Branch to Poplar Creek may be discernable in Poplar Creek biota, although no definite conclusions can be drawn from single sampling. As noted in Sect. 4.3.5, such an impact is not apparent in Poplar Creek sunfish.

The concentrations of PCBs measured in caged clams in Mitchell Branch indicate that this stream is among the most highly PCB contaminated aquatic sites on ORR (Table 4.5). Levels of PCB contamination in clams from Mitchell Branch in 1988-90 were similar to those measured in clams from White Oak Creek/White Oak Lake, Bear Creek, and EFPC immediately downstream from the Oak Ridge Y-12 Plant. PCB concentrations in resident sunfish at these sites (Table 4.3) are similar to those observed in caged clams following a 4-week exposure.

4.3.7 Other Organics in Caged Clams

Some chemicals, such as polycyclic aromatic hydrocarbons (PAHs) do not accumulate appreciably in fish because of their rapid conversion into more soluble metabolites. Because clams such as *Corbicula* have less developed capabilities for detoxifying organic contaminants than fish, these chemicals can accumulate substantially in clams if they are present in solution or in particles filtered while feeding. Analysis of clams for semi-volatile organic compounds and PAHs did not reveal any substances not also found at similar concentrations in reference stream clams, except for the detection of very low concentrations of PAHs in 1990

Table 4.4. Concentrations of polychlorinated biphenyls (PCBs) and other organic contaminants in caged clams (*Corbicula fluminea*) maintained in Mitchell Branch for 4 weeks

Site	Date	PCB ^a ($\mu\text{g/g}$ wet wt)	Other organics ^a ($\mu\text{g/g}$ wet wt)
MIK 0.2	April 1988	0.34 \pm 0.07	BLD ^{b,c} (below limit of detection)
MIK 0.2	April 1989	0.69 \pm 0.02 (2)	
MIK 0.2	July 1989	0.98 \pm 0.10	BLD ^d
MIK 0.2	April 1990	0.91 \pm 0.06 (2)	aldrin, 0.045 chlordan ^e , 0.007 acenaphthene ^e , 0.008 anthracene ^e , 0.001 benz(a)anthracene ^e , 0.009 chrysene ^e , 0.004 fluorene ^e , 0.022 Others BLD ^f
PCK 8.2	April 1988	0.12 \pm 0.00 (3)	
PCK 6.9	April 1988	0.19 \pm 0.01 (3)	
Reference sites	April 1988	0.05 \pm 0.00 (3)	BLD ^{b,c}
	April 1989	0.12 \pm 0.01 (2)	
	July 1989	<0.02	BLD ^d
	April 1990	<0.09	BLD ^f

^aValues are mean \pm 1 SE, with the number of samples in parentheses.

^bBis(2-ethylhexyl)phthalate, diethylphthalate, benzoic acid, and bis(2-chloroisopropyl)ether were detected in reference site samples and MIK samples.

^cDiethylphthalate and benzoic acid were detected in reference site samples and MIK samples.

^dDiethylphthalate was detected in reference site samples and MIK samples.

^eThese trace concentrations of these substances were reported at levels far below the analytical limit of quantification, but at higher concentrations than found in reference site samples. Confirmation that these substances were indeed present is impossible at such low levels.

^fDiethylphthalate and dibutylphthalate were detected in reference site and MIK samples.

Note: MIK = Mitchell Branch kilometer; PCK = Poplar Creek kilometer.

Table 4.5. Range of mean PCB concentrations measured in caged clams (*Corbicula fluminea*) at PCB-contaminated sites on Oak Ridge Reservation, 1988-90

Site	PCBs ($\mu\text{g/g}$, wet wt)
Mitchell Branch, km 0.2	0.34-0.98
Bear Creek, km 4.5	0.32-1.01
East Fork Poplar Creek, km 23.4	0.18-0.47
White Oak Creek, km 2.6	0.23-0.89
White Oak Lake	0.41-0.83

(Table 4.4). Improvements in analytical detection limits in 1990 enabled the detection of lower concentrations of PAHs than was possible in previous years. Concentrations of several PAHs were clearly higher in clams held in Mitchell Branch than in clams at the reference sites; however, the concentrations were very low, the highest being fluorene ($0.022 \mu\text{g/g}$). Given the presence of coal pile runoff and hydrocarbon contamination in Mitchell Branch, it is not surprising to find traces of PAHs in clams. This finding corroborates the detection of elevated levels of mixed function oxidases in fish in Mitchell Branch (Sect. 5.3.1) because PAHs stimulate the production of such detoxification enzymes (Payne and Penrose 1975).

Analysis of clams for hydrophobic pesticides detected aldrin in one of two clam samples at a concentration of $0.04 \mu\text{g/g}$. This agricultural pesticide is no longer sold in the United States, and its occurrence at the Oak Ridge K-25 Site is inexplicable. Similar concentrations of aldrin were reported in one clam sample from Bear Creek and in a reference site clam sample that had been spiked with PCBs, PAH, and phthalates to determine

percentage recovery of analytes through the analytical procedure. Aldrin was not present in the spike, nor was it detected in the unspiked reference site clams. Thus, it appears likely that the aldrin reported in one Mitchell Branch clam sample was a result of a positive interference in the analytical procedure. Annual monitoring for pesticide contamination will continue in Mitchell Branch; if aldrin contamination is present, it should appear in future samples.

4.4 CONCLUSIONS

Mercury and PCBs are the two primary contaminants that accumulate above background levels in Mitchell Branch biota. From 1987 through 1990, the levels of mercury were not excessive relative to the FDA limit and were similar to those in fish in nearby Poplar Creek. However, the significance of Mitchell Branch as a source of the elevated Hg levels observed in its fish, or in fish from lower Poplar Creek, remains unclear. Elevated concentrations of other metals were not found in fish from Mitchell Branch.

The PCB monitoring data were more conclusive and clearly showed that Mitchell Branch was a source of PCB contamination to its biota. Some fish in Mitchell Branch contained PCB concentrations in excess of the 2 $\mu\text{g/g}$ FDA action level. Significant PCB contamination ($>1 \mu\text{g/g}$) was also evident in channel catfish and gizzard shad collected in Poplar Creek a short distance downstream of Mitchell Branch. Mitchell Branch undoubtedly is a source of PCB contamination to Poplar Creek, but its significance as a source of PCB contamination to the biota in the Clinch River arm of Watts Bar Reservoir relative to other possible sources (EFPC, White Oak Creek, Bear Creek, and Poplar Creek sediment), cannot be ascertained from these data.

Caged clams placed in Mitchell Branch accumulated very low concentrations of PAHs and substantial concentrations of PCBs. Oil and coal inputs to the stream probably account for the trace amounts of PAHs, which in turn are probably the causes of the elevated concentrations of detoxification enzymes in fish reported in Sect. 5.3.1. A discontinued agricultural insecticide, aldrin, was detected in one of two clam samples analyzed in 1990, but its detection in two other unrelated samples suggest that it may be the result of an analytical interference.

4.5 FUTURE STUDIES

The most significant contaminant bio-accumulation problem in Mitchell Branch appears to be PCBs. A key objective of future studies will be to identify and characterize the PCB source(s) in Mitchell Branch (e.g., specific ongoing discharges, sediment contamination, and episodic releases). Placement of clam cages further upstream near possible PCB discharges, along with sediment sampling of Mitchell Branch may help in locating the PCB source(s).

Resident fish and caged clams will continue to be used for monitoring the accumulation of metals and organics in Mitchell Branch. Bluegill will continue to be collected from sites on Poplar Creek to further evaluate the role of Mitchell Branch as a source of mercury and PCBs in that system. To evaluate the significance of Mitchell Branch as a source of mercury to its biota, caged fish may be introduced and monitored for it in conjunction with the biological indicator studies (Sect. 5). Alternatively, resident species unlikely to be immigrants from Poplar Creek, such as creek chubs or blacknose dace, may be analyzed for mercury if populations are large enough that such collections would not substantially affect them.

5. BIOLOGICAL INDICATORS OF CONTAMINANT-RELATED STRESS

S. M. Adams and W. D. Crumby

5.1 INTRODUCTION

This report summarizes the results of the 1989 bioindicator study conducted in Mitchell Branch and three reference streams. Two approaches were used to evaluate the current health status of fish in Mitchell Branch: (1) analysis of functional biological response groups and (2) analysis of integrative bioindicator groups. The functional group approach involved analysis of five major groups of biotic responses to determine if differences in fish health existed between sites. The integrative bioindicator analysis involved the use of all bioindicators together within a multivariate context to investigate holistic responses of fish to stress and to aid in the identification of causative agents or mechanisms that may be affecting the health of fish populations in Mitchell Branch.

5.2 METHODS

5.2.1 Sampling Procedures

Sampling was conducted during summer of 1989 in Mitchell Branch (above weir) and three reference streams, including Brushy Fork located between Oliver Springs and Clinton (Anderson County, Tennessee), Hinds Creek between Clinton and Norris (Anderson County), and Paint Rock Creek (Loudon County, Tennessee). At each site, redbreast sunfish (*Lepomis auritus*) of all sizes and age groups were collected by electroshocking. Blood samples were taken from 12 to 15

of the larger male fish within 2 min after collection by puncturing the caudal vessels with a 20-gauge needle. Blood samples of ~0.7 mL were obtained from all fish by using unheparinized 3-mL vacutainers (Becton, Dickson, & Co.). Each tube was labeled with a fish identification number and placed on ice for transport to the laboratory. All fish were processed for population-level analysis by recording their lengths and weights and removing a sample of scales for age and growth information. Fish not used for the blood and bioindicator analysis were returned to the stream alive.

5.2.2 Analytical Procedures

Total lengths and weights were recorded for fish transported from the field, and observations were also made on the general condition of the fish, such as presence or absence of disease, body and/or mouth sores, external parasites, and general appearance. Following sacrifice, the liver and spleen were removed from each fish for further analysis. A 100-mg section of liver was placed in a 20-mL scintillation vial with 5-mL of Bouin's fixative for histopathological analysis. The remaining viscera (minus liver and spleen) were excised from the body cavity, and their total weight was recorded after all food material was removed from the stomach and intestine. The liver and visceral somatic-indices were calculated by dividing their respective weights by total body weight. The condition factor (K) of each fish was calculated as $K = 10^5 W/L^3$,

where W is body weight (g) and L is total length (cm).

5.2.2.1 Serum analyses

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

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

5.2.2.2 Histopathological analyses

The following histopathological analyses were performed by the School of Veterinary Medicine, University of

California-Davis: (1) percentage of tissue occupied by parasites, (2) percentage of liver composed of necrotic parenchyma, (3) percentage of tissue composed of macrophage aggregates, and (4) percentage of liver occupied by functional parenchyma. These analyses were performed by using methods described by Hinton and Couch (1984).

5.2.2.3 Detoxification enzymes

Microsome isolation. Fish hepatic microsomes were prepared by differential centrifugation (McKee et al. 1983) with several modifications. Fish were sacrificed by severing the spinal cord, and the livers were immediately removed and blotted dry. Each liver sample was placed in ice-cold buffer (0.25 mM sucrose, 0.1 M tris, pH 7.4). The tissues were minced and then homogenized in 5 volumes of buffer with a motor-driven Potter-Elvehjem glass and Teflon homogenizer. The homogenates were centrifuged in a J-21B Beckman centrifuge at 3000 g^* for 10 min and at 10,000 g for 20 min. The resulting supernatants were then centrifuged at 105,000 g for 60 min in a Beckman L3-50 ultracentrifuge. Microsomal pellets were resuspended in 0.1 M tris buffer, 1 mM EDTA, and 20% glycerol at pH 7.4 by sonication for 10 to 15 s with a Braun Sonic 1510 at 50 W. All operations were performed at 0 to 4°C. The microsomes were frozen with liquid nitrogen and stored at -120°C; no significant change in 7-ethoxyresorufin O-deethylase (EROD) activity was detected after 6 months of storage at this temperature. The activity of fish microsomes stored under these conditions have been reported to be stable for 1 year (Forlin and Anderson 1985).

*Italic lower case g denotes standard acceleration due to gravity ($\sim 9.8 \text{ m}\cdot\text{s}^{-1}\cdot\text{s}^{-1}$).

Enzyme assays. The activity of EROD was measured fluorimetrically at 30°C (Burke and Mayer 1974) and expressed as moles of resorufin per minute per milligram of microsomal protein. The final reaction buffer contained 80 mM HEPES buffer (pH 7.8), 5 mM magnesium acetate, 1.0 μ M 7-ethoxyresorufin, 250 μ M of reduced nicotinamide adenine dinucleotide phosphate (NADPH), and 100 μ M EDTA. The concentration of total protein used in the enzyme assay ranged from 0.2 to 1.0 mg/mL, depending on the activity of the sample. Proteins were measured by the Bio-Rad (Richmond, California) reagent method (Bradford 1976) with a Centrifugal Fast Analyzer System in which bovine serum albumin was used as a standard. Cytochrome P-450 and cytochrome b_5 content were each measured by their characteristic oxidized and reduced spectra. Cytochrome P-450 samples were oxidized with carbon monoxide and reduced with sodium dithionite (Johannesen and DePierre 1978). Cytochrome b_5 was reduced with NADH, and the b_5 assays were conducted prior to the P-450 analysis (Stegeman et al. 1979). NADPH-cytochrome c reductase was assayed spectrophotometrically by the reduction of cytochrome c using an extinction coefficient of 21.1 cm/mM.

5.2.2.4 Population characteristics

Scale samples were used to estimate the age and growth rates of redbreast sunfish in Mitchell Branch, Hinds Creek, and Brushy Fork. Approximately ten scales were removed from each fish by using a small pair of metal forceps. Scales were taken from the left side of the body near the tip of the pectoral fin and stored in small coin envelopes. Total length (centimeter), weight (gram), species, sample site, date, and tag number (if applicable) were recorded on each envelope.

Fish not used for blood samples and bio-indicator analyses were returned to the stream alive after the scale sample was taken.

In the laboratory, scales from each fish were arranged in rows between two cellulose acetate slides. The slides were pressed together by using a hand-cranked laboratory press (Wildlife Supply Co.), which left an impression of each scale on one of the slides. Scales of the smallest fish (4–6 cm total length) were not pressed because they did not make an adequate impression on the cellulose acetate slides. Instead, they were arranged between two slides, which were taped together and read directly.

Scale impressions were analyzed by a single individual who used microfiche machine with a magnification of 43 \times . Scales were measured and annulus marks identified according to the techniques outlined by Jerald (1983). The scale radius and the distance from the focus to each annulus were measured to the nearest 1 mm. All scale samples from Brushy Fork and Mitchell Branch were analyzed, but because the Hinds Creek sample was so large, no more than 15 fish were read for any 1-cm size group. If there were 15 or less fish in a size group, all fish in that group were analyzed. Quality assurance of the scale samples was maintained by having 20% of the scale samples from each site analyzed by a second individual.

Fish were divided into 1-cm size groups based on total length (TL), and length-frequency histograms were developed for each stream. Data from the scale analyses were used to assign fish to age groups. Mean TL (in centimeters) and mean weight (in grams) at time of capture were calculated for each age group.

A length-weight relationship of redbreast sunfish in each stream was developed according to the formula: $\log W = \log_{10} a + b \log_{10} L$; where W is

the weight at time of capture, b is the regression coefficient, $\log a$ is the intercept of the line on the Y-axis, and L is the total length (TL) of the fish at the time of capture (Bagenal 1978). The regression coefficient b is the functional slope of the length-weight relationship and was calculated using a GLM procedure from SAS (1985b).

Estimates of true growth were calculated in accordance with the methods of Ricker (1975). The true growth rate G is the instantaneous rate of increase in weight for the last complete year of growth. To estimate G , the following procedures were followed:

1. The age of each fish was determined from scales, and measurements to successive annuli were recorded.
2. A relationship of scale size to fish TL was established.
3. The back-calculation procedure of Carlander (1981) was used to estimate fish TL at the start and close of the last complete year of growth for each age group.
4. The natural logarithm of the TLs determined in (3) was calculated for each fish, and initial lengths were then subtracted from final lengths. The remaining values represented the instantaneous rate of increase in length for each fish.
5. The instantaneous rates of increase in length for each age group were averaged and multiplied by b (calculated in length-weight relationship). The resulting value of G is the mean instantaneous rate of increase in weight (true growth) for each age group during the last complete year of growth.

After values for G underwent natural log transformation, they were compared by age class by using an ANOVA in which site was the main effect (PROC GLM,

SAS 1985b). Site differences were then separated with a Tukey studentized range test with significant differences being accepted at $p < 0.05$.

5.2.3 Statistical Procedures

An ANOVA was used to test for site and seasonal effects, and their interaction on each bioindicator variable. If the ANOVA rejected a multisample null hypothesis of equal means, then Dunnett's test (Zar 1984) was used to test for significant differences in response parameters in fish from Mitchell Branch relative to those for fish from the reference sites. The significance level for rejecting the hypothesis of equal means was set at $p < 0.05$.

To determine the integrated response of fish to the environmental conditions at each sampling site, all the bioindicator variables were considered jointly within a multivariate context by using canonical discriminant analysis (PROC CANDISC, SAS 1985b). This method provides a graphical representation of the positions and orientations of the various site responses relative to each other. In addition, this method derives linear combinations of all variables and can identify statistically significant differences among treatment means even when single variables may not indicate such differences. A variable selection procedure available in SAS (PROC STEPWISE, SAS 1985b) was used to identify the variables that contributed most to the discrimination among sites based on the integrated parameters. This variable selection procedure considered all possible combinations of the observed values and, for any specified subset size, selected those variables having the best discriminating power. Tests for homogeneity of variance of individual response variables between sites were

conducted by using Levene's test (Sokal and Rohlf 1981); this procedure uses an F-distribution test that compares the ratios of the variances from two independent sample populations.

5.3 RESULTS AND DISCUSSION

Three levels of analyses were conducted to evaluate the effects of water quality on the health of fish in Mitchell Branch. First, individual parameters were analyzed to compare the response of fish in Mitchell Branch with those at reference sites. Second, the integrated bioindicator analysis determined the overall response of fish at sites known to have degraded water quality. Third, age and growth analyses were used to compare fish populations among streams.

5.3.1 Individual Parameter Response

Individual parameter responses were grouped into five functional categories representing indicators of (1) detoxification enzyme induction, (2) organ dysfunction, (3) histopathology, (4) overall fish health or condition, and (5) nutritional status. These categories reflect gradients in both ecological relevance and time-course of responses to a stressor, such as a contaminant (Fig. 5.1). The variables in categories (1) and (2) are short-term response indicators and have relatively low ecological relevance, whereas indicators in groups (3) to (5) are longer-term response variables and integrate environmental conditions over longer time scales.

5.3.1.1 Detoxification enzymes

The activity or levels of liver detoxification enzymes are used to indicate exposure to various xenobiotics, such as

PAHs, PCBs, and pesticides (Payne and Penrose 1975). Many of these compounds commonly occur in industrial and municipal effluents and are accumulated within living tissues (Ahokas et al. 1976, Neff 1978, Brown et al. 1986).

Of the five detoxification enzymes measured, only EROD was significantly lower at all three reference sites compared with measurements at Mitchell Branch (Fig. 5.2). Significant accumulations of PCBs were found in fish and clams (Sects. 4.3.5 and 4.3.6), and low concentrations of PAHs were observed in clams sampled from Mitchell Branch, documenting that PAHs and PCBs occur in this system and may contribute to the elevated EROD values in Mitchell Branch fish. Of the detoxification enzymes measured, EROD has been shown to be the best indicator of toxicant exposure in fish collected from other contaminated systems (Adams 1993). Cytochrome b_5 was also lower in fish from the three reference sites relative to Mitchell Branch, but this difference was not significant. The levels of reduced nicotinamide adenine dinucleotide (NADH) and NADPH in fish at the reference sites were highly variable and did not differ significantly from the levels in fish from Mitchell Branch.

5.3.1.2 Organ dysfunction

Serum albumin, creatinine, ALT, and BUN were used as indicators of organ dysfunction. Serum albumin promotes the transport of insoluble or sparingly soluble compounds, such as fatty acids, bilirubin, and hormones, across the cell membranes. This protein is also involved with osmotic regulation and can be used as a reserve source of protein and amino acids. Serum albumin values are affected by many types of diseases and physiological disturbances. Concentrations of serum albumin were significantly elevated only at Brushy Fork,

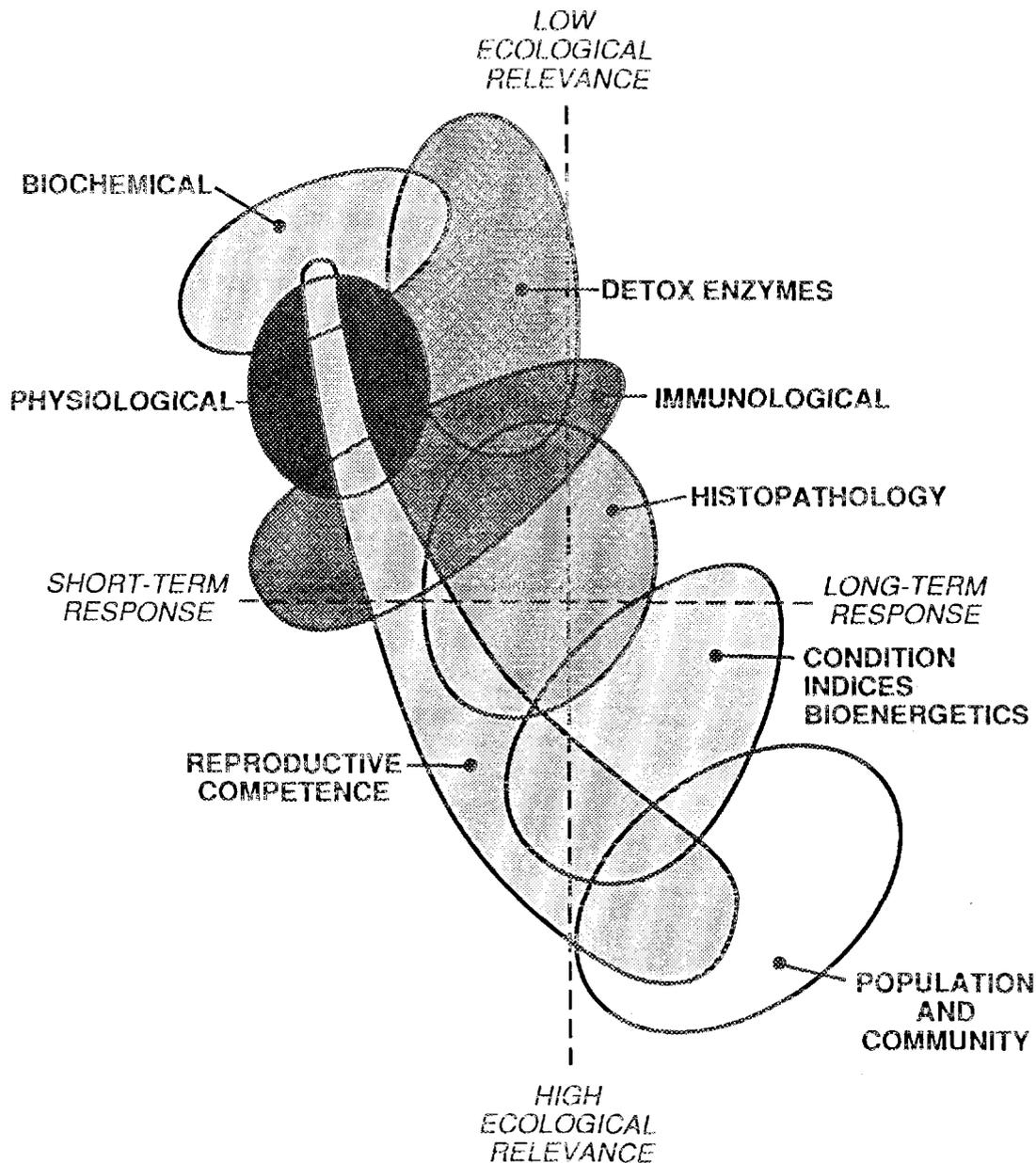


Fig. 5.1. Levels of biological response in fish to contaminant stress, illustrating the continuum of these responses along gradients of response time and ecological relevance.

possibly indicating some level of liver damage resulting from parasitic infestation (Fig. 5.3).

Elevated creatinine levels are typically used as an indicator of kidney damage or malfunction (Tietz 1986). Creatinine

values did not differ significantly among the four streams, thus, indicating that kidney function in Mitchell Branch redbreast sunfish is not being impaired.

The transferase enzyme ALT is generally used as an indicator of liver

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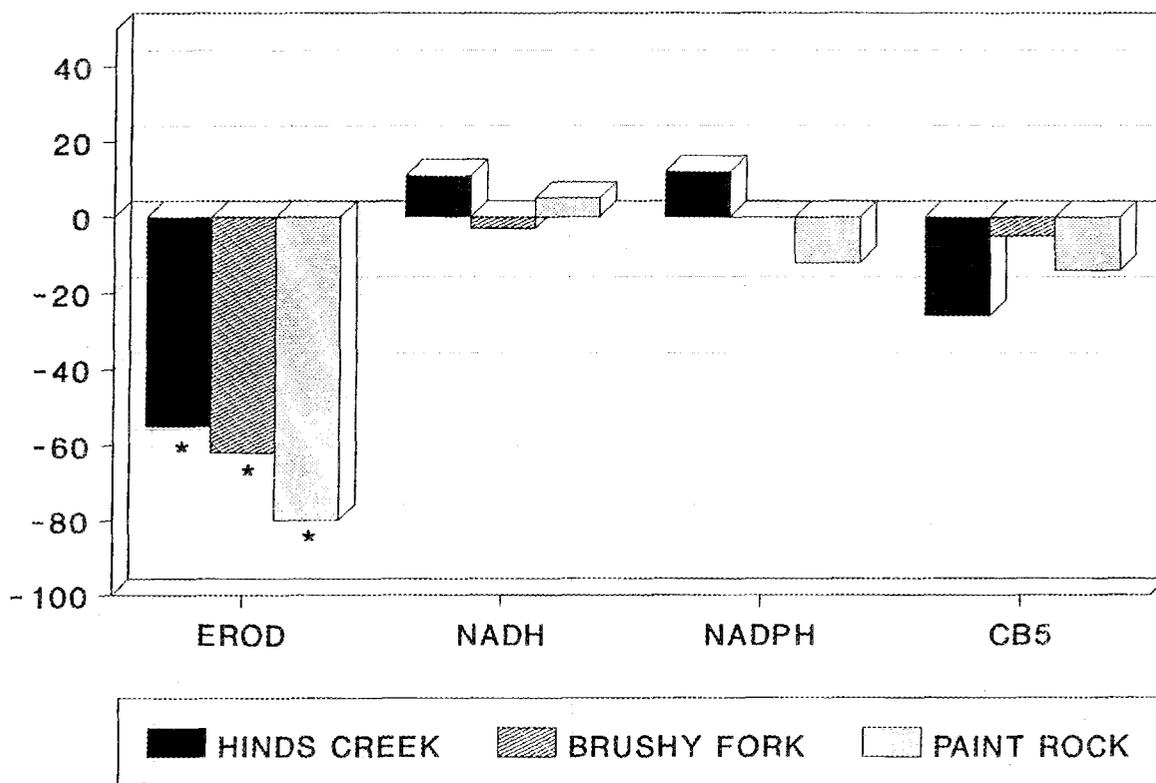


Fig. 5.2. Relative differences in the response of detoxification enzymes for redbreast sunfish from each of the reference sites compared with Mitchell Branch (the zero line). Values above or below the zero line indicate that the response for fish from each of the reference streams was higher or lower, respectively, than that for fish from Mitchell Branch. Asterisks indicate values that were significantly different ($p < 0.05$) from Mitchell Branch.

damage, with elevated levels reflecting cirrhosis or obstructive jaundice. ALT activity in fish at all reference sites was slightly lower than that of fish from Mitchell Branch. However, these differences were not significant (Fig. 5.3).

BUN is the major end product of protein nitrogen metabolism in mammals and is generally used in the medical profession to diagnose renal disorders. Fish, however, produce a more diverse array of compounds as nitrogenous waste including, but not limited to, ammonia and urea. In addition to being excreted by the kidneys, some ammonia and urea is also excreted through the gills. Therefore, in

diagnosing the effects of stress on fish, abnormal levels of urea in the blood may serve as an important indicator of gill dysfunction. In fish from Hinds Creek and Brushy Fork, BUN levels were significantly lower than in fish from Mitchell Branch. BUN levels in fish from Paint Rock Creek were lower than in fish from Mitchell Branch, but the difference was not statistically significant (Fig. 5.3). Because creatinine, an indicator of kidney dysfunction, was not elevated in Mitchell Branch fish, it is possible that water quality conditions in Mitchell Branch may be affecting gill function and not kidney function in these fish.

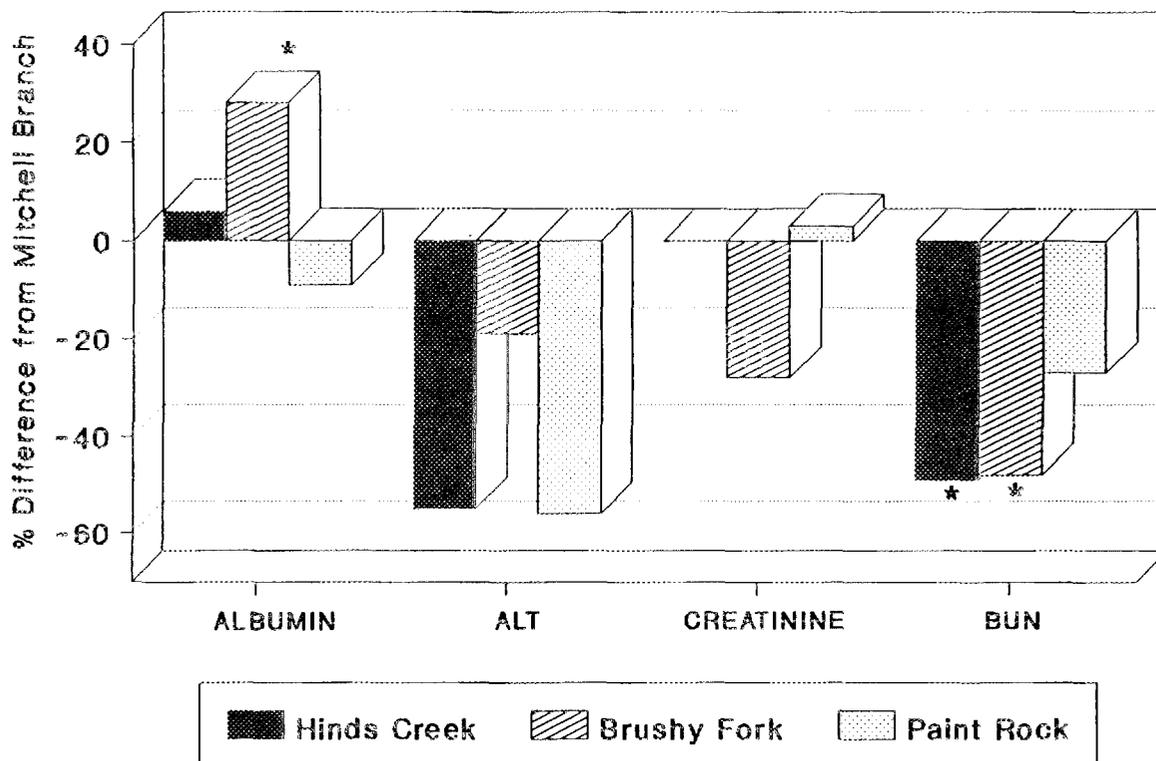


Fig. 5.3. Relative differences in the response of the organ dysfunction indicators for redbreast sunfish from each of the reference sites compared with Mitchell Branch (the zero line). Values above or below the zero line indicate that the response for fish from each of the reference streams was higher or lower, respectively, than that for fish from Mitchell Branch. Asterisks indicate values that were significantly different ($p < 0.05$) from Mitchell Branch. ALT = alanine aminotransferase, BUN = blood urea nitrogen.

5.3.1.3 Histopathological condition

Indicators of histopathological condition in Mitchell Branch fish showed differences compared with reference fish, but these differences were not significant (Fig. 5.4). The percentage of the liver composed of encysted parasites (LPARS) was lower in Hinds Creek and Brushy Fork than in Mitchell Branch but was slightly elevated in Paint Rock Creek. The percentage of liver occupied by melanophage aggregates (LMACA), which are indicators of diseased or infected areas in the tissue, was lower in Hinds Creek and Brushy Fork than Mitchell Branch but also was slightly

elevated in Paint Rock Creek redbreast. The percentage of liver actually occupied by functional parenchyma (LFPMA) was similar at all sites. There does seem, however, to be some indication of liver impairment in Mitchell Branch fish as indicated by elevated parasite loads and melanophage aggregates.

5.3.1.4 Condition indices

Three condition indices were measured as indicators of the general health of fish. The visceral-somatic index (VSI) provides information on energy

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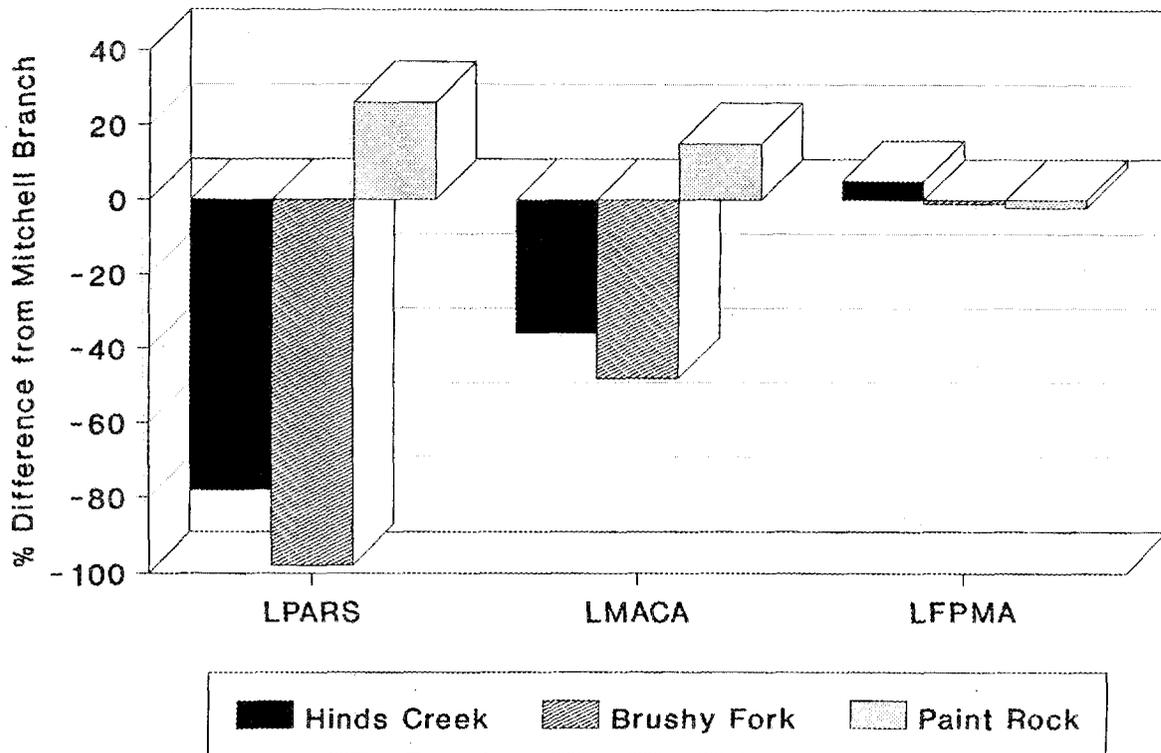


Fig. 5.4. Relative differences in histopathological condition for redbreast sunfish from each of the reference sites compared with Mitchell Branch (the zero line). Values above or below the zero line indicate that the response for fish from each of the reference streams was higher or lower, respectively, than that for fish from Mitchell Branch. Asterisks indicate values that were significantly different ($p < 0.05$) from Mitchell Branch. LPARS = liver parasites, LMACA = melanophage aggregates, LFPMA = functional parenchyma in liver.

stored as lipids in the mesenteries of the viscera. These lipids are important for long-term energy use and gonad maturation (Adams and McLean 1985). Additionally, the VSI generally reflects the level of total body lipids, which is used to indicate overall fat storage and general nutritional status of the fish. The VSI in redbreast from all three reference sites was significantly higher than in fish from Mitchell Branch (Fig. 5.5). Lower fat levels in Mitchell Branch fish may indicate that metabolic stress caused by contaminant exposure is greater in these fish and/or that less food is available.

The liver-somatic index (LSI) reflects both short-term nutritional status and metabolic energy demands (Heidinger and Crawford 1977, Adams and McLean 1985). The LSI is also sensitive to toxicant stress, and liver enlargement resulting from hyperplasia (increase in cell number) and hypertrophy (increase in cell size) has been reported in fish exposed to toxic compounds (Fletcher et al. 1982, Heath 1987, Addison 1984). The latter may be applicable to redbreast in Mitchell Branch because the LSI was higher at this site than for redbreast from all three reference streams, although the difference was not statistically significant (Fig. 5.5).

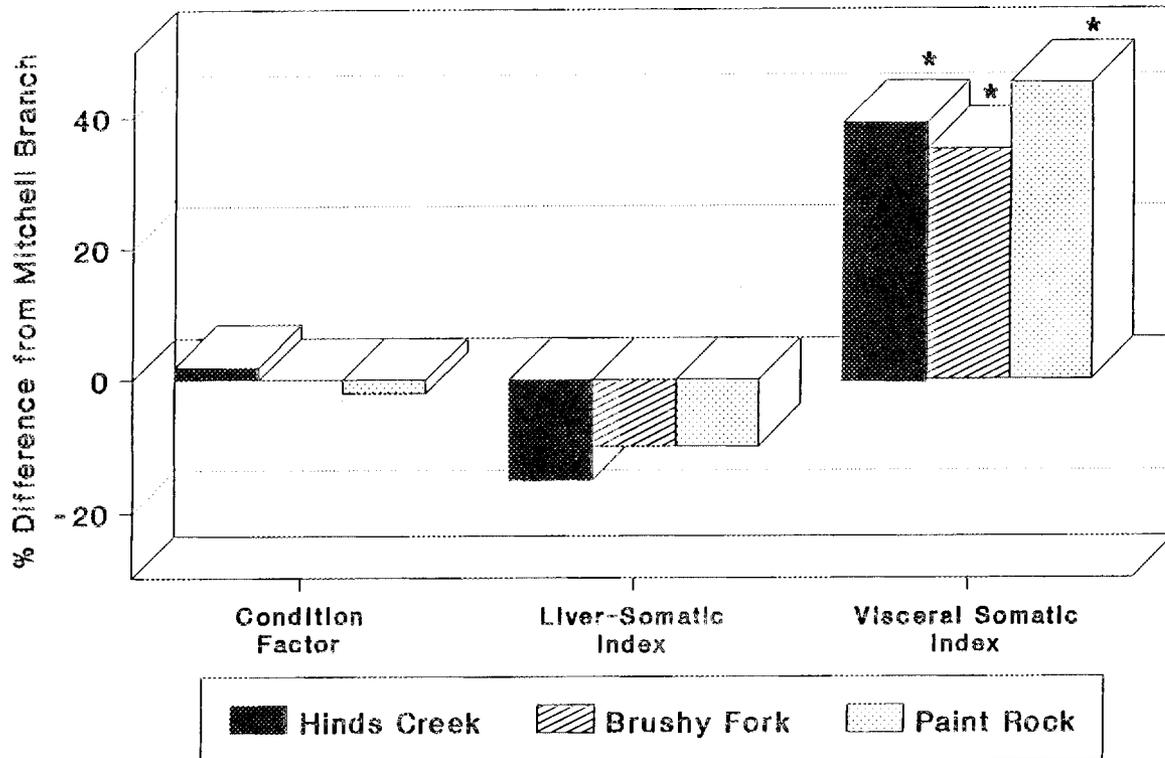


Fig. 5.5. Relative differences in the condition indices for redbreast sunfish from each of the reference sites compared with Mitchell Branch (the zero line). Values above or below the zero line indicate that the response for fish from each of the reference streams was higher or lower, respectively, than that for fish from Mitchell Branch. Asterisks indicate values that were significantly different ($p < 0.05$) from Mitchell Branch.

The condition factor (CDFC) is a generalized indicator of the overall health or "plumpness" of a fish and can reflect the integrated effect of both nutritional status and metabolic impairment caused by stress. The condition factor is relatively insensitive to changes in lipid stores (Adams and McLean 1985). This feature of the index may explain why the CDFC values were similar for fish from Mitchell Branch and the three reference sites.

5.3.1.5 Nutritional indices

An evaluation of the nutritional status of the organism is important for interpret-

ing the nature of the effects of contaminant stress on fish. Three indicators were used as measures of nutritional condition, including the percentage of the stomach and intestine containing food and the level of serum triglycerides. Fish from Hinds Creek and Paint Rock Creek had at least 45% more food in their digestive systems than fish from Mitchell Branch, whereas fish from Brushy Fork had slightly less food (Fig. 5.6). None of these differences, however, were statistically significant.

Serum triglycerides were much higher in Brushy Fork and Paint Rock Creek fish than they were in fish from Mitchell Branch, indicating either higher feeding levels (food quantity) or better nutritional

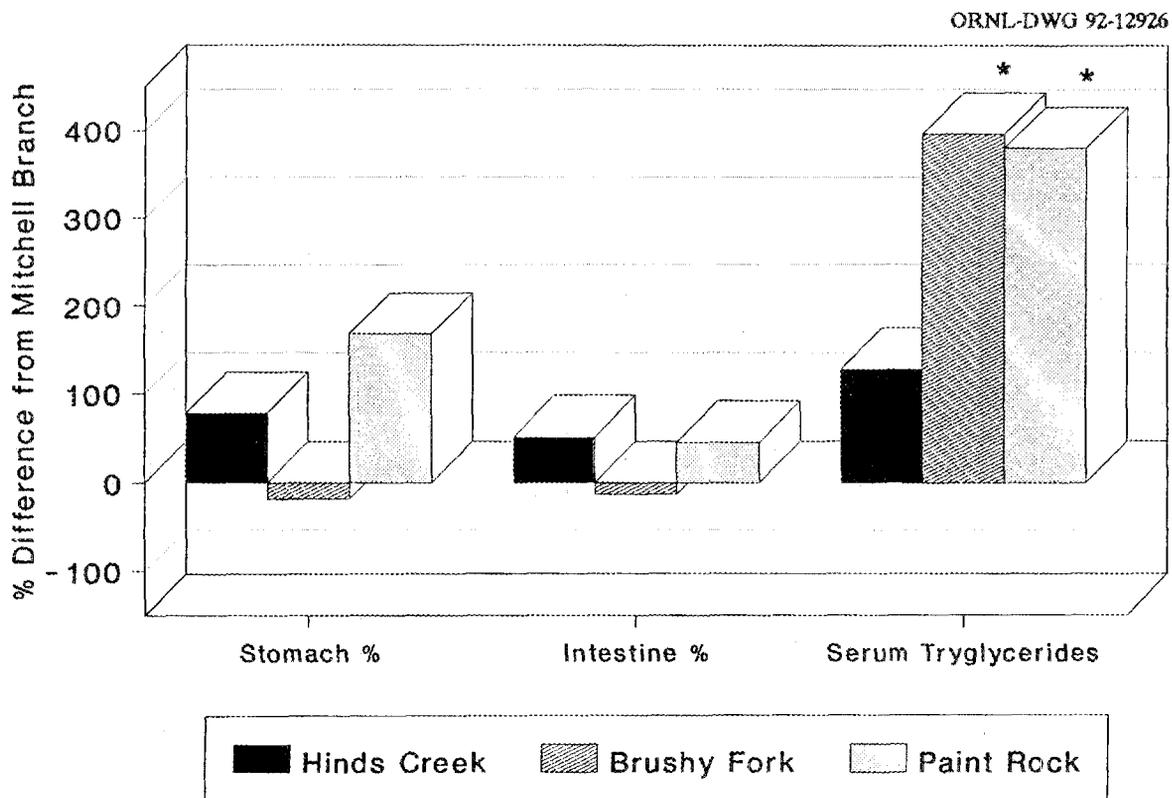


Fig. 5.6. Relative differences in the response of the nutritional indices for redbreast sunfish from each of the reference sites compared with Mitchell Branch (the zero line). Values above or below the zero line indicate that the response for fish from each of the reference streams was higher or lower, respectively, than the same response for fish from Mitchell Branch. Asterisks indicate those values that were significantly different ($p < 0.05$) from Mitchell Branch.

content (food quality) of available food for fish from these reference sites (Fig. 5.6). The higher levels of triglycerides in fish from the reference sites suggest that these fish were in better nutritional condition than fish from Mitchell Branch.

5.3.2 Integrated Bioindicator Analysis

Canonical discriminant analysis can be an informative approach for determining the overall response of fish to stress. This method includes all the bioindicators within a multivariate context and provides a graphical representation of the positions and orientations of the sites relative to

each other. This integrated approach to evaluating the effects of water quality on fish populations has the primary advantage of providing an integrative assessment of the health of fish based on multiple rather than individual indicators.

The most obvious feature of the site responses shown in Fig. 5.7 is that the integrated response of fish from the three reference streams is distinct from that of redbreast from Mitchell Branch. The integrated responses of fish from Brushy Fork and Hinds Creek are not significantly different, as shown by overlapping of their 95% confidence radii ($p < 0.05$). As indicated by the differences in the linear distances between the site means

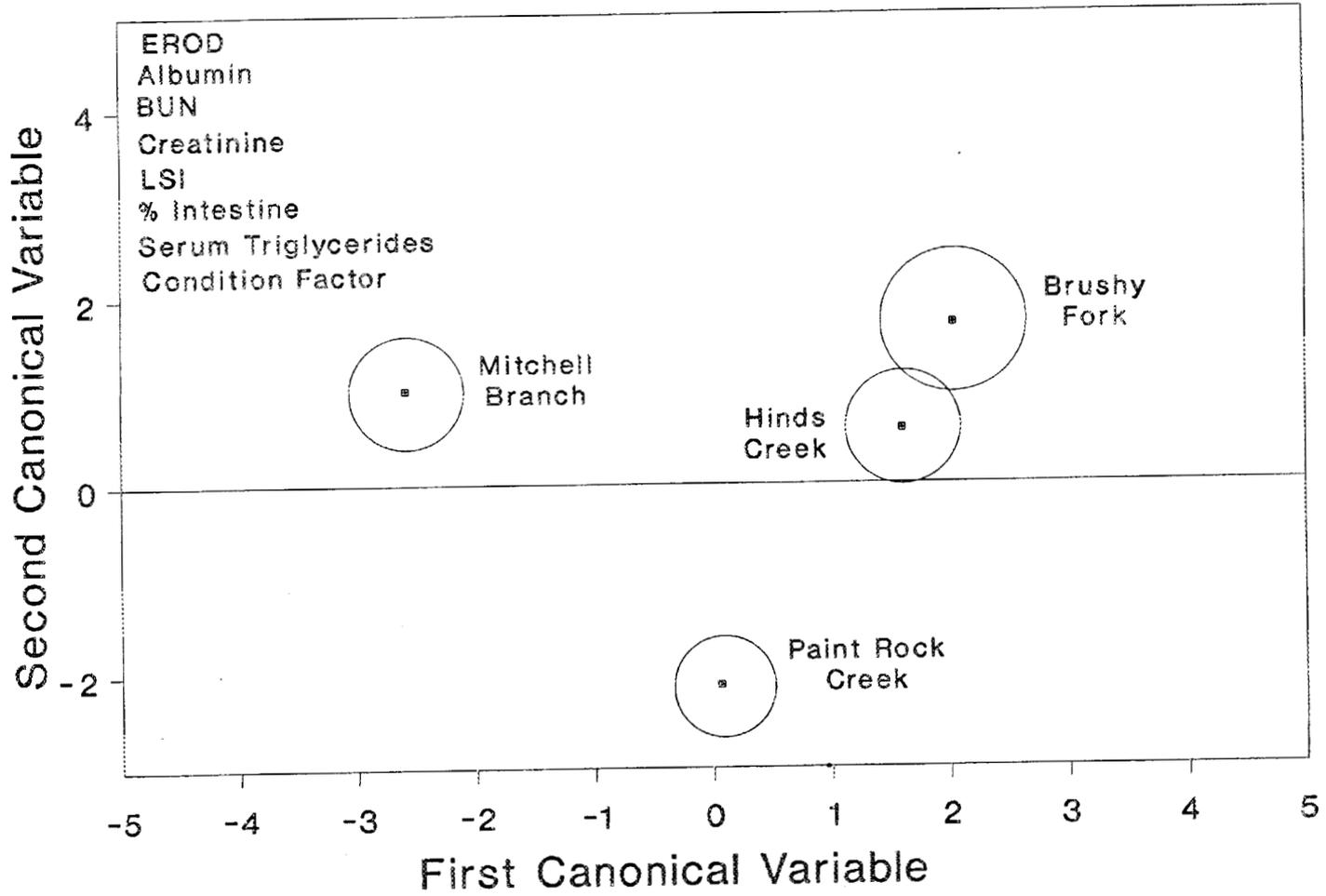


Fig. 5.7. Segregation of integrated health responses for redbreast sunfish from three reference sites and Mitchell Branch. Circles represent site means and the 95% confidence radii of the site means. Also shown are the variables used for discriminating among these sites.

(midpoints of circles), redbreast from Paint Rock Creek were the most similar to redbreast from Mitchell Branch, whereas fish from Brushy Fork were least similar to fish from Mitchell Branch.

Multivariate selection. A multivariate selection procedure was used to identify the individual variables that best discriminated between the health status of fish at the various sites. The variables examined that were the most important in discriminating among sites are indicated on Fig. 5.7. These eight variables consist of representative indicators from four of the five functional groups (see Sect. 5.3), including one detoxification enzyme, three organ dysfunction indicators, two condition indices, and two nutritional indices. This analysis illustrates the importance of including bioindicators at several levels of biological organization when evaluating the integrated responses of fish to environmental stress.

5.3.3 Population-Level Analysis

Length-frequency histograms indicated that the population structure of redbreast sunfish at Mitchell Branch is different from that at the reference sites (Fig. 5.8). Only 25 fish could be collected from Mitchell Branch, all of which were greater than 11 cm in total length. This atypical distribution of sizes in the Mitchell Branch population may indicate a problem with reproduction or recruitment. The habitat in Mitchell Branch appears inadequate for successful spawning. Like most species of sunfish, redbreast prefer sand and gravel over other substrate types, such as silt or detrital material, for nest construction (Davis 1972). However, much of the stream bottom at Mitchell Branch is covered with silt, with accumulations several centimeters deep. Even if hatching were successful, predation by larger fish in the system could limit survival of young.

An alternative explanation for lack of smaller fish at Mitchell Branch might have to do with the immigration of fish from Poplar Creek. The larger fish may swim past or over the weir in times of high water flow. For example, water levels were observed to be 0.5 to 1 m over the top of the K-1700 weir on two occasions during heavy rains in June 1989. The only apparent contradiction to this immigration hypothesis is that in 1989, electrofishing samples below the weir and downstream in Poplar Creek yielded very few redbreast sunfish, none over 10 cm. The small number of redbreasts at Mitchell Branch may be directly related to the size of the sample area. From the weir to the upstream boundary of the sample site is a distance of ~100 m, which may limit the number and biomass of fish this stream section can support in terms of food and habitat resources.

Although 227 fish were collected at Hinds Creek, only 159 were randomly selected for scale analysis. All scale samples from Brushy Fork (91) and Mitchell Branch (25) were analyzed. Fish from 1 to 7 years old were observed from these three streams, but most were from age classes 1 through 3 (Table 5.1). In age classes 4-7, sample sizes were quite small, and statistical comparisons were not possible. This was also true of age 1 fish in Mitchell Branch. Thus, statistical comparisons were made only for growth rates of fish in age classes 2 and 3, although data from other age groups are displayed in Figs. 5.9 through 5.11.

Sunfish from Mitchell Branch were generally larger at each age than those at the reference sites (Figs. 5.9 and 5.10). Because stream area is limited and substrate is not ideal for benthic food production, habitat does not seem to be conducive to favorable growth rates. Lack of competition for resources could be a reason for the larger size and weight of fish at Mitchell Branch. Electrofishing

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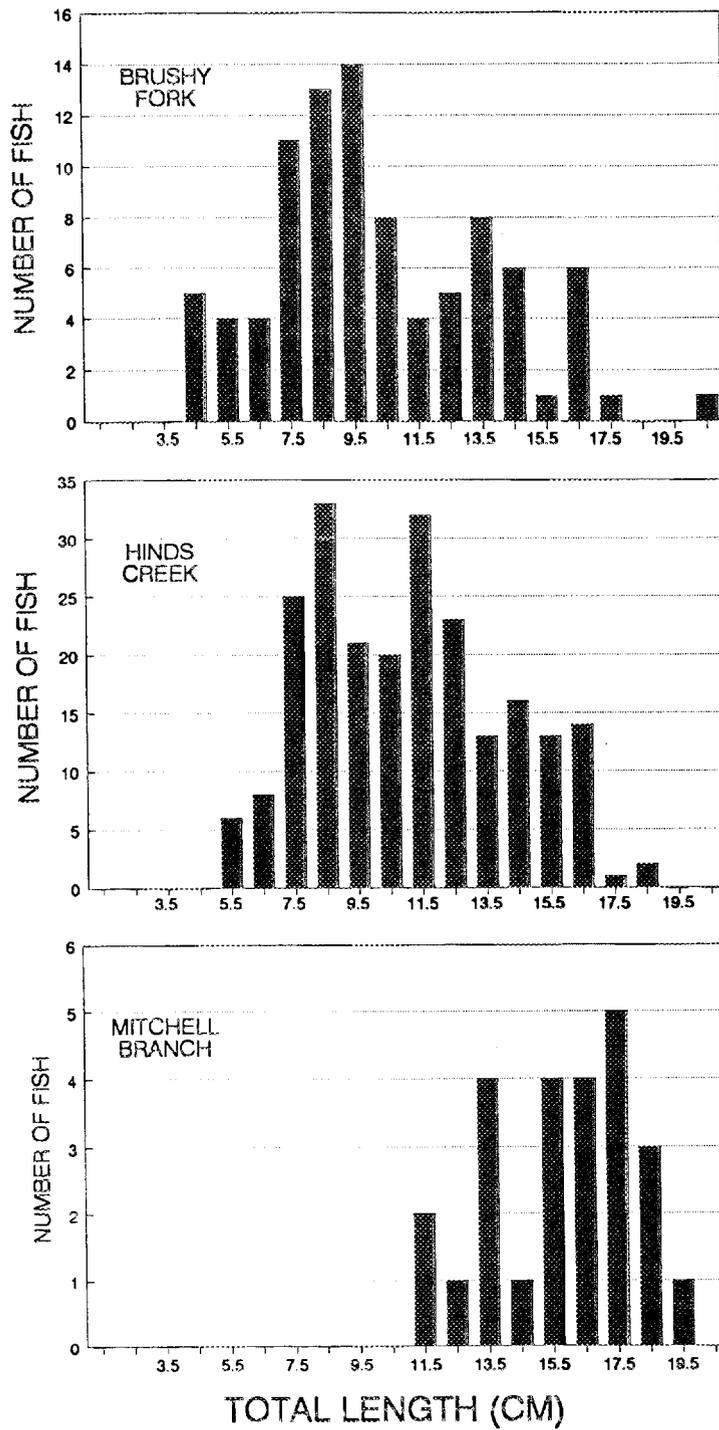


Fig. 5.8. Length frequency of redbreast sunfish collected in Mitchell Branch, Hinds Creek, and Brushy Fork during August 1989. Bars are plotted at the midpoints of 1-cm size groups.

Table 5.1. Number of redbreast sunfish scale samples examined for each age group at each sample site

Sample site	Age	Sample size
Mitchell Branch	1	$n = 2$
	2	$n = 13$
	3	$n = 8$
	4	$n = 1$
	5	$n = 1$
		Total = 25
Brushy Fork	1	$n = 32$
	2	$n = 23$
	3	$n = 28$
	4	$n = 3$
	5	$n = 3$
	6	$n = 2$
		Total = 91
Hinds Creek	1	$n = 38$
	2	$n = 53$
	3	$n = 58$
	4	$n = 6$
	5	$n = 0$
	6	$n = 3$
	7	$n = 1$
		Total = 159

samples revealed that no competing species of fish were present in substantial numbers. Only a few small bluegill and green sunfish were found; therefore, redbreast basically compete with each other for available food. Redbreast sunfish at both reference sites must also compete with bluegill, rockbass, suckers, and several species of *Notropis*. Mitchell Branch redbreast could also have attained their size in the larger Poplar Creek system before immigrating into Mitchell Branch.

The mean instantaneous growth rate (G) of age 1 redbreast sunfish in 1988 was significantly higher ($p < 0.05$) at Mitchell

Branch than at the reference sites (Fig. 5.11). However, no significant difference in the growth rate was found among the reference sites. Age 2 fish in Mitchell Branch grew at a significantly lower rate in 1988 than did fish at either of the reference sites. As for age 1 fish, there were no significant differences in growth rates of age 2 fish among the reference sites. This pattern suggests that in 1988 there could have been unequal availability of food resources for different size fish in Mitchell Branch. The smaller age 1 fish grew well, but the larger fish were apparently less able to benefit from

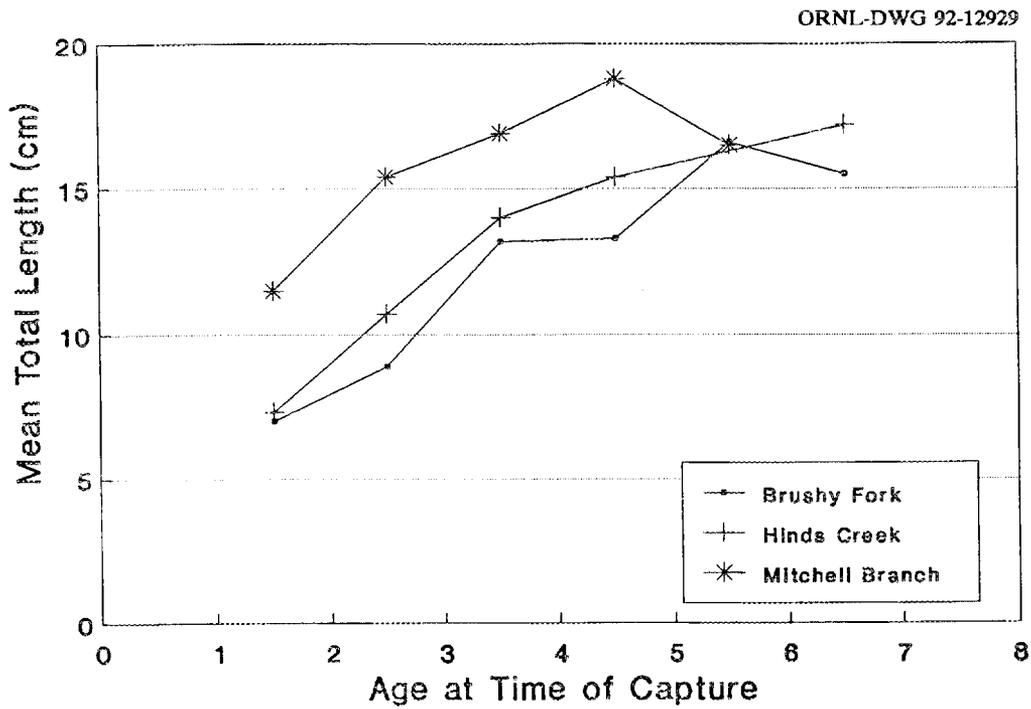


Fig. 5.9. Mean total length of redbreast sunfish in each age group in August 1989 at the three sample sites.

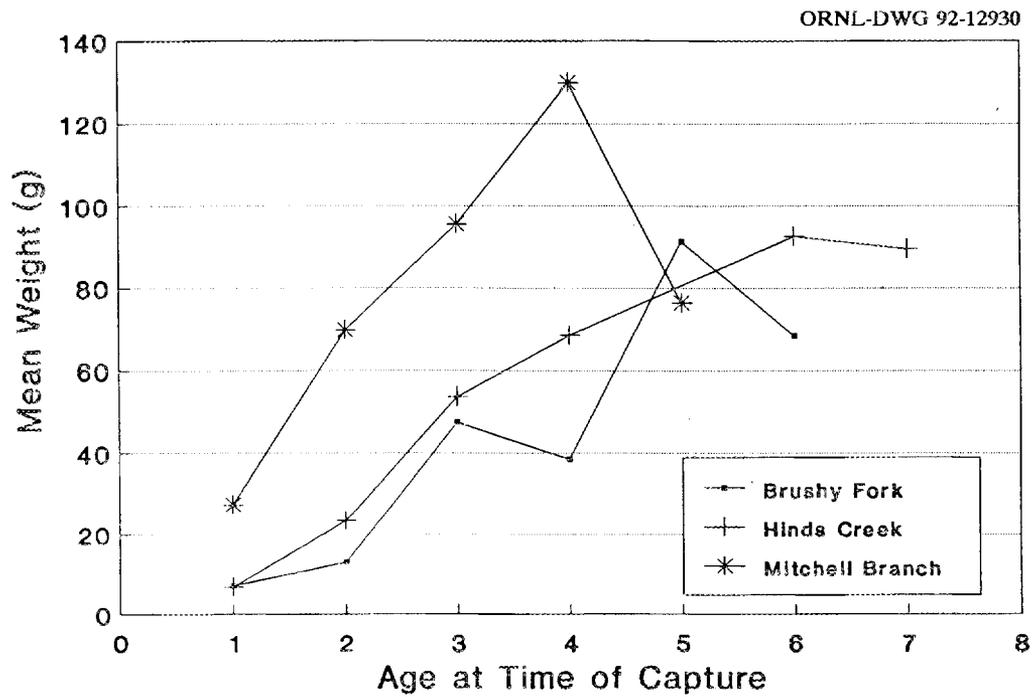


Fig. 5.10. Mean weight of redbreast sunfish in each age group August 1989 at the three sample sites.

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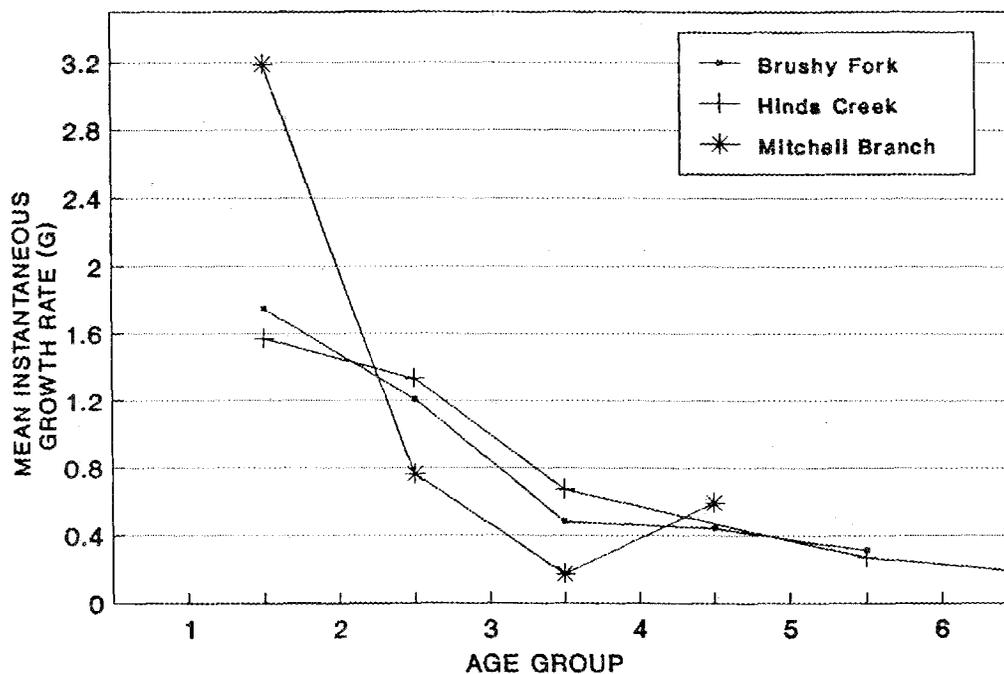


Fig. 5.11. Mean instantaneous growth rate (G) of redbreast sunfish in 1988 at the three sample sites.

the same resources. Because of the need for larger prey items and the limited area of habitat, competition may have been greater between larger fish.

5.3.4 Summary and Synthesis

Several conclusions can be drawn from the comprehensive analysis of the 1989 bioindicator data from Mitchell Branch. Redbreast sunfish in Mitchell Branch appear to be physiologically stressed. On a short-term basis, the effects of contaminant exposure are reflected in gill and possibly liver dysfunction for fish inhabiting Mitchell Branch. Longer-term stress is manifested by changes in histopathological condition and overall condition or health. The reproductive competence of the redbreast sunfish population in Mitchell Branch may

have been compromised, as evidenced by the lack of small individuals in the population. Some of the stress responses may have been the result of the indirect effects of water quality manifested through the food chain.

The integrated response of fish from Mitchell Branch was distinctly different from that of fish from the three reference streams. Although the integrated response was not the same for all three reference sites, they were all dissimilar to Mitchell Branch by approximately the same amount, indicating impaired health of fish in this stream.

When integrated, bioindicators representative of four functional groups permitted Mitchell Branch fish to be statistically separated from fish at the reference sites. This analysis demonstrates the importance of incorporating bioindicators from several levels of biological

organization into biological monitoring programs. Such an integration is needed to effectively evaluate the effects of contaminant-related stress on fish populations.

5.4 FUTURE STUDIES

The annual bioindicator study of fish in Mitchell Branch will continue with one major change initiated during the 1990 sampling period. At that time, reproductive studies were initiated to assess the effects of contaminant stress on reproductive success. Reproductive competence will be evaluated based on measurements of gonad condition (e.g., quality and

quantity of eggs), sizes and timing of clutches, hormone levels, and gonad histology.

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

6. INSTREAM ECOLOGICAL MONITORING

J. G. Smith and M. G. Ryon

The objectives of the instream ecological monitoring task (Task 4 of BMAP as described in Loar et al. 1991b) are to (1) characterize spatial and temporal patterns in the distribution and abundance of the benthic macroinvertebrate and fish populations and (2) document the effects of new pollution abatement facilities on community structure and function. This task consists of two components: (1) benthic macroinvertebrate studies (Subtask 4a) and fish population studies (Subtask 4b); results to date for these studies are presented in Sects. 6.1 and 6.2 respectively.

6.1 BENTHIC MACROINVERTEBRATES (*J. G. Smith*)

6.1.1 Introduction

During the initial year of BMAP (August 1986 through July 1987) the benthic invertebrate community of the middle and lower reaches of Mitchell Branch exhibited characteristics indicative of moderately to severely degraded water quality (Smith et al. 1993). These affected reaches were generally characterized by low taxonomic diversity and richness and the absence or low relative abundance of some major taxonomic groups that are relatively intolerant of pollution (e.g., Ephemeroptera, Plecoptera, and Trichoptera). During the second year of BMAP (August 1987 through July 1988), benthic invertebrate studies continued as in the first year with the specific objectives of (1) further characterizing the benthic macroinvertebrate community in

Mitchell Branch and documenting the impacts on this community from past and current Oak Ridge K-25 Site operations and (2) continuing efforts to identify causal factors responsible for any observed adverse impacts. This report includes results of samples collected from Mitchell Branch from August 1987 through January 1988; thus, in the absence of a complete year of data, conclusions drawn from these data are considered tentative.

6.1.2 Materials and Methods

Benthic macroinvertebrates were sampled at approximately monthly intervals from August 1987 through July 1988 at six sites in Mitchell Branch (see Fig. 2.1); the upstream-most site (MIK 1.43) served as a reference site. Three random quantitative samples were collected from each site with a Surber bottom sampler (0.09 m²) fitted with a 363- μ m-mesh collection net. Each replicate sample was placed in a pre-labeled, glass jar and preserved in 80% ethanol; the ethanol was replaced with fresh ethanol within 1 week of collection. The laboratory procedures used to process these samples are described in Smith et al. (1993).

Supplemental information on water quality and stream characteristics were also recorded at the time of sampling. Temperature, conductivity, dissolved oxygen, pH, and turbidity were measured with an Horiba Model U-7 Water Quality Checker. Water depth, location within the riffle area (distance from permanent head-stakes on the stream bank), relative stream velocity (visually determined as very slow,

slow, moderate or fast), and substrate type based on a modified Wentworth particle size scale (Loar et al. 1985) were recorded for each sample.

All calculations and statistical analyses were performed with the use of the Statistical Analysis System (SAS 1985a, 1985b). The Shannon-Wiener index (H') was used to calculate taxonomic diversity of benthic macroinvertebrates at each site (Pielou 1977):

$$H' = - \sum P_j (\log_2 P_j),$$

where P_j is the proportion of the benthic invertebrate community made up of species j . The H' values of 3 or greater are typical of clean-water communities, whereas values of 1 to 3 are usually associated with moderate pollution and values of less than 1 characterize heavily polluted water (Platts et al. 1983).

Statistical comparisons were performed on transformed data [$\log_{10}(X + 1)$] (Elliott 1977). Mean values for density; biomass; number of taxa (taxonomic richness); number of Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT richness); and taxonomic diversity were compared separately with a two-way ANOVA with site and date as the main effects. Significant site differences ($p < 0.05$) were then separated with a Tukey studentized range test. Between-year comparisons within each site were made with a one-way ANOVA with year as the main effect.

6.1.3 Results

6.1.3.1 Taxonomic composition

A checklist of the benthic invertebrates collected from Mitchell Branch from August 1987 through January 1988 is given in Appendix E, Table E.1. Over 143 taxa, of which 124 were insects, were collected

in quantitative samples from Mitchell Branch. Ten orders of insects were collected, including Collembolla (spring-tails), Ephemeroptera (mayflies), Odonata (dragonflies and damselflies), Plecoptera (stoneflies), Hemiptera (true bugs), Megaloptera (alderflies and fishflies), Trichoptera (caddisflies), Coleoptera (beetles), Hymenoptera (wasps), and Diptera (true flies). The order Diptera was the most taxonomically rich group, with 77 representative taxa, of which 55 were from the family Chironomidae. The remaining insect orders were represented by nine or fewer taxa. Several insect taxa were not collected from sites downstream of the reference site (MIK 1.43). Particularly notable at these downstream sites was the absence or rare occurrence of most or all representative taxa that are generally intolerant of pollution, such as the Ephemeroptera, Plecoptera, and Trichoptera.

The noninsect taxa collected included Turbellaria (flatworms or planarians), Nematoda (roundworms), Oligochaeta (aquatic earthworms), Copepoda (copepods), Ostracoda (seed shrimp), Isopoda (aquatic sow bugs), Amphipoda (sideswimmers), Decapoda (crayfish), Hydracarina (water mites), Gastropoda (snails), and Bivalvia (mussels and clams) (Table E.1). As for the insects, several noninsect taxa (e.g., Isopoda, Amphipoda, Bivalvia) collected at MIK 1.43 were not collected at those sites further downstream.

6.1.3.2 Density and biomass

Mean density and biomass of the benthic macroinvertebrates at each sampling site in Mitchell Branch during the first 6 months of the second year of BMAP are presented in Table 6.1. Both density and biomass were significantly higher at MIK 1.43 than at each downstream

Table 6.1. Benthic macroinvertebrate density, biomass, taxonomic richness, EPT richness, and taxonomic diversity in Mitchell Branch, August 1986–January 1987 (Year 1) and August 1987–January 1988 (Year 2)^a

Site	Density (number/0.1 m ²)	Biomass (mg/0.1 m ² wet wt)	Richness (taxa/sample)	EPT ^b Richness (taxa/sample)	Diversity (H')
MIK 1.43					
Year 1	88.7 ± 26.9 ^c	138.1 ± 38.9 ^d	20.1 ± 3.6 ^c	3.8 ± 0.7 ^c	3.5 ± 0.3 ^d
Year 2	298.5 ± 63.5	267.6 ± 70.7	32.3 ± 3.0	5.6 ± 0.7	3.8 ± 0.1
MIK 0.86					
Year 1	34.9 ± 13.4	310.6 ± 165.0 ^e	5.7 ± 1.1	0.2 ± 0.1	1.6 ± 0.1
Year 2	28.2 ± 5.8	30.5 ± 12.0	7.4 ± 1.0	0.4 ± 0.2	2.0 ± 0.2
MIK 0.78					
Year 1	30.6 ± 8.2	21.5 ± 6.0	5.4 ± 1.0	0.3 ± 0.1	1.5 ± 0.2
Year 2	46.6 ± 22.1	18.7 ± 7.9	6.3 ± 1.2	0	1.6 ± 0.3
MIK 0.71					
Year 1	1.1 ± 0.3 ^f	2.3 ± 1.4	0.6 ± 0.1 ^c	0	0 ^e
Year 2	10.5 ± 4.5	2.8 ± 1.4	2.3 ± 0.5	0	0.8 ± 0.3
MIK 0.54					
Year 1	30.9 ± 12.9 ^d	37.0 ± 18.6	2.2 ± 0.4 ^c	0	0.7 ± 0.2 ^e
Year 2	105.8 ± 48.1	69.8 ± 16.6	6.6 ± 1.6	0	1.4 ± 0.3
MIK 0.45					
Year 1	21.2 ± 7.4 ^c	89.1 ± 38.7 ^d	1.9 ± 0.2 ^c	0.1 ± 0.1	0.5 ± 0.1
Year 2	270.2 ± 57.0	146.0 ± 47.3	7.2 ± 1.1	0.1 ± 0.1	0.6 ± 0.1

^aValues are the means ± 1 SE, with $n = 18$

^bEphemeroptera, Plecoptera, and Trichoptera.

^cSignificant between-year difference, $p < 0.0001$.

^dSignificant between-year difference, $p < 0.05$.

^eSignificant between-year difference, $p < 0.01$.

^fSignificant between-year difference, $p < 0.001$.

Note: MIK = Mitchell Branch kilometer.

site except for MIK 0.45 (Table 6.2). Although biomass at MIK 0.45 did not differ statistically from biomass at MIK 1.43, the biomass at MIK 1.43 was almost two times greater. MIK 0.71 exhibited the greatest difference in density and biomass from MIK 1.43; at MIK 0.71

density and biomass were about 28 times and 96 times lower respectively. Densities at MIKs 0.54 and 0.45 were considerably higher than at MIKs 0.86 and 0.78; however, only the density at MIK 0.45 was significantly higher than at MIKs 0.86 and 0.78.

Table 6.2. Statistical comparisons of mean benthic macroinvertebrate density, biomass, taxonomic richness (number of taxa), EPT richness, and taxonomic diversity for Mitchell Branch, August 1987-January 1988

Parameter/site ^a					
Density					
MIK 1.43	MIK 0.45	<u>MIK 0.54</u>	<u>MIK 0.78</u>	<u>MIK 0.86</u>	<u>MIK 0.71</u>
Biomass					
<u>MIK 1.43</u>	<u>MIK 0.45</u>	<u>MIK 0.54</u>	<u>MIK 0.86</u>	MIK 0.78	<u>MIK 0.71</u>
Richness					
<u>MIK 1.43</u>	<u>MIK 0.86</u>	MIK 0.45	<u>MIK 0.54</u>	<u>MIK 0.78</u>	<u>MIK 0.71</u>
EPT richness ^b					
		<u>MIK 1.43</u>	<u>MIK 0.86</u>	<u>MIK 0.45</u>	
Diversity					
<u>MIK 1.43</u>	<u>MIK 0.86</u>	<u>MIK 0.78</u>	<u>MIK 0.54</u>	<u>MIK 0.71</u>	<u>MIK 0.45</u>

^aSites are arranged in order from highest to lowest values from left to right. Sites not joined by lines are significantly different ($p < 0.05$), based on Tukey's studentized range test (HSD). $n = 18$ for each site.

^bNo EPT taxa were collected from MIKs 0.78, 0.71, and 0.54 in quantitative samples.

Density and biomass varied considerably between sampling periods at most sites, particularly MIKs 1.43, 0.45, and 0.54 (Figs. 6.1 and 6.2). Few similarities in patterns of temporal change were noted among sites; this condition reflected between-site differences in taxonomic composition and community structure.

Between-year (August through January only) comparisons within each site revealed that, with few exceptions, values for density and biomass were higher during the second year than in the first year of BMAP (Table 6.1). These differences were statistically significant at all sites, except for density at MIKs 0.86 and 0.78

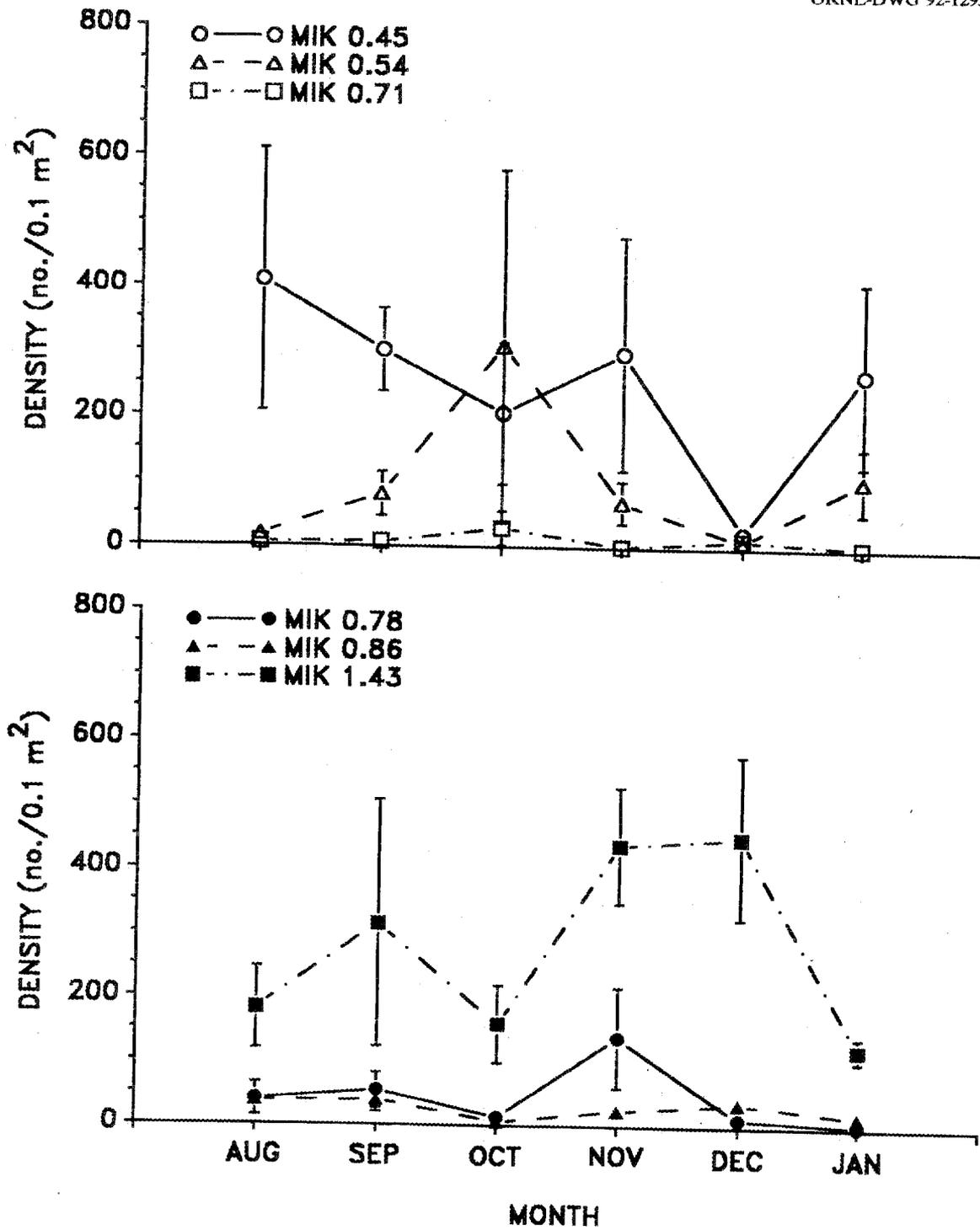


Fig. 6.1. Mean monthly density of benthic macroinvertebrates in Mitchell Branch, August 1987-January 1988. Values are the mean \pm 1 SE.

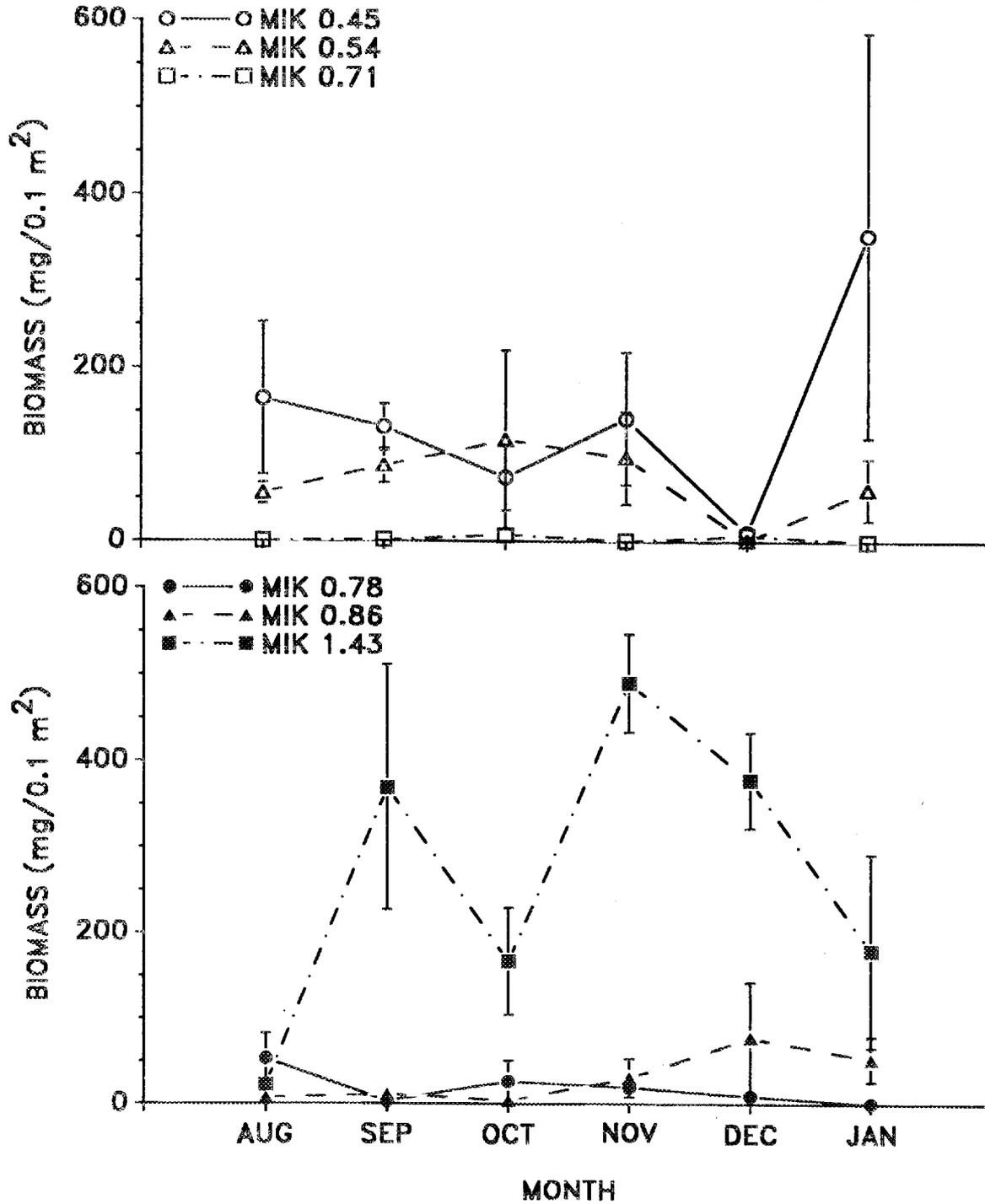


Fig. 6.2. Mean monthly biomass of benthic macroinvertebrates in Mitchell Branch, August 1987–January 1988. Values are the mean \pm 1 SE.

and biomass at MIKs 0.78, 0.71, and 0.54 (Table 6.1). The most dramatic change in biomass occurred at MIK 0.86; the most dramatic change in density occurred at MIK 0.45. Biomass can be greatly influenced by the occasional collection of a few large taxa such as Decapoda (crayfish) and Tipulidae (Diptera: crane flies). The significant reduction in biomass at MIK 0.86 during the second year, with only a minor change in density, shows that large taxa were either absent or collected infrequently. The large increase in density at MIK 0.45, coupled with a slight change in biomass, was probably the result of (1) higher densities of small taxa (e.g., oligochaetes and chironomids) that add little to the biomass and (2) the infrequent occurrence of larger taxa (e.g., Decapoda and Tipulidae).

6.1.3.3 Dominant taxa*

Considerable spatial and temporal differences were observed in taxonomic composition of the benthic community in Mitchell Branch. Insight into the possible reasons for these differences may be gained from considering the dominant taxonomic groups because these taxa usually account for most of the between-site differences.

The most dominant taxa at all sites in Mitchell Branch were oligochaetes and chironomids; these two taxa combined for more than 80% of the total density at all sites, except for MIKs 1.43 and 0.78, where they accounted for more than 60% of the total density (Table 6.3). Oligochaetes

became more dominant with distance downstream from MIK 1.43, whereas the percentage of chironomids tended to decrease with distance downstream from MIK 1.43. In addition to their numerical dominance, these two groups also contributed considerably to the total biomass at all sites; their combined contribution to biomass was greater than 40% at all sites except for MIK 1.43.

Other taxa also contributed substantially to either the density or biomass of the invertebrate community at some sites, or some taxa [e.g., Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa] provided useful information for assessing the water quality of the stream because of their overall sensitivity to changes. Dipterans (excluding Chironomidae) were moderately abundant at most sites, accounting for up to 7.9% of the total density and up to 22.2% of the total biomass (Table 6.3). EPT taxa comprised a substantial part of the total density and/or biomass at MIK 1.43 only (Table 6.3). These taxa were not collected in quantitative samples from MIKs 0.78, 0.71, or 0.54, and they contributed very little to either density or biomass at MIKs 0.86 and 0.45.

6.1.3.4 Community structure

Taxonomic richness. Spatial patterns in taxonomic richness were similar to those observed for density and biomass (Table 6.1). The site having the greatest number of taxa per sample was MIK 1.43, where richness was more than four times greater than at any of the other sites. The site with the fewest number of taxa per sample was MIK 0.71, where there was a 15-fold difference from that for MIK 1.43. Richness at the remaining four sites was similar and statistically indistinguishable, whereas richness at MIKs 1.43 and 0.71 was significantly greater and less,

*For the benthic macroinvertebrate section of this report, the term dominant is used synonymously with numerical dominance. Taxa were considered numerically dominant if they were collected at 50% or more of the study sites and comprised $\geq 10\%$ of the average density at two or more sites.

Table 6.3. Relative density (mean percentage \pm 1 SE) and biomass (mean percentage \pm 1 SE) of selected macroinvertebrate taxa in Mitchell Branch, August 1987-January 1988

Taxon	Site ^a					
	MIK 1.43	MIK 0.85	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Oligochaeta						
Density	14.8 \pm 3.7	23.5 \pm 8.4	33.8 \pm 10.6	46.4 \pm 13.6	59.4 \pm 8.5	89.3 \pm 2.6
Biomass	3.4 \pm 0.7	18.3 \pm 9.1	32.6 \pm 14.0	39.5 \pm 12.0	62.9 \pm 6.7	85.2 \pm 3.9
Chironomidae						
Density	48.0 \pm 7.1	60.6 \pm 11.3	32.3 \pm 11.4	35.9 \pm 14.7	23.3 \pm 8.1	6.4 \pm 2.7
Biomass	8.9 \pm 4.9	25.1 \pm 7.5	16.1 \pm 11.6	35.1 \pm 13.9	7.6 \pm 4.8	2.1 \pm 1.0
Diptera ^b						
Density	4.6 \pm 0.6	5.9 \pm 1.7	2.4 \pm 1.1	7.9 \pm 4.2	6.3 \pm 2.5	1.8 \pm 0.7
Biomass	12.6 \pm 6.1	22.2 \pm 9.9	5.3 \pm 3.2	13.9 \pm 8.0	7.5 \pm 3.5	0.7 \pm 0.3
Ephemeroptera						
Density	6.0 \pm 1.0	1.2 \pm 0.8	0	0	0	0
Biomass	4.0 \pm 1.2	1.8 \pm 1.6	0	0	0	0
Plecoptera						
Density	12.4 \pm 5.5	0.4 \pm 0.2	0	0	0	0
Biomass	5.0 \pm 2.2	3.4 \pm 3.4	0	0	0	0
Trichoptera						
Density	1.6 \pm 0.6	0.1 \pm 0.1	0	0	0	<0.1
Biomass	14.2 \pm 5.6	<0.1	0	0	0	<0.1

^aMIK = Mitchell Branch kilometer.

^bExcludes Chironomidae.

respectively, than at all other sites (Table 6.2).

Most of the sites exhibited temporal patterns in richness similar to those of density and biomass, with few similarities between sites (Fig. 6.3). One striking difference between MIK 1.43 and the

downstream sites was that richness values of less than 20 taxa per sample were never observed at MIK 1.43, whereas at the remaining 5 sites, richness never exceeded 12 taxa per sample during any sampling period and was usually less than 10 taxa per sample. Furthermore, variability

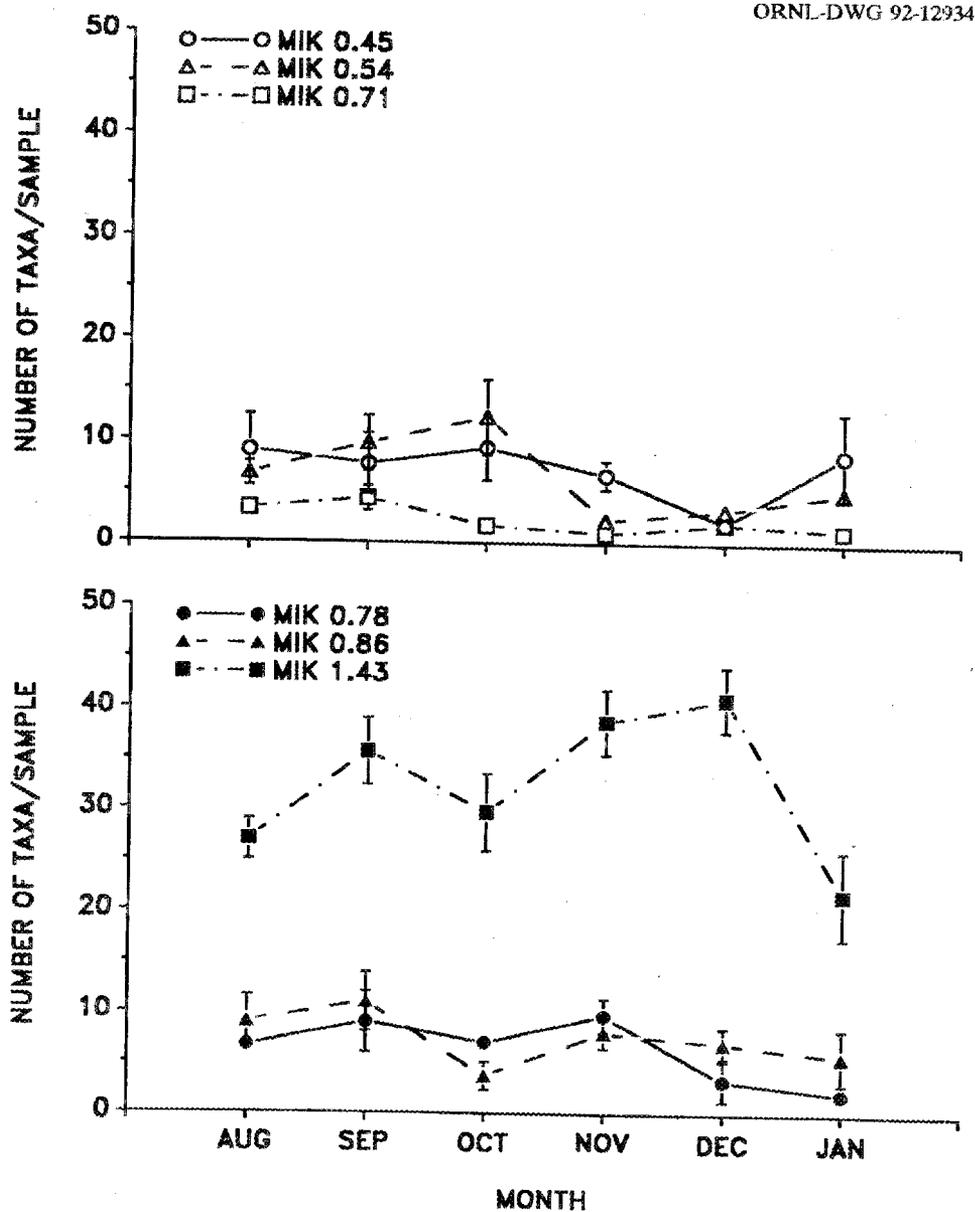


Fig. 6.3. Mean monthly taxonomic richness of benthic macroinvertebrates in Mitchell Branch, August 1987-January 1988. Values are the mean \pm 1 SE.

between replicate samples was generally much greater at MIK 1.43 than at the other sites during each sampling period, which may reflect a more complex community structure at this site (i.e., variability between replicates should decrease as the number of taxa that can potentially be collected decreases).

Comparisons between years showed that richness values at all sites were higher during the second year of the monitoring program than they were during the first year (Table 6.1). These differences were very highly significant ($p < 0.0001$) at all sites except MIKs 0.86 and 0.78, where the values did not differ significantly between years.

EPT richness. During the second year EPT taxa were not obtained in quantitative samples from MIKs 0.78, 0.71, and 0.54 (Table 6.1). EPT taxa were rare at MIKs 0.45 and 0.86, where they were collected in only 2 and 4 of the 6 months, respectively (Fig. 6.4), while at MIK 1.43 the number of EPT taxa per sample was never less than three and exceeded five in 4 of 6 months. Statistical comparison of the three sites having EPT taxa showed that there were significantly more EPT taxa at MIK 1.43 than at the two downstream sites and that the number of EPT taxa at the two downstream sites did not differ (Table 6.2). Between-year differences were minimal in EPT richness at all

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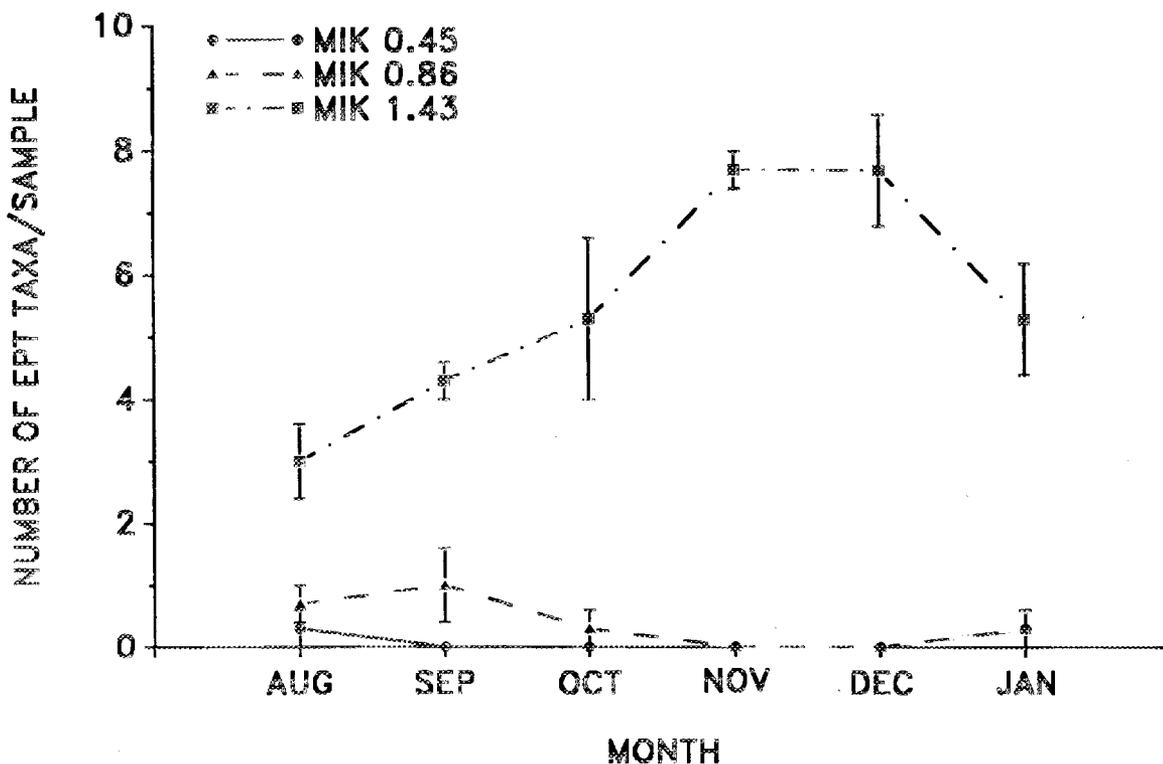


Fig. 6.4. Mean monthly richness of Ephemeroptera, Plecoptera, and Trichoptera (EPT richness) in Mitchell Branch, August 1987–January 1988. Values are the mean ± 1 SE.

sites except MIK 1.43, where the value for the second year was significantly greater than for the first year (Table 6.1).

Diversity. Spatial patterns in taxonomic diversity were similar to those observed for density, biomass, and taxonomic richness, with the highest value obtained from MIK 1.43 and the lower values from the downstream sites (Tables 6.1 and 6.2). Diversity was significantly greater at MIK 1.43 than at all other sites, whereas differences in diversity at MIKs 0.86, 0.78, and 0.54 were not significant. Unlike the other parameters, the lowest mean value for diversity was found at MIK 0.45; however, mean diversity at this site did not differ significantly from mean diversity at MIK 0.71.

Temporally, MIK 1.43 exhibited less dramatic fluctuations in diversity than the other sites, and never fell below 3.0 (Fig. 6.5). At the other sites, however, diversity exhibited more dramatic monthly fluctuations and generally remained at 2.0 or less. All sites exhibited increases in diversity between years, but the increases were significant at only MIKs 1.43 and 0.54 (Table 6.1).

6.1.4 Discussion

The benthic invertebrate community of Mitchell Branch within the boundaries of the Oak Ridge K-25 Site continued to show evidence of severely degraded water quality during the first 6 months of the second year of BMAP. Relative to the reference site in upper Mitchell Branch (MIK 1.43), density, biomass, taxonomic richness, EPT richness, and taxonomic diversity were all substantially lower at the five downstream sites. The most notable differences between MIK 1.43 and the downstream sites were in taxonomic and EPT richness. Taxonomic richness was at least four times higher at MIK 1.43 than at any of the other sites, and EPT taxa, which

generally tend to be sensitive to changes in water quality (Wiederholm 1984), were either absent or very rarely collected at the sites downstream of MIK 1.43.

As was found in the first year of BMAP (Smith et al. 1993), the most severely affected site in Mitchell Branch was MIK 0.71. Values for density, biomass, taxonomic richness, and EPT richness were all significantly lower at this site than at all other sites. Also, no EPT taxa were collected from this site, and the number of taxa collected per sample (richness) was at least 2.7 times less than at any of the other sites.

The extent of impact on the benthic invertebrate community at the remaining four affected sites (MIKs 0.86, 0.78, 0.54, and 0.45) was somewhat less than at MIK 0.71. Although differences in the invertebrate community existed between these four sites (e.g., differences in relative density and biomass of the dominant taxa), differences in the extent of impact were not readily discernable. Relative to other affected streams on the ORR, the benthic community at these sites clearly reflected highly stressed conditions. For example, in EFPC (Smith 1992a) and in streams in the White Oak Creek watershed (Smith 1992b), the most highly stressed benthic communities are usually made up almost exclusively of chironomids and/or oligochaetes, whereas less-pollution-tolerant taxa such as Ephemeroptera and Trichoptera are rare or absent. As water quality improves in these streams with distance downstream of effluent discharges, the relative abundance of the chironomids and oligochaetes decreases as the relative abundance of Ephemeroptera and Trichoptera increases. In relatively unaffected sites, the latter two taxonomic groups and Plecoptera generally contribute substantially to the total density. In Mitchell Branch, EPT taxa were rare or absent from all sites except for MIK 1.43, and the chironomids and oligochaetes

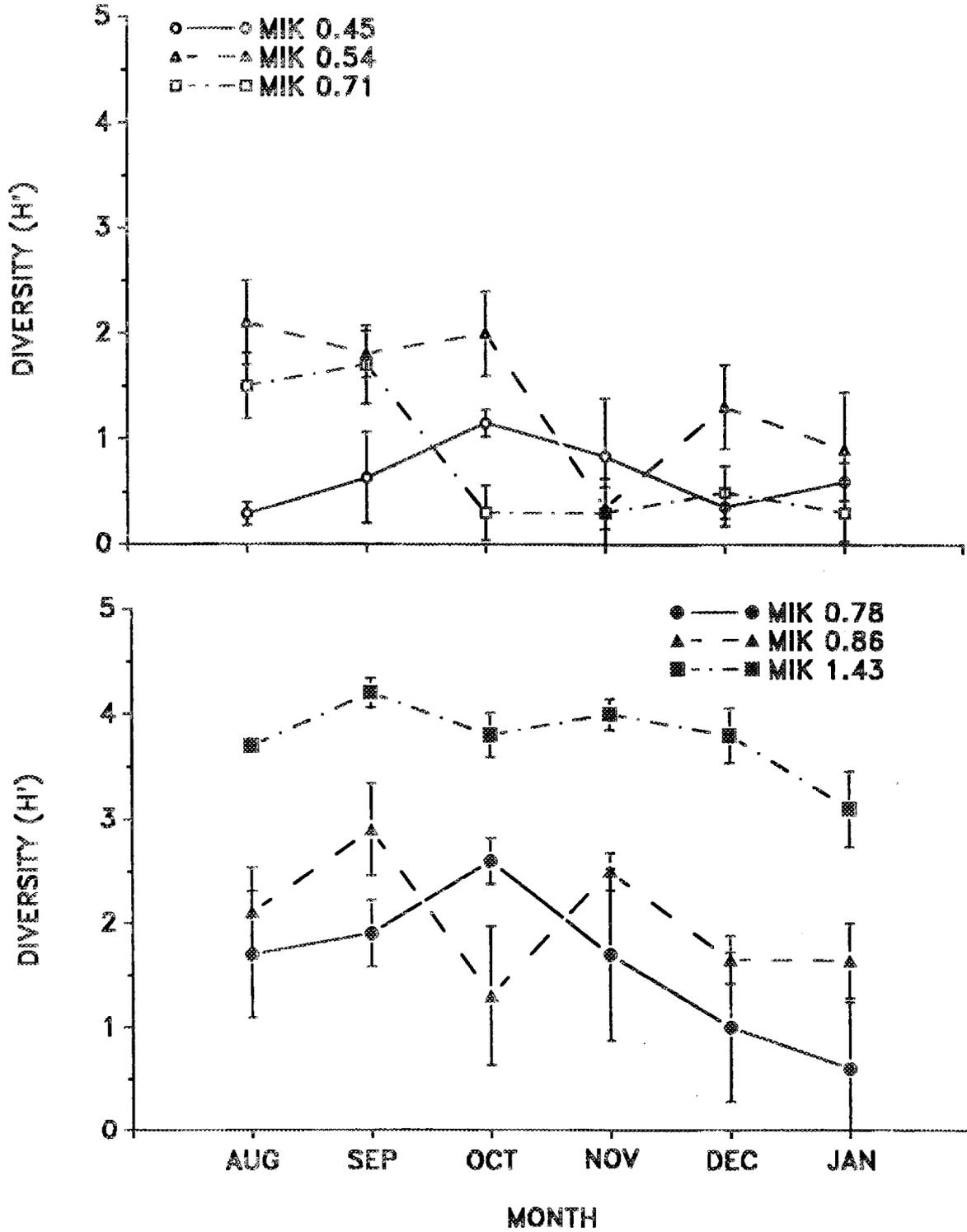


Fig. 6.5. Mean monthly taxonomic diversity (H') of benthic macroinvertebrates in Mitchell Branch, August 1987-January 1988. Values are the mean ± 1 SE.

accounted for over 65% of the total density at all affected sites.

Between-year changes in the benthic community of each site were examined for trends indicative of recovery. Most sites exhibited increases in density, biomass, taxonomic richness, and taxonomic diversity, and in many cases, these increases were statistically significant. EPT richness, on the other hand, changed little between years at any site except for MIK 1.43, where the value was significantly higher during the second year.

Benthic invertebrate communities exhibit natural year-to-year changes in composition, richness, relative abundance of taxa, etc. (e.g., McElravy et al. 1989). However, numerous factors including the biota present, the location and types of effluent discharges, the extent and type of streamside vegetation, substrate composition, and other abiotic factors make it unlikely that even sites as close to each other as those in Mitchell Branch would exhibit similar changes between years (e.g., substantial increases at most sites in density and taxonomic richness). Thus, the magnitude of most of the observed between-year changes in Mitchell Branch was probably not the result of natural variability alone, but to other factors, such as improving or worsening conditions, and/or between-year differences in laboratory sample processing efficiency (i.e., the percentage of organisms removed from a sample). A preliminary analysis of some quality assurance checks of sample processing efficiency of JAYCOR personnel indicated that processing efficiency has improved from consistently <70% by the Tennessee Valley Authority, the former processing subcontractor, to consistently >90% by JAYCOR, the current processing subcontractor (J. G. Smith, ESD, ORNL, unpublished data). Thus, at least some of the between-year differences observed within each site may have been

the result of differences in sample processing efficiency.

Even though processing efficiency of the samples differed between years, this difference did not appear to have much effect on distinguishing spatial trends (i.e., similar spatial trends were evident in Mitchell Branch during both years). However, differences in processing efficiency did reduce the ability to detect between-year differences within sites, which reduced the ability to detect possible recovery. In an attempt to identify changes that may be indicative of recovery, the data for each affected site in each year were first "normalized" to the reference site by comparing each parameter value (i.e., density, biomass, taxonomic richness, and EPT richness) from each affected site with that of MIK 1.43 and then dividing the largest of the two values (i.e., affected vs reference site) by the smallest value to obtain a difference factor between the sites. For illustration, the resulting values were designated as negative when the parameter value was highest at the affected site and positive when the parameter value was highest at the reference site. Assuming that (1) processing efficiency had no effect on the magnitude of difference between sites within each year and (2) the reference site remained relatively undisturbed, the magnitude of change between years was then determined for each parameter by subtracting the difference factor for the second year (August 1987-January 1988) from the difference factor for the first year (August 1986-January 1987).

Thus, for a site that exhibited no between-year change relative to its reference site, the resulting value was zero, whereas a positive or negative change between years could be indicative of improving or worsening conditions respectively. For example, in the first year mean density at MIKs 0.86 and 1.43, the reference site, was 34.9 and 88.7 individuals per

0.1 m², respectively, giving a difference factor of 2.5. In the second year, mean densities at these sites were 28.2 and 298.5 individuals per 0.1 m², respectively, giving a difference factor of ~10.6. The magnitude of change between the two years in density at MIK 0.86 would then be the first year difference factor (2.5) minus the second year difference factor (10.6), which would equal -8.1. This value would indicate that, relative to the reference site, density declined at MIK 0.86 during the second year, which may be indicative of worsening conditions.

As would be expected under natural conditions, between-year changes occurred in most parameters at most sites (Fig. 6.6). Differences in parameter values between the reference site and MIKs 0.86 and 0.78 tended to be greater during the second than the first year, whereas the trend was generally just the opposite at MIKs 0.54 and 0.45. However, the magnitude of between-year changes at these four affected sites tended to be minor (i.e., magnitude of change <10). Thus, the data suggest that these sites changed little between years and that these changes could be, at least in part, the results of natural variability. The most dramatic between-year changes occurred at MIK 0.71. This site showed substantial increases in both density and taxonomic richness (magnitude of change >20) and a substantial decline in biomass (magnitude of change of 35.6). Although the magnitude of change in density and taxonomic richness implies that improvements may have occurred at this site since the first year of BMAP, records on sample processing efficiency do not support this. Records on processing efficiency of samples indicated that in most cases lower efficiency was associated with samples having low numbers of organisms (<100) (J. G. Smith, ESD, ORNL, unpublished data). This was probably because most of the organisms collected at this site are very

small (i.e., chironomids) and thus more easily overlooked. For a site such as MIK 0.71, where very low numbers of organisms were collected (~10 individuals per sample), differences in sample processing efficiency could result in substantially greater magnitudes of change between years than for a site having much higher numbers of organisms.

No substantial changes were evident in the benthic community between the first 6 months of the first and second years of BMAP. This finding indicates that neither significant improvements nor further degradation have occurred in Mitchell Branch during this period. As hypothesized in the first report on Mitchell Branch, siltation is probably one of the main perturbations, particularly at MIKs 0.86 and 0.78, which are upstream of all major effluent discharges. Construction activities along Mitchell Branch have resulted in the removal of riparian vegetation and the exposure of surface soils, both of which can contribute to erosion and thus increase the silt load in the stream. Removal of riparian vegetation also increases the rate of surface water runoff, which in turn affects stream discharge during periods of rain and further reduces habitat stability within the stream. A substratum dominated by mud and silt tends to favor oligochaetes (Whitley 1982). The substratum in Mitchell Branch becomes increasingly covered with and/or dominated by silt with increasing distance from the headwaters, which may partially explain why oligochaetes become increasingly dominant with distance downstream.

Although siltation appears to have an important impact on the lower three sites in Mitchell Branch, the structure and composition of the benthic invertebrate community indicate the presence of toxicants in addition. This was particularly evident at MIK 0.71, where total density, biomass, and taxonomic richness were all very low, a condition characteristic of

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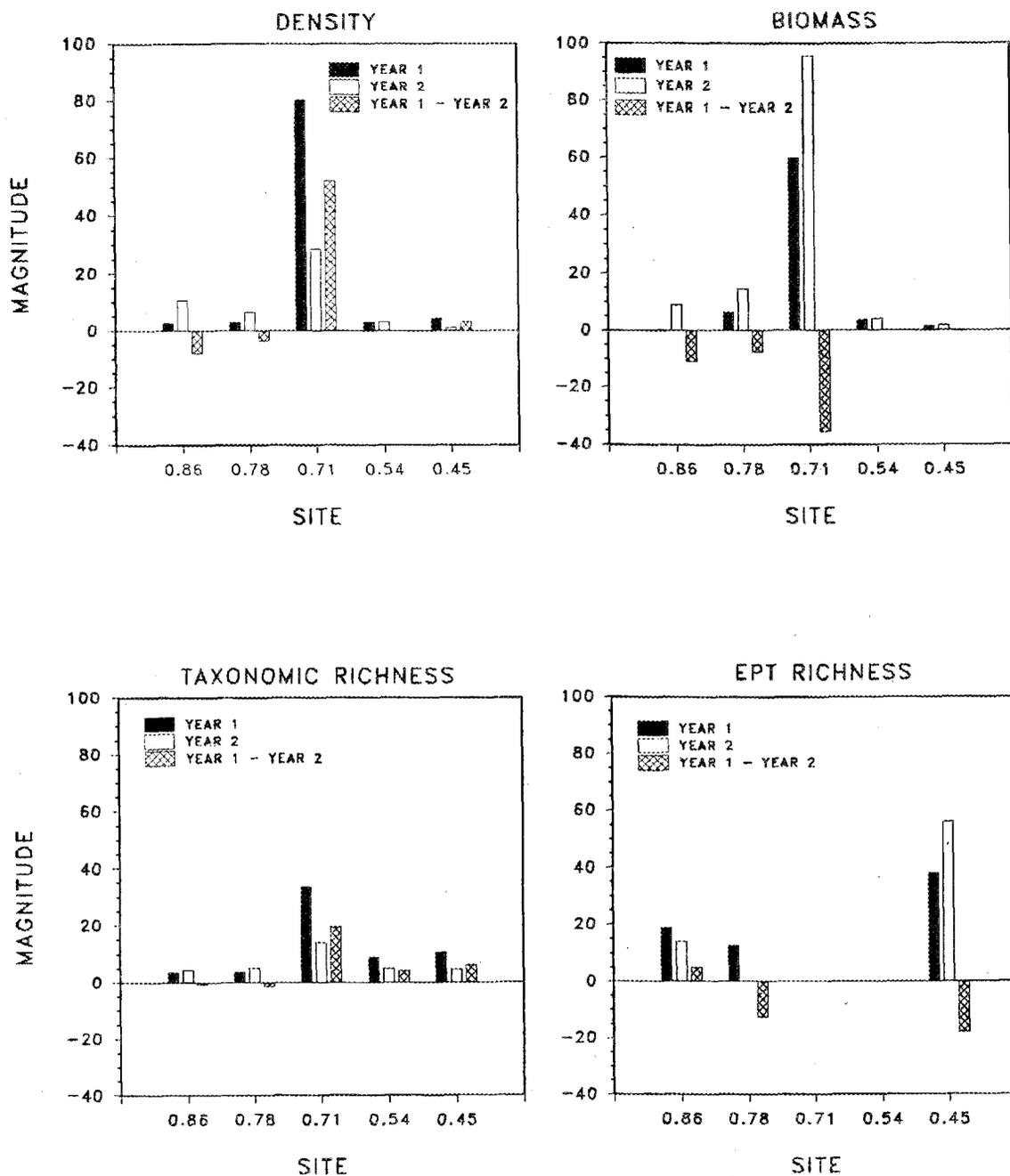


Fig. 6.6. Difference factor between each downstream site and the reference site (Mitchell Branch kilometer 1.43) (Year 1, August 1986–January 1987; Year 2, August 1987–January 1988) and the magnitude of change within each site between Year 1 and Year 2 (crosshatched bars) for mean density, biomass, taxonomic richness, and richness of Ephemeroptera, Plecoptera, and Trichoptera (EPT richness) in Mitchell Branch.

benthic communities exposed to toxicants (e.g., Wiederholm 1984). Based on the results obtained in the toxicity monitoring program (Sect. 3; see also Sect. 3 in Smith et al. 1993), chlorine is the most likely toxicant affecting the invertebrate community at these sites, originating primarily from SD 170. The dominance of taxa that are capable of producing several generations per year (i.e., oligochaetes and chironomids), suggests that the benthic community at these sites is probably exposed intermittently to the toxicant(s). These taxa probably have resistant (e.g., egg) or unexposed (i.e., terrestrial adult) stages in the stream at most times. Thus, between toxic releases such organisms are capable of rapid recolonization. This "intermittent release" hypothesis is supported by water quality data collected concurrently with the toxicity monitoring program, which found that high concentrations of chlorine occurred periodically (Sect. 3.2.3).

Other factors (e.g., elevated temperatures, flow augmentation, and elevated concentrations of metals and organics) identified and discussed in the first report continue to be potential perturbations (Smith et al. 1993), and reductions in *Ceriodaphnia* fecundity at all sites downstream of MIK 1.0 provide evidence of chronic toxicity (Sect. 3.2). However, it is not currently possible to determine the extent of influence of these factors from the available data.

6.1.5 Conclusions

The structure and composition of the benthic invertebrate community in Mitchell Branch within the boundary of the Oak Ridge K-25 Site continue to indicate that water quality in this portion of the stream is severely degraded. Maximum impact occurs just downstream of SD 170, but within a short distance minor improve-

ment is evident; thus, SD 170 clearly continues to be the major source of impact.

Several factors are probably involved in the degradation of the invertebrate community, but the two most important factors appear to be silt and the presence of one or more toxicants (e.g., chlorine). Excessive quantities of silt were evident at all sites and may have been the major cause of impact to the invertebrate community upstream of SD 170. Silt also appears to be an important cause of impact to the invertebrate community downstream of SD 170, but the presence of one or more toxicants also appears to be affecting this portion of the stream.

6.1.6 Future Studies

The initial benthic invertebrate monitoring program in which the sampling frequency was monthly was completed in July 1988; the sampling frequency was then reduced to quarterly. Qualitative samples will continue to be collected annually during the spring of each year at the non-reference sites, but a qualitative sample will no longer be collected from the reference site unless the site shows evidence of impact. Data analyses in future reports will continue to incorporate information obtained from other monitoring programs on the ORR where appropriate, and efforts to obtain more sensitive parameters to help ascertain the status of the benthic community, such as EPT richness, will also continue.

6.2 FISHES (*M. G. Ryon*)

6.2.1 Introduction

Fish population and community studies can be used to assess the ecological effects of water quality and habitat. These

studies offer several advantages over other indicators of environmental quality (see Fausch et al. 1990, Karr et al. 1986, Karr 1987, Karr 1991) and are relevant to any evaluation of the biotic integrity of streams such as Mitchell Branch. Fish communities, for example, comprise several trophic levels, with species that are at or near the end of food chains. Consequently, they integrate the direct effects of water quality and habitat changes on primary producers (periphyton) and consumers (benthic invertebrates) that are used for food. Because of these trophic interrelationships, the well-being of fish populations has often been used as an index of water quality (e.g., Weber 1973; Greeson et al. 1977; Karr et al. 1991). Moreover, statements about the condition of the fish community are understood by the general public (Karr 1981).

The initial objectives of the instream fish monitoring task (Subtask 4b of BMAP, as described in Loar et al. 1991) were to (1) characterize spatial and temporal patterns in distribution and abundance of fishes in Mitchell Branch and (2) document possible effects on fish community structure and function resulting from implementation of the K-25 Site Water Pollution Control Program.

6.2.2 Methods

The fish community in Mitchell Branch was evaluated at five sites (Fig. 2.1 and Table 2.8), four of which reflect reaches potentially affected by the Oak Ridge K-25 Site effluents (Sect. 2.2.3). The most upstream site, MIK 1.43, served as an unaffected reference. An additional, nearby reference site (GCK 2.4) was sampled also (Fig. 2.2). The affected Mitchell Branch sites were sampled four times during 1988, twice in 1989, and once in the spring of 1990 (Table 6.4). Grassy Creek was sampled three times during

1988, twice in 1989, and once in the spring of 1990. The other reference site, MIK 1.43, was dropped from the sampling program in the fall of 1988 because the shallow water did not provide fish habitat similar to the downstream sites (Table 6.4). The absence of similar habitat made fish community comparisons invalid.

All sampling was conducted with the use of a Smith-Root Model 15A backpack electrofisher. The unit has a self-contained, gasoline-powered generator capable of delivering up to 1200 V of pulsed direct current. The pulse frequency and the output voltage can be varied, but generally a pulse frequency of 90 to 120 Hz and a voltage of less than 400 V was used. The circular (ring) electrode at the end of the fiberglass anode pole was fitted with a nylon net (0.64-cm mesh) to allow the operator to assist in collecting stunned fish.

After 0.64-cm-mesh seines were placed across the upper and lower boundaries of the fish sampling site to restrict fish movement, a two- to three-person sampling team electrofished through the site in an upstream direction, making three consecutive passes. Stunned fish were collected and separated by pass in buckets prior to processing. If the number of fish captured during the first pass was extremely low or zero, only one pass was made. Depending on the turbidity of the water, the consecutive passes could not always be made immediately. Rather, fish were processed after each pass to allow sufficient time for the water to clear before another pass was started.

Following the electrofishing, fish were anesthetized with MS-222 (tricaine methanesulfonate), identified, measured to the nearest 0.1 cm (total length), and weighed to the nearest 0.1 g (for fish less than 100 g) or 1 g (for fish greater than 100 g) by using Pesola spring scales. At sites with high fish densities, individuals were recorded by 1-cm size classes and species. After 25 individuals of a

Table 6.4. Length, mean width, mean depth, area, and pool/riffle ratio (P/R) of fish sampling sites on Mitchell Branch and Grassy Creek, a reference stream, January 1988-May 1990

Site ^a	Date	Length (m)	Width (m)	Depth (cm)	Area (m ²)	P/R
MIK 0.45	1/12/88	30	1.8	18	54	NM
	3/29/88	31	1.8	18	56	NM
	7/25/88	30	1.8	15	54	NM
	11/28/88	31	2.0	24	62	NM
	5/15/89	29	1.8	15	52	0.5
	10/30/89	33	1.7	11	56	3.1
	5/15/90	29	1.9	10	55	0.3
MIK 0.54	1/12/88	42	1.3	10	55	NM
	3/29/88	44	1.3	12	57	NM
	7/25/88	40	1.3	9	52	NM
	11/28/88	42	1.2	13	50	NM
	5/15/89	40	1.2	11	48	0 ^b
	10/30/89	36	1.4	6	50	1.4
	5/15/90	42	1.4	8	59	0.2
MIK 0.71	1/12/88	50	1.0	10	50	NM
	3/29/88	51	0.9	14	46	NM
	7/25/88	NS ^c				
	11/28/88	58	1.1	22	64	NM
	5/15/89	55	0.9	15	50	0.9
	10/30/89	56	1.0	9	56	1.0
	5/15/90	47	1.3	11	61	0.3
MIK 0.78	1/12/88	44	0.7	9	31	NM
	3/29/88	44	0.9	13	40	NM
	7/25/88	45	1.1	7	50	NM
	11/28/88	48	1.0	14	48	NM
	5/15/89	43	1.0	11	43	0.5
	10/30/89	43	0.9	9	39	1.3
	5/15/90	43	1.2	9	52	0.5
MIK 1.43	1/22/88	31	0.9	6	28	NM
	3/29/88	31	1.0	4	31	NM
	7/25/88	31	0.4	3	12	NM
GCK 2.4	1/22/88	59	1.8	12	106	NM
	3/24/88	59	1.3	8	76	NM
	7/25/88	NS				
	10/14/88	38 ^d	1.4	11	52	NM
	4/06/89	60	1.9	13	114	0.8
	10/23/89	61	1.4	9	85	1.8
	4/19/90	61	1.6	8	98	0.8

^aMIK = Mitchell Branch kilometer; GCK = Grassy Creek kilometer.

^bStream section contained only pools.

^cSite not sampled during this period.

^dLength reflects only pools; riffle areas were dry and not measured.

Note: NM = not measured for this sample.

species-size class were measured and weighed, additional members of that size class were only measured. Length-weight regressions were later used to estimate missing weight data (Railsback et al. 1989). Other data recorded (if possible to determine), included sex, reproductive state, disposition (i.e., dead or kept for laboratory identification and inclusion in a reference collection), and the presence of any abnormalities (e.g., external parasites, skeletal deformities, etc.). After processing fish from all passes, the fish were allowed to fully recover from the anesthetic and were returned to the stream within the sampling site. Any additional mortality occurring as a result of processing was noted at that time.

Also recorded were data on selected physical and chemical parameters of the stream site and on the sampling effort. A Horiba Model U-7 Water Quality Checker was used to measure pH, temperature, dissolved oxygen, and conductivity. An HF Instruments Model DRT-15 turbidimeter was used to measure turbidity. The duration of the electrofishing effort was recorded, and a visual estimate was made of percentage cloud cover. Following completion of fish sampling, the length, widths (at 5-m intervals), and depths (at three points on the width transect) of the sampling reach were measured at each site.

Species population estimates were calculated by using the three-pass removal method of Carle and Strub (1978). Biomass was estimated by multiplying the population estimate by the mean weight per individual. Annual production was estimated at each site by using a size-frequency method (Garman and Waters 1983). Total numbers, biomass, and production were divided by the surface area (m^2) of the sampling site to calculate density, biomass, and annual production per unit area. For each sampling date, surface area was estimated by multiplying the length of sampling reach by the mean

width, based on measurements taken at 5-m intervals (Table 6.4). The procedures used for calculating density, biomass, and production are given in more detail in Railsback et al. (1989).

Condition factors measure the degree of plumpness of a fish as an index of relative health (Bennett 1970). A condition factor (K) was calculated for individual fish according to the formula

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

with weight in grams and total length in centimeters (Hile 1936). Fish weights estimated by regression were not used in the calculation of condition factors. Comparisons of condition factors between sites and between sampling periods were made with an ANOVA on untransformed data (PROC GLM, SAS 1985b) because the condition factors exhibited homogeneity of the variance (PROC UNIVARIATE, SAS 1985a). If the ANOVA indicated significant differences in condition factors between groups, the Tukey test was performed to identify those groups that were significantly different ($p < 0.05$).

6.2.3 Results and Discussion

Surveys made from 1988 to 1990 continued to show declines in fish population densities and biomass that were seen at most sites during the first year of BMAP (Ryon 1993), with the declines resulting in the absence of fish populations in much of Mitchell Branch. However, a slight increase in number of fish species was observed in 1990 in contrast to the earlier sampling period.

6.2.3.1 Species composition and richness

During the first 2 years of sampling (October 1986 to October 1993), the fish

community of Mitchell Branch was a simple, three-species complex (Table 6.5). The complex consisted of the blacknose dace, *Rhinichthys atratulus*, the creek chub, *Semotilus atromaculatus*, and the redbreast sunfish, *Lepomis auritus* (Ryon 1993). All three species appear to be insensitive to many habitat and water quality stresses and would not be considered intolerant species in an impact assessment methodology such as the Index of Biotic Integrity (Karr et al. 1986). During the last two sampling periods, two additional species were collected in Mitchell Branch (Table 6.5). The central stoneroller, *Camptostoma anomalum*, and the striped shiner, *Luxilus chrysocephalus*, were found in low numbers at MIK 0.78. In comparison with similar-sized, relatively undisturbed area streams, the fish species richness in Mitchell Branch was typical of that found in headwater streams (Ryon and Loar 1988). In upper Grassy Creek at GCK 2.4, the striped shiner, white sucker (*Catostomus commersoni*), banded sculpin (*Cottus carolinae*), and green sunfish (*L. cyanellus*) occurred with blacknose dace and creek chub during the same period. The faunal differences in these two streams may reflect temperature or water quality differences between the sites, although only the sculpin is considered to be an intolerant species (Karr et al. 1986).

6.2.3.2 Population densities

Surveys from 1988 to 1990 indicated that fish populations had disappeared from sites other than MIK 0.78 (Table 6.5). The only fish taken outside of this site was collected in the May 1990 sample at the upper end of MIK 0.71. This large creek chub may have been displaced from a section just upstream of SD 170. The effluent from SD 170 enters Mitchell Branch in the lower reach of fish sampling site MIK 0.78. In the 1988 to 1990 surveys,

no fish were collected in this site below where this effluent enters Mitchell Branch. This effluent is consistently toxic to fathead minnows in laboratory toxicity tests (Table 3.6). Episodic releases of chlorine (Sect. 3.2.3) may prevent fish from establishing permanent residency in the section of Mitchell Branch below the SD 170 outfall and above the embayment near Poplar Creek. The greatest impact on the benthic invertebrate community also occurred at this site (Table 6.2).

The 1988-90 surveys also documented declines in the fish population at MIK 0.78 above SD 170. In the first year's surveys (1986 to 1987), the population densities at MIK 0.78 averaged 3.2 fish per square meter (Ryon 1993). During 1988, the average population density dropped to 1.5 fish per square meter before increasing somewhat to 2.3 fish per square meter in 1989-90 samples (Table 6.5). Also noticeable in the 1988-90 sampling period was the gradual disappearance of redbreast sunfish from the population. In comparison with the reference site at GCK 2.4, the fish density at MIK 0.78 was slightly higher (1.6 vs 1.2 fish per square meter). Both reference sites and MIK 0.78 suffered from low-water conditions as a result of a drought in 1987-88. Another factor that may have affected fish densities was a fish kill in Mitchell Branch in November 1988 that was traced to SD 170 (J. M. Loar, ESD, ORNL, March 17, 1989, personal communication). Although the kill occurred downstream of the section of MIK 0.78 where fish were absent, it probably adversely affected immigration into the site from the lowermost reaches of Mitchell Branch downstream of the toxic zone.

6.2.3.3 Population biomass

Fish biomass values (in grams wet weight per square meter) paralleled the

Table 6.5. Fish densities (per square meter) in Mitchell Branch and Grassy Creek, a reference stream, January 1988–April/May 1990

Species ^a	MIK 0.45	MIK 0.54	MIK 0.71	MIK 0.78	MIK 1.43 ^b	GCK 2.4
<i>January 1988</i>						
Blacknose dace				0.5		0.4
Creek chub				1.9	<0.1	0.1
Redbreast sunfish				0.2		
White sucker						<0.1
Total	NF ^c	NF	NF	2.6	<0.1	0.5
<i>March 1988</i>						
Blacknose dace				0.2		0.5
Creek chub				0.9		0.1
White sucker						<0.1
Total	NF	NF	NF	1.1	NF	0.6
<i>July 1988</i>						
Blacknose dace				0.2		
Creek chub				0.3		
Total	NF	NF	NF	0.5	NF	NS ^d
<i>October/November 1988</i>						
Blacknose dace				0.5		0.9
Creek chub				1.0		0.7
Redbreast sunfish				0.1		
Total	NF	NF	NF	1.6	NS ^d	1.6
<i>April/May 1989</i>						
Blacknose dace				0.7		0.1
Creek chub				1.3		0.1
Striped shiner						<0.1
White sucker						<0.1
Total	NF	NF	NF	2.0	NS	0.2

Table 6.5 (continued)

Species ^a	MIK 0.45	MIK 0.54	MIK 0.71	MIK 0.78	MIK 1.43 ^b	GCK 2.4
<i>October 1989</i>						
Blacknose dace				1.3		2.2
Creek chub				1.5		0.5
Striped shiner				0.1		
Central stoneroller				0.3		
White sucker						<0.1
Banded sculpin						<0.1
Green sunfish						<0.1
Total	NF	NF	NF	3.2	NS	2.8
<i>April/May 1990</i>						
Blacknose dace				0.7		0.9
Creek chub			<0.1	1.0		0.3
Central stoneroller				0.1		
Striped shiner						<0.1
White sucker						<0.1
Banded sculpin						<0.1
Total	NF	NF	<0.1	1.8	NS	1.3

^aSpecies are blacknose dace, *Rhinichthys atratulus*; creek chub, *Semotilus atromaculatus*; striped shiner *Luxilus chrysocephalus*; central stoneroller, *Campostoma anomalum*; redbreast sunfish, *Lepomis auritus*; green sunfish, *L. cyanellus*; banded sculpin *Cottus carolinae*; and white sucker, *Catostomus commersoni*.

^bReference site on upper Mitchell Branch.

^cNF = no fish taken in sample.

^dNS = not sampled.

Note: MIK = Mitchell Branch kilometer; GCK = Grassy Creek kilometer.

trends observed in population density (Table 6.6). Biomass at MIK 0.78 declined about 50% between the first year's surveys (mean of 10.6) and the surveys of 1988-90 (mean of 5.6), further indicating that adverse impacts affected fish populations in Mitchell Branch. When compared with the reference site on Grassy Creek (GCK 2.4), biomass at MIK 0.78 was high.

6.2.3.4 Annual production

Annual production (g wet wt·m⁻²·year⁻¹) of individual species and the fish community was calculated for MIK 0.78 and GCK 2.4 from spring to spring for 1987-90. In general, production increased each year for both individual species and the whole community (Table 6.7). The

Table 6.6. Fish biomass (in grams per square meter) in Mitchell Branch and Grassy Creek, a reference stream, from January 1988–April/May 1990

Species ^a	MIK 0.45	MIK 0.54	MIK 0.71	MIK 0.78	MIK 1.43 ^b	GCK 2.4
<i>January 1988</i>						
Blacknose dace				0.4		0.4
Creek chub				1.8	<0.1	0.1
Blacknose dace				0.4		0.4
Creek chub				5.0	<0.1	0.5
Redbreast sunfish				0.5		
White sucker						0.2
Total	NF ^c	NF	NF	5.9	<0.1	1.1
<i>March 1988</i>						
Blacknose dace				0.2		0.3
Creek chub				4.1		0.2
White sucker						0.6
Total	NF	NF	NF	4.3	NF	1.1
<i>July 1988</i>						
Blacknose dace				<0.1		
Creek chub				0.2		
Total	NF	NF	NF	0.2	NF	NS ^d
<i>October/November 1988</i>						
Blacknose dace				0.4		0.9
Creek chub				2.1		1.9
Redbreast sunfish				3.4		
Total	NF	NF	NF	5.9	NS	2.8
<i>April/May 1989</i>						
Blacknose dace				1.4		0.2
Creek chub				7.2		0.4
Striped shiner						0.1
White sucker						0.2
Total	NF	NF	NF	8.6	NS	0.9

Table 6.6 (continued)

Species ^a	MIK 0.45	MIK 0.54	MIK 0.71	MIK 0.78	MIK 1.43 ^b	GCK 2.4
<i>October 1989</i>						
Blacknose dace				1.4		1.5
Creek chub				4.5		2.1
Striped shiner				0.1		
Stoneroller				1.6		
White sucker						0.6
Banded sculpin					0.2	
Green sunfish						0.7
Total	NF	NF	NF	7.6	NS	5.1
<i>April/May 1990</i>						
Blacknose dace				1.2		1.1
Creek chub			0.3	4.9		1.6
Stoneroller				0.5		
Striped shiner						0.5
White sucker					<0.1	
Banded sculpin						0.3
Total	NF	NF	0.3	6.6	NS	3.5

^aSpecies are blacknose dace, *Rhinichthys atratulus*; creek chub, *Semotilus atromaculatus*; striped shiner *Notropis chrysocephalus*; stoneroller, *Camptostoma anomalum*; redbreast sunfish, *Lepomis auritus*; green sunfish, *L. cyanellus*; banded sculpin *Cottus carolinae*; and white sucker, *Catostomus commersoni*.

^bReference site on upper Mitchell Branch.

^cNF = no fish taken in sample.

^dNS = not sampled.

Note: MIK = Mitchell Branch kilometer; GCK = Grassy Creek kilometer.

exceptions were a decline in production of redbreast sunfish at MIK 0.78 after 1987-88 and a dip in production for creek chub and blacknose dace at GCK 2.40 in 1988-89. The pattern of increasing production may be part of a general recovery from drought conditions in 1987 and 1988.

Total production at MIK 0.78 was 7 to 17 times higher than at GCK 2.4. The higher productivity of the disturbed Mitchell Branch site may reflect increased

primary production resulting from nutrient enrichment and/or canopy removal. Elwood et al. (1981) demonstrated increased primary production following phosphorus enrichment of a nonpolluted stream. Such an enrichment may be occurring in Mitchell Branch as a result of plant discharges. Minshall (1978) presented some data for nonpolluted streams showing that mean annual gross primary production was greater in open

Table 6.7. Annual production (in grams wet weight, per square meter per year ± 1 SE) and production to biomass (P/B) ratios, in parenthesis, in Mitchell Branch and Grassy Creek from spring 1987 to spring 1990

Species ^b	Production period	Site ^a			
		MIK 0.78		GCK 2.4	
		Production	P/B ^c	Production	P/B
Blacknose dace	1987-1988	0.18 \pm 0.04	(1.3)	0.06 \pm 0.03	(0.2)
	1988-1989	0.83 \pm 0.05	(1.0)	-0.02 \pm 0.06	(0.1)
	1989-1990	1.38 \pm 0.05	(1.1)	0.41 \pm 0.05	(0.6)
Creek chub	1987-1988	2.75 ^d	(0.6)	0.17 \pm 0.03	(0.3)
	1988-1989	3.90	(0.7)	0.10	(0.3)
	1989-1990	4.83	(0.8)	0.19	(0.2)
Redbreast sunfish	1987-1988	0.13	(0.4)		
	1988-1989	0.0			
	1989-1990	0.0			
Stoneroller	1987-1988	0.0			
	1988-1989	0.0			
	1989-1990	0.65 \pm 0.17	(2.4)		
White sucker	1987-1988			-0.02	(<0.1)
	1988-1989			0.19	(0.5)
	1989-1990			0.31	(3.1)
Total ^e	1987-1988	3.06	(0.6)	0.18	(0.1)
	1988-1989	4.73	(0.7)	0.37	(0.4)
	1989-1990	6.85	(0.9)	0.91	(0.4)

^aMIK = Mitchell Branch kilometer; GCK = Grassy Creek kilometer.

^bSpecies are blacknose dace, *Rhinichthys atratulus*; creek chub, *Semotilus atromaculatus*; striped shiner *Luxilus chrysocephalus*; central stoneroller *Campostoma anomalum*; redbreast sunfish, *Lepomis auritus*; and white sucker, *Catostomus commersoni*.

^cBiomass (B) is a mean of beginning and ending spring values for that production period.

^dStandard error could not be calculated because some size classes have only one fish per class.

^eTotals include miscellaneous species (green sunfish *L. cyanellus*, and banded sculpin, *Cottus caroliniae*) not listed.

streams than in streams with a closed canopy. The Grassy Creek canopy is more closed than that of Mitchell Branch. Higher primary production should sustain higher levels of secondary production, with

greater numbers of benthic invertebrates available for consumption by fish species. However, secondary production of benthic invertebrates in 1986 sampling of MIK 0.78 was generally lower than other published

values (Smith 1993). Secondary production estimates for MIK 0.78 were also lower than production estimates for unaffected headwater streams in the Oak Ridge area (Smith 1993). Data to determine whether this trend in benthic production continues for 1988-90 are not yet available.

New fish species found at MIK 0.78 in 1989-90 demonstrate that immigration (also see Sect. 5.3.3) provides an added component of the production value.

Although new species were also added to the fauna at GCK 2.4, their occurrence was not as sustained as that at MIK 0.78.

The annual fish production values for MIK 0.78 are in the middle of the range of other published production in

warmwater streams (Table 6.8). The MIK 0.78 production was generally higher than that reported by Neves and Pardue (1983) for second-order streams in Virginia but lower than production in second-order streams in Kentucky (Lotrich 1973, Small 1975). The same trend held for comparisons of species. The production rate of blacknose dace was very similar between MIK 0.78 (0.18 to 1.38 g wet wt·m⁻²·year⁻¹) and Virginia streams (0.30 to 0.38 g wet wt·m⁻²·year⁻¹). For creek chub, the production rate at MIK 0.78 (2.75 to 4.83 g wet wt·m⁻²·year⁻¹) was lower than in Kentucky streams (7.28 g wet wt·m⁻²·year⁻¹, Lotrich 1973). The production to biomass (P/B) ratio at MIK 0.78 was 0.6 to 0.9;

Table 6.8. Annual production of fish communities in warm water streams in the Southeast

Stream (state)	Stream order	Production (g·m ⁻² ·year ⁻¹ wet wt)	Reference ^a
J. Carpenter Branch (Ky.)	1st	8.55	Lotrich 1973
Clemmons Creek (Ky.)	2nd	10.55	Lotrich 1973
Steeles Run (Ky.)	2nd	12.0 ^b	Small 1975
	2nd	15.8 ^b	
Guys Run (Va.)	2nd	2.84	Neves and Pardue 1983
	2nd	3.16	
	2nd	3.96	
Little Cooter Prairie (Ga.)	^c	17.07	Freeman and Freeman 1985

^aComplete reference citations may be found in Sect. 7 of this document (J. G. Smith et al., 1993, *Second Report on the Oak Ridge K-25 Site Biological Monitoring and Abatement Program for Mitchell Branch*, ORNL/TM-12150, Oak Ridge National Laboratory, Oak Ridge, Tenn.).

^bValues converted from dry wt by using the conversion factor in T. F. Waters 1977, "Secondary production in inland waters," *Adv. Ecol. Res.* 10:91-164.

^cBlackwater swamp; part of the Okefenokee Swamp.

these values were much lower than the other published value, 3.2, for a blackwater swamp in Georgia (Freeman and Freeman 1985).

6.2.3.5 Condition factors

Comparisons of condition factors (*K*) should provide information on the relative well-being of the fish because those with more weight per length have a higher condition factor (Everhart et al. 1975). Data were available for a statistical comparison of condition factors for blacknose dace and creek chub between MIK 0.78 and GCK 2.4 during January 1988, March 1988, October/November 1988, April/May 1989, October 1989, and April/May 1990.

For the blacknose dace and the creek chub, condition factors were significantly greater at MIK 0.78 than at GCK 2.4 in four of six comparisons (Table 6.9). Both species demonstrated significant differences at the same sampling periods. However, a consistent trend for improvement or decline was not apparent over the 1988-90 period (also see Sect. 5.3.1.4). The higher condition factors of fish in Mitchell Branch vs Grassy Creek suggest that individual fish surviving stressful conditions in Mitchell Branch generally remained in good health. However, caution must be exercised in the interpretation of condition factors because they may be relatively insensitive to environmental conditions or nutritional status (Loar et al. 1985).

Table 6.9. An analysis of variance comparison of fish condition factors between Mitchell Branch and Grassy Creek, January 1988-April/May 1990

Species	Sampling date	Site ^a		<i>p</i> ^b
		MIK 0.78	GCK 2.4	
Blacknose dace	January 1988	1.00	0.97	0.40
	March 1988	1.17 ^c	0.83	0.0001
	Oct/Nov 1988	0.95 ^c	0.85	0.004
	April/May 1989	1.07	1.06	0.90
	October 1989	1.15 ^c	0.89	0.0001
	April/May 1990	1.04 ^c	0.89	0.0001
Creek chub	January 1988	1.03	0.97	0.10
	March 1988	1.17 ^c	0.96	0.005
	Oct/Nov 1988	0.99 ^c	0.92	0.04
	April/May 1989	1.18	1.10	0.07
	October 1989	1.02 ^c	0.91	0.0001
	April/May 1990	1.16 ^c	0.98	0.0001

^aMIK = Mitchell Branch kilometer; GCK = Grassy Creek kilometer.

^b*P* = probability level.

^cMIK 0.78 values that are statistically significantly different from GCK 2.4 values.

Seasonal comparisons of condition factors did not show a trend for blacknose dace but did indicate for the creek chub that the spring samples were statistically greater ($p < 0.05$) than the other sampling periods. These spring increases for creek chubs may reflect preparations for spawning.

6.2.4 Conclusions

During the 1988-90 sampling period, fish populations in Mitchell Branch revealed substantial, adverse impacts. Species richness, density, and biomass of fish in Mitchell Branch were all low, with the extent of impact related spatially to operations at the Oak Ridge K-25 Site. Additionally, a steady decline in the robustness of the fish community was observed. The absence of fish below SD 170, which enters Mitchell Branch at MIK 0.76, strongly suggests that toxic effluents are entering the stream at this site. Toxicants may have also entered through discharges from the K-1407-E/F ponds, SD 180, and/or SD 190. Although residual chlorine is a toxicant in Mitchell Branch (see Sect. 3), other stressors such as siltation could also contribute to the observed effects on the fish populations.

6.2.5 Future Studies

Earlier plans for assessing the fish community in Mitchell Branch involved quarterly sampling of the fish populations for the first 2 years (Loar et al. 1991, Ryon 1993). Because fish populations in most of Mitchell Branch have disappeared, sampling was restricted to the spring and fall periods. Future sampling will continue on this frequency. Data indicate an adverse impact from MIK 0.78 downstream to MIK 0.45, but because fish are taken in an impounded area of lower Mitchell Branch, an attempt will be made to define the downstream edge of toxic conditions in lower Mitchell Branch.

Plans to assess impacts at the individual level, by examining fecundity or feeding patterns, will be implemented only if population densities and biomass stabilize. Because immigration of individuals and species into Mitchell Branch from Poplar Creek is occurring, further qualitative sampling will be conducted to assess immigration. This sampling may be a cooperative effort with sampling for the bioaccumulation task.

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Appendix A

WATER QUALITY DATA FOR MITCHELL BRANCH

Table A.1. Median and maximum concentrations (in milligrams per liter unless noted otherwise) of National Pollutant Discharge Elimination System parameters in the water from lower Mitchell Branch

Parameter	K-1700 ^a							
	1986		1987		1988		1989	
	Median	Max	Median	Max	Median	Max	Median	Max
Aluminum, total	0.1900	2.8000	0.1300	12.0000	0.1800	8.1000	0.2500	9.4000
Arsenic, total	NA ^b	NA	NA	NA	NA	NA	<0.0050 ^c	^d
Barium, total	<0.1	^d	NA	NA	NA	NA	<0.1000	^d
Beryllium, total	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
Beta, total pCi/L	NA	NA	NA	NA	NA	NA	18.5000	36.0000
Boron, total	0.0410	^d	NA	NA	NA	NA	0.0720	^d
Cadmium	<0.0020	0.0020	0.0020	0.0120	<0.0020	0.0026	<0.0020	0.0030
Cesium	100.0000	^e	100.0000	100.0000	0.0000	100.0000	0.0000	0.0000
Chemical oxygen demand	7.0000	7.0000	7.5000	38.0000	7.7500	688.0000	<5.0000	159.0000
Chromium	0.0100	0.0180	0.0100	0.0230	<0.0100	0.0180	<0.0100	0.0250
Cobalt, total	NA	NA	NA	NA	NA	NA	<0.1000	^d
Copper	NA	NA	NA	NA	NA	NA	0.0110	^d
Flow (L/s)	NA	NA	NA	NA	23.1000	43.6000	25.1000	58.5000
Fluoride	0.2000	1.0000	0.4000	6.0000	0.7000	14.0000	0.3000	1.7000
Iron, total	NA	NA	NA	NA	NA	NA	2.0200	3.2000
Lead	0.0040	0.0070	0.0040	0.0360	<0.0040	0.0230	<0.0040	0.0130
Magnesium, total	NA	NA	NA	NA	NA	NA	5.9000	^d
Manganese, total	NA	NA	NA	NA	NA	NA	0.0930	^d
Mercury, total	<0.0002	0.0002	0.0002	0.0010	<0.0002	0.0007	<0.0002	0.00065
Molybdenum, total	NA	NA	NA	NA	NA	NA	<0.0100	^d
Neptunium	<1.0000	^e	<1.0000	<1.0000	0.1200	1.0000	0.0950	0.1600
Nickel	NA	NA	NA	NA	NA	NA	<0.0500	^d
Nitrate as N	0.4400	5.7200	0.4500	7.6800	0.3300	3.7000	0.4000	4.0000
Oil and grease	2.0000	4.0000	<2.0000	2.0000	<2.0000	5.5000	<2.0000	<2.0000

Table A.1 (continued)

Parameter	K-1700 ^a							
	1986		1987		1988		1989	
	Median	Max	Median	Max	Median	Max	Median	Max
pH	NA	NA	NA	NA	7.7000	8.8000	7.6000	8.6000
Phenols, total	NA	NA	3.0000	^e	NA	NA	NA	NA
Phosphorus, total	NA	NA	NA	NA	NA	NA	<0.2000	^d
Plutonium	1.0000	^b	1.0000	2.0000	-0.04	37.0000	0.0750	0.5900
Selenium, total	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	0.0059	<0.0050	0.0050
Silver	<0.0100	<0.0100	<0.0100	<0.0100	<0.0100	0.0240	<0.0100	0.0100
Technetium (pCi/L)	110.0000	110.0000	149.0000	229.0000	140.0000	295.0000	63.0000	157.0000
Temperature	NA	NA	NA	NA	16.5500	32.0000	15.5000	30.3000
Titanium, total	<0.0030	^c	NA	NA	NA	NA	0.0570	^d
Total dissolved solids	NA	NA	522.0000	1264.0000	NA	NA	318.0000	996.0000
Total suspended solids	5.0000	64.0000	7.0000	292.0000	4.0000	129.0000	4.0000	50.0000
Turbidity (NTU) ^f	6.0000	79.0000	4.05	180.0000	4.0500	300.0000	7.4000	1400.0000
Uranium	0.0270	0.0470	0.0230	0.0350	0.0380	0.6300	0.0270	0.2070
Zinc	<0.0200	0.0420	0.0200	0.3200	<0.0200	0.0620	<0.0200	0.0750

^aLocated at Mitchell Branch kilometer (MIK) 0.12.

^bNA = not available.

^cLess than (<) values were assigned if more than 50% of the initial observations had (<) values.

^dOne observation.

^eTwo observations.

^fNTU = nephelometric turbidity units.

Note: For information on sampling frequency and type, see Table A.8 in this report.

Table A.2. Median and maximum concentrations (in milligrams per liter) of organics routinely monitored in the water from lower Mitchell Branch

Parameter	K-1700 ^c							
	1986		1987		1988		1989	
	Median	Max	Median	Max	Median	Max	Median	Max
1 1 1-Trichloroethane	4.0000	12.0000	2.0000	7.0000	2.0000	8.0000	2.0000	39.0000
1 1 2 2-Tetrachloroethane	<5.0000 ^b	5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
1 1-Dichloroethane	<5.0000	5.0000	3.0000	5.0000	<5.0000	35.0000	<5.0000	6.0000
1 1-Dichloroethene	5.0000	5.0000	2.0000	5.0000	<5.0000	5.0000	<5.0000	14.0000
1 2-Dichloroethane	<5.0000	<5.0000	5.0000	48.0000	<5.0000	<5.0000	<5.0000	<5.0000
2-Butanone	<10.0000	16.0000	9.0000	20.0000	65.5000	100.0000	<10.0000	°
2-Hexanone	<10.0000	10.0000	2.0000	13.0000	NA ^d	NA	<10.0000	°
Acetone	10.0000	25.0000	10.0000	200.0000	15.0000	48.0000	<10.0000	°
Acrolein	NA	NA	5.0000	100.0000	NA	NA	NA	NA
Benzene	5.0000	10.0000	<5.0000	<5.0000	<5.0000	5.0000	<5.0000	5.0000
Bromodichloromethane	<5.0000	5.0000	<5.0000	5.0000	<5.0000	5.0000	<5.0000	5.0000
Bromoform	<5.0000	5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Carbon tetrachloride	<5.0000	5.0000	<5.0000	<5.0000	<5.0000	5.0000	<5.0000	<5.0000
Chlorobenzene	5.0000	5.0000	<5.0000	<5.0000	<5.0000	5.0000	<5.0000	5.0000
Chloroethane	<10.0000	10.0000	<10.0000	10.0000	<10.0000	<10.0000	<10.0000	10.0000
Chloroform	5.0000	9.0000	4.0000	14.0000	5.0000	17.0000	3.0000	10.0000
Chloromethane	<10.0000	10.0000	<10.0000	10.0000	<10.0000	<10.0000	<10.0000	10.0000
Cis-1 2-dichloroethene	NA	NA	NA	NA	28.0000	°	NA	NA
Cis-1 3-dichloropropene	<5.0000	5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Dibromochloromethane	<5.0000	5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Dibromomethane	1.0000	2.0000	37.0000	°	NA	NA	NA	NA

Table A.2 (continued)

Parameter	K-1700 ^a							
	1986		1987		1988		1989	
	Median	Max	Median	Max	Median	Max	Median	Max
Ethylbenzene	5.0000	5.0000	<5.0000	<5.0000	<5.0000	5.0000	<5.0000	<5.0000
Freon 113	3.0000	5.0000	2.0000	920.0000	7.0000	10.0000	NA	NA
Methylene chloride	5.0000	5.0000	5.0000	6.0000	<5.0000	5.0000	<5.0000	5.0000
Tetrachloroethane	5.0000	13.0000	2.0000	29.0000	2.0000	5.0000	2.0000	98.0000
Toluene	2.0000	5.0000	1.0000	49.0000	<5.0000	5.0000	<5.0000	5.0000
Trans-1 2-dichloroethene	40.0000	76.0000	43.0000	140.0000	38.5000	63.0000	30.0000	50.0000
Trans-1 3-dichloropropene	<5.0000	5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	32.0000
Trichloroethane	60.0000	86.0000	55.0000	510.0000	43.0000	87.0000	34.0000	72.0000
Vinyl chloride	10.0000	13.0000	8.0000	17.0000	6.0000	10.0000	6.0000	10.0000
Xylenes, total	2.0000	5.0000	1.0000	81.0000	1.0000	5.0000	<5.0000	^c

^aLocated at Mitchell Branch kilometer (MIK) 0.12.

^bLess than values (<) were assigned if more than 50% of the initial observations had (<) values.

^cOne observation.

^dNA = Not available.

^eTwo observations.

Note: For information on sampling frequency and type, see Table A.8 in this report.

Table A.3. Median and maximum concentrations (in milligrams per liter unless noted otherwise) of National Pollutant Discharge Elimination System parameters in the water from K-1407-B pond

Parameter	K-1407-B*					
	1986 ^b		1987 ^c		1988 ^d	
	Median	Max	Median	Max	Median	Max
Alpha, total (pCi/L)	NA ^e	NA	7.0000	29.2000	5.0000	f
Aluminum, total	0.2300	0.6200	0.2000	2.1000	0.2350	1.6000
Ammonia (as N)	<0.2000 ^g	0.5000	<0.2000	0.7400	<0.2000	2.9000
Antimony, total	<0.0500	0.0620	<0.0500	0.1300	<0.0500	0.2700
Arsenic, total	0.0050	0.0070	<0.0050	0.0100	0.0050	0.0210
Barium, total	<0.1000	<0.1000	<0.1000	<0.1000	<0.1000	<0.1000
Beryllium, total	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	0.0011
Beta, total (pCi/L)	78.0000	93.0000	16.1000	52.9000	19.0000	f
Boron, total	0.1200	0.1600	0.0765	0.1900	0.1050	12.0000
Bromide	3.0000	6.0000	<2.0000	3.0000	<2.0000	3.7000
Cadmium	<0.0020	<0.0020	<0.0020	0.0500	<0.0020	0.0250
Cesium	<100.0000	^h	<100.0000	<100.0000	0.0000	100.0000
Chemical oxygen demand	14.0000	66.0000	11.0000	146.0000	16.0000	56.0000
Chloride	255.0000	366.0000	288.0000	1095.0000	476.5000	994.0000
Chlorine, total residual	<0.1000	<0.1000	<0.1000	0.5000	<0.1000	0.7000
Chromium	<0.0100	0.0130	<0.0100	1.9000	<0.0100	0.0210
Cobalt, total	<0.1000	0.1800	<0.1000	<0.1000	<0.1000	<0.1000
Copper	0.0042	0.0910	<0.0040	2.0000	0.0043	0.0450
Cyanide	0.0060	0.0070	<0.0020	0.1000	0.0060	0.1980
Flow (L/s)	NA	NA	NA	NA	1.3000	165.5000
Fluoride	0.3500	3.0000	0.8000	21.0000	2.5000	57.0000
Iron, total	1.5000	3.8000	0.4350	5.0000	0.3800	7.0000
Kjeldahl nitrogen	0.6000	1.3000	0.6000	1.6200	0.8500	4.4200
Lead	<0.0040	0.0070	<0.0040	0.0680	<0.0093	0.2100
Magnesium, total	23.0000	27.0000	15.5000	35.0000	20.0000	27.0000
Manganese, total	0.1800	0.2700	0.0560	2.0000	0.0580	0.6300
Mercury, total	<0.0002	<0.0002	<0.0002	0.0010	<0.0002	0.0023
Molybdenum, total	<0.0100	0.0140	<0.0100	0.0250	<0.0100	0.0360
Neptunium	<1.0000	^h	<1.0000	<1.0000	0.2650	1.0000
Nickel	0.2300	0.8600	<0.0500	2.0000	<0.0500	1.6000
Nitrate (as N)	0.6300	22.3000	0.5500	26.2000	<0.5500	15.0000
Nitrogen (total organic)	NA	NA	NA	NA	0.9500	1.4000
Oil and grease	<2.0000	3.0000	<2.000	3.0000	<2.0000	51.0000
pH (pH units)	NA	NA	8.1000	8.4000	7.9000	8.9000
Phenols, total	0.0010	0.0040	<0.0010	0.0250	0.0025	0.0300
Phosphorus, total	NA	NA	<0.2000	1.4000	<0.2000	8.4000
Plutonium	<1.0000	^h	<1.0000	1.4000	-0.0200	1.0000
Polychlorinated biphenyls ($\mu\text{g/L}$)						
Aroclor-1254	BD ⁱ	BD	<1.0000	1.2000	<1.0000	1.9000

Table A.3 (continued)

Parameter	K-1407-B*					
	1986 ^b		1987 ^c		1988 ^d	
	Median	Max	Median	Max	Median	Max
Selenium, total	<0.0500	0.0500	<0.0050	<0.0050	<0.0050	<0.0050
Silver	<0.0100	<0.0100	<0.0100	0.0140	<0.0100	0.0210
Sulfate (as SO ₄)	585.0000	859.0000	339.0000	752.0000	507.0000	1410.0000
Sulfide (as S)	<1.0000	<1.0000	1.0000	1.0000	<1.0000	2.0000
Sulfite (as SO ₃)	<2.0000	3.0000	2.0000	7.0000	<2.0000	4.0000
Technetium (pCi/L)	276.5000	444.0000	<214.5000	327.0000	222.0000	3004.0000
Temperature (°C)	NA	NA	NA	NA	23.0000	33.0000
Thallium, total	<0.0100	<0.0100	<0.0100	<0.0100	<0.0100	0.0100
Tin, total	<0.0100	<0.0100	<0.0100	0.0150	<0.0100	0.0280
Titanium, total	<0.0030	0.0093	<0.0030	0.0230	<0.0030	0.0280
Total dissolved solids	1594.0000	1922.0000	1196.0000	3204.0000	1680.0000	14032.0000
Total organic carbon	19.0000	54.0000	4.3000	63.0000	11.8500	43.0000
Total suspended solids	12.5000	398.0000	21.0000	21.0000	9.0000	52.0000
Uranium	0.0510	0.1620	0.0455	0.1820	0.1900	4.9400
Zinc	0.0240	0.0320	<0.0200	2.0000	<0.0200	0.1100

*A holding pond.

^bSampling period November–December.

^cSampling period March–December, January and February not available.

^dSampling does not include January or October.

^eNA = not available.

^fOne observation.

^gLess than values (<) were assigned if more than 50% of the initial parameters had (<) values.

^hTwo observations.

ⁱBD = below detection limit of <1.0 ug/L.

Note: For information concerning sampling frequency and type, see Table A.8 in this report.

Table A.4. Median and maximum concentrations (in milligrams per liter) of organics routinely monitored in the water from K-1407-B pond

Parameter	K-1407-B ^a					
	1986		1987		1988	
	Median	Max	Median	Max	Median	Max
1 1 1-Trichloroethane	2.0000	62.0000	<5.0000	25.0000	5.0000	16.0000
1 1 2 2-Tetrachloroethane	<5.0000 ^b	50.0000	<5.0000	<5.0000	<5.0000	<5.0000
1 1-Dichloroethane	<5.0000	50.0000	<5.0000	5.0000	<5.0000	9.0000
1 1-Dichloroethene	<5.0000	50.0000	<5.0000	5.0000	<5.0000	7.0000
1 2-Dichloroethane	<5.0000	50.0000	<5.0000	5.0000	<5.0000	<5.0000
2-Butanone	<10.0000	100.0000	10.0000	34.0000	8.0000	20.0000
2-Hexanone	<10.0000	100.0000	1.0000	6.0000	7.0000	^c
Acetone	20.0000	550.0000	26.0000	1200.0000	35.5000	610.0000
Acrolein	NA ^d	NA	<5.0000	100.0000	NA	NA
Benzene	3.5000	50.0000	<5.0000	8.0000	<5.0000	5.0000
Bromodichloromethane	<5.0000	50.0000	<5.0000	<5.0000	<5.0000	5.0000
Bromoform	<5.0000	50.0000	<5.0000	5.0000	<5.0000	5.0000
Carbon tetrachloride	<5.0000	50.0000	<5.0000	5.0000	<5.0000	<5.0000
Chlorobenzene	5.0000	50.0000	<5.0000	5.0000	<5.0000	5.0000
Chloroethane	<10.0000	100.0000	<10.0000	10.0000	10.0000	10.0000
Chloroform	5.0000	50.0000	4.0000	20.0000	2.0000	5.0000
Chloromethane	<10.0000	100.0000	<10.0000	10.0000	<10.0000	10.0000
Cis-1 2-dichloroethene	NA	NA	NA	NA	8.0000	10.0000
Cis-1 3-dichloropropene	<5.0000	50.0000	<5.0000	<5.0000	<5.0000	5.0000
Dibromochloromethane	<5.0000	50.0000	<5.0000	<5.0000	<5.0000	<5.0000
Dibromomethane	1.0000	2.0000	1.0000	^e	NA	NA
Ethylbenzene	5.0000	50.0000	<5.0000	<5.0000	<5.0000	5.0000
Freon 113	2.0000	6.0000	2.0000	18.0000	5.0000	5.0000
Methylene chloride	<5.0000	50.0000	5.0000	240.0000	<5.0000	5.0000
Tetrachloroethane	18.0000	120.0000	3.0000	23.0000	<5.0000	12.0000
Toluene	4.0000	50.0000	3.0000	5.0000	<5.0000	5.0000
Trans-1 2-dichloroethene	11.0000	50.0000	7.0000	120.0000	11.0000	240.0000
Trans-1 3-dichloropropene	<5.0000	50.0000	<5.0000	<5.0000	<5.0000	<5.0000
Trichloroethane	33.5000	56.0000	14.0000	50.0000	12.0000	91.0000
Vinyl chloride	<10.0000	100.0000	<10.0000	10.0000	<10.0000	17.0000
Xylenes, total	3.0000	50.0000	1.0000	43.0000	1.0000	^e

^aA holding pond.

^bLess than values (<) were assigned if more than 50% of the initial observations had (<) values.

^cTwo observations.

^dNA = Not available.

^eOne observation.

Note: For information concerning sampling frequency and type, see Table A.8 in this report.

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Table A.5. Median and maximum concentrations (in milligrams per liter unless noted otherwise) of National Pollutant Discharge Elimination System parameters in the water from K-1407-E/F pond

Parameter	K-1407-E/F ^a			
	1988		1989	
	Median	Max	Median	Max
Aluminum, total	<0.1000	0.4700	<0.1000	0.5500
Antimony, total	<0.0500	^c	NA ^d	NA
Arsenic, total	<0.0050	^c	<0.0050	<0.0050
Barium, total	<0.1000	<0.1000	<0.1000	<0.1000
Beryllium, total	<0.0010	<0.0010	<0.0010	<0.0010
Boron, total	0.0750	0.1500	0.0580	0.1300
Cadmium	<0.0030	<0.0030	<0.0020	0.0030
Chromium	<0.0100	<0.0100	<0.0100	0.1000
Cobalt, total	<0.1000	<0.1000	<0.1000	<0.1000
Copper	<0.0040	0.0130	0.0160	0.0640
Flow (L/S)	2.7523	5.8372	2.1403	12.1352
Iron, total	0.6800	0.9700	0.5100	4.7000
Lead	0.0270	0.0500	<0.0040	0.0500
Magnesium, total	18.0000	63.0000	12.0000	24.0000
Manganese, total	0.1350	0.2000	0.0835	0.1400
Mercury, total	<0.0002	^c	NA	NA
Molybdenum, total	<0.0100	<0.0100	<0.0100	<0.0100
Nickel	<0.0500	<0.0500	<0.0500	6.3000
Oil and grease	<2.0000	<2.0000	<2.0000	2.7000
pH (pH units)	8.0000	9.1000	7.8000	11.5000
Phosphorus, total	<0.2000	<0.2000	<0.2000	^c
Selenium, total	<0.0050	^c	<0.0050	<0.0050
Silver	<0.0100	<0.0100	<0.0100	0.1000
Sulfate (as SO ₄)	914.0000	1100.0000	634.0000	1710.0000
Temperature	8.4000	14.7000	16.9000	33.3000
Titanium, total	0.0110	0.0150	<0.0030	0.0180
Total suspended solids	7.0000	10.0000	8.0000	65.0000
Turbidity (NTU) ^f	NA	NA	2.4500	3.3000
Zinc	<0.0200	0.0260	<0.0200	0.3200

^aHolding ponds.

^bLess than (<) values were assigned if more than 50% of the initial observations had (<) values.

^cOne observation.

^dNA = not available.

^eTwo observations.

^fNephelometric turbidity units.

Note: For information concerning sampling frequency and type, see Table A.8 in this report.

Table A.6 (continued)

Parameter	K-1407-J ^a					
	Nov-Dec 1988		Nov-Dec 1989		Year 1989	
	Median	Max	Median	Max	Median	Max
Silver	<0.0100	<0.0100	<0.0100	0.0100	<0.0100	0.0100
Sulfate (as SO ₄)	654.0000	1200.0000	307.5000	471.0000	447.0000	1500.0000
Sulfide (as S)	<1.0000	2.0000	<1.0000	<1.0000	<1.0000	<1.0000
Sulfite (as SO ₃)	<2.0000	<2.0000	<2.0000	<2.0000	<2.0000	<2.0000
Technetium, pCi/L	NA	NA	1304.0000	^c	526.0000	1719.0000
Temperature	9.3000	100.0000	7.8000	11.5000	13.1000	31.9000
Thallium, total	<0.0100	<0.0100	<0.0100	<0.0100	<0.0100	<0.0100
Tin, total	<0.0100	0.0100	<0.0100	0.0100	<0.0100	0.0310
Titanium, total	0.0150	0.0440	0.0080	0.0560	0.0100	0.2200
Total dissolved solids	2619.0000	6924.0000	720.0000	1604.0000	1358.0000	4096.0000
Total organic carbon	12.0000	120.0000	5.0000	8.0000	8.0000	20.0000
Total suspended solids	11.5000	23.0000	9.0000	118.0000	9.0000	359.0000
Uranium	1.0950	2.8000	0.6380	0.9070	0.3380	0.5940
Zinc	0.0500	0.4600	0.0360	0.2000	0.0465	0.3600

^aA basin.

^bNA = not available.

^cOne observation.

^dLess than values (<) were assigned if more than 50% of the initial observations had (<) values.

^eTwo observations.

^fSIE.

Note: For information concerning sampling frequency and type, see Table A.8 in this report.

Table A.7. Median and maximum concentrations (in milligrams per liter unless noted otherwise) of organics routinely monitored in the water from K-1407-J

Parameter	K-1407-J ^a					
	Nov-Dec 1988		Nov-Dec 1989		Year 1989	
	Median	Max	Median	Max	Median	Max
1 1 1-Trichloroethane	<5.0000 ^b	5.0000	6.0000	140.0000	<5.0000	140.0000
1 1 2 2-Tetrachloroethane	<5.0000	<5.0000	<5.0000	5.0000	<5.0000	5.0000
1 1-Dichloroethane	<5.0000	5.0000	<5.0000	20.0000	<5.0000	36.0000
1 1-Dichloroethene	<5.0000	<5.0000	5.0000	100.0000	<5.0000	100.0000
1 2-Dichloroethane	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Benzene	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Bromodichloromethane	<5.0000	5.0000	<5.0000	5.0000	<5.0000	5.0000
Bromoform	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Carbon tetrachloride	<5.0000	19.0000	<5.0000	5.0000	<5.0000	5.0000
Chlorobenzene	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	5.0000
Chloroethane	<10.0000	<10.0000	<10.0000	<10.0000	<10.0000	<10.0000
Chloroform	5.0000	13.0000	5.0000	11.0000	3.0000	11.0000
Chloromethane	<10.0000	<10.0000	<10.0000	<10.0000	<10.0000	<10.0000
Cis-1 3-dichloropropene	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Dibromochloromethane	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Ethylbenzene	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	5.0000
Methylene chloride	<5.0000	5.0000	<5.0000	5.0000	<5.0000	8.0000
Tetrachloroethane	13.0000	96.0000	11.0000	240.0000	5.0000	240.0000
Toluene	<5.0000	5.0000	<5.0000	5.0000	<5.0000	5.0000
Trans-1 2-dichloroethene	<5.0000	6.0000	22.0000	200.0000	<5.0000	200.0000
Trans-1 3-dichloropropene	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000	<5.0000
Trichloroethane	3.0000	17.0000	60.0000	1400.0000	5.0000	1400.0000
Vinyl chloride	<10.0000	<10.0000	<10.0000	12.0000	<10.0000	12.0000

^aA basin.

^bLess than values (<) were assigned if more than 50% of the initial observations had (<) values.

Note: For information concerning sampling frequency and type, see Table A.8 in this report.

Table A.8. Sampling frequency and sample type for regularly monitored water quality parameters as outlined in the National Pollutant Discharge Elimination System Permit

Parameter	Sampling frequency ^a	Sample type ^b
Acid compounds	5	3
Alpha, total	5	4
Aluminum, total	2	1
Ammonia (as N)	1	1
Antimony, total	2	1
Arsenic, total	1	1
Barium, total	2	1
Base/neutral	5	3
Beryllium, total	2	1
Beta, total	5	4
Boron, total	2	1
Bromide	1	1
Cadmium	2	1
Cesium	5	4
Chemical oxygen demand	3	1
Chloride	1	1
Chlorine, total residual	1	1
Chromium	2	1
Cobalt, total	2	1
Copper	1	1
Cyanide	1	2
Flow	6	NA ^c
Fluoride	3	1
Iron, total	2	1
Lead	2	1
Magnesium, total	2	1
Manganese, total	2	1
Mercury, total	2	1
Molybdenum, total	2	1
Neptunium	5	4
Nickel	1	1
Nitrate (as N)	2	1
Nitrogen, total organic	1	1
Oil and grease	2	2

Table A.8 (continued)

Parameter	Sampling frequency ^a	Sample type ^b
pH	6	2
Phenols, total	1	2
Phosphorus, total	1	1
Plutonium	5	4
Polychlorinated biphenyls	1	1
Selenium, total	2	1
Silver	1	1
Sulfate (as SO ₄)	1	1
Sulfide (as S)	1	1
Sulfite (as SO ₃)	1	1
Surfactants	1	1
Technetium	5	4
Temperature	3	2
Thallium, total	1	1
Tin, total	1	1
Titanium, total	2	1
Total dissolved solids	3	1
Total organic carbon	2	1
Total suspended solids	3	1
Turbidity	3	2
Uranium	1	4
Volatile compounds	4	2
Zinc	2	1

^a1 = once per week; 2 = twice per week; 3 = four times per week; 4 = five times per week; 5 = monthly; and 6 = continuous sampling.

^b1 = 24 h per composite; 2 = grab; 3 = 72 h grab per composite; 4 = once per week; 24 h composite was analyzed for uranium.

^c(NA) Not applicable.

Note: The weekly composites were compiled into a monthly sample and analyzed for Cs, Pu, Np, and Tc. Isotopic analysis was conducted on uranium any week that values were above 0.02 mg/L.

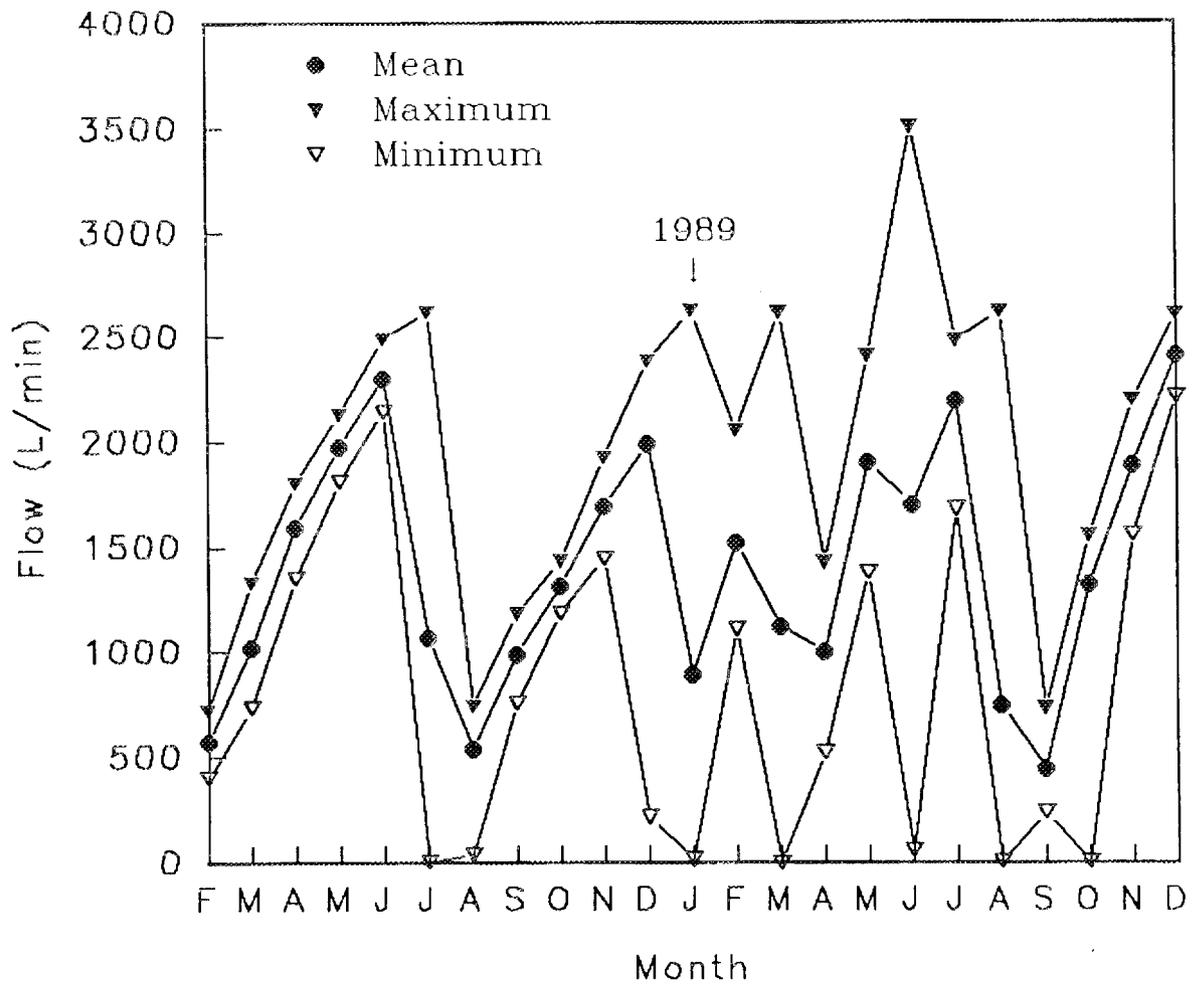


Fig. A.1. Mean monthly streamflow for Mitchell Branch at Mitchell Branch kilometer 0.12 (National Pollutant Discharge Elimination System monitoring station K-1700) below Oak Ridge K-25 Site. Maximum and minimum values for each month are also shown. Source: National Pollutant Discharge Elimination System Monthly Reports.

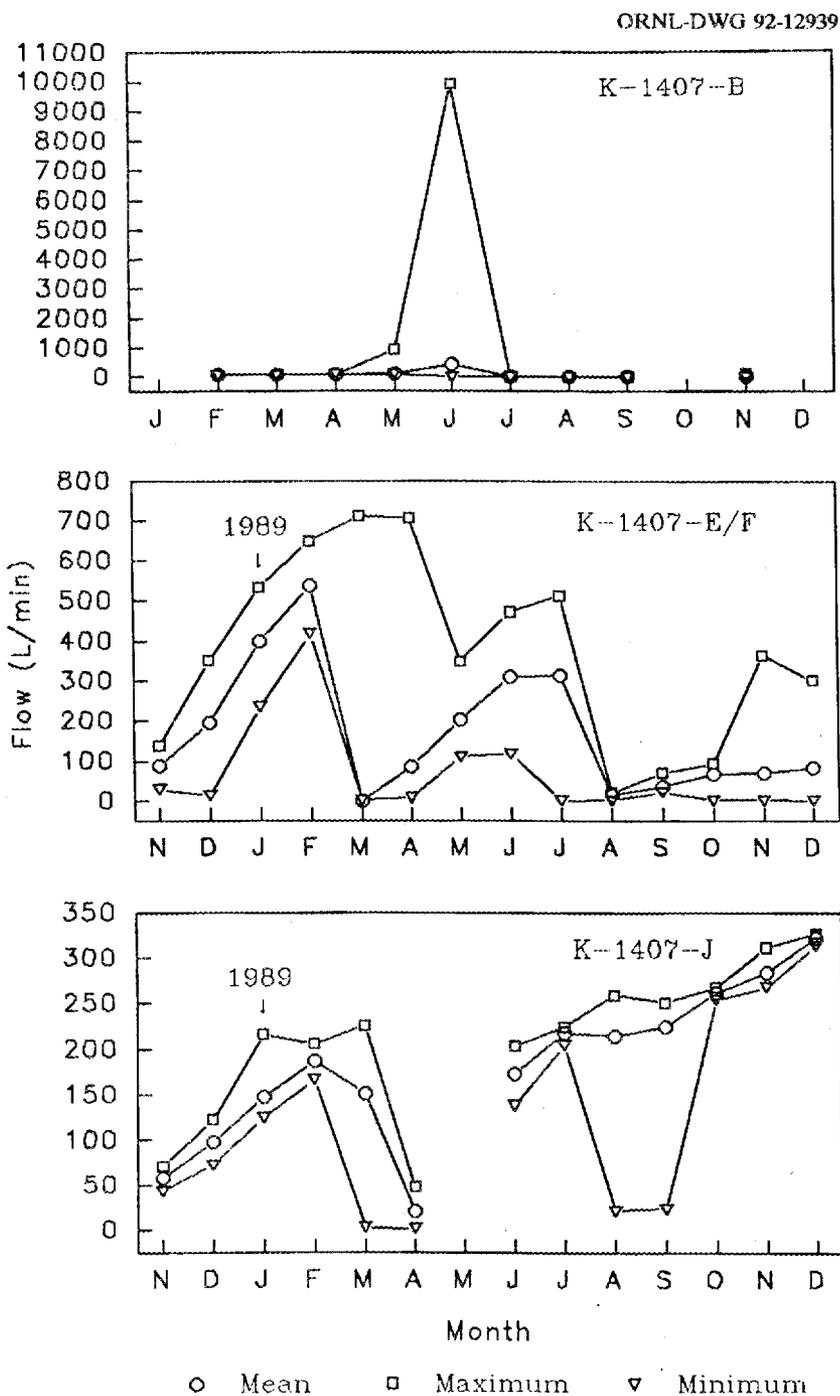


Fig. A.2. Mean monthly discharge from pond K-1407-B for 1988 and from K-1407-E/F and K-1407-J for November 1988–December 1989. Data were not available for K-1407-J from May to June 1989. Pond K-1407-B closed October 1988 and was replaced with K-1407-E/F (nonhazardous effluent) and K-1407-J (hazardous effluent).

Appendix B

**MEAN CONCENTRATIONS OF SELECTED PARAMETERS
IN MITCHELL BRANCH WATER OBTAINED BY THE
OAK RIDGE K-25 SITE PROCESS SUPPORT
DEPARTMENT IN CONJUNCTION WITH
THE TOXICITY TESTS CONDUCTED
DURING JULY 1988 THROUGH
DECEMBER 1989**

Table B.1. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during July 1988

Parameter	Mitchell Branch kilometer		
	1.0	0.54	0.45
Aluminum	0.56 (0.22-1.10)	7.3 (0.58-14)	7.5 (0.95-14)
Calcium	40 (36-47)	60 (43-77)	69 (65-72)
Chloride	15 (4-21)	66 (43-77)	83 (74-92)
Dissolved solids	259 (222-306)	387 (286-488)	471 (450-492)
Fluoride	0.13 (0.1-0.2)	0.35 (0.2-0.5)	0.4 (0.4-0.4)
Iron	0.79 (0.4-1.4)	3.6 (1.6-5.6)	4.7 (3.1-6.2)
Magnesium	9.6 (9.3-10.0)	11.0 (11-12)	11.5 (11-12)
Manganese	0.06 (0.032-0.12)	0.11 (0.08-0.14)	0.17 (0.16-0.17)
Nickel	0.09 (0.08-0.12)	0.19 (0.15-0.22)	0.28 (0.20-0.35)
Silicon	2.1 (0.7-3.8)	6.1 (1.2-11)	8.9 (2.8-15)
Sodium	20 (15-30)	34 (20-48)	41 (37-45)
Sulfate	33 (8-46)	108 (65-151)	141 (118-163)
Suspended solids	13 (3-30)	143 (6-280)	138 (13-262)

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-288, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Two samples were taken from Mitchell Branch kilometer (MIK) 0.54 and 0.45, and 3 were taken from MIK 1.0.

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Table B.2. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during September 1988

Parameter	Mitchell Branch kilometer				
	1.43	0.71	0.54	0.45	0.12
Aluminum	0.45 (0.21-0.64)	0.57 (0.55-0.58)	0.64 (0.27-1.0)	0.41 (0.18-0.79)	0.25 (0.09-0.68)
Calcium	27 (26-27)	38 (36-39)	83 (53-150)	78 (56-130)	70 (53-97)
Chloride	1 (1-1)	25 (24-26)	106 (56-218)	78 (37-168)	58 (28-119)
Dissolved solids	150 (136-172)	242 (240-244)	595 (348-1116)	514 (324-922)	463 (342-664)
Fluoride	BD ^a (<0.1)	0.15 (BD-0.2)	0.89 (0.4-2.0)	0.73 (0.4-1.6)	0.66 (0.4-1.1)
Iron	0.45 (0.23-0.65)	0.60 (0.57-0.63)	0.77 (0.38-1.1)	0.58 (0.26-0.96)	0.39 (0.16-0.76)
Magnesium	13.7 (13-14)	9.3 (9.3-9.3)	11.7 (9.9-15)	12.3 (11-15)	11.7 (10-13)
Manganese	0.05 (0.03-0.07)	0.08 (0.06-0.10)	0.11 (0.06-0.14)	0.12 (0.08-0.16)	0.14 (0.03-0.23)
Nickel	BD (<0.01)	0.03 (0.03-0.03)	0.07 (0.04-0.14)	0.05 (0.04-0.11)	0.043 (0.02-0.07)
Silicon	4.0 (3.7-4.4)	2.4 (2.3-2.4)	3.1 (2.7-3.5)	1.9 (0.8-2.6)	2.4 (0.5-4.1)
Sodium	0.9 (0.7-1.0)	22 (21-22)	75 (37-150)	62 (33-120)	51 (25-80)
Sulfate	2 (2-2)	48 (47-49)	187 (93-414)	174 (97-333)	143 (75-239)
Suspended solids	28 (6-60)	9 (6-11)	16 (5-22)	8 (2-13)	2 (1-6)

^aBD = below detection.

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-288, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken at each locations.

Table B.3. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during November 1988

Parameter	Mitchell Branch kilometer					
	1.43	1.00	0.71	0.54	0.45	0.12
Aluminum	1.01 (0.15-2.8)	0.11 (BD ^a -0.35)	0.89 (BD-5.4)	0.26 (0.05-0.78)	0.26 (0.04-0.70)	0.29 (0.13-0.44)
Calcium	22 (21-24)	55 (30-65)	57 (41-86)	54 (40-87)	60 (40-81)	70 (45-98)
Chloride	BD (<10)	11 (BD-15)	71 (13-296)	74 (17-187)	74 (14-187)	104 (18-279)
Dissolved solids	121 (102-146)	253 (200-274)	639 (220-1752)	640 (250-1636)	547 (238-1252)	358 (116-1620)
Fluoride	BD	0.20 (0.1-0.3)	2.2 (0.2-9.8)	2.1 (0.2-9.0)	1.5 (0.2-6.2)	2.2 (0.3-8)
Iron	0.99 (0.22-2.30)	0.23 (0.09-0.46)	0.74 (0.08-3.9)	0.39 (0.16-1.2)	0.41 (0.14-0.69)	0.51 (0.36-0.76)
Magnesium	13.1 (12-14)	15.6 (9-18)	12.0 (11-14)	13.0 (11-15)	12.6 (11-14)	13.3 (12-16)
Manganese	0.09 (0.05-0.16)	0.05 (0.03-0.15)	0.07 (0.04-0.12)	0.12 (0.09-0.20)	0.16 (0.12-0.25)	0.24 (0.18-0.35)
Nickel	BD (<0.01)	0.01 (BD-0.01)	0.03 (BD-0.12)	0.03 (BD-0.07)	0.02 (BD-0.03)	0.02 (BD-0.04)
Silicon	5.4 (4.5-8.6)	3.5 (1.8-4.1)	3.4 (2-9.3)	2.9 (2.4-3.5)	2.8 (2.5-3.2)	2.9 (2.4-3.3)
Sodium	1 (0.96-1.8)	10 (6.5-23)	148 (17-540)	140 (20-490)	105 (16-340)	136 (20-420)
Sulfate	BD (<10)	56 (51-60)	134 (26-468)	169 (51-412)	142 (49-309)	184 (48-404)
Suspended solids	37 (5-115)	3 (BD-6)	18 (BD-96)	5 (BD-16)	5 (1-15)	5 (3-9)

^aBD = below detection.

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP October-December 1988*, Report No. K/QT-0311, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken at each location.

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Table B.4. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during January 1989

Parameter	Mitchell Branch kilometer					
	1.43	1.00	0.71	0.54	0.45	0.12
Aluminum	0.36 (0.22-0.58)	0.41 (0.29-0.65)	0.32 (0.08-0.54)	0.30 (0.16-0.52)	0.22 (0.11-0.38)	0.20 (0.05-0.35)
Calcium	13 (11-14)	25 (23-27)	38 (33-62)	44 (37-64)	46 (39-65)	52 (43-70)
Chloride	4 (BD ^a -10)	2 (2-2)	19 (8-79)	25 (10-74)	24 (10-72)	27 (11-78)
Dissolved solids	63 (56-74)	125 (112-142)	229 (190-432)	251 (190-410)	254 (204-408)	280 (214-470)
Fluoride	BD (<0.1)	BD	0.14 (BD-0.40)	0.10 (0.1-0.1)	0.11 (0.1-0.2)	0.20 (0.2-0.2)
Iron	0.38 (0.26-0.62)	0.41 (0.34-0.52)	0.32 (0.15-0.49)	0.35 (0.2-0.55)	0.45 (0.32-0.63)	0.50 (0.35-0.68)
Magnesium	7.0 (5.9-8)	9.9 (8.6-11)	10.0 (9.3-12)	10.4 (9.7-12)	10.9 (10-12)	11.7 (11-13)
Manganese	0.05 (0.05-0.07)	0.06 (0.04-0.11)	0.07 (0.06-0.08)	0.19 (0.16-0.24)	0.25 (0.20-0.29)	0.28 (0.22-0.34)
Nickel	BD (<0.01)	0.01 (BD-0.01)	0.05 (0.02-0.23)	0.05 (0.01-0.22)	0.04 (BD-0.19)	0.05 (0.01-0.22)
Silicon	3.8 (3.6-4.1)	3.9 (3.7-4.4)	3.5 (2.8-4.2)	3.5 (3.2-3.8)	3.4 (3.0-4.0)	3.4 (3.0-3.6)
Sodium	1 (1.0-1.2)	2 (1.9-2.4)	17 (9.7-54)	21 (11-53)	20 (10-52)	23 (10-63)
Sulfate	6 (BD-10)	14 (13-14)	44 (32-106)	51 (34-100)	54 (36-102)	21 (6-103)
Suspended solids	4.3 (3-7)	3.1 (2-5)	3.1 (1-5)	4.6 (2-7)	3.6 (2-6)	3.3 (1-5)

^aBD = below detection.

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP January-March 1989*, Report No. K/QT-312, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken from each location.

Table B.5. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during March 1989

Parameter	Mitchell Branch kilometer					
	1.43	1.00	0.71	0.54	0.45	0.12
Aluminum	BD ^a (<0.5)	0.60 (BD-1.2)	0.53 (BD-0.74)	0.58 (BD-0.88)	0.70 (BD-1.2)	0.80 (0.53-0.92)
Calcium	13 (12-15)	26 (23-28)	43 (31-70)	49 (37-76)	52 (43-75)	55 (47-80)
Chloride	BD (<10)	BD	28 (BD-91)	31 (BD-96)	27 (BD-82)	35 (15-92)
Dissolved solids	76 (66-84)	110 (102-118)	231 (146-412)	269 (186-482)	275 (206-472)	299 (215-524)
Fluoride	BD (<0.1)	0.11 (BD-0.2)	0.11 (BD-0.20)	0.11 (0.1-0.2)	0.11 (BD-0.2)	0.23 (0.2-0.4)
Iron	0.22 (BD-0.48)	0.11 (BD-0.2)	0.48 (0.24-1.2)	0.44 (0.33-0.58)	0.44 (0.33-0.57)	0.53 (0.4-0.6)
Magnesium	6.9 (6-7.6)	9.3 (8.2-10)	10.5 (8.6-14)	11.2 (9.4-15)	11.9 (11-15)	12.9 (12-16)
Manganese	0.05 (BD-0.15)	0.04 (BD-0.06)	0.14 (0.08-0.20)	0.21 (0.18-0.25)	0.31 (0.19-0.39)	0.27 (0.22-0.35)
Nickel	BD (<0.25)	BD	0.25 BD-0.28	BD	BD	BD
Silicon	5.3 (4.4-8.4)	5.1 (4.6-6.2)	5.0 (4.2-6.8)	4.7 (4.2-5.5)	5.2 (4.2-6.7)	4.2 (3.4-5.7)
Sodium	4 (1.8-8.7)	5 (4.6-6.2)	27 (9.9-81)	31 (13-78)	32 (14-69)	31 (14-73)
Sulfate	BD (<10)	11 (10-12)	49 (22-136)	48 (BD-116)	52 (32-114)	56 (32-124)
Suspended solids	6.4 (3-11)	4.1 (2-7)	12.7 (BD-57)	9.7 (5-15)	6.6 (3-9)	4.0 (2-6)

^aBD = below detection.

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP January-March 1989*, Report No. K/QT-312, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken from each location.

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Table B.6. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during May 1989

Parameter	Mitchell Branch kilometer					
	1.43	1.00	0.71	0.54	0.45	0.12
Aluminum	0.86 (0.32-3.0)	0.91 (0.44-3.0)	0.42 (0.1-2.0)	0.75 (0.03-4.0)	0.98 (0.11-4.9)	1.05 (0.27-5.4)
Calcium	17 (15-17)	29 (21-30)	39 (20-78)	43 (20-83)	48 (20-89)	54 (17-79)
Chloride	2 (1-2)	3 (2-3)	34 (3-113)	34 (3-115)	30 (8-87)	43 (6-88)
Dissolved solids	91 (86-94)	141 (108-152)	234 (102-476)	273 (138-550)	263 (98-514)	227 (84-378)
Fluoride	BD ^a (<0.1)	BD	BD	0.13 (0.1-0.2)	0.16 (0.1-0.2)	0.36 (0.2-1.3)
Iron	1.00 (0.46-3.3)	1.11 (0.59-2.9)	0.58 (0.19-2.6)	0.96 (0.2-4.4)	0.34 (0.24-5.7)	1.26 (0.53-5.4)
Magnesium	8.8 (5.2-9.9)	10.9 (5.5-13)	8.9 (4.5-12)	9.3 (4.5-13)	9.4 (3.7-13)	9.7 (3.3-12)
Manganese	0.16 (0.07-0.36)	0.20 (0.14-0.3)	0.06 (0.05-0.11)	0.10 (0.09-0.12)	0.17 (0.1-0.25)	0.21 (0.12-0.25)
Nickel	BD (<0.25)	BD	0.01 (BD-0.018)	0.01 (BD-0.021)	0.02 (BD-0.025)	0.01 (BD-0.017)
Silicon	4.8 (4.3-6.6)	5.0 (4.5-6.7)	3.1 (2.6-4.5)	3.7 (2.7-7.5)	4.0 (2.7-8.3)	4.2 (3.1-9.2)
Sodium	1 (0.9-2.1)	3 (1.7-2.8)	22 (2.7-50)	22 (2.4-51)	22 (5.2-49)	31.7 (4-553)
Sulfate	3 (3-5)	14 (9-17)	64 (12-213)	67 (15-217)	70 (20-195)	78 (18-152)
Suspended solids	16.3 (3-74)	14.9 (4-48)	10.1 (BD-46)	12.1 (BD-69)	20.9 (BD-88)	15.4 (1-86)

^aBD = below detection.

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP April-June 1989*, Report No. K/QT-313, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken from each location.

Table B.7. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during July 1989

Parameter	Mitchell Branch kilometer					
	1.43	1.00	0.71	0.54	0.45	0.12
Aluminum	NM ^a	0.50 (0.15-0.85)	0.10 (0.07-0.18)	0.29 (0.08-1.3)	0.36 (0.11-0.36)	0.22 (0.10-0.4)
Calcium	19 (19-20)	42 (34-79)	61 (44-79)	58 (43-73)	64 (55-75)	67 (56-87)
Chloride	1 (1-1)	8 (2-45)	35 (15-57)	32 (21-56)	32 (23-44)	40 (20-92)
Dissolved solids	117 (96-136)	209 (158-450)	336 (246-460)	366 (250-660)	348 (246-580)	349 (242-610)
Fluoride	BD ^b (<0.1)	0.13 (BD-0.3)	0.27 (0.2-0.041)	0.13 (0.1-0.2)	0.19 (0.1-0.3)	0.21 (0.2-0.3)
Iron	0.49 (0.41-0.61)	0.99 (0.51-1.7)	0.17 (0.1-0.5)	0.54 (0.18-2.4)	0.80 (0.35-2.2)	0.57 (0.21-0.88)
Magnesium	11.1 (11-12)	14.1 (13-15)	12.4 (11-14)	11.9 (11-13)	13 (12-13)	12.7 (12-132)
Manganese	0.10 (0.07-0.13)	0.30 (0.18-0.61)	0.13 (0.1-0.19)	0.19 (0.10-0.41)	0.28 (0.24-0.28)	0.30 (0.11-0.35)
Nickel	0.01 (BD-0.02)	0.01 (BD-0.02)	0.02 (0.02-0.04)	0.02 (0.01-0.05)	0.02 (0.02-0.03)	0.02 (0.02-0.03)
Silicon	4.1 (3.9-4.3)	4.6 (3.6-5.2)	3.6 (3.5-3.7)	3.8 (3.3-5.3)	3.8 (3.5-4.6)	3.5 (3.4-3.8)
Sodium	0.8 (0.7-0.8)	8 (12-44)	31 (16-53)	35 (15-110)	32 (13-87)	40 (46-228)
Sulfate	2 (2-2)	30 (13-131)	92 (43-166)	105 (48-246)	101 (48-246)	126 (46-228)
Suspended solids	13 (3-41)	14 (2-33)	2 (BD-4)	15 (BD-96)	12 (1-45)	5 (1-8)

^aNM = not measured.

^bBD = below detection.

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-342, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken from each location.

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Table B.8. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during September 1989

Parameter	Mitchell Branch kilometer					
	1.43	1.00	0.71	0.54	0.45	0.12
Aluminum	0.95 (0.2-1.9)	1.01 (0.47-1.6)	1.32 (0.22-2.9)	1.90 (0.11-8.1)	1.44 (0.08-6.2)	3.5 (0.16-14)
Calcium	20 (17-22)	35 (30-41)	50 (12-94)	56 (30-85)	56 (28-77)	53 (24-74)
Chloride	2 (1-2)	2 (2-3)	30 (4-116)	32 (10-96)	27 (7-73)	16 (6-32)
Dissolved solids	121 (104-134)	176 (138-204)	262 (110-524)	277 (150-480)	301 (140-418)	250 (116-362)
Fluoride	BD ^a (<0.1)	0.12 (BD-0.2)	0.16 (BD-0.53)	0.19 (BD-0.3)	0.20 (0.1-0.3)	0.24 (0.2-0.3)
Iron	1.1 (0.4-2.2)	1.10 (0.7-1.7)	0.16 (0.28-4.1)	2.24 (0.14-9.6)	2.01 (0.3-8.3)	4.07 (0.45-16)
Magnesium	10.0 (8.5-13)	11.4 (6.3-17)	10.2 (4.7-12)	10.6 (6.3-12)	10.3 (5.3-12)	10.4 (5.5-13)
Manganese	0.12 (0.07-0.17)	0.14 (0.07-0.33)	0.14 (0.07-0.3)	0.14 (0.08-0.24)	0.19 (0.13-0.28)	0.27 (0.17-0.34)
Nickel	BD (<0.01)	BD	0.03 (0.01-0.06)	0.02 (BD-0.03)	0.02 (0.02-0.03)	0.03 (0.02-0.04)
Silicon	5.0 (4.2-6.1)	5.2 (4.1-6)	5.2 (3.1-7.6)	6.0 (3.1-13)	5.2 (3.4-9.8)	8.1 (3.7-17)
Sodium	1 (0.7-2.0)	3 (2.1-3.8)	16 (2.8-41)	18 (7.1-38)	14 (5.5-26)	10 (3.7-17)
Sulfate	5 (2-7)	21 (14-27)	52 (2.8-41)	61 (33-107)	58 (27-90)	48 (24-75)
Suspended solids	20 (6-46)	11 (BD-30)	27 (BD-88)	41 (BD-185)	43 (4-222)	74 (6-319)

^aBD = below detection.

Source: M. A. McGaha, 1989, *Toxicity Monitoring at ORGDP July-September 1988*, Report No. K/QT-342, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken from each location.

Table B.9. Mean (range in parentheses) concentrations of selected parameters (in milligrams per liter) in Mitchell Branch water obtained by the Oak Ridge K-25 Site Process Support Department in conjunction with the toxicity test conducted during December 1989

Parameter	Mitchell Branch kilometer					
	1.43	1.00	0.71	0.54	0.45	0.12
Aluminum	0.21 (0.13-0.41)	0.29 (0.18-0.51)	0.22 (0.12-0.33)	0.26 (0.17-0.38)	0.20 (0.15-0.31)	0.46 (0.24-0.65)
Calcium	14 (13-15)	26 (24-28)	40 (32-65)	46 (37-66)	48 (39-70)	54 (44-76)
Chloride	1 (1.1-1.2)	2 (1.5-3)	23 (8-91)	26 (9.4-79)	28 (12-82)	34 (18-87)
Dissolved solids	51 (34-82)	136 (108-162)	238 (159-456)	259 (178-424)	273 (206-444)	302 (230-468)
Fluoride	BD ^a (<0.1)	BD	BD	BD	BD	
Iron	0.29 (0.22-0.45)	0.32 (0.23-0.46)	0.21 (0.14-0.35)	0.26 (0.19-0.39)	0.35 (0.31-0.45)	0.7 (0.44-0.94)
Magnesium	7.6 (6.8-8.2)	9.8 (8.9-11)	10.2 (9.1-13)	11 (9.5-13)	11 (10-13)	12 (11-14)
Manganese	0.13 (0.12-0.14)	0.14 (0.13-0.18)	0.08 (0.07-0.1)	0.15 (0.14-0.17)	0.22 (0.19-0.24)	0.28 (0.23-0.31)
Nickel	BD (<0.01)	0.01 (BD-0.03)	0.01 (BD-0.02)	BD	0.02 (0.02-0.04)	0.02 (0.02-0.05)
Silicon	3.9 (3.6-5)	4.0 (3.7-4.4)	3.2 (2.9-3.5)	3.6 (3.2-4)	3.3 (3-3.7)	3.6 (3.2-4)
Sodium	1 (0.9-1.1)	2 (1.7-2.0)	16 (6.6-62)	21 (8.2-58)	19 (7.5-53)	21 (10-56)
Sulfate	4 (3.2-4.1)	11 (10-15)	41 (21-124)	49 (26-102)	54 (30-111)	55 (29-114)
Suspended solids	7.4 (4-12)	6 (2-8)	2 (BD-2)	4.9 (2-10)	6.8 (2-12)	13 (5-22)

^aBD = below detection.

Source: J. L. Shoemaker, 1990, *Toxicity Monitoring at ORGDP October-December 1989*, Report No. K/QT-373, Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee.

Note: Seven samples were taken from each location.

Appendix C

**RESULTS OF QUALITY ASSURANCE/QUALITY CONTROL
ANALYSES OF FISH SAMPLES**

C. RESULTS OF QUALITY ASSURANCE/QUALITY CONTROL ANALYSES OF FISH SAMPLES

Ten pairs of duplicate samples of fish muscle were analyzed for mercury. The mean coefficient of variation (CV) between sample pairs was 21%, with a mean standard deviation (SD) of 0.07 $\mu\text{g/g}$. The mean absolute difference between duplicate samples was 0.10 $\mu\text{g/g}$. Multiple analyses of EPA reference fish ($n = 6$) averaged ($\pm\text{SD}$) 2.54 \pm 0.09 $\mu\text{g/g}$, vs an expected value of 2.52 $\mu\text{g/g}$. Mercury concentrations in sunfish ($n = 10$) from reference sites averaged 0.07 \pm 0.02 $\mu\text{g/g}$.

The results of PCB analyses were somewhat more variable than those from mercury analyses. The mean coefficient of variation between 11 sample pairs was 36%, with a mean standard deviation of 0.15 $\mu\text{g/g}$. The mean absolute differ-

ence between sample pairs was 0.36 $\mu\text{g/g}$. Recoveries of mixtures of PCB-1254 and PCB-1260 spiked into samples of uncontaminated fish and clams averaged 105 \pm 9%. PCB concentrations in fish ($n = 11$) from the uncontaminated reference sites averaged 0.05 \pm 0.05 $\mu\text{g/g}$.

Other quality assurance samples, including split sample exchanges among laboratories at TVA, Tennessee Department of Health and Environment, and EPA, are run as part of BMAP studies for ORNL and the Y-12 Plant. Results of these evaluations are applicable to K-25 site samples and are available in Loar (1991b, 1992, 1993) and Hinzman et al. 1993.

Appendix D

**CONCENTRATIONS OF MERCURY AND PCBs
IN INDIVIDUAL FISH FROM MITCHELL
BRANCH, POPLAR CREEK, AND
NEARBY REACHES OF THE
CLINCH RIVER**

Table D.1. Concentrations of mercury and PCBs (micrograms per gram wet weight) in resident fish from Mitchell Branch and nearby reaches of Poplar Creek and the Clinch River, 1987-1990

Site ^a	Date	Spp ^b	Sex ^c	No. ^d	Wt ^e	Lgth ^f	Hg	PCB ^g	1254 ^h	1260 ⁱ
PCK 6.9	06/01/87	REDBRE	M	7025	141.5	19.5	0.38	0.46	0.17	0.29
PCK 6.9	06/01/87	REDBRE	F	9177	48.4	13.6	0.19	0.18	0.06	0.12
PCK 6.9	06/01/87	REDBRE	F	9157	41.0	13.3	0.27	0.34	0.16	0.18
PCK 6.9	06/01/87	REDBRE	F	7638	15.2	9.6		0.40	0.12	0.28
PCK 6.9	06/01/87	BLUGIL	F	7052	93.0	17.6	0.57	0.32	0.13	0.19
PCK 6.9	06/01/87	BLUGIL	F	7054	71.1	15.6	0.57	0.05	0.03	0.02
PCK 6.9	06/01/87	BLUGIL	M	7061	102.1	17.0	0.38	0.19	0.04	0.15
PCK 6.9	06/01/87	BLUGIL	M	7056	85.3	16.7	0.24	0.09	0.02	0.07
PCK 6.9	06/01/87	BLUGIL	F	7057	104.2	17.3	0.65	0.53	0.15	0.38
PCK 6.9	06/01/87	BLUGIL	F	7055	63.5	15.3	0.27	0.29	0.18	0.11
PCK 6.9	06/01/87	BLUGIL	F	7077	75.5	15.9	0.38	0.14	0.07	0.07
PCK 6.9	06/01/87	BLUGIL	F	7078	42.8	13.8	0.33	0.11	0.03	0.08
PCK 10.4	11/18/87	BLUGIL	M	6348	77.2	16.0	0.05	0.11	0.03	0.08
PCK 10.4	11/18/87	BLUGIL	F	6349	59.1	15.0	0.13	0.04	0.02	0.02
PCK 10.4	11/18/87	BLUGIL	F	6350	72.0	16.0	0.09	0.04	0.02	0.02
PCK 10.4	11/18/87	BLUGIL	F	6351	71.3	15.0	0.10	0.07	0.03	0.04
PCK 10.4	11/18/87	BLUGIL	F	6352	39.0	13.5	0.09	0.03	0.02	0.01
PCK 10.4	11/18/87	BLUGIL	M	6353	37.7	12.8	0.10	0.06	0.02	0.04
PCK 10.4	11/18/87	BLUGIL	F	6354	45.4	13.5	0.16	0.08	0.04	0.04
PCK 10.4	11/18/87	BLUGIL	F	6355	64.6	15.4	0.11	0.02	0.01	0.02
PCK 8.2	11/18/87	BLUGIL	M	6340	59.4	14.6	0.22	0.21	0.13	0.08
PCK 8.2	11/18/87	BLUGIL	F	6341	45.9	14.2	0.69	0.25	0.10	0.15
PCK 8.2	11/18/87	BLUGIL	M	6342	28.1	11.8	0.38	0.27	0.12	0.15
PCK 8.2	11/18/87	BLUGIL	F	6343	40.5	13.2	0.51	0.15	0.06	0.09
PCK 8.2	11/18/87	BLUGIL	F	6344	84.1	17.3	0.60	0.14	0.07	0.07
PCK 8.2	11/18/87	BLUGIL	M	6345	91.5	17.2	0.33	0.36	0.12	0.24
PCK 8.2	11/18/87	BLUGIL	F	6346	57.0	14.7	0.50	0.12	0.04	0.08
PCK 8.2	11/18/87	BLUGIL	F	6347	66.5	15.3	0.21	0.11	0.05	0.06
PCK 6.9	11/18/87	BLUGIL	M	6332	60.7	15.0	0.28	0.22	0.12	0.10
PCK 6.9	11/18/87	BLUGIL		6333	41.7	13.9	0.66	0.35	0.01	0.34
PCK 6.9	11/18/87	BLUGIL	M	6334	45.3	13.5	0.50	0.20	0.11	0.09
PCK 6.9	11/18/87	BLUGIL	M	6335	55.9	14.4	0.42	0.25	0.12	0.13
PCK 6.9	11/18/87	BLUGIL	M	6336	48.6	14.2	0.26	0.18	0.05	0.13
PCK 6.9	11/18/87	BLUGIL	M	6337	71.6	15.4	0.33	0.29	0.13	0.16
PCK 6.9	11/18/87	BLUGIL	F	6338	70.7	15.8	0.50	0.14	0.06	0.08
PCK 6.9	11/18/87	BLUGIL	F	6339	95.9	17.2	0.48	0.47	0.09	0.38
PCK 1.6	12/08/87	BLUGIL	M	7043	135.9	18.9	0.22	0.17	0.06	0.11
PCK 1.6	12/08/87	BLUGIL	M	7044	97.9	18.0	0.35	0.18	0.08	0.10
PCK 1.6	12/08/87	BLUGIL	M	7045	85.8	16.7	0.30	0.13	0.05	0.08
PCK 1.6	12/08/87	BLUGIL	F	7046	76.1	16.2	0.06	0.10	0.06	0.04
PCK 1.6	12/08/87	BLUGIL	F	7047	56.2	15.2	0.11	0.17	0.08	0.09
PCK 1.6	12/08/87	BLUGIL	M	7048	99.5	18.2	0.09	0.14	0.02	0.12
PCK 1.6	12/08/87	BLUGIL	M	7049	58.7	16.3	0.07	0.37	0.11	0.28
PCK 1.6	12/08/87	BLUGIL	M	7050	109.5	18.8	0.06	0.12	0.03	0.09

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Table D.1 (continued)

Site ^a	Date	Spp ^b	Sex ^c	No. ^d	Wt ^e	Lgth ^f	Hg	PCB ^g	1254 ^h	1260 ⁱ
CRK 15.0	11/18/87	BLUGIL	F	6324	57.3	15.3	0.10	0.06	0.03	0.03
CRK 15.0	11/18/87	BLUGIL	M	6325	73.6	15.9	0.11	0.10	0.03	0.07
CRK 15.0	11/18/87	BLUGIL	M	6326	33.9	12.8	0.06	0.07	0.04	0.03
CRK 15.0	11/18/87	BLUGIL	F	6327	65.0	15.1	0.11	0.06	0.03	0.03
CRK 15.0	11/18/87	BLUGIL	M	6328	57.6	14.8	0.11	0.06	0.02	0.04
CRK 15.0	11/18/87	BLUGIL	F	6329	65.1	15.3	0.31	0.04	0.01	0.03
CRK 15.0	11/18/87	BLUGIL	M	6330	67.4	15.5	0.10	0.06	0.02	0.04
CRK 15.0	11/18/87	BLUGIL	M	6331	69.5	15.8	0.23	0.15	0.01	0.14
MHL 38	11/11/87	BLUGIL	M	6308	72.2	15.9	0.03	0.03	0.02	0.01
MHL 38	11/11/87	BLUGIL	F	6309	51.9	15.0	0.10	0.02	0.02	0.01
MHL 38	11/11/87	BLUGIL	M	6310	53.9	14.8	0.03	0.06	0.06	0.01
MHL 38	11/11/87	BLUGIL	M	6311	61.0	14.8	0.03	0.03	0.01	0.02
MHL 38	11/11/87	BLUGIL	F	6312	34.7	13.0	0.03	0.02	0.02	0.01
MHL 38	11/11/87	BLUGIL	M	6313	31.3	12.5	0.03	0.02	0.01	0.01
MHL 38	11/11/87	BLUGIL	F	6314	108.8	18.0	0.04	0.02	0.02	0.01
MHL 38	11/11/87	BLUGIL	M	6315	72.8	16.1	0.02	0.04	0.03	0.01
PCK 10.4	11/16/88	BLUGIL	M	7594	88.1	16.6	0.03	0.01	0.01	0.001
PCK 10.4	11/16/88	BLUGIL	F	7595	66.5	15.5	0.05	0.01	0.001	0.01
PCK 10.4	11/16/88	BLUGIL	M	7590	105.4	18.3	0.05	0.02	0.02	0.001
PCK 10.4	11/16/88	BLUGIL	F	7591	54.5	14.6	0.24	0.26	0.13	0.13
PCK 10.4	11/16/88	BLUGIL	F	7593	73.2	15.7	0.03	0.03	0.03	0.001
PCK 10.4	11/16/88	BLUGIL	M	7597	90.0	16.0	0.05	0.03	0.02	0.01
PCK 10.4	11/16/88	BLUGIL	M	7592	72.1	15.6	0.04	0.01	0.01	0.001
PCK 10.4	11/16/88	BLUGIL	M	7596	67.9	15.8	0.13	0.01	0.001	0.01
PCK 8.2	11/16/88	BLUGIL	M	7580	80.8	16.2	0.23	0.59	0.45	0.14
PCK 8.2	11/16/88	BLUGIL	M	7581	112.3	17.8	0.33	0.55	0.49	0.06
PCK 8.2	11/16/88	BLUGIL	M	7582	86.3	16.8	0.34	0.49	0.40	0.09
PCK 8.2	11/16/88	BLUGIL	M	7583	63.8	15.0	0.40	0.18	0.14	0.04
PCK 8.2	11/16/88	BLUGIL	F	7584	75.7	16.0	0.54	0.22	0.13	0.09
PCK 8.2	11/16/88	BLUGIL	M	7585	63.8	15.0	0.32	0.11	0.07	0.04
PCK 8.2	11/16/88	BLUGIL	M	7586	69.1	15.2	0.54	0.24	0.17	0.07
PCK 8.2	11/16/88	BLUGIL	F	7587	98.4	17.4	0.59	0.08	0.05	0.03
PCK 6.9	11/16/88	BLUGIL	F	7570	73.4	16.6	0.54	0.17	0.14	0.03
PCK 6.9	11/16/88	BLUGIL	M	7571	93.9	17.1	0.09	0.14	0.12	0.02
PCK 6.9	11/16/88	BLUGIL	M	7572	54.7	15.0	0.28			
PCK 6.9	11/16/88	BLUGIL	M	7573	49.6	14.2	0.36	0.39	0.25	0.14
PCK 6.9	11/16/88	BLUGIL	F	7574	51.2	14.5	0.39	0.17	0.12	0.05
PCK 6.9	11/16/88	BLUGIL	M	7575	67.9	15.8	0.31	0.40	0.23	0.17
PCK 6.9	11/16/88	BLUGIL	M	7576	60.2	15.3	0.09	0.08	0.06	0.02
PCK 6.9	11/16/88	BLUGIL	F	7577	50.2	14.2	0.27	0.15	0.12	0.03
PCK 1.6	11/16/88	BLUGIL	F	7650	68.2	15.9	0.24	0.14	0.11	0.03
PCK 1.6	11/16/88	BLUGIL	F	7651	49.2	14.6	0.24	0.23	0.16	0.07
PCK 1.6	11/16/88	BLUGIL	M	7652	78.8	16.0	0.15	0.30	0.26	0.04
PCK 1.6	11/16/88	BLUGIL	M	7653	55.2	14.2	0.14	0.39	0.37	0.02
PCK 1.6	11/16/88	BLUGIL	M	7654	70.0	15.5	0.06	0.06	0.05	0.01
PCK 1.6	11/16/88	BLUGIL	F	7655	59.2	14.8	0.16	0.11	0.09	0.02

Table D.1 (continued)

Site ^a	Date	Spp ^b	Sex ^c	No. ^d	Wt ^e	Lgth ^f	Hg	PCB ^g	1254 ^h	1260 ⁱ
PCK 1.6	11/16/88	BLUGIL	M	7656	59.6	14.6	0.11	0.18	0.12	0.06
PCK 1.6	11/16/88	BLUGIL	F	7657	61.2	14.5	0.22	0.14	0.09	0.05
CRK 15.0	11/16/88	BLUGIL	M	7643	90.5	17.5	0.08	0.06	0.03	0.03
CRK 15.0	11/16/88	BLUGIL	M	7648	69.9	15.6	0.10	0.12	0.08	0.04
CRK 15.0	11/16/88	BLUGIL	M	7647	53.5	14.5	0.02	0.17	0.15	0.02
CRK 15.0	11/16/88	BLUGIL	M	7641	83.6	16.2	0.16	0.01	0.001	0.01
CRK 15.0	11/16/88	BLUGIL	M	7640	54.8	15.0	0.20	0.22	0.22	0.001
CRK 15.0	11/16/88	BLUGIL	M	7645	104.5	17.8	0.16	0.17	0.16	0.01
CRK 15.0	11/16/88	BLUGIL	M	7642	109.6	17.8	0.28	0.08	0.02	0.06
CRK 15.0	11/16/88	BLUGIL	M	7644	78.2	16.4	0.11	0.05	0.03	0.02
PCK 6.9	03/07/90	BLUGIL	M	5230	80.3	17.3	0.68	0.10	0.001	0.10
PCK 6.9	03/07/90	BLUGIL	M	5231	63.4	16.0	0.43	0.12	0.04	0.08
PCK 6.9	03/07/90	BLUGIL	M	5232	114.8	19.4	0.33	0.17	0.001	0.17
PCK 6.9	03/07/90	BLUGIL	M	5233	69.4	16.4	0.43	0.14	0.07	0.07
PCK 6.9	03/07/90	BLUGIL	M	5234	42.6	14.2	0.33	0.07	0.03	0.04
PCK 6.9	03/07/90	BLUGIL	M	5235	83.1	17.4	0.42	0.07	0.001	0.07
PCK 6.9	03/07/90	BLUGIL	M	5236	59.2	15.5	0.30	0.15	0.09	0.06
PCK 6.9	03/07/90	BLUGIL	M	5237	58.5	15.4	0.51	0.17	0.05	0.12
PCK 8.2	03/07/90	BLUGIL	M	5240	71.0	16.3	0.14	0.08	0.03	0.05
PCK 8.2	03/07/90	BLUGIL	M	5241	70.7	16.0	0.33	0.07	0.03	0.04
PCK 8.2	03/07/90	BLUGIL	M	5242	89.5	17.4	0.43	0.02	0.001	0.02
PCK 8.2	03/07/90	BLUGIL	M	5243	77.5	16.7	0.30	0.06	0.02	0.04
PCK 8.2	03/07/90	BLUGIL	F	5244	65.4	15.9	0.65	0.05	0.02	0.03
PCK 8.2	03/07/90	BLUGIL	M	5245	71.3	15.8	0.36	0.11	0.01	0.10
PCK 8.2	03/07/90	BLUGIL	M	5246	81.5	17.2	0.24	0.02	0.01	0.01
PCK 8.2	03/07/90	BLUGIL	M	5247	100.8	18.0	0.45	0.15	0.05	0.10
MIK 0.2	05/06/87	REDBRE	I	MB1	5.7	6.8	0.18			
MIK 0.2	05/06/87	REDBRE	I	MB2	8.6	8.0	0.16			
MIK 0.2	05/06/87	REDBRE	I	MB4	4.5	6.5	0.17			
MIK 0.2	05/06/87	REDBRE	I	MB7	4.4	6.5	0.18			
MIK 0.2	05/06/87	REDBRE	I	MB8	4.4	6.1	0.18			
MIK 0.2	05/06/87	REDBRE	I	MB10	3.3	5.9	0.15			
MIK 0.2	05/06/87	REDBRE	I	MB12	2.5	5.3	0.17			
MIK 0.2	03/23/88	BLUGIL	F	7912	120.3	17.5	0.40			
MIK 0.2	03/23/88	REDBRE	M	7913	9.3	7.9	0.09			
MIK 0.2	03/23/88	REDBRE	I	7914	4.0	5.9	0.13			
MIK 0.2	03/23/88	REDBRE	I	7917	3.3	5.7	0.14			
MIK 0.2	03/23/88	REDBRE	I	7921	3.3	5.5	0.12			
MIK 0.2	03/23/88	REDBRE	I	7922	3.1	5.5	0.11			
MIK 0.2	03/29/89	REDBRE	M	7377	10.4	8.1	0.12			
MIK 0.2	03/29/89	REDBRE	F	7378	3.0	5.2	0.22			
MIK 0.2	03/29/89	REDBRE	M	7379	7.1	7.1	0.17			
MIK 0.2	03/29/89	REDBRE	M	7479	37.8	13.0	0.20	3.27	2.73	0.54
MIK 0.2	03/29/89	REDBRE	F	7450	34.6	12.0	0.18	0.97	0.79	0.18
MIK 0.2	03/29/89	REDBRE	M	7451	38.8	12.9	0.18	1.70	1.21	0.49
MIK 0.2	03/29/89	REDBRE	F	7453	36.3	12.9	0.21	1.93	1.52	0.41
MIK 0.2	03/29/89	REDBRE	F	7454	46.7	13.0	0.22	0.50	0.38	0.12
MIK 0.2	03/29/89	REDBRE	M	7452	11.3	8.4		2.41	2.22	0.19

Table D.1 (continued)

Site ^a	Date	Spp ^b	Sex ^c	No. ^d	Wt. ^e	Lgth ^f	Hg	PCB ^g	1254 ^h	1260 ⁱ
MIK 0.2	03/29/89	REDBRE	M	7455	19.1	9.9		0.91	0.79	0.12
MIK 0.2	03/29/89	REDBRE	M	7456	32.1	11.6		1.16	0.82	0.34
MIK 0.2	01/11/90	REDBRE	M	5210	68.3	15.9	0.38	0.85	0.39	0.46
MIK 0.2	01/11/90	REDBRE	F	5211	68.5	15.5	0.31	0.45	0.36	0.09
MIK 0.2	01/11/90	REDBRE	M	5212	36.1	13.0	0.22	1.55	1.09	0.46
MIK 0.2	01/16/90	REDBRE	F	5217	59.7	16.0	0.39	0.23	0.18	0.05
MIK 0.2	01/11/90	REDBRE	M	5213	6.9	7.0	0.11			
MIK 0.2	01/16/90	REDBRE	F	5200	38.6	13.4	0.26	0.71	0.51	0.20
MIK 0.2	01/16/90	REDBRE	F	5201	29.5	12.6	0.25	0.87	0.57	0.30
MIK 0.2	01/16/90	REDBRE	F	5218	20.7	11.8	0.21	0.74	0.57	0.17
MIK 0.2	01/16/90	REDBRE	M	5202	15.0	9.9		1.00	0.73	0.27
MIK 0.2	01/16/90	BLUGIL	F	5219	72.6	16.5	0.31	0.74	0.46	0.28

^aSite: PCK = Poplar Creek kilometer, MIK = Mitchell Branch kilometer, CRK = Clinch River kilometer, MHL = Melton Hill Lake. Numerical designation is the distance from the mouth of the stream.

^bSpecies (SPP): Bluegil = bluegill (*Lepomis macrochirus*), and Redbre = redbreast sunfish (*Lepomis auritus*).

^cSex: M = male, F = female, and I = immature.

^dNo. = Fish identification number.

^eWt = Fish weight, grams.

^fLgth = Fish total length, centimeters.

^gPCB = Total PCBs (sum of PCB-1254 and PCB-1260) in fish axial muscle, micrograms per gram wet weight.

^h1254 = PCB-1254 (Arochlor 1254) in fish axial muscle, micrograms per gram wet weight.

ⁱ1260 = PCB-1260 (Arochlor 1260) in fish axial muscle, micrograms per gram wet weight.

Table D.2. Detection limits of organic compounds
(in micrograms per gram, wet weight)

Compound	Detection limit ^a
Capillary column GC/MS	
Phenol	<2.0
Bis(2-chloroethyl) ether	<2.0
2-Chlorophenol	<2.0
1,3-Dichlorobenzene	<2.0
1,4-Dichlorobenzene	<2.0
Benzyl alcohol	<2.0
1,2-Dichlorobenzene	<2.0
2-Methylphenol	<2.0
Bis(2-chlorodisopropyl)ether	<2.0
4-Methylphenol	<2.0
N-nitroso-di-N-propylamine	<2.0
Hexachloroethane	<2.0
Nitrobenzene	<2.0
Isophorone	<2.0
2-Nitrophenol	<2.0
2,4-Dimethylphenol	<2.0
Benzoic acid	<10.0
Bis(2-chloroethoxy)methane	<2.0
2,4-Dichlorophenol	<2.0
1,2,4-Trichlorobenzene	<2.0
Naphthalene	<2.0
4-Chloroaniline	<2.0
Hexachlorobutradiene	<2.0
4-Chloro-3-methylphenol	<2.0
2-Methylnaphthalene	<2.0
Hexachlorocyclapentadiene	<2.0
2,4,6-Trichlorophenol	<2.0
2,4,5-Trichlorophenol	<10.0
2-Chloronaphthalene	<2.0
2-Nitroaniline	<10.0
Dimethylphthalate	<2.0
Acenaphthalene	<2.0
3-Nitroaniline	<10.0
Acenaphthene	<2.0
2,4-Dinitrophenol	<10.0
Nitrophenol	<10.0

Table D.2. (continued)

Compound	Detection limit ^a
Dibenzofuran	<2.0
2,4-Dinitrotoluene	<2.0
2,6-Dinitrotoluene	<2.0
Diethylphthalate	<2.0
4-Chlorophenyl-phenylether	<2.0
Fluorene	<2.0
4-Nitroaniline	<10.0
4,6-Dinitro-2-methylphenol	<10.0
N-nitrosodiphenylamine	<2.0
4-bromophenyl-phenylether	<2.0
Hexachlorobenzene	<2.0
Pentachlorophenol	<10.0
Phenanthrene	<2.0
Anthracene	<2.0
Di-N-butylphthalate	<2.0
Fluoranthene	<2.0
Pyrene	<2.0
Butylbenzylphthalate	<2.0
3,3-Dichlorobenzidene	<10.0
Benz[<i>a</i>]anthracene	<2.0
Bis(2-ethylhexyl)phthalate	<2.0
Chrysene	<2.0
Di-N-octylphthalate	<2.0
Benzo[<i>b</i>]fluoranthene	<2.0
Benzo[<i>k</i>]fluoranthene	<2.0
Benzo[<i>a</i>]pyrene	<2.0
Indeno[1,2,3- <i>cd</i>]pyrene	<2.0
Dibenz[<i>a,h</i>]anthracene	<2.0
Benzo[<i>g,h,i</i>]perylene	<2.0

Capillary column GC/ECD

Alpha-bhc	<0.02
Beta-bhc	<0.04
Delta-bhc	<0.04
Gamma-bhc	<0.02
Heptachlor	<0.04
Aldrin	<0.04

Table D.2. (continued)

Compound	Detection limit ^a
Heptachlor epoxide	<0.04
Endosulfan i	<0.04
Dieldrin	<0.04
4,4'-Dde	<0.04
Endrin	<0.2
Endosulfan ii	<0.08
4,4'-Ddd	<0.2
Endosulfan sulfate	<0.2
4,4'-Ddt	<0.08
Endrin ketone	<0.4
Methoxychlor	<0.2
Alpha chlordane	<0.04
Gamma chlordane	<0.04
Toxaphene	<2
HPLC with fluorescence detection	
Naphthalene	<0.12
Acenaphthene	<0.03
Phenanthrene	<0.01
Anthracene	<0.10
Fluoranthene	<0.2
Pyrene	<0.003
Benz[a]anthracene	<0.001
Benzo[b]fluoranthene	<0.03
Benzo[k]fluoranthene	<0.02
Benzo[a]pyrene	<0.004
Dibenz[a,h]anthracene	<0.002
Benzo[g,h,i]perylene	<0.005
Indeno[1,2,3-cd]pyrene	<1.0
Chrysene	<0.003

^aOptimum detection limit in 10-g sample. Sample weights varied between 5 and 10 g, so detection limits were higher in some samples. Concentrations of contaminants approximately ten times lower than these levels can be detected but are outside the limits of reliable quantification.

Appendix E

**CHECKLIST OF BENTHIC MACROINVERTEBRATE
TAXA COLLECTED FROM MITCHELL BRANCH
AUGUST 1987 THROUGH JANUARY 1988**

Table E.1. Checklist of benthic macroinvertebrate taxa collected from Mitchell Branch, August 1987 through January 1988

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Turbellaria						
Planariidae	X ^a		X		X	
Planariidae?					X	
Nematoda	X	X	X		X	X
Oligochaeta	X	X	X	X	X	X
Crustacea						
Copepoda	X	X	X			X
Ostracoda			X			
Isopoda						
Asellidae						
<i>Asellus</i>	X					
<i>Lirceus</i>	X					
Amphipoda						
Gammaridae						
<i>Crangonyx</i>	X					
Decapoda						
Cambaridae	X					
Hydracarina	X					
Hydracarina?					X	
Insecta						
Collembola					X	
Entomobryomorpha	X	X	X	X	X	X
Sminthuridae	X		X			X
Ephemeroptera						
Baetidae						
<i>Baetis</i>	X	X				
<i>Cloeon</i>	X					
Ephemerellidae						
<i>Eurylophella</i>	X					
Ephemeridae						
<i>Ephemera</i>	X					
Ephemeridae?	X					
Leptophlebiidae	X					
<i>Habrophlebiodes</i>	X					
<i>Habrophlebiodes?</i>	X					

Table E.1 (continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Ephemeroptera (con.t)						
<i>Lepthophlebia</i>	X					
<i>Paraleptophlebia</i>	X					
<i>Paraleptophlebia?</i>	X					
Siphonuridae						
<i>Ameletus</i>	X					
Odonata						
Anisoptera	X					
Aeshnidae						
<i>Boyeria vinosa</i>			X			
Cordulegastridae						
<i>Cordulegaster</i>	X					
<i>Cordulegaster?</i>	X					
Gomphidae	X			X		
<i>Lanthus?</i>	X					
<i>Stylogomphus</i>						
<i>albistylus</i>	X					
Libellulidae						
<i>Libellula?</i>						X
Libellulidae?	X					
Zygoptera				X		X
Calopterygidae						
<i>Calopteryx</i>	X		X			
Coenagrionidae			X	X	X	X
<i>Argia</i>			X	X	X	X
<i>Enallagma</i>					X	X
<i>Enallagma/Ischnura</i>			X		X	
<i>Enallagma?</i>						X
<i>Ischnura?</i>						X
Coenagrionidae?	X			X		
Plecoptera						
Capniidae		X				
<i>Allocapnia</i>	X					
Leuctridae						
<i>Leuctra</i>		X				
Nemouridae						
<i>Amphinemura</i>	X					
<i>Nemoura</i>		X				
<i>Nemoura?</i>	X					

Table E.1 (continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Plecoptera (cont.)						
Perlidae		X				
Perlodidae	X					
<i>Clioperla clio</i>	X					
<i>Isoperla?</i>	X					
Perlodidae?	X					
Hemiptera						
Microveliidae						
<i>Microvelia</i>	X					
<i>Microvelia?</i>				X	X	
Notonectidae						
<i>Notonecta</i>					X	
Megaloptera						
Corydalidae	X					
<i>Nigronia</i>						
<i>fasciatus</i>	X					
Sialidae						
<i>Sialis</i>	X					
Trichoptera						
Hydropsychidae						X
<i>Diplectrona</i>						
<i>modesta</i>	X	X				
Limnephilidae	X					
<i>Goera</i>	X					
<i>Neophylax</i>	X					
<i>Pycnopsyche</i>	X					
Molannidae						
<i>Molanna</i>	X					
Phryganeidae?						X
Coleoptera						
Dryopidae						
<i>Helichus</i>	X					
Dytiscidae?						X
Elmidae	X					
<i>Dubiraphia</i>	X					
<i>Optioservus</i>	X	X	X			X
<i>Stenelmis</i>	X	X	X	X		

Table E.1 (continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Coleoptera (cont.)						
Hydrophilidae	X					
Hymenoptera		X				
Diptera		X				
Ceratopogonidae	X	X	X	X	X	X
<i>Atrichopogon</i>			X	X		
Chaoboridae						
<i>Chaoborus</i>					X	X
Chironomidae	X	X		X	X	X
Tanypodinae	X	X	X		X	X
<i>Ablabesmyia</i>		X				
<i>Ablabesmyia?</i>			X			X
<i>Labrundinia</i>			X		X	
<i>Larsia</i>		X				
<i>Larsia?</i>	X	X			X	
<i>Natarsia</i>	X					
<i>Nilotanypus</i>		X				
<i>Procladius</i>					X	X
<i>Tanypus</i>					X	X
<i>Telopelopia?</i>	X					
<i>Thienemannimyia</i> gp	X	X	X			X
<i>Thienemannimyia?</i>		X	X			
<i>Zavreliomyia</i>	X					
<i>Zavreliomyia?</i>	X					
Tanypodinae?	X					
Podonominae						
<i>Paraboreochlus</i>	X					
Diamesinae						
<i>Syndiamesa</i>		X				
<i>Syndiamesa?</i>		X				
Orthocladiinae	X		X			
<i>Brillia?</i>	X					
<i>Cardiocladius?</i>						X
<i>Chaetocladius</i>		X	X		X	X
<i>Corynoneura</i>	X	X	X		X	X
<i>Cricotopus/</i>						
<i>Orthocladius</i>	X	X	X	X	X	X
<i>Cricotopus/</i>						
<i>Orthocladius?</i>	X				X	X

Table E.1 (continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Chironomidae (cont.)						
<i>Diplocladius</i>	X					
<i>Gymnometriocnemus?</i>					X	
<i>Heleniella</i>	X					
<i>Heterotrissocladius</i>	X					
<i>Limnophyes</i>			X			
<i>Limnophyes?</i>			X		X	
<i>Nanocladius?</i>	X					
<i>Parakiefferiella</i>	X					
<i>Parakiefferiella?</i>	X					
<i>Parametriocnemus</i>	X	X		X		X
<i>Parametriocnemus?</i>	X					X
<i>Psilometriocnemus?</i>	X					
<i>Rheocricotopus?</i>	X					
<i>Smittia</i>		X				X
<i>Thienemanniella</i>		X				
Orthocladiinae?	X					
Chironominae	X				X	X
Chironomini						
<i>Chironomus</i>			X		X	X
<i>Chironomus/</i> <i>Goeldichironomus</i>					X	
<i>Chironomus?</i>			X	X	X	
<i>Cryptochironomus</i>	X				X	X
<i>Cryptotendipes</i>	X					
<i>Cryptotendipes?</i>	X					
<i>Dicrotendipes</i>					X	X
<i>Endochironomus</i>	X					
<i>Endochironomus?</i>	X					
<i>Glyptotendipes</i>				X	X	X
<i>Goeldichironomus</i>		X			X	X
<i>Goeldichironomus?</i>	X				X	
<i>Microtendipes</i>	X	X	X		X	X
<i>Microtendipes?</i>				X		
<i>Paralauterborniella</i>	X					
<i>Paralauterborniella?</i>	X					
<i>Paratendipes</i>	X	X				
<i>Phaenopsectra</i>	X					
<i>Polypedilum</i>	X	X	X	X		
<i>Stictochironomus</i>	X		X		X	
Chironominae?	X		X			

Table E.1 (continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Chironomidae (cont.)						
Tanytarsini	X					
<i>Cladotanytarsus?</i>					X	
<i>Micropsectra</i>	X					
<i>Micropsectra?</i>	X	X	X			X
<i>Paratanytarsus</i>					X	
<i>Rheotanytarsus</i>	X	X				X
<i>Rheotanytarsus?</i>		X				
<i>Stempellina</i>	X					
<i>Stempellinella</i>	X	X				
Tanytarsini?	X					
Culicidae						
<i>Anopheles</i>	X					
Dolichopodidae?	X					
Dixidae						
<i>Dixa</i>		X				
Empididae	X					
<i>Hemerodromia</i>	X	X			X	X
Psychodidae						
<i>Pericoma</i>	X					
Simuliidae						
<i>Prosimulium</i>	X					
<i>Simulium</i>	X		X		X	X
Stratiomyidae						
<i>Stratiomys</i>			X			X
Tabanidae						
<i>Chrysops</i>	X					
Tipulidae	X	X			X	
<i>Hexatoma</i>	X					
<i>Limnophila</i>	X					
<i>Limnophila?</i>	X					
Limoniinae		X				
<i>Ormosia?</i>	X					
<i>Pilaria</i>	X					
<i>Pilaria?</i>	X					
<i>Pseudolimnophila</i>	X					
<i>Pseudolimnophila?</i>	X					
<i>Tipula</i>	X	X		X		X
<i>Tipula</i>						
<i>abdominalis</i>	X					
Tipulidae?	X					

Table E.1 (continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Mollusca						
Gastropoda	X				X	
Ancylidae						
<i>Ferrissia</i>	X					
Lymnaeidae						X
Lymnaeidae?			X			X
Physidae						
<i>Physella</i>			X			
Bivalvia						
Sphaeriidae	X					
<i>Pisidium</i>	X					
<i>Pisidium?</i>	X					
<i>Sphaerium</i>	X					

*An "X" denotes that the taxon was collected at least once in quantitative samples.

Note: MIK = Mitchell Branch kilometer.

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