DOE/OR/01-1393/V1&D3

Remedial Investigation/Feasibility Study of the Clinch River/Poplar Creek Operable Unit

Volume 1. Main Text

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This volume incorporates minor changes in response to regulators' comments about the D2 issue of this document. Vertical lines in the outside margins in the Executive Summary and in Chaps. 1, 5, 7, 8, 9, and 10 indicate passages in which such changes have been made.

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ORNL/ER-315/V1&D3

Energy Systems Environmental Restoration Program

Remedial Investigation/Feasibility Study of the Clinch River/Poplar Creek Operable Unit

Volume 1. Main Text

Date Issued—June 1996

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Oak Ridge, Tennessee
(under subcontract DE-AC05-93OR22028)

Prepared for the
U.S. Department of Energy
Office of Environmental Management
under budget and reporting code EW 20

Environmental Management Activities at the
OAK RIDGE Y-12 PLANT
Oak Ridge, Tennessee 37831-8169
managed by
LOCKHEED MARTIN ENERGY SYSTEMS, INC.
for the
U.S. DEPARTMENT OF ENERGY
under contract DE-AC05-84OR21400

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ABBREVIATIONS

AET	apparent effect threshold
ALT	alanine aminotransferase
ANC	American Nuclear Corporation
ANOVA	analysis of variance
ARARs	applicable or relevant and appropriate requirements
AWQC	ambient water quality criteria
BEDS	Biological Effects Database for Sediments
BEIAS	Biomedical and Environmental Information Analysis
BMAP	Biological Monitoring and Abatement Program
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CCC	chronic continuous criterion
CFR	*Code of Federal Regulations*
CLP PQL	Contract Laboratory Program Practical Quantification Limit
CNF	Central Neutralization Facility
COC	chemical of concern
COPC	contaminant of potential concern
COPEC	contaminants of potential ecological concern
CR-ERP	Clinch River Environmental Restoration Program
CRM	Clinch River mile
CR/PC	Clinch River/Poplar Creek
CRRI	Clinch River Remedial Investigation
CV	chronic value
DNA	deoxyribonucleic acid
DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
DQO	data quality objective
EC	effective concentration
ECR	excess cancer risk
EFPC	East Fork Poplar Creek
EMP	Environmental Monitoring Program
EPA	U.S. Environmental Protection Agency
EPRI	Electric Power Research Institute
EPT	ephemeroptera, plecotera, tricoptera
EQ_Part	equalibrium partitioning benchmark
ER	Environmental Restoration
ER-L	effects range—low
ER-M	effects range—medium
EROD	7-ethoxyresorufin O-deethylase
FDA	Food and Drug Administration
FFA	Federal Facility Agreement
FS	Feasibility Study
FY	fiscal year
GI	gastrointestinal
GSI	gonadal-somatic index

HAI	Health Assessment Index
HEAST	Health Effects Assessment Summary Tables
HI	hazard index
HQ	hazard quotient
IAEA	International Atomic Energy Agency
i.p.	intraperitoneal
IRIS	Integrated Risk Information System
LC	lethal concentration
LCL_{95}	95% lower confidence limit
LCV	lowest chronic value
LEL	lowest effect level
LET	linear energy transfer
LHS	Latin Hypercube Sampling
LLW	low-level waste
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed-effect concentration
LSI	liver-somatic index
LTV	lowest test value
M&M	monitoring and maintenance
MCL	maximum contaminant level
MOE	Ministry of Environment (Ontario)
msl	mean sea level
NAWQC	national ambient water quality criteria
NCP	National Contingency Plan
NCRP	National Council on Radiation Protection
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NOEL	no-observed-effect level
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
ORGDP	Oak Ridge Gaseous Diffusion Plant
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
ORREMP	Oak Ridge Reservation Enviornmental Monitoring Program
OU	operable unit
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyl (commercially marketed as Aroclor mixtures)
PCM	Poplar Creek mile
PLE	product limit estimate
PNL	Pacific Northwest Laboratory
PORTS	Portsmouth Gaseous Diffusion Plant
QA	quality assurance
RA	remedial action
RAO	remedial action objectives
RCRA	Resource Conservation and Recovery Act
RD	remedial design
RfC	reference concentration

RfD	reference dose
RI/FS	remedial investigation/feasibility study
RGO	remedial goal option
ROD	Record of Decision
RME	reasonable maximum exposure
SAV	secondary acute value
s.c.	subcutaneous
SCV	secondary chronic value
SF	slope factor
SI	sensitivity index
SQAG	Sediment Quality Assessment Guideline
SQC	sediment quality criteria
SSNAEL	site-specific no apparent effects level
SWSA	solid waste storage area
T&E	threatened and endangered
TBC	to be considered
TCLP	Toxicity Characteristics Leaching Procedure
TDEC	Tennessee Department of Environment and Conservation
TIE	toxicity evaluation and identification
TKN	total Kjeldahl nitrogen
TOC	total organic carbon
TRE	trivalent rare earths
TRM	Tennessee River mile
TSCA	Toxic Substances Control Act
TU	toxic unit
TVA	Tennessee Valley Authority
TWRA	Tennessee Wildlife Resources Agency
UCB	upper confidence bound
UCL_{95}	95% upper confidence limit
USACE	U.S. Army Corps of Engineers
USDA	U.S. Department of Agriculture
VSI	visceral-somatic index
WAC	waste acceptance criteria
WBR	Watts Bar Reservoir
WOC	White Oak Creek
%GI	percent gastrointestinal absorption efficiency
ΣTU	sum of toxic units

EXECUTIVE SUMMARY

This report presents the findings of an investigation into contamination of the Clinch River and Poplar Creek near the U.S. Department of Energy's (DOE's) Oak Ridge Reservation (ORR) in eastern Tennessee. For more than 50 years, various hazardous and radioactive substances have been released to the environment as a result of operations and waste management activities at the ORR. In 1989, the ORR was placed on the National Priorities List (NPL), established and maintained under the federal Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA). Under CERCLA, NPL sites must be investigated to determine the nature and extent of contamination at the site, assess the risk to human health and the environment posed by the site, and, if necessary, identify feasible remedial alternatives that could be used to clean the site and reduce risk. To facilitate the overall environmental restoration effort at the ORR, CERCLA activities are being implemented individually as distinct operable units (OUs). This document is the combined Remedial Investigation and Feasibility Study Report for the Clinch River/Poplar Creek OU.

This report is organized into five volumes, the first of which presents the main text. Chapter 1 describes the regulatory setting, and Chapter 2 broadly portrays the environmental setting. Chapter 3 depicts the operational and release history of the site and characterizes in detail the nature and extent of contamination. Chapter 4 briefly identifies other regulatory requirements that are applicable or appropriate to the site. Chapters 5 and 6 assess the risk to human health and the environment, respectively. Chapter 7 explains the purpose and organization of the feasibility study. Chapter 8 defines remedial action objectives for the site; identifies pathways and contaminants of concern; and screens general response actions, potential remedial technologies, and process options. Chapter 9 develops remedial alternatives based on the remedial action objectives, the screened technologies, and representative process options. Chapter 10 analyzes, evaluates, and compares the remedial alternatives. Chapter 11 lists the references cited in the main text.

Volumes 2–5 consist of appendices that contain supporting data and information. Volume 2 characterizes the biota on the ORR (Appendix A) and summarizes data related to contaminant concentrations in water (Appendix B), in sediment (Appendix C), and in biota (Appendix D). Volume 3 presents information related to the human health risk assessment (Appendix E) and the ecological risk assessments (Appendix F). Volume 4 focuses on the feasibility study, detailing the selection of remedial process options (Appendix G) and providing the basis for the cost estimates for each remedial alternative (Appendix H). Volume 4 (Appendix I) additionally presents the applicable or relevant and appropriate requirements (ARARs), which help define the extent of the remedial response. Volume 5 is a compilation of data from individual studies that were conducted as part of the overall remedial investigation. As such, the volume addresses the quality assurance objectives for measuring the data (Appendix J) and presents selected historical data (Appendix K), data from several discrete water characterization studies (Appendix L), data supporting the sediment characterization (Appendix M), and data related to several biota characterization studies (Appendix N).

BACKGROUND

The ORR is a 34,600-acre tract of land in Anderson and Roane counties, Tennessee. It is administered by DOE, and it houses three main facilities: the Oak Ridge Y-12 Plant, the K-25 Site (formerly known as the Oak Ridge Gaseous Diffusion Plant), and Oak Ridge National Laboratory (ORNL). Each facility was created in the early 1940s as part of the U.S. government's war effort. The K-25 Site was used for the large-scale production of enriched uranium until its shutdown in 1985. The Y-12 Plant had several missions, but it primarily manufactured nuclear weapons components; production there ended in 1992. ORNL was initially a pilot-scale plant for the production of plutonium, but its post-war mission has centered on nuclear reactor research and the production of radionuclides for use in medicine and science. In addition to these operations, each plant has housed large support operations, including maintenance shops; waste treatment, storage, and disposal areas; steam plants; storm and sewer drains; and infrastructure.

The Clinch River/Poplar Creek OU is located adjacent to the ORR and consists of the Clinch River and several tributary embayments in Melton Hill and Watts Bar reservoirs. Both reservoirs are large multipurpose impoundments created and maintained by the Tennessee Valley Authority. The OU extends from the upstream boundary of the ORR at Clinch River mile (CRM) 49 in Melton Hill Reservoir, downstream to the mouth of the Clinch River in Watts Bar Reservoir at Kingston. It also includes several embayments that extend up tributary streams, including the McCoy Branch embayment of Melton Hill Reservoir and the Poplar Creek embayment [up to Poplar Creek mile (PCM) 5.5] of Watts Bar Reservoir. Originally, the OU included all of Watts Bar Reservoir downstream of the confluence of the Clinch and Tennessee rivers, but this area was segregated into a new OU (the lower Watts Bar Reservoir OU) in 1994, and a CERCLA Record of Decision was reached in 1995. No action-based remedial alternatives were implemented in lower Watts Bar Reservoir. The OU is currently being monitored to ensure that exposure to contaminants remains low.

REMEDIAL INVESTIGATION

The remedial investigation had two primary objectives: (1) to characterize the nature and extent of contamination, and (2) to assess the baseline risk to human health and the environment. Under CERCLA, if site risks are too high, remedial action is generally warranted. This investigation was implemented in a phased approach. First, existing environmental data were used to develop a preliminary site model, which considered the known or suspected contaminant sources, the physical characteristics of the site, and the environmental fate of various contaminants. An initial round (Phase 1) of limited sampling of water, sediment, and fish was then conducted (in 1989) to confirm these historical data and to refine the site model. A much more extensive sampling effort (Phase 2) was conducted in 1994 to more definitively meet the objectives.

Nature and Extent of Contamination

Several contaminant sources were included in the site model. The waters of Poplar Creek were known to receive effluent from the Y-12 Plant (and the City of Oak Ridge) via East Fork Poplar Creek (EFPC), which enters the Poplar Creek at PCM 5.5. Large quantities of elemental mercury were released from the Y-12 Plant in the late 1950s, and small quantities currently continue to escape from contaminated buildings, equipment, and soils. Increased levels of mercury, therefore, were predicted in water, sediment, and biota of Poplar Creek downstream of EFPC. (Contamination at the Y-12 Plant and contamination in the EFPC floodplain have been addressed separately in efforts at other ORR OUs). Other contaminants known to have been released from the Y-12 Plant include uranium, polychlorinated biphenyls (PCBs), and several metals. Poplar Creek also has historically received a variety of effluents from the K-25 Site, through which the creek flows. Numerous metals, uranium, PCBs, laboratory chemicals, and organic solvents are thought to have been released from the site. In addition, the downstream reaches of Poplar Creek formerly received coal ash from the K-770 steam plant at the K-25 Site, and sediment at this location was expected to contain elevated levels of several metals, particularly arsenic.

The contaminants of potential concern in the Clinch River below Melton Hill Dam are primarily man-made radionuclides, by-products of nuclear fission. Contaminants were released to the Clinch River via White Oak Creek, which enters at CRM 20.8. Studies in the 1960s demonstrated that water-soluble radionuclides were rapidly and greatly diluted upon entering the Clinch River and were quickly transported downstream, with little loss of contaminant mass (i.e., they remained in solution). However, those contaminants that adsorbed to particulate matter became bound to particles of suspended sediment and accumulated in areas of sediment deposition. Earlier studies indicated that the principal radionuclide of potential concern in Clinch River sediment at the beginning of this investigation was ^{137}Cs, which is strongly particle-associated and has a relatively long (30-year) half-life. Because peak releases of ^{137}Cs from ORNL occurred at the same time as peak releases of mercury from the Y-12 Plant, peak concentrations of each were known to co-occur in the lower Clinch River, buried under several inches of cleaner sediment. Although one would have expected sediment in the Clinch River below White Oak Creek to contain the highest levels of ^{137}Cs and other radionuclides, there was actually very little sediment in this portion of the river, most having been scoured and transported downstream by the periodic high-volume releases of water from Melton Hill Dam, located approximately 2 miles upstream.

Current contaminant releases from ORNL are much lower than those of the 1950s and 1960s and are largely due to leaching or runoff from waste disposal areas. Most of these areas are no longer in use and are themselves the focus of environmental restoration efforts at ORNL.

Fly ash from the Y-12 steam plant was formerly disposed of in a settling pond located near the headwaters of McCoy Branch on Chestnut Ridge. As a result, several contaminants associated with coal ash, particularly arsenic, were known to be present at elevated levels in surface water and sediment in the McCoy Branch embayment of Melton Hill Reservoir. Because the embayment is bisected by a road built on fill material, conditions were expected to be worse in the upper embayment, whose water had limited mixing (via a culvert) with waters of the lower embayment and the main reservoir.

In addition to these ORR-specific concerns, it was known that fish collected on and near the reservation contained more PCBs than fish found at most upstream reference areas. PCBs had been used at each of the three facilities. The ORR as a whole has likely been a source of PCBs to the environment. However, PCBs have been widely used in transformers and in industrial operations, and numerous potential sources exist throughout eastern Tennessee. The identification of sources is difficult because PCBs bioaccumulate in fish and other organisms to much greater levels than in water or sediment, where they are largely undetected. The extent to which the ORR had contributed to the problem was unclear.

The knowledge of these site conditions was used to guide the remedial investigation. Much of the sampling focused on Poplar Creek, where the combination of multiple sources and site conditions (e.g., areas of significant sedimentation, less water volume than in the Clinch River) were expected to result in some of the highest levels of contamination. Sampling in the Clinch River focused on fish and sediment, media in which contaminants tend to accumulate. Sediment sampling was limited in the Clinch River between Melton Hill Dam and Poplar Creek because sediment was scarce there.

The results of the site characterization phase of the remedial investigation were consistent with the site model. The nature and extent of contamination were evaluated by identifying those study reaches in which levels of any contaminant in water, sediment, or biota were elevated in comparison with levels in upstream reference reaches. The nature and extent of contamination are described as follows.

- **Arsenic in surface water and sediment of upper McCoy Branch Embayment.** Average concentrations of arsenic (4.1 μg/L) in surface water exceeded the state of Tennessee's recreation-based Ambient Water Quality Criterion. This criterion is designed to protect persons who regularly consume fish taken from a particular body of water. In sediment, elevated levels of arsenic, vanadium, and boron were found throughout McCoy Branch Embayment, but concentrations were highest in the upper embayment.

- **Radionuclide levels in water, sediment, and biota of the Clinch River downstream of White Oak Creek.** Average gross alpha and gross beta levels and mean activities of ^{90}Sr and ^{3}H in surface water were a factor of ten higher than reference values. These data were extremely variable, probably as a result of the extreme variability in flow below Melton Hill Dam. A conservative evaluation of the radionuclide concentrations indicated that, even immediately below White Oak Creek, the state's Ambient Water Quality Criterion for protection of domestic supplies was not exceeded.

  Levels of ^{137}Cs were elevated in Clinch River sediment below the mouth of White Oak Creek. This radionuclide has a strong affinity for particles, particularly the clay minerals that make up a significant portion of Clinch River sediment. However, the discharge of water from Melton Hill Dam resulted in the scouring of most of the sediment from this portion of the river, creating larger inventories of ^{137}Cs in the lower Clinch River, where sedimentation is greater.

  Bluegill sunfish and largemouth bass collected in 1989 from the Clinch River near the mouth of White Oak Creek contained ^{137}Cs at levels 100 times that of fish from upstream reference areas. Catfish were found to have levels approximately ten times that of reference areas. Although elevated, these levels were not thought to pose a significant risk to persons or wildlife consuming these fish, and thus radionuclide analysis was discontinued after the initial round of sampling. However, the species-specific baseline human health risk assessment has identified ^{137}Cs as a contaminant of concern in largemouth bass. Additional bass and sunfish will be collected to determine whether concentrations have dropped since the Phase 1 data were collected.

- **Mercury in surface water, sediment, and biota of Poplar Creek downstream of East Fork Poplar Creek.** Average mercury concentrations in surface water, sediment, and biota were significantly elevated in Poplar Creek downstream of EFPC in comparison with average values upstream of EFPC. Elevated concentrations (up to 0.19 $\mu g/L$) measured in Poplar Creek surface water below EFPC and in the Clinch River downstream of Poplar Creek exceed the state's Criterion Continuous Concentration (CCC) (0.012 $\mu g/L$). This criterion is designed to protect aquatic life from chronic exposure to mercury. Although also elevated above reference values, mean mercury levels in fish did not exceed the Federal Drug Administration's action level (1.0 mg/kg) in any species sampled. Several individual largemouth bass, however, had mercury levels that exceeded this value. Increased body burdens of mercury were also found in benthic organisms living in Poplar Creek, in heron eggs and chicks from a rookery near Poplar Creek, and in laboratory mink fed a diet high in fish from Poplar Creek. A decreasing gradient of biological effects, as measured by a suite of physiological and physical indices, was found to extend from upper Poplar Creek downstream through the Clinch River. This gradient in effects can be roughly correlated with a decreasing gradient in fish body burdens of mercury and PCBs in the downstream direction.

- **Metals and radionuclides in the sediment of Poplar Creek.** In addition to mercury, contaminants in Poplar Creek sediment that were elevated above reference levels were silver, arsenic, boron, cadmium, copper, chromium, nickel, vanadium, ^{238}U, ^{235}U, ^{234}U, ^{99}Tc, ^{137}Cs, and ^{60}Co. PCBs (Aroclor 1254), rarely detected in sediment anywhere in the system, were detected in Poplar Creek. As with mercury, concentrations of copper, cadmium, and chromium increased immediately below EFPC and likely represented releases from the Y-12 Plant. Concentrations of silver, nickel, ^{99}Tc, and the uranium isotopes were elevated below K-25 discharge points, and copper and chromium concentrations in this area were substantially increased above the already elevated levels found below EFPC. Increased levels of arsenic, vanadium, and boron were found in lower Poplar Creek and were associated with an area where the disposal of coal ash from the K-770 steam plant historically took place. The increased levels of ^{137}Cs and ^{60}Co were restricted to the last mile of Poplar Creek and are thought to be caused by backflow from the Clinch River, which regularly takes place as a result of reservoir operations.

- **PCBs in fish of Poplar Creek and the Clinch River.** Mean PCB levels in largemouth bass were highest in Poplar Creek. Although no bass were available from the reference reach of Poplar Creek, concentrations were still greater than at most other study and reference sites.

  Mean concentrations of PCBs in catfish were highest in Phase 1 samples collected from the White Oak Creek Embayment (now part of a separate OU) and in fish from the Clinch River immediately downstream. Levels were significantly increased over those in Melton Hill and Norris Reservoir catfish. Mean concentrations in catfish from Poplar Creek below the confluence with EFPC were greater than those in catfish from above the confluence. Mean total PCB concentrations in largemouth bass did not exceed the FDA action level (2.0 mg/kg) at any location. The mean concentration in catfish did not exceed this action level at any location (except at the White Oak Creek Embayment). However, individual fish from the Clinch River and Poplar Creek had concentrations that exceeded this level.

The PCBs detected in fish flesh were almost exclusively Aroclor 1254 and Aroclor 1260. An analysis of individual PCB congeners in catfish did not reveal any patterns that could explain additional sources of PCB contamination.

Risk Assessment

The data used to characterize the nature and extent of contamination were also used to meet the second objective of the remedial investigation, risk assessment. The baseline risk assessment contained in this report consists of a human health risk assessment and an ecological risk assessment.

Risk to human health was evaluated for seven exposure scenarios, each of which contained one or more pathways through which exposure actually occurs. The seven scenarios were (1) the use of surface water as an untreated drinking water source, (2) the consumption of fish, (3) the use of the reservoir shoreline during winter drawdown, (4) swimming, (5) the hunting and consumption of waterfowl that frequent the ORR, (6) the dredging and subsequent land disposal of sediment, and (7) the use of surface water for irrigation. In each scenario, risk from carcinogens was assessed by assuming a 30-year exposure duration, and risk from noncarcinogens was assessed by assuming a 6-year exposure period. Under CERCLA, media whose pathways result in either a cumulative excess cancer risk of 1.0E−04 or a Hazard Quotient of 1.0 (a measure of noncarcinogenic exposure) generally warrant remedial action at the site.

The human health risk assessment evaluated the risk from each contaminant for which sufficient data existed to obtain a representative concentration. Therefore, the human health risk assessment identified certain analytes whose presence did not appear to be the result of the ORR operations. Because these contaminants might have contributed significantly to overall risk, they generally were included in the risk assessment.

Thirty-five potential contaminants of concern were identified in Clinch River and Poplar Creek water, sediment, and fish. The majority of contaminants were found in deep sediment, and the greatest risks were identified through the agricultural pathways. Of the contaminants identified, only 2 in water, 7 in fish, and 19 in sediment were clearly site-related.

Eight contaminants of concern, all noncarcinogens, were identified in surface water in the OU. Five were identified in the drinking water scenario; the non-site-related analyte manganese drove the risk in all reaches except in upper McCoy Branch, where arsenic contributed most of the risk. Seven contaminants of concern were identified in the irrigation scenario, and two were identified in the swimming scenario (Poplar Creek only). The two contaminants identified in the swimming scenario (Di-n-octylphthalate and Aroclor 1254, also identified in the irrigation scenario) were detected infrequently and therefore might not be contaminants of concern. In general, the number of contaminants of concern in each scenario was greatest in Poplar Creek and least in McCoy Branch.

Seven contaminants of concern were identified in the shoreline-use scenario, which was based on contaminant concentrations in near-shore sediment only. Melton Hill Reservoir is managed in such a way that no prolonged drawdown occurs; therefore, assessment of this scenario was not conducted for the area. Noncarcinogenic risk was common throughout near-shore areas along both the Clinch River and Poplar Creek, and it was driven almost exclusively by manganese via inhalation of resuspended sediment. A significant (<1.0E−04) excess cancer risk existed in one subreach of Poplar Creek, primarily because of chromium exposure via the inhalation pathway.

Eleven contaminants of concern were found in fish. Most of the contaminants were organic compounds (PCBs and pesticide residues) and were found in catfish and largemouth bass from both the Clinch River and Poplar Creek. The excess cancer risk from the consumption of catfish exceeded 1.0E−03 in all study reaches, primarily as a result of Aroclor 1260. The excess cancer risk from the ingestion of largemouth bass was generally equal to or less than one half of that from the ingestion of catfish. Arsenic and Aroclor 1260 were the primary contributors to carcinogenic risk from the ingestion of largemouth bass. Several radionuclides were also identified as carcinogenic contaminants of concern, but they generally contributed only a small portion of the total risk. The exception was ^{137}Cs, which contributed a significant portion of the risk associated with the ingestion of largemouth bass from the Clinch River immediately below the mouth of White Oak Creek. Contaminants of concern that were important noncarcinogens included mercury, Aroclor 1254, and chlordane in one or more species in Poplar Creek and the Clinch River.

In the dredging scenario, 31 contaminants of concern were identified in sediment. This scenario assessed the risk from contaminant exposure that would occur if dredge spoil were placed on land where it was accessible to humans. Several direct exposure pathways were evaluated, as were several agricultural scenarios in which contaminant concentrations in produce, milk, and beef were modeled from sediment contaminant concentrations.

Of the direct pathways, external exposure to gamma-emitting radionuclides in spoil from throughout most of the Clinch River (including Melton Hill Reservoir) and at the mouth of Poplar Creek would result in an excess cancer risk greater than 1.0E−04. At all locations downstream of White Oak Creek, the primary contributor to risk via external exposure was ^{137}Cs. In Melton Hill Reservoir, the risk was primarily due to ^{60}Co from a non-DOE source (now closed) on Braden Branch. In all reaches for which data were available, manganese in spoil posed the greatest risk to adults and children via the inhalation of resuspended sediment. Barium similarly posed ubiquitous risk but generally only to children. In addition to this noncarcinogenic risk, the inhalation of resuspended spoil from lower Poplar Creek would result in an excess cancer risk of 2.1E−04, primarily due to arsenic and chromium. Finally, the incidental ingestion of arsenic and mercury in spoil from lower Poplar Creek would be potentially harmful to children.

In the three agricultural pathways evaluated under the dredging scenario, the milk and meat ingestion pathways showed the most carcinogenic potential. Evaluation of the majority of the reaches for which data were available indicated that the ingestion of milk and beef produced with vegetation grown on dredge spoil would result in an excess cancer risk greater than 1.0E−04 and that many reaches had risk values an order of magnitude greater. The contaminants responsible for the majority of this risk were members of a class of ubiquitous contaminants known as polycyclic aromatic hydrocarbons. In particular, benzo(a)pyrene and dibenz(a,h)anthracene drove this risk. In addition, in those reaches where it was detected, Aroclor 1260 contributed significantly to the risk. By contrast, the excess cancer risk in vegetables was generally lower, exceeding the 1.0 E−04 threshold in only two locations; in these locations the risk was driven by different analytes than in the other two pathways. In Poplar Creek adjacent to the K-25 Site, ^{99}Tc was the primary contributor to risk, although nine other analytes at this location were also of concern. At one location in the Clinch River, the organic analyte N-nitroso-di-n-propylamine was identified as a contaminant of concern but was detected in only one of three samples from that reach.

Evaluation of every subreach for which there were data indicated that one or more of the agricultural pathways posed an unacceptable risk under the dredging scenario. Fourteen noncarcinogenic contaminants of concern were identified. Risk in the milk and meat pathways was frequently driven by mercury and Aroclor 1254. Risk in the vegetable ingestion pathway was frequently driven by manganese, except in the Poplar Creek subreaches, where mercury was the concern.

The ecological baseline risk assessment estimated the ecological risk due to contaminants in the Clinch River/Poplar Creek OU. Seven assessment endpoints were evaluated during the assessment: (1) reduced species richness or abundance or the increased frequency of gross pathologies in fish; (2) reduced species richness or abundance of benthic macroinvertebrate communities; (3) reduced abundance or production of piscivorous wildlife populations; (4) reduced abundance or production of flying insectivorous wildlife populations; (5) reduced production in terrestrial plant communities; (6) reduced abundance or production of terrestrial wildlife populations; and (7) reduced viability of any individuals of a threatened or endangered species. For each endpoint, the reduction in the parameter was required to be 20% or more and to be the result of toxicity.

Three lines of evidence were used in the ecological risk assessment. First, the site and media-specific contaminant data used in the site characterization were evaluated against a series of benchmark values (e.g., the no-observed-effects level) to determine whether concentrations were great enough to cause adverse effects. Second, site-specific toxicity data were used to determine whether these levels were actually causing a toxic effect at a particular site. Finally, site-specific biological survey data (species richness and abundance) were used to help assess whether any toxicity was actually having an impact at the population or community level. When all lines of evidence were not available for each of the endpoints, risk assessment was usually based on contaminant data alone. The assessment for the fish endpoint used data on fish pathologies and fecundity as a fourth line of evidence.

The fish community in Poplar Creek was found to be at significant risk from episodically high concentrations of several metals (copper, mercury, nickel, and silver). Toxicity to fish was assessed by using several test protocols and organisms. Poplar Creek water was toxic to Japanese medaka and redbreast sunfish embryos, but not to fathead minnows or *Ceriodaphnia*. The fish community of Poplar Creek exhibited decreased species richness and abundance in comparison with a reference site with similar habitat (Bull Run Creek embayment of Melton Hill Reservoir). The results of the ecological risk assessment for fish indicated that, while individual fish were probably suffering some physiological impacts immediately below WOC, the fish community was not being significantly impacted in the Clinch River. In McCoy Branch, adverse impacts could not be ruled out, but data were unavailable for some of the lines of evidence (data on fish community, pathology, and fecundity).

The benthic macroinvertebrate community of Poplar Creek was identified as being at significant risk from several metals (arsenic, mercury, nickel, and silver) and PCBs in surface sediment. The benthic community contained fewer species and had less abundance of organisms than the communities at other sites. The toxicity data did not reveal consistently toxic effects, but in at least one test from each site a toxic response was observed in test organisms of at least one species. The benthic communities of the Clinch River and the McCoy Branch embayment were found not to be significantly impacted by contaminants.

Risks to piscivorous wildlife were assessed by using two avian species (great blue heron and osprey) and two mammalian species (mink and river otter). Two lines of evidence, biomonitoring data and contaminant data in whole fish, were available for assessing risk to heron and osprey. Partly because of their wide foraging behavior, osprey were found to be not at risk from contaminants even though mercury levels in Poplar Creek fish exceeded benchmark values. Although the data indicated that individual heron feeding exclusively in certain portions of Poplar Creek might be at risk, the local populations of the avian species were not expected to be impacted. This conclusion was supported by surveys of the reproductive success of osprey and great blue heron in Poplar Creek: the surveys found high reproduction and no increase in deformities. Mink were not identified as being at risk from contaminants in either Poplar Creek or the Clinch River. However, individual river otter feeding in Poplar Creek near the mouth of EFPC would be expected to have a significant risk of impaired reproduction. Although river otter do not currently exist within the OU, they have recently been reintroduced into east Tennessee, and the natural expansion of their range is expected to lead to their re-establishment on the ORR. Because the otter is a state threatened species, impacts to individual otter would be considered significant.

The risk to insectivorous wildlife was assessed by using one avian species, the rough-winged swallow, and two mammalian species, the gray and little brown bats. In each case, only one line of evidence (contaminant concentrations in benthic insects) was available for the assessment. These data indicate that a colony of rough-winged swallows feeding in Poplar Creek near the mouth of EFPC could suffer impaired reproductive potential as a result of mercury exposure, but the magnitude of the effects could not be evaluated. Populations of neither species of bat were found to be at risk. Although mercury concentrations in Poplar Creek were high enough to put individual bats at risk if they were to forage exclusively within this area, the foraging range of bats is great enough that this possibility is unlikely.

The ecological risk assessment for terrestrial wildlife was based on a dredging scenario much like that used in the human health risk assessment. The assessment examined risk both to the plant community that would develop on dredge spoil and to a herbivorous mammal (eastern cottontail) foraging there. A single line of evidence (contaminant concentrations in sediment) was available for use in the assessment. In the sediment of one or more study reaches, 12 metals were found at concentrations that exceeded benchmarks indicative of plant toxicity. The greatest number of metals were found in lower Poplar Creek, which contained arsenic, boron, cadmium, chromium, mercury, nickel, selenium, uranium, and vanadium. Benchmark numbers for plants were not available for a large number (i.e., 37) of the organic compounds; therefore, these analytes could not be evaluated. Populations of cottontails foraging on future spoil from lower Poplar Creek or from the Clinch River immediately downstream of Poplar Creek would be at significant risk of impaired reproduction due to levels of mercury and cadmium. A number of other analytes from these (and most other) reaches might pose a risk to individual cottontails foraging on future spoil; however, population-level effects from these contaminants would not be expected.

Ecological risk from radiation exposure was assessed separately. Risks to many of the same endpoint species were assessed, including a benthic organism (a mayfly), epibenthic (bottom-dwelling) fish, piscivorous wildlife (great blue heron), and a terrestrial herbivore (eastern cottontail). The assessment used the contaminant data for radionuclides to calculate the total radiation dose to these organisms. The DOE limit of 1.0 rad/day was used to assess acceptable exposures for most organisms; the International Atomic Energy Agency's recommended limit of 0.1 rad/day for terrestrial organisms was used to assess exposure for cottontail rabbits. None of the calculated doses approached the appropriate benchmarks above; thus, the radiological contaminants in the various environmental media of the OU did not pose a significant ecological risk.

Based on the findings of the site characterization and risk assessments, a number of pathway and media-specific remedial goal options (RGOs) were developed for individual analytes, which represent concentrations corresponding to an ARAR or to an acceptable human health or ecological risk level. RGOs were developed only for those reaches containing one or more environmental media that exceeded one of the criteria and thus indicated the potential need for remedial action in that reach. In almost every reach studied, RGOs were developed for one or more analytes for each of the three media evaluated.

FEASIBILITY STUDY

On the basis of the remedial investigation, a feasibility study was conducted to identify remedial alternatives that would be effective in reducing contaminant concentrations or reducing or eliminating exposure and that could be feasibly implemented. The overall approach taken in the feasibility study was to focus on remedies for site-related contaminants as identified in the site characterization. Therefore, remedies were not evaluated for those reaches in which risk was primarily the result of non-site-related contaminants (e.g., manganese-driven risks in surface sediment or in water). In addition, no remedies were evaluated for surface water contaminants. Remediation of surface water is best effected at the source of the contamination, which in each case is primarily in upstream OUs. Moreover, remediation of the large volumes of flowing water is not practical. The use of institutional controls is the only remedy considered for limiting human exposure to fish. Therefore, active remedies evaluated in the feasibility study focus on site-related contaminants of concern in sediment.

Four alternatives are evaluated in the feasibility study. Alternative 1, the no-action alternative, is required by CERCLA to be evaluated. As the name implies, this alternative would be easily implemented at no cost. However, the risk assessments indicate that the no-action alternative would not be protective of human health or the environment.

Alternative 2 consists of the use of institutional controls and advisories to reduce exposure to contaminants in fish and sediment. Although not empowered under CERCLA, the state of Tennessee, the Tennessee Valley Authority, and the U.S. Army Corps of Engineers each have separate regulatory authority to regulate activities that could result in the disturbance of sediment in the Clinch River and Poplar Creek. Each of these agencies is party to an interagency agreement with DOE that requires the multiagency review, on a case-by-case basis, of all permit applications that propose activities with the potential to disturb sediment. The state of Tennessee currently issues fish consumption advisories warning the public to avoid or limit consumption of certain species of fish in which contaminant levels are unacceptably high. The present worth cost of implementing alternative 2 for 30 years, including future monitoring and administrative costs, is estimated to be \$3.6M.

Alternative 3 incorporates the institutional controls described in alternative 2 and in addition proposes the combined containment and removal of contaminated sediment from Poplar Creek. The presence of several contaminants in the sediment of Poplar Creek posed a risk to human health or benthic organisms. Seven separate locations in Poplar Creek would be remediated, through the use of a combination of sediment containment technologies in the near-shore areas (bottom elevation >733 ft msl) and sediment removal technologies in deep water areas (bottom elevation <733 ft msl). A total area of 388,800 ft^2 is proposed for containment, and a total of 179,250 yd^3 for removal (the top 3 ft of sediment), at a total present worth cost of $109.6M. Removal of the deep sediment would also address the potential for future risks to human health and the environment in the dredging scenario.

Alternative 4 also incorporates the institutional controls of alternative 2, but in addition it proposes the removal of contaminated sediment from Poplar Creek. Removal of those sediments in Poplar Creek that pose a human health or ecological risk would require the removal and safe disposal of approximately 226,500 yd^3 of sediment, at a total present worth cost of approximately $123.5M. Because benthic habitat extends bank-to-bank, addressing the existing ecological risk by removing sediment would also address both the existing human health risk in the near-shore scenario and the potential for future human health risk and ecological risk in the dredging scenario.

1. INTRODUCTION

1.1 REGULATORY INITIATIVE

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

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

The CR/PC Remedial Investigation (CRRI) was originally begun under authority of Sect. 3008(v) of RCRA. Although these requirements still apply, CERCLA is the primary regulatory driver because it allows for the consideration of a wider range of contaminants, including radionuclides, polychlorinated biphenyls (PCBs, commercially marketed as Aroclor mixtures, such as Aroclor 1254, etc.), and contaminants in wastes released before the enactment of RCRA in 1976. The FFA parties intend to meet the RCRA corrective action requirements, as well as any applicable or relevant and appropriate requirements of other environmental law, through the CERCLA process.

1.2 OAK RIDGE RESERVATION ENVIRONMENTAL RESTORATION PROGRAM

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

1.3 CLINCH RIVER ENVIRONMENTAL RESTORATION PROGRAM

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

At the public's request, in October 1993, the FFA parties decided that the WBR downstream of the Clinch River should become a distinct OU for the purpose of accelerating the remedial decision in that portion of the reservoir (DOE 1994c). The results of the lower WBR RI/FS are discussed by DOE (1995). In keeping with the original phased approach, the remainder of the original OU, consisting of the Clinch River and Poplar Creek downstream of the ORR, was studied in more detail under Phase 2 of the CRRI.

1.4 OBJECTIVES AND SCOPE OF THIS REPORT

The objectives of this report are to (1) describe the current nature and extent of the contamination in the CR/PC system resulting from releases from the ORR (Chap. 3), (2) quantify the current and future risk in the CR/PC system to human health (Chap. 5) and the environment (Chap. 6) resulting from these contaminants, and (3) identify and evaluate remedial action alternatives that are feasible for application in the Clinch River or Poplar Creek (Chaps. 7–10). Data collected during Phase 1 and Phase 2 of the CRRI, in conjunction with certain existing data, are used to accomplish each of these objectives.

1.5 ORGANIZATION OF THIS REPORT

This report is organized into five volumes, the first of which presents the main text. Chapter 1 describes the regulatory setting, and Chapter 2 broadly portrays the environmental setting. Chapter 3 depicts the operational and release history of the site and characterizes in detail the nature and extent of contamination. Chapter 4 briefly identifies other regulatory requirements that are applicable or appropriate to the site. Chapters 5 and 6 assess the risk to human health and the environment, respectively. Chapter 7 explains the purpose and organization of the feasibility study. Chapter 8 defines remedial action objectives for the site; identifies pathways and contaminants of concern; and screens general response actions, potential remedial technologies, and process options. Chapter 9 develops remedial alternatives based on the remedial action objectives, the screened technologies, and representative process options. Chapter 10 analyzes, evaluates, and compares the remedial alternatives. Chapter 11 lists the references cited in the main text.

Volumes 2–5 consist of appendices that contain supporting data and information. Volume 2 contains Appendices A–D, which characterize the biota on the ORR (Tables A1–A3) and summarize data related to contaminant concentrations in water (Tables B1–B7), in sediment (Tables C1–C6), and in biota (Tables D1–D8). Volume 3 contains Appendices E and F, which present information related to human health risk assessment (Tables E1–E67) and ecological risk assessments (Tables F1.1–F1.11, F2.1, F3.1–F3.3, F4.1–F4.20, F5.1–F5.20, F6.1–F6.6, F8.1–F8.3). Volume 4 contains Appendices G–I, which focus on the feasibility study. Appendix G details the selection of remedial process options, while Appendix H provides the basis for the cost estimates for each remedial alternative. Appendix I presents the applicable or relevant and appropriate requirements (ARARs), which help define the extent of the remedial response. Volume 5, consisting of Appendices J–N, is a compilation of data from individual studies that were conducted as part of the overall remedial investigation. Appendix J addresses the quality assurance objectives for measuring the data. Appendix K presents selected historical data (Tables K1–K8), Appendix L contains data from several discrete water characterization studies (Tables L1–L28), Appendix M provides data supporting the sediment characterization (Tables M1–M9), and Appendix N contains data related to several biota characterization studies (Tables N1–N30).

1.6 SCHEDULE

In conjunction with this RI/FS, DOE is preparing a Proposed Plan that will summarize the findings of the RI/FS and identify a preferred remedial alternative. A public meeting will be held to present the Plan, and a minimum 30-day comment period will follow. DOE, EPA, and TDEC will jointly select the final remedy after consideration of all public comments. The Proposed Plan is scheduled for release later in 1996.

DOE will document the selection of the final remedy in a formal Record of Decision (ROD), to be signed by EPA and TDEC. This ROD is scheduled for completion in late 1997. If the ROD results in contaminants being left in place at the site, Sect. 121(c) of CERCLA requires review of the remedial decision at least every 5 years to ensure that the selected remedy is protective of human health and the environment. In such an event, DOE may implement a long-term monitoring plan in the Clinch River and Poplar Creek to gather the data necessary to make this determination. If this review indicates that the remedy is not protective of human health and the environment, further remedial action will be required.

2. CHARACTERIZATION OF ENVIRONMENTAL SETTING

2.1 GEOGRAPHY

The area of the Clinch River under investigation is located in eastern Tennessee downstream of the ORR (Fig. 2.1). The OU is large, extending almost 34 miles from the mouth of the Clinch River at Kingston, Tennessee, to the upstream limit of the ORR at Clinch River mile (CRM) 43.7 near the city of Oak Ridge (Fig. 2.2 and Plate A). It includes the Poplar Creek embayment of WBR from the creek's mouth at CRM 12.0 upstream to its confluence with East Fork Poplar Creek (EFPC) at Poplar Creek mile (PCM) 5.5.

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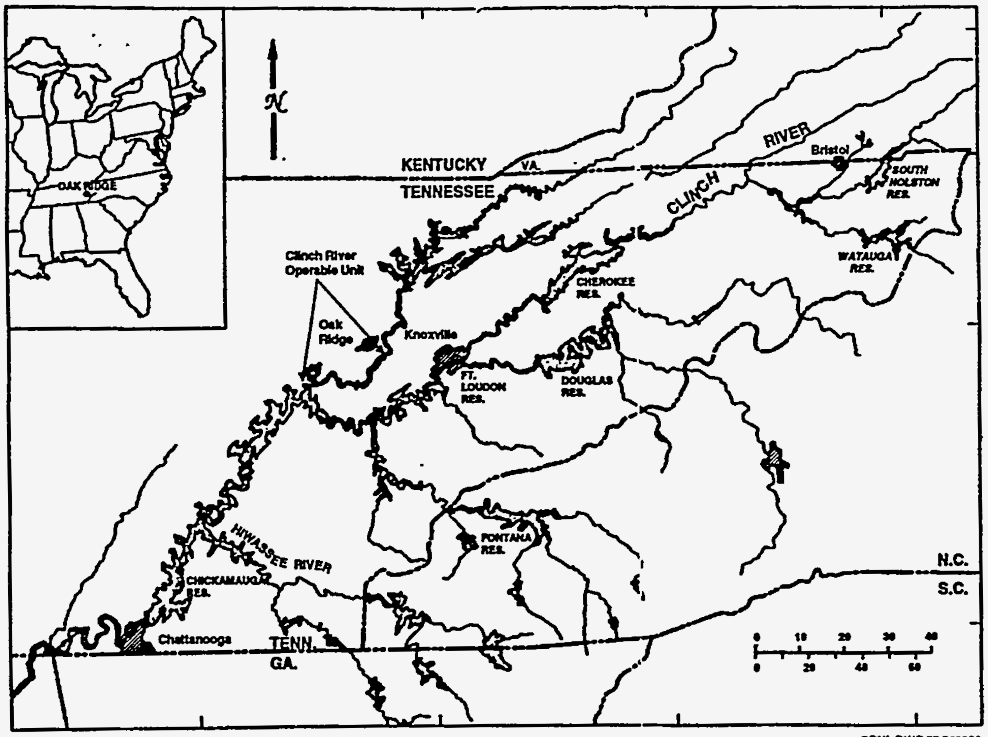

Fig. 2.1. Location of the Clinch River/Poplar Creek Operable Unit in eastern Tennessee.

The ORR is a 34,600-acre tract located within the city limits of Oak Ridge in Anderson and Roane counties. The Clinch River forms the southern and eastern boundaries of the ORR; Poplar Creek flows directly through the reservation. The Clinch River also forms the boundary between Knox and Anderson counties for a portion of its length in the study area.

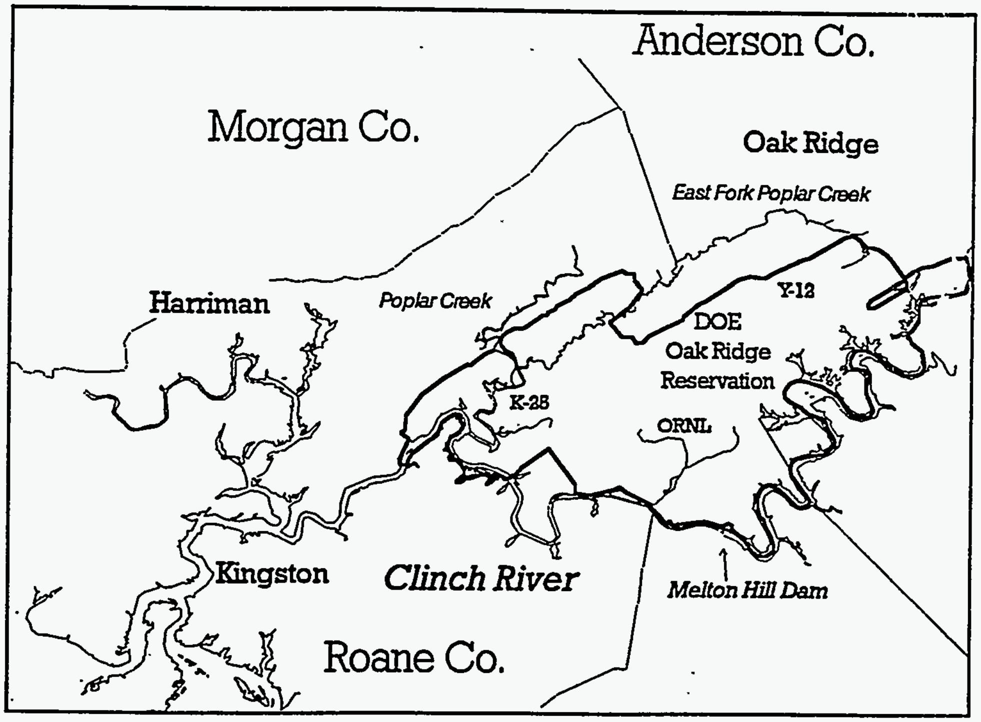

Fig. 2.2. The Clinch River/Poplar Creek Operable Unit in relation to the U.S. Department of Energy's Oak Ridge Reservation.

The area of the Clinch River under study includes portions of two Tennessee Valley Authority (TVA) reservoirs. Downstream of Melton Hill Dam, located at CRM 23.2, the study area is considered part of the WBR. Upstream of Melton Hill Dam, the study area is in Melton Hill Reservoir.

2.2 DEMOGRAPHY

The ORR lies within the city limits of Oak Ridge (pop. 27,310) but is situated southwest of the residential portion of the city. The largest population center near Oak Ridge is Knoxville (pop. 165,121), located ~20 miles to the east. The nearest community downstream of the ORR is Kingston (pop. 4552), situated at the confluence of the Clinch and Tennessee rivers ~9 river miles downstream of the ORR. A total of 115,477 persons live in Anderson and Roane counties. An additional 560,931 persons reside in the 10-county area that borders the two counties. The population distribution within a 50-mile radius of the ORR is depicted in Fig. 2.3. Land use along the Clinch River is primarily agricultural and residential in the valleys and woodland on the ridges (for example, see Fig. 2.18).

There are currently three potable water intakes on the Clinch River within the study area (Plate A). In Melton Hill Reservoir, the city of Oak Ridge has a municipal intake at CRM 41.5 near the Scarboro area, and the West Knox County Utility District has an intake at CRM 36.2 near Melton Hill Park. In the WBR, DOE's K-25 Site has a potable water intake at CRM 14.5 at the mouth of Grassy Creek. The municipal intake for the city of Kingston is located on the Tennessee River [Tennessee River mile (TRM) 568.4] immediately upstream of its confluence with the Clinch. Depending on the flow conditions resulting from reservoir operations, effluents from the ORR could reach the intake. Several intakes are located in Melton Hill Reservoir upstream of the ORR, including a pumping station for the West Knox

County Utility District at CRM 46.1, an Anderson County Utility Board intake at CRM 52.5, and municipal intakes for the city of Clinton at CRM 56.9 and 64.8. No domestic intakes are located on Poplar Creek within the ORR or downstream of it.

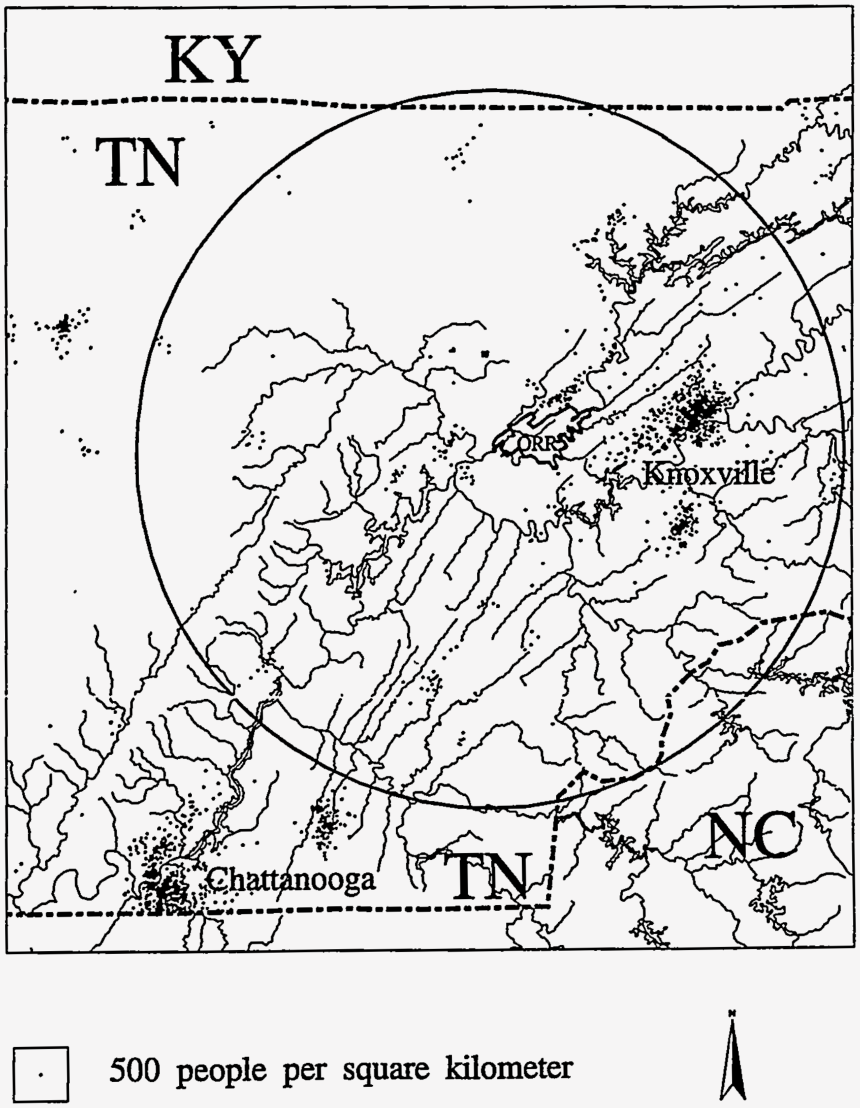

Fig. 2.3. Population density and distribution within 50 miles of the Oak Ridge Reservation.

Popular recreational uses of the Clinch River within the study area include fishing, swimming, skiing, and boating. Public recreational facilities on Melton Hill Reservoir within the study area include Guinn Road Park and Melton Hill Park (Plate A). TVA's Melton Hill Dam Public Use Area provides access to the Clinch River both above and below Melton Hill Dam. In WBR, the only public recreation areas are Kingston City Park and Southwest Point Park at Kingston. The 300-acre Kingston Wildlife Management Area is located across from Kingston at the mouth of the Emory River. The only commercial recreational facility within the study area is a campground on WBR at CRM 17.

2.3 CLIMATE

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

Winds in the area are influenced primarily by the surrounding topography. Prevailing winds are usually southwesterly during the day, moving up the Tennessee River Valley. Nighttime winds generally move down the valley, to the southwest. Strong winds are rare; wind speeds <7.4 mph occur 75% of the time (Energy Systems 1993b).

Precipitation in the area averages between 50 and 55 in./year. The wettest months are December through March, when slow-moving fronts may result in low-intensity storms of long duration. Another peak occurs in summer, when convective currents create high-intensity, short-duration, localized thunderstorms. The driest period is typically autumn, when slow-moving high pressure cells may suppress rain for extended periods. Average annual evapotranspiration is estimated to be from 32 to 35 in., or 66% of rainfall, which leaves an average of 20 to 23 in. available for runoff and infiltration. Evapotranspiration is greatest during the growing season, when it often exceeds the rate of precipitation and results in soil moisture deficits. In winter, precipitation exceeds evapotranspiration, resulting in increased water available for runoff and groundwater recharge (Energy Systems 1993b).

Rainfall in 1994 (when data for this RI were being collected) was ~65.6 in., considerably above normal. More than half of this total (35.19 in.) fell from January through April and resulted in higher-than-normal flows during much of the sampling period.

2.4 TOPOGRAPHY, GEOLOGY, AND SOILS

2.4.1 Topography

The ORR and the CR/PC lie in the Ridge and Valley Province (Fig. 2.4). This province is bordered on the east by the Blue Ridge Province and on the west by the Cumberland Plateau Province. The Ridge and Valley Province extends for 1200 miles from the Canadian St. Lawrence Valley to the Gulf Coastal Plain of Alabama. It is characterized by a succession of northeast trending ridges of various widths. The topography is reflective of the underlying geology, in which ridges are underlain by less soluble cherty limestones, dolomites, and sandy shales, while the valleys are underlain by more soluble limestones, dolomites, and shales. Extensive folding and thrusting throughout the Province has resulted in the characteristic belted pattern of rock formations (Energy Systems 1993b).

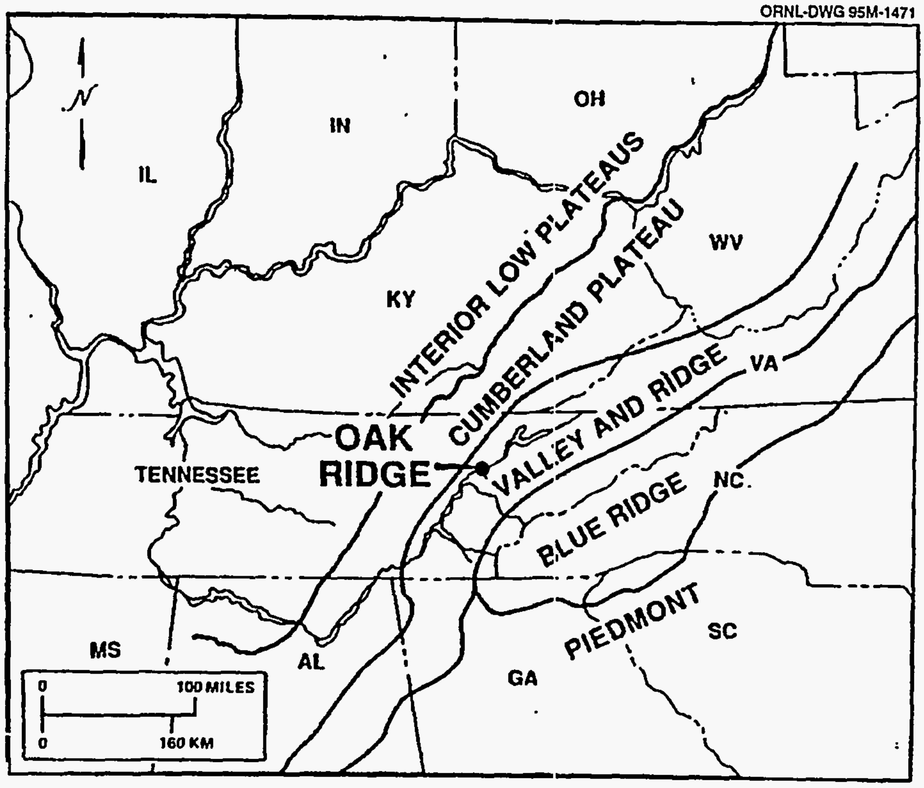

Fig. 2.4. Regional physiographic map depicting the location of the Oak Ridge Reservation and the Clinch River/Poplar Creek Operable Unit in the Valley and Ridge Province. *Source:* Martin Marietta Energy Systems, Inc. 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31. Oak Ridge National Laboratory, Oak Ridge, Tenn.

2.4.2 Geology

The ORR and the CR/PC are underlain by several different geologic formations or groups of formations, all of sedimentary origin (Fig. 2.5). Energy Systems (1993b) describes the generalized stratigraphy as including, in ascending order, the lower Cambrian Rome Formation, the Cambrian Conasauga Group, the Cambrian-Ordovician Knox Group, and the middle Ordovician Chickamauga Group. Younger upper Ordovician to Mississipian rocks are exposed locally in the cores of two synclines north of the White Oak Mountain thrust fault. Although minor carbonate beds are found throughout the Conasauga Group bedrock, the principal carbonate formations on the ORR are the upper Conasauga Group Maynardville Limestone, Chickamauga Group limestones, and the Knox Group. Other formations on the ORR are characterized by silty sandstones, siltstones, limey siltstones, and shales (Energy Systems 1993b).

The northeast trending ridges of the study area result from a succession of northeast-trending thrust faults that have structurally stacked and replicated the rocks of the geologic units (Fig. 2.6). Most of the prominent ridges are underlain by the Rome or Knox groups, whereas many of the valleys are underlain by the Chickamauga or Conasauga groups. The major faults in the area were formed during the Permian-Pennsylvanian age and are not active structures (Energy Systems 1993b). Competent bedrock in each of these formations is generally overlain by a mantle of regolith, a zone of weathered, unconsolidated materials that form in place through chemical and physical weathering of the underlying bedrock. Above the regolith is a relatively thin layer of soil or alluvial sediment.

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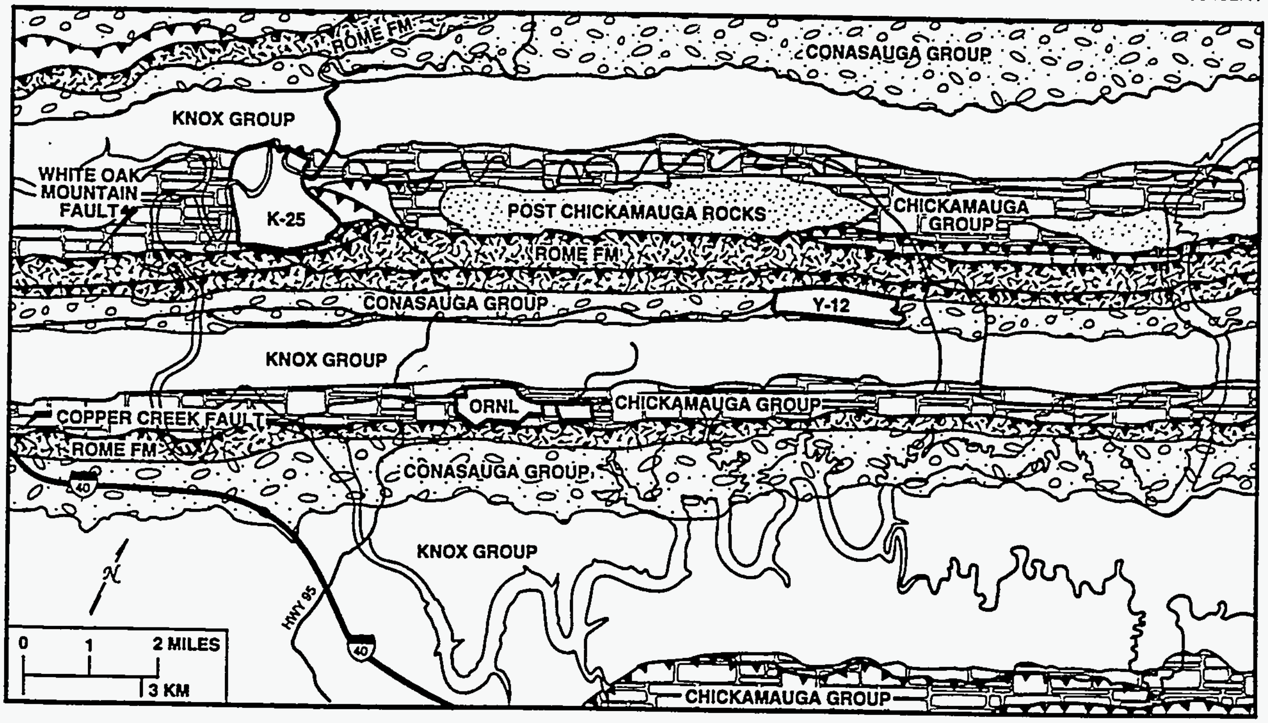

Fig. 2.5. Geologic map of the Oak Ridge Reservation. *Source:* Martin Marietta Energy Systems, Inc. 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31. Oak Ridge National Laboratory, Oak Ridge, Tenn.

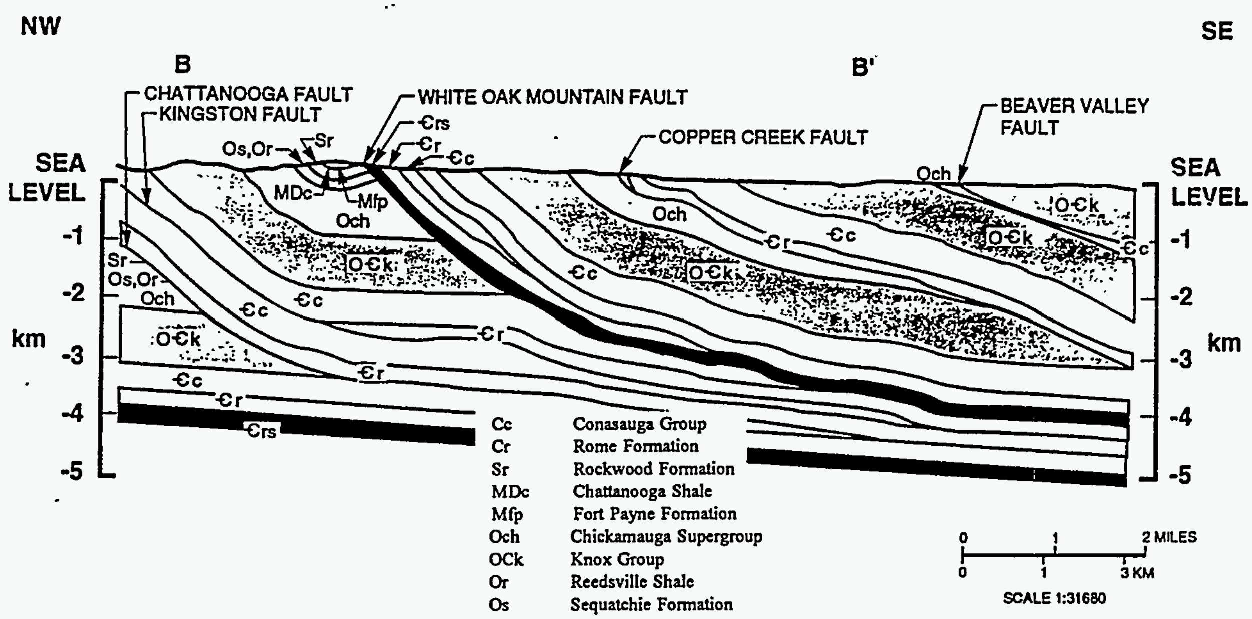

Fig. 2.6. Generalized geologic cross-section of the Oak Ridge Reservation. *Source:* Martin Marietta Energy Systems, Inc. 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31. Oak Ridge National Laboratory, Oak Ridge, Tenn.

2.4.3 Soils

The land along the Clinch River study area is primarily overlain by residual soils that have weathered in place. The most extensive soils in the study area are the Fullerton, Bodine, Talbott, Colbert, and LeHew soils (USDA 1981). In general, soils can be correlated with the underlying geologic unit that serves as the parent material. The deep, well-drained Fullerton and Bodine soils of Chestnut Ridge, Blackoak Ridge, and Copper Ridge have been derived from the Knox Group, whereas the shallower, loamier soils of Pine Ridge and Haw Ridge have formed from the Rome Formation. The Talbott and Colbert soils, characterized by plastic clay subsoils, predominate in the valleys underlain by the Chickamauga Limestone, such as Bethel Valley and much of EFPC Valley. In valleys underlain by the Conasauga shale, such as Bear Creek Valley and Melton Valley, the moderately deep, well-drained Sequoia soils are found. Alluvial soils are also present to a limited extent within the study area, primarily on high river terraces of ancient river floodplains and in narrow tracts along modern streams. The characteristics of any soil are highly localized and can vary widely even within a soil type.

2.4.4 Groundwater

Groundwater conditions on the ORR are summarized here to acquaint the reader with the environmental setting. Groundwater contamination on the ORR, including the potential for contaminant migration to the Clinch River, is being addressed separately from the CR/PC OU (DOE 1994).

A conceptual model of groundwater occurrence has recently been formulated for the ORR (Solomon et al. 1992; Moore and Toran 1992). The hydrology of the area (Fig. 2.7) can be considered in two broad hydrologic units: (1) the Knox aquifer and (2) the aquitards (Energy Systems 1993b). The Knox aquifer consists of the Knox Group and the underlying Maynardville Limestone of the Conasauga Group (Energy Systems 1993b), both of which consist of massive carbonate rocks. Although the matrix of this unit exhibits low primary porosity and permeability, flow is controlled by a combination of secondary fractures and solution conduits. Large volumes of water may move relatively long distances in this unit. The Knox aquifer is the primary source of base flow for many streams in the study area. All large springs on the ORR issue from the Knox unit (Energy Systems 1993b), and some wells penetrating the larger solution conduits within this unit yield 1000 gal/min. The Knox aquifer has been described as the most important aquifer in eastern Tennessee (DeBuchananne and Richardson 1956).

The remaining geologic units underlying the majority of the ORR and Clinch River (the Rome Formation, the Conasauga Group below the Maynardville Limestone, and the Chickamauga Group) constitute the aquitards (Energy Systems 1993b). These units are all of low primary porosity and permeability; secondary permeability is usually much reduced compared to the Knox aquifer. These units are more likely to produce wells of lower yield than the Knox, and base flow to streams is also typically much reduced (Energy Systems 1993b).

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

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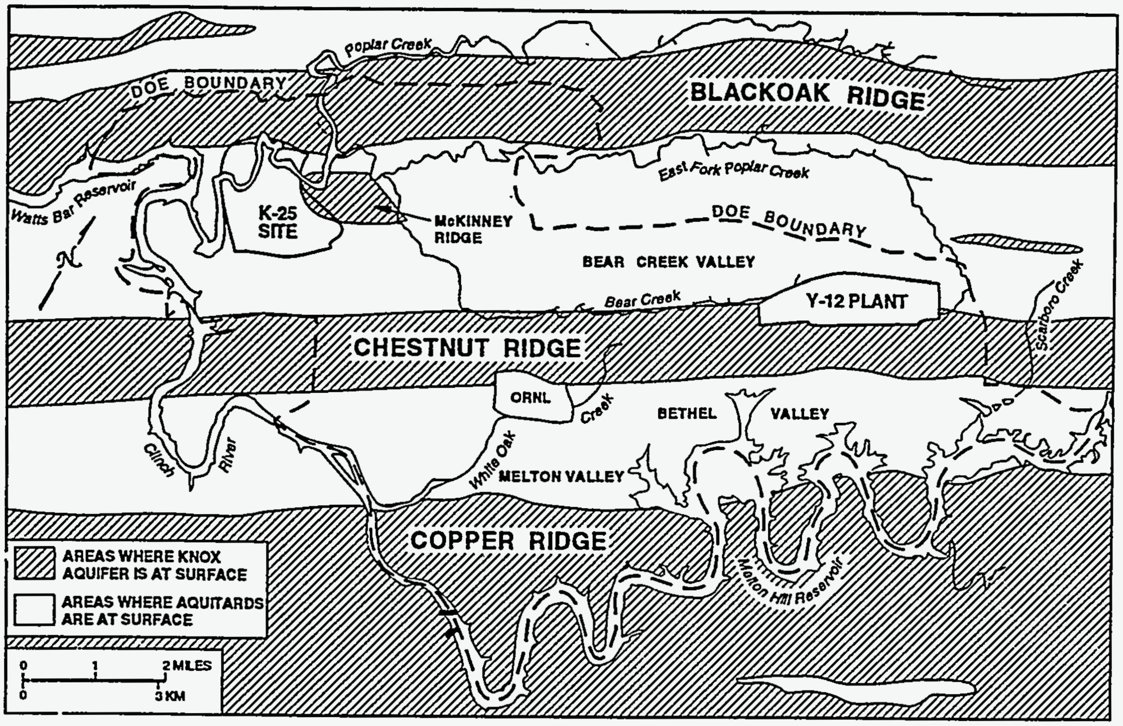

Fig. 2.7. Location of aquitards and the Knox Aquifer on the Oak Ridge Reservation. *Source:* Martin Marietta Energy Systems, Inc. 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31. Oak Ridge National Laboratory, Oak Ridge, Tenn.

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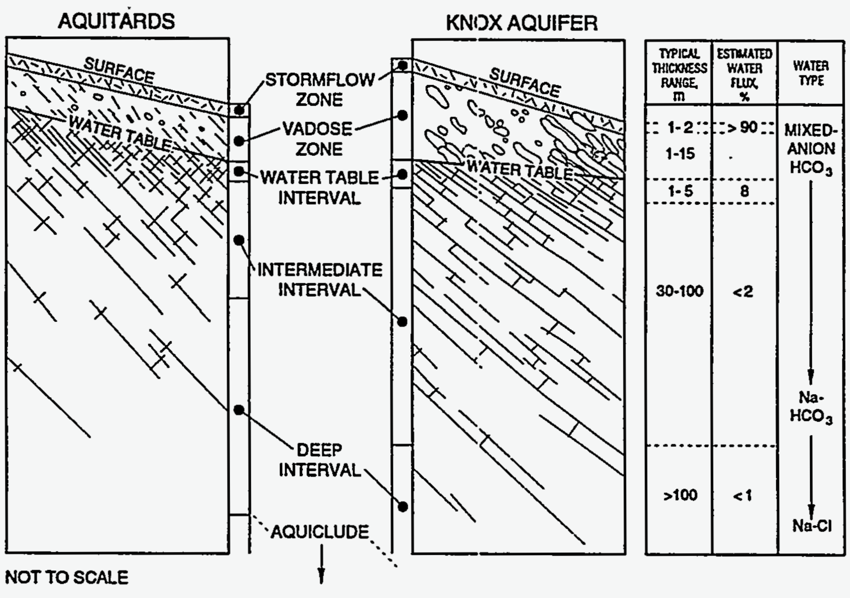

Fig. 2.8. Schematic representation of vertical relationships characteristic of flow zones on the Oak Ridge Reservation: estimated thickness, water flux, and water type. *Source:* Martin Marietta Energy Systems, Inc. 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31. Oak Ridge National Laboratory, Oak Ridge, Tenn.

200 m), the quality of the water changes from freshwater to brine; this change is sometimes used to delineate the functional basement of this zone (Energy Systems 1993b).

On the ORR, as well as in the surrounding region, the majority (95%) of the active subsurface flow moves via the shallow stormflow zone to nearby surface streams. The vadose zone transmits water vertically to the saturated zone, as is necessary to recharge the saturated zone. Saturated zone flow in the aquitards occurs primarily (95%) in the upper 50 to 100 ft; consequently, flow routes are short, and discharge moves to nearby surface streams. In the Knox aquifer, a few solution conduits may be as much as 2 miles long (Energy Systems 1993b).

2.5 SURFACE WATER

2.5.1 Physical Characteristics

The Clinch River rises near Tazewell, Virginia, and flows 351.6 miles to the southwest before entering the Tennessee River at Kingston. It drains an area of 4413 mile^2 (Fig. 2.9), primarily within the River and Valley Province. There are two major impoundments on the Clinch River. Norris Reservoir extends from Norris Dam (CRM 79.8) 73 miles upstream on the Clinch River and 53 miles on the Powell River. Norris Reservoir is a deep storage impoundment providing hydropower, flood control, and augmentation of downstream flow during drought conditions. Melton Hill Reservoir, impounded in 1963, extends from Melton Hill Dam at CRM 23.2 upstream 44 miles to Clinton, Tennessee. Melton Hill Reservoir is a smaller, shallower reservoir used primarily for navigation and the generation of hydropower during peak demand periods. Downstream of Melton Hill Dam, the river constitutes an arm of WBR, impounded in 1942 by TVA's Watts Bar Dam on the Tennessee River (TRM 529.9). The CR/PC study area consists of the Clinch River arm of WBR and portions of Melton Hill Reservoir.

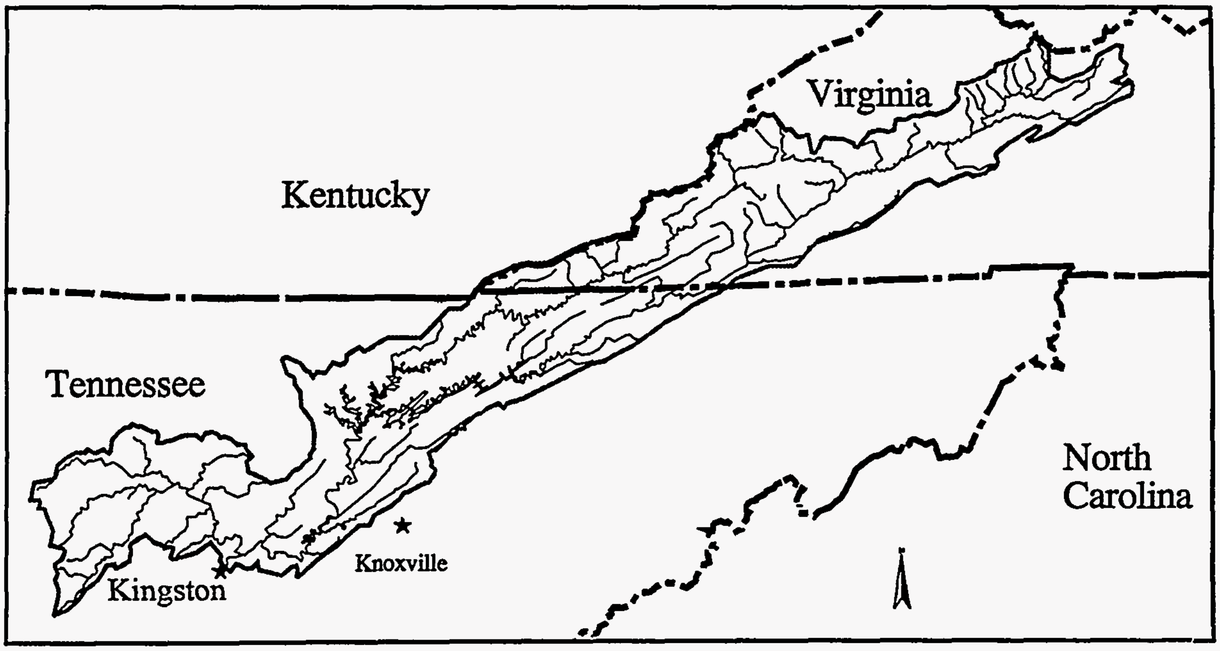

Fig. 2.9. The Clinch River drainage basin—4413 square miles in Virginia and Tennessee.

The Clinch River arm of WBR extends ~23 miles from the mouth of the Clinch River to Melton Hill Dam (Fig. 2.2). It is 2100 surface acres in size and contains ~39,000 acre-feet of water at full pool [elevation 741 ft above mean sea level (msl)]. Average annual discharge at the mouth is ~6800 cfs, and the average hydraulic retention time is ~3 days. Most of the flow (75%) in this stretch of the river is released from Melton Hill Dam, with an average annual discharge of 4400 cfs. Principal tributaries to the Clinch River arm of WBR are the Emory River, which enters at CRM 4.0, and Poplar Creek, which enters at CRM 12.0. The Emory River discharges ~1020 cfs on average, or about 20% of the flow in the Clinch River at Kingston. Most of its 865-mile^2 watershed is located in the Cumberland Plateau. Poplar Creek has a drainage area of 135.9-mile^2 and an average annual flow at its mouth of 228 cfs, or about 4% of the flow at Kingston. The remaining flow comes from minor tributaries, direct precipitation, and groundwater flow.

Poplar Creek and White Oak Creek are the streams of primary interest that have historically transported contaminants from the ORR to the Clinch River arm of WBR. Poplar Creek receives effluents from the Y-12 Plant (via EFPC) and the K-25 Site. Above the ORR, Poplar Creek receives municipal discharges from Oak Ridge (again via EFPC) and Oliver Springs, as well as non-point-source discharges from agricultural and mining activities. White Oak Creek, which enters at CRM 20.8, has its headwaters on the ORR. It drains 6 mile^2 of the ORR, including ORNL, and has an average annual flow of 13.8 cfs.

The Clinch River arm of WBR is primarily riverain throughout most of its length. More lakelike (lacustrine) conditions, characterized by slowly moving waters, are found downstream of the Emory River, extending up to and including parts of Poplar Creek during summer. Because of reduced surface water velocity, lacustrine areas are subject to thermal and chemical stratification. This reduced flow also leads to the accumulation of organic matter, particle-associated contaminants, and other finer grained materials in the sediments. During colder months, the riverine conditions persist throughout the Clinch River. Because these areas exhibit both riverine and lacustrine conditions, they may best be thought of as transition areas.

The WBR is managed by TVA to provide flood control, navigation, water supply, aquatic habitat, and recreation. During the winter months, the surface of the reservoir is maintained at an elevation of 735 ft above msl to provide flood storage capacity for spring runoff. In mid April, the water level is gradually raised to the summer pool elevation of 741 ft to provide recreational benefits and to improve shoreline aquatic habitat (Fig. 2.10). The relationship between water elevation, surface area, and volume in the Clinch River arm of WBR (excluding Poplar Creek and the Emory River) is depicted in Fig. 2.11.

The flow rate below Melton Hill Dam varies greatly, often hourly, because the primary role of the dam is to generate electricity to meet peak power demands. This practice produces discharges up to 17,500 cfs during peak demand, followed by periods of zero discharge, resulting in rapid changes in surface water elevation in the upper portions the Clinch River (Fig. 2.12).

Melton Hill Reservoir, situated on 44 miles of the Clinch River between Watts Bar and Norris reservoirs, borders the ORR on the east and to the south. Melton Hill may be characterized as a "run-of-the-river" reservoir because of its close resemblance to river morphometry, its small surface area (5700 acres), small volume (120,000 acre-feet), and short water-retention time (16 to 17 days) relative to its length (Fehring and Meinert 1993). Full pool in Melton Hill is 795 ft above msl, and intermittent water-level fluctuations occur in accordance with the system-wide reservoir management needs of the TVA. Unlike Watts Bar, there is no seasonal drawdown of Melton Hill.

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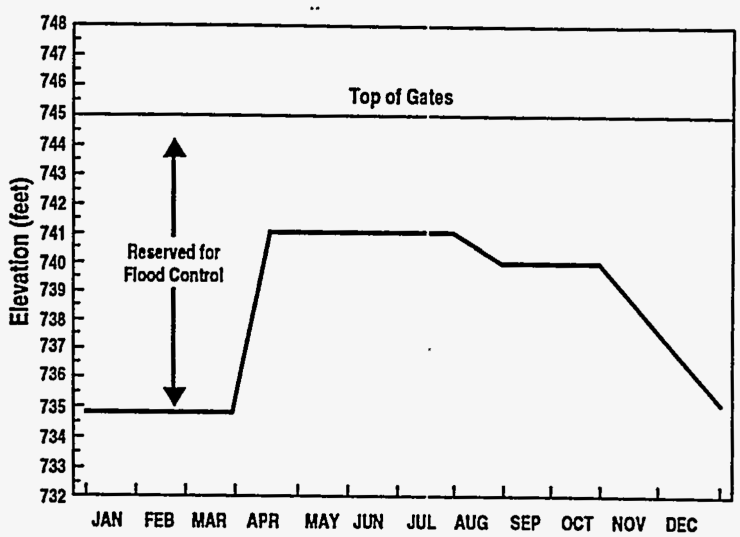
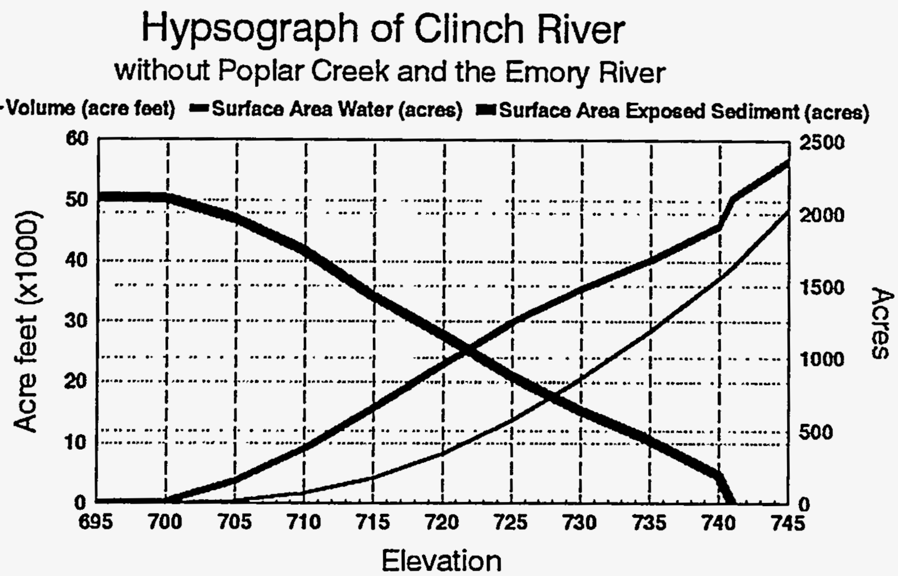


Fig. 2.10. Seasonal water-level elevations in Watts Bar Reservoir as maintained by the Tennessee Valley Authority.



	695	700	705	710	715	720	725	730	735	740	741	745
Volume (acre feet)	0.00057	0.02018	0.34203	1.65881	4.18597	8.19479	13.7216	20.4934	28.3657	37.284	39.2488	48.6196
Surface Area Water (acres)	0.5	12.35	150.569	373.018	654.368	952.615	1239.95	1470.62	1672.95	1913.11	2109.7	2359.54
Surface Area Exposed Sediment (acres)	2109.65	2097.35	1959.13	1736.68	1424.33	1157.08	869.75	639.08	436.75	196.59	0	

Fig. 2.11. Relationship between surface water elevation, volume, and exposed sediments in the Clinch River arm of Watts Bar Reservoir.

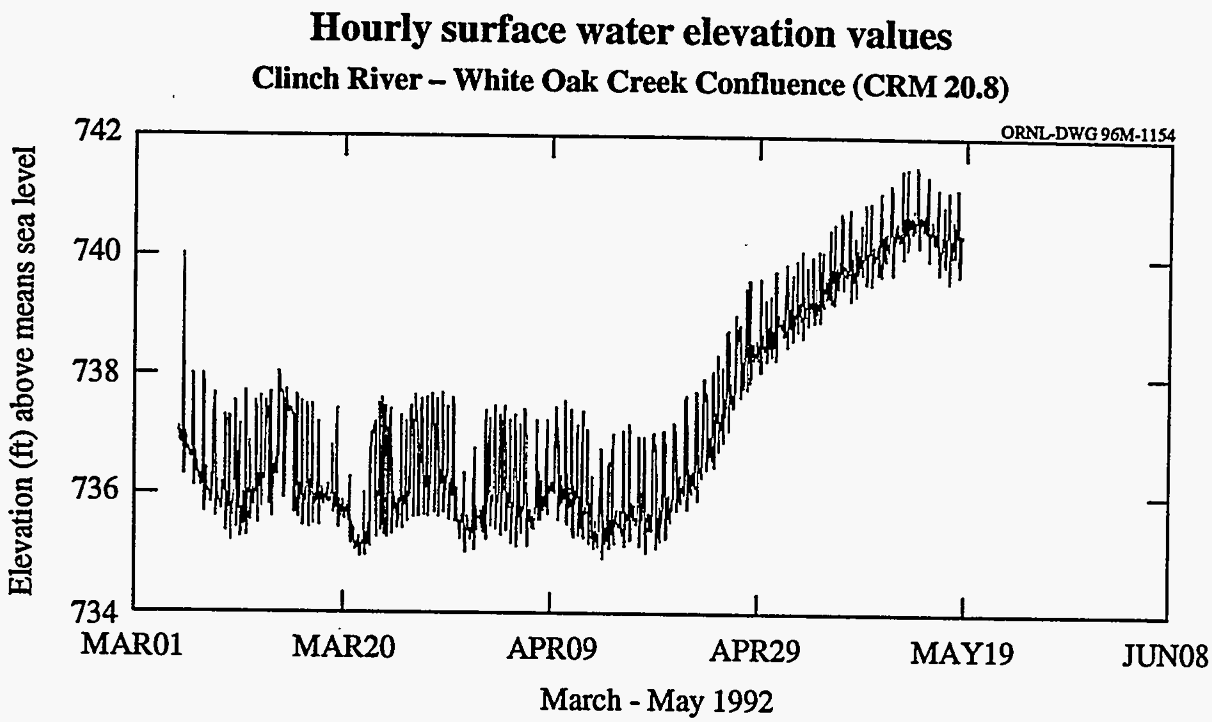


Fig. 2.12. Variation in water level elevations typical of the Clinch River immediately below Melton Hill Dam.

Although it is narrow and relatively shallow, Melton Hill Reservoir exhibits lacustrine qualities for the lower 15 to 18 miles of its length. Transition conditions persist from the Scarboro Creek confluence (CRM 41.1) to the upper end of the Bull Run Steam Plant (CRM 49). The reservoir is exclusively riverine from this point to the end of maintained navigation at CRM 66. This trend from riverine through transition to lacustrine is also observed for the off-channel embayments of the streams that drain the ORR. The situation is most pronounced in the Scarboro and McCoy Branch embayment areas. Both upper and lower embayments exhibit lake-like qualities.

Norris Reservoir, with an average annual discharge of 4200 cfs, contributes 95% of the flow into Melton Hill Reservoir. The remainder comes principally from Bull Run Creek and other minor tributaries. Several small tributaries to Melton Hill originate on the ORR and include Scarboro Creek, McCoy Branch, Walker Branch, and Bearden Creek, all of which originate on the south slope of Chestnut Ridge. The tributaries of principal interest that drain the ORR are Scarboro Creek and McCoy Branch. Wastes from the Y-12 Plant were formerly discharged into Kerr Hollow Quarry, from which they flow via Kerr Hollow Branch to Scarboro Creek, which enters the river at CRM 41.1. Fly ash from the Y-12 steam plant was formerly disposed of in Rogers Quarry, which discharges into McCoy Branch, which flows into the Clinch at CRM 37.3. The average annual discharge of each stream is <2 cfs.

2.5.2 Water Quality

Waters of the Clinch River and Poplar Creek are circumneutral, moderately alkaline, and of medium hardness. Water quality parameters measured during this RI are within the ranges for natural surface waters and those historically reported for the Clinch River and Poplar Creek (Table 2.1).

Differences in several water quality parameters may be expected between the waters of the Clinch River and those of Poplar Creek, due to differences in watershed lithologies, land use, and as a result of impoundments on the Clinch River (Fig. 2.13). For example, although principal cations in both streams are calcium and magnesium and the principal anions are carbonate-bicarbonate, sulfate is historically of increased importance in Poplar Creek, ostensibly because of coal mining activities on the Cumberland Plateau (Energy Systems 1993b). Averages and ranges for these values observed during the RI are presented in Table 2.1.

Table 2.1. Minimum, maximum, and mean values for selected water quality parameters in Clinch River and Poplar Creek[a]

Water quality parameter	Clinch River		Poplar Creek	
	Historical values[b]	CRRI[c] values	Historical values[d]	CRRI values
pH	7–8.6 (7.8)	7.4–8.2 (7.7)	6.4–8.1 (7.5)	5.3–7.9 (7.2)
Conductivity (μmhos/cm)	156–312 (244.3)	131–266 (211)	98–360 (228.7)	92–318 (211)
Alkalinity (mg/L $CaCO_3$)	54–121 (99.8)	55–120 (90.0)	20–150 (76.3)	25–130 (75.0)
Hardness (mg/L Ca)	71–140 (112)	75.3–131 (101.3)	63–146 (116)	2.3–149.5 (95.9)
Dissolved oxygen (mg/L)	5.2–12.5 (9.1)	6.3–11.2 (8.5)	6.3–13.1 (8.0)	4.7–12.5 (8.1)
Ca (mg/L)	22–40 (33.7)	18.1–35.9 (27)	17–40 (30.2)	0.03–39.3 (25.7)
Mg (mg/L)	3.8–11 (9.3)	5.3–10 (7.5)	5–13 (9.1)	0.01–12.9 (7.4)
SO_4 (mg/L)	7–27 (19.3)	13–25 (17.4)	16–72 (36.5)	16–46 (26.1)
NO_2/NO_3 (mg/L)	0.1–0.7 (0.4)	0.2–0.8 (0.5)	0.5–0.6 (0.53)	0.02–2.7 (0.55)
Fl (mg/L)	0.01–0.2 (0.1)	0.05–0.27 (0.09)	0–3 (0.36)	0.05–0.1 (0.12)
Cl (mg/L)	2–7 (3.8)	2.3–10.2 (3.5)	0.9–4 (2.2)	0.85–55 (5.8)

[a]Comparison of historically recorded values (values from STORET; sample size varies from <10 to >150, depending on parameter) to values measured during the Clinch River Remedial Investigation. The mean values are indicated in parentheses.
[b]Melton Hill Dam Tailrace, U.S. Geological Survey data.
[c]CRRI = Clinch River Remedial Investigation.
[d]Poplar Creek above East Fork Poplar Creek confluence.

Several water quality parameters [e.g., temperature, dissolved oxygen , conductivity, alkalinity, total suspended particles, turbidity, hardness] are naturally influenced by season. During this RI, several parameters varied significantly by season but not by location.

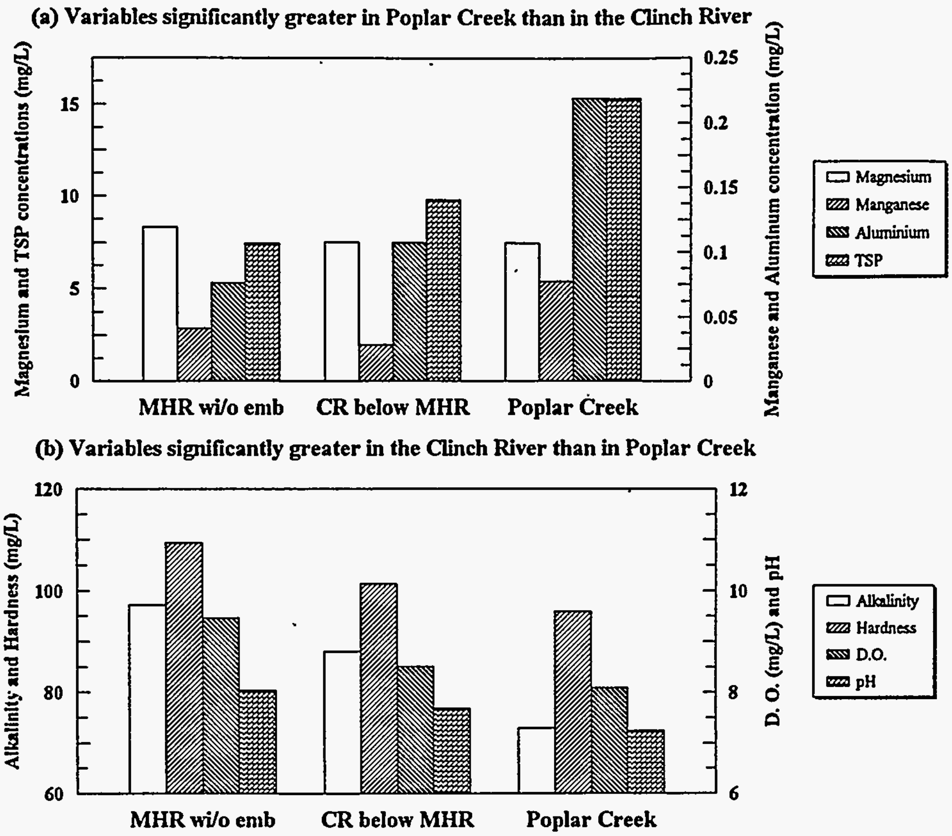


Fig. 2.13. Water quality differences between Clinch River and Poplar Creek.

Water quality parameters did not vary significantly by location in the Clinch River. Within Poplar Creek, the only parameter that did vary by location was chloride concentration, which was significantly higher ($p < 0.05$) in reach 3.3 below the K-25 Central Neutralization Facility (CNF) outfall. The outfall is currently being extended (by pipeline) to the mouth of Poplar Creek to increase effluent dilution and avoid adverse ecological effects in Poplar Creek that could result from elevated salinity during low flow conditions.

2.6 SEDIMENT

The WBR arm of the Clinch River contains an estimated 2900 acre-ft of sediment, most of which is downstream of Brashear Island (CRM 9.9, Fig. 2.2). TVA silt-range data indicate that before construction of Melton Hill Dam, the deposition of sediment was relatively uniform from the mouth of the river upstream to the current location of the dam. After completion of the dam in 1963, much of the sediment upstream of Poplar Creek had been eroded, and deposition downstream of Brashear Island increased dramatically. It is probable that high-volume, pulsed discharges from Melton Hill Dam (Sect. 2.4) resulted in the scouring of sediment immediately below the dam and its redeposition downstream of Brashear Island.

Sediment input to the Clinch River is also affected by TVA reservoirs. Norris Reservoir, with a sediment retention factor of 95% (Trimble and Carey 1984), effectively reduces the sediment basin at the mouth of the Clinch River from 4413 mile^2 to 1501 mile^2. Melton Hill Reservoir retains ~60% of the sediment entering from upstream and further reduces the local sediment basin to 1070 mile^2. The Emory River and Poplar Creek watersheds comprise 81% and 13%, respectively, of the sediment basin and serve as the primary sediment sources to the Clinch River arm of WBR. Discharges from Melton Hill Dam, although still containing a significant quantity of sediment, carry finer grained sediment that is less susceptible to deposition in the Clinch River than the coarser grained sediment carried during high flow by the Emory River and Poplar Creek.

White Oak Creek, although historically a significant source of contaminated sediment, has always been a small contributor of total sediment to the Clinch River by virtue of its small watershed. Currently, its capacity to transport sediment to the Clinch River is reduced by White Oak Dam and a recently completed sediment retention structure (coffer dam) at its mouth.

The major forces driving sedimentation processes in the Watts Bar arm of the Clinch River are (1) releases from Melton Hill Dam, (2) the operation of WBR, (3) storm events, and (4) the shape of the river bottom. WBR is operated to maintain a summer pool elevation at 741 msl, with a weekly 1-ft variation for mosquito control (Fig. 2.10). During winter drawdown the exposed shoreline is subject to repeated "washing" from boat waves and also by the rapid water-level fluctuations resulting from power generation at Melton Hill Dam (Sect. 2.5). The shape of the river bottom also influences sediment deposition patterns. The reservoir bottom can be divided into two areas: the preimpoundment river channel and the overbank area, or inundated floodplain (Fig. 2.14). The channel portion of the reservoir has historically been thought of as an area which has little long-term deposition and which functions as a sediment "pipeline" to the lower reservoir (Struxness et al. 1967). However, the presence of sediments within the channel, particularly in the lower reaches, suggests that this scenario is not entirely accurate. As flow decreases in the lower reaches of the river, sediment accumulation increases. Additionally, the washing of sediment from the near-shore areas effectively winnows sediment from the shallow area into the deep water area. The sinuous shape of the reservoir in the Clinch River arm also influences sediment deposition patterns in cross section; more sediment accumulates in the straight portions of the Clinch River than in sharp bends. The particle size distribution of sediments also strongly influences where sediments deposit. For instance, coarser sediments (sands) are typically found on the inside of river bends. Large sand spits below Jones Island (CRM 19.8) and Grubb Island (CRM 18.2) are likely depositional areas that predate the construction of Melton Hill Dam. The current deposition zone between Brashear Island (CRM 9.9) and Campbells Bend (CRM 11.5) is probably mostly sediment from Poplar Creek. Spatially and temporally varying rainfall and flow conditions obviously affect these generalizations.

A hydro-acoustic survey of the Clinch River and the lower portion of Poplar Creek was performed to gather current data regarding sediment type, thickness, and depth to sediment. Approximately 80 miles of survey lines were used in characterizing 20 miles of the river. Because the survey instrumentation required 12 ft of water to operate, the survey was restricted to the channel areas and to those near-shore areas that were safe to navigate. The survey revealed a continuum of sediment densities ranging from 1.1 (slightly denser than water) to greater than 2.5 (rock) (Table 2.2).

A map of surface sediment density as interpolated from the survey data is presented in Fig. 2.15. It should be emphasized that this figure represents a possible model of the surface sediment conditions; model data indicate that there are seasonal patterns of erosion and deposition within the Clinch River (Sect. 3.4.6). Because the hydro-acoustic survey was performed during very high flow conditions (water elevation up to 744 ft), it is likely that there was a large amount of soft flocculent and detrital matter rolling along the bottom of the river, increasing the proportions of softer sediment types at the time of the survey. Despite this, the softer surface sediment types are scattered in patches along the main channel and are intermixed with denser material. The shoreline area is dominated by denser sediments.

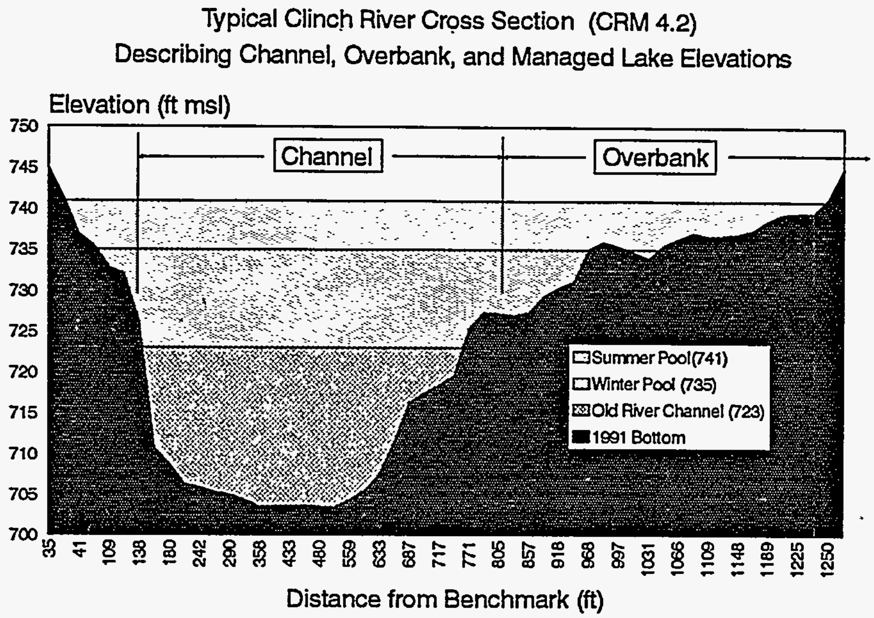


Fig. 2.14. Typical Clinch River cross section (Clinch River mile 4.2) depicting the river channel, overbank areas, and managed surface water elevations.

Table 2.2. Density ranges for various descriptive sediment type classes[a]

Density range (g/cm^3 wet)	Sediment description
1.0–1.2	Fluid mud, muds
1.2–1.4	Clays, silty clays
1.4–1.6	Clayey silt, silt
1.6–1.8	Coarse silt, clayey silty sands, very fine sand
>1.8	Sands (loose or compacted), gravel, moderate to stiff clay, hardpan clay, rock

Source: Hamilton, E. L. 1972. "Compressional-Wave Attenuation in Marine Sediments." *Geophysics* 37:620–626.

Figure 2.16 presents the percentage of each surface sediment density class for two areas within the river: (1) the near-shore area, or that area between winter and summer pool (elevation 735–741 msl), and (2) the area below the winter pool. The near-shore area accounts for 25% of the surface area and is dominated by firmer sediment types. Seventy-five percent of this near-shore area is composed of the densest two categories. The area below the 735-ft winter pool elevation is just the opposite. Only 25% of the sediment falls within the densest two categories; this is not surprising, because the near shore is exposed to wave action for part of the year. The wave action effectively washes fine sediment into deeper water.

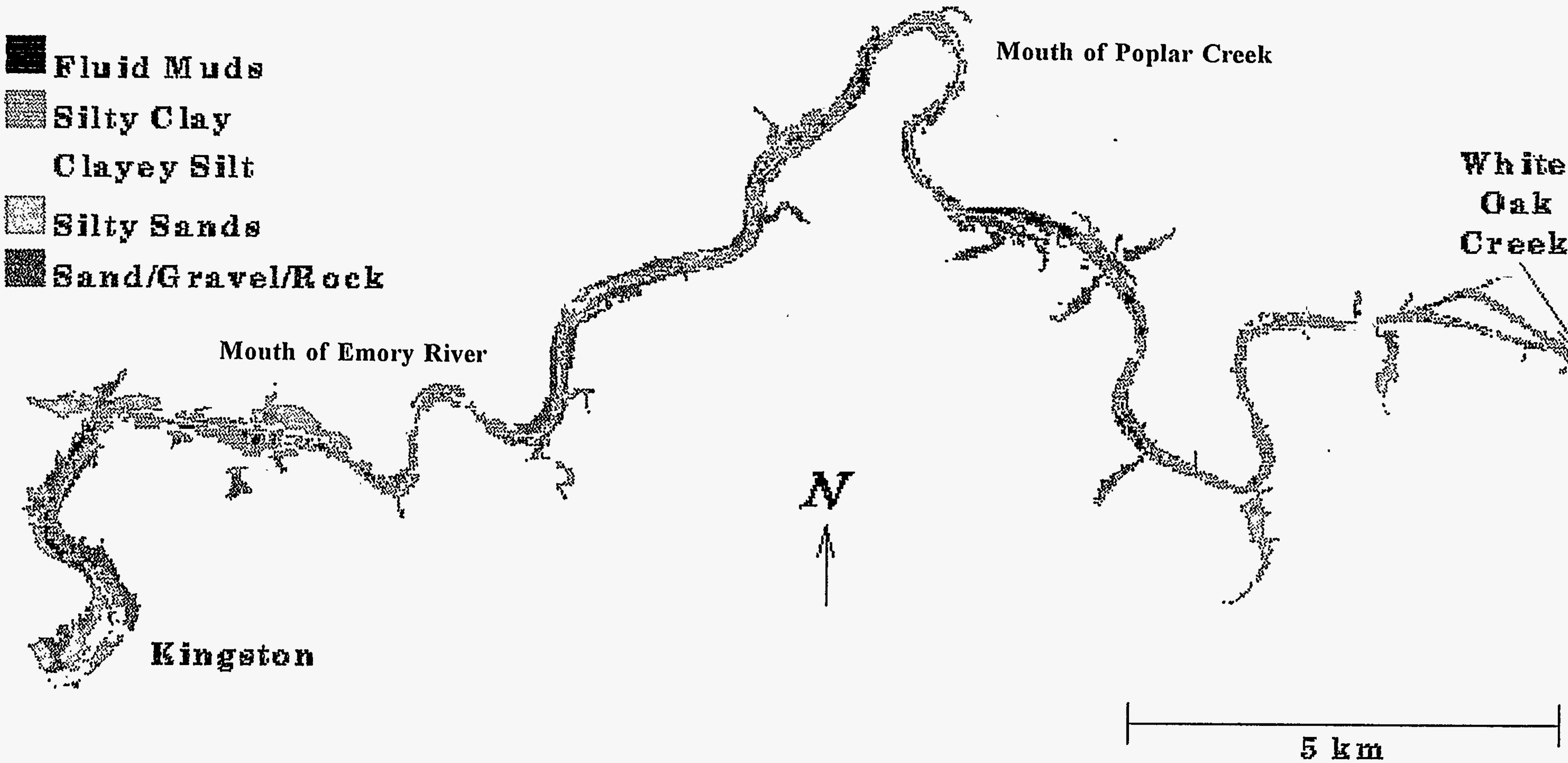


Fig. 2.15. Density of surface sediment in the Clinch River downstream of the Oak Ridge Reservation.

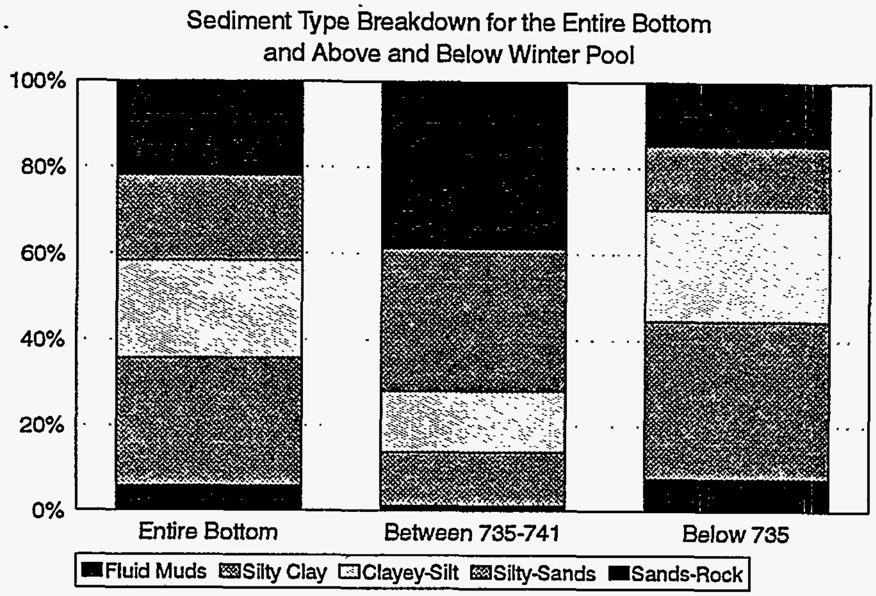


Fig. 2.16. Relative proportions of sediment types in the Clinch River downstream of the Oak Ridge Reservation.

Microtopography also has an influence on deposition pattern. The geologic formations in the area are often exposed, and they cross the river bottom at various angles. The tilted nature and the varying erosional characteristics of these formations make for a "jagged" bottom in which the harder formations stick up and the softer formations are eroded away. The jagged bottom provides numerous locations for sediment deposition because of local low spots and changes in water flow. The result is a highly variable sediment deposition pattern, which is illustrated in the side-scan sonar output from the hydro-acoustic survey (Fig. 2.17). This figure, a plan view of a small section of the river, is a gray-scale representation of river bottom hardness. The dark features are hard rock bottom, and the light features represent softer material. Geologic formations and the associated deposition zone are readily apparent.

Two major construction operations have affected the distribution of sediments in the Clinch River. (1) In 1952, as part of the construction of the Kingston coal-fired steam plant, an intake channel was dredged in the Emory River, resulting in the removal of ~44,000 yd^3 of material. In the same year, a discharge channel was dredged in the Clinch River, and ~124,587 yd^3 of material was removed. In the fall of 1955, an underwater dam (weir) was constructed in the Clinch River at CRM 3.9 to facilitate the movement of colder Clinch River water up the Emory River for cooling water supply. The weir was constructed of 17,000 tons of quarry-run limestone rock dumped by barge in a line perpendicular to flow. The dam was constructed to have a crest at the 722-ft-msl elevation, which is 13 ft below the winter pool elevation. (2) In 1962, the Clinch River between Grubb Island and Melton Hill Dam of which was dredged during construction of the dam. An estimated 454,600 yd^3 of material was removed, of which approximately half was sediment and half was rock. All dredged material was placed on Grubb and Jones islands and along nearby shorelines.

Side-Scan Sonar output from US Army Corps of Engineers Survey

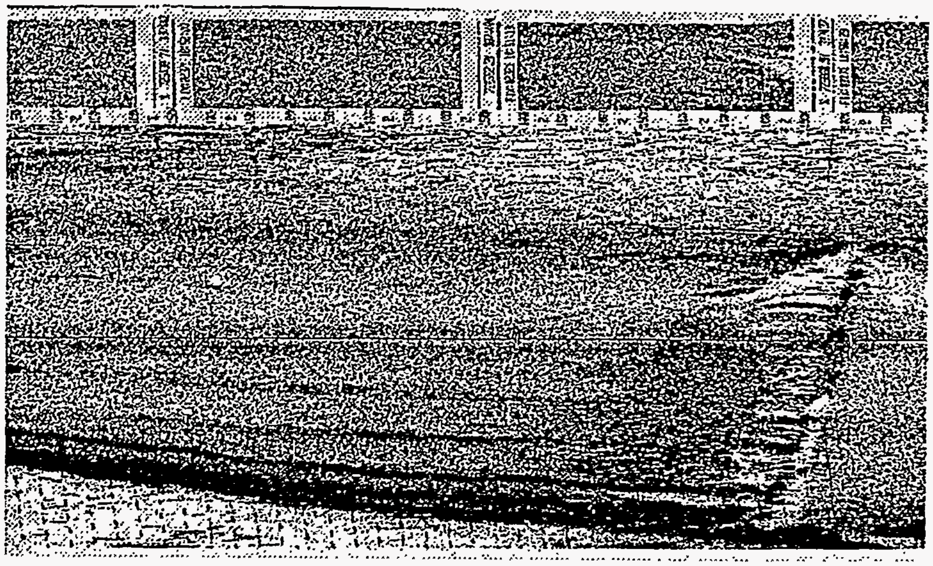


Fig. 2.17. Sonographic image of a portion of the Clinch River bottom. Darker shades represent denser sediments. The structure at the right side of the image is the weir at Clinch River mile 3.9 that deflects cooler Clinch River water to Tennessee Valley Authorty's Kingston Steam Plant.

2.7 ECOLOGY

2.7.1 Aquatic Resources

The aquatic ecosystem of the Clinch River and lower Poplar Creek can be summarized by a simplified food web diagram illustrating the principal energy flow pathways and environmental factors that regulate these pathways (Fig. 2.18). The components of this diagram are discussed in the following text.

The Clinch River arm of WBR is a highly elongated riverain impoundment that receives discharges from Melton Hill Reservoir immediately upstream. Varying hydrodynamic regimes such as flow and velocity, temperature fluctuations, and habitat availability, are the main environmental factors that influence and regulate the structure and function of the aquatic food web in the Clinch River and lower Poplar Creek. The combination of cold hypolimnetic discharges, hydropower generation, and the serial arrangement (Norris-Melton Hill to Watts Bar) of these impoundments creates complex hydrodynamic and physicochemical conditions that affect the structure and function of the biological communities in the Clinch River and Poplar Creek systems (Soballe et al. 1992). The ecological structure and functioning of biotic communities in the Clinch River and Poplar Creek are regulated by water residence time, both directly through effects of water renewal on the plankton and other components of the biota and indirectly through effects on other limnological variables, such as nutrient loading, water depth, turbidity, and mixing regime. The longitudinal ebb and flow of water and the vertical shear created by daily pulsations in reservoir discharges affect material and energy fluxes, plankton dynamics, abundance and distribution of benthic organisms, and nekton ecology. Although lower Poplar Creek can be described as an embayment of upper WBR, it actually exhibits many of the hydrodynamic and physicochemical attributes of estuarine systems, including fluctuations in water level, density gradients, and, in particular, the upstream and downstream movement of water (Loar et al. 1981).

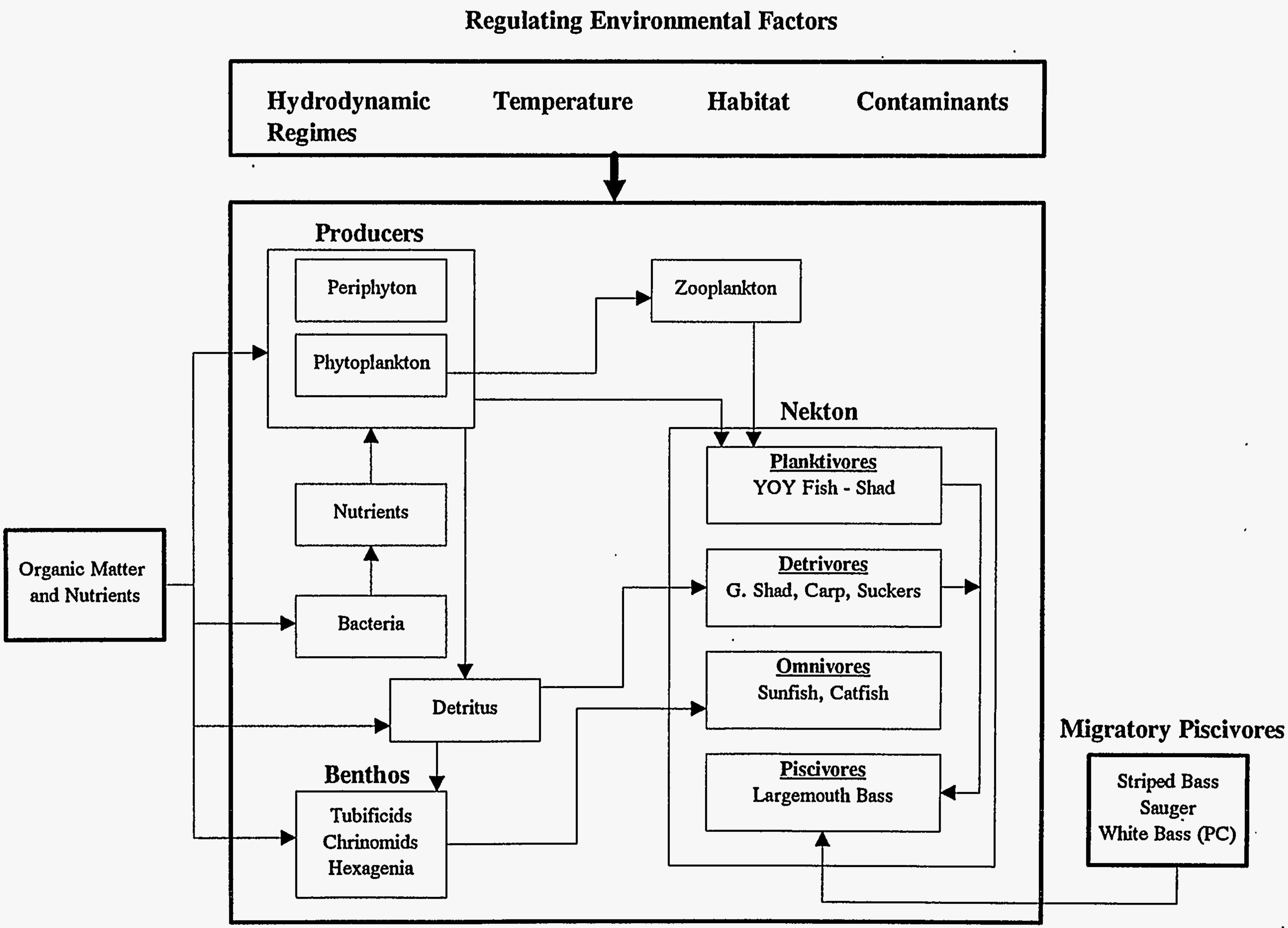


Fig. 2.18. Simplified aquatic ecosystem diagram for the Clinch River and lower Poplar Creek illustrating the principal energy flow pathways and environmental factors that regulate these pathways.

Water residence time influences planktonic communities directly through the effects of water renewal and indirectly by its relationships with other important limnological variables, such as nutrient and light availability, turbidity, and mixing regime (Adams and Hackney 1992). Short residence times generally lower overall system productivity and preclude the establishment of phytoplankton and zooplankton communities. In rapidly flushed systems such as the Clinch, plankton organisms are quickly washed downstream. Therefore, the relative importance of periphyton production to the total autotrophic production within the system is increased. In the Clinch River and Poplar Creek, the food web may depend more on allochthonous material transported from within the watershed and on littoral periphyton production than on phytoplankton production. Even though the lower reaches of Melton Hill Reservoir are primarily lotic in nature, allowing for plankton populations to develop, their contribution to the downstream food chain in the Clinch River is probably minimized because of the hypolimnetic discharge from Melton Hill. Plankton production may also be suppressed in the Clinch River by phosphorus limitation in this system (Elser and Kimmel 1985). In addition, the relative importance of allochthonous material as an energy base to the food web in the Clinch River may be minimized because of the low degree of land-water interconnectivity and the relatively small size of the local watershed (Adams and Hackney 1992). The contribution of macrophyte production to the energy base of the food web is also minimal in these two systems. Even though Eurasian water milfoil is relatively abundant in lower Watts Bar, it is sparse in the Clinch River and has rarely been reported in Poplar Creek. The energy base, therefore, for the food web (both autochthonous and allochthonous) in the Clinch River and Poplar Creek systems is probably relatively low in comparison with other lotic systems. Reduced energy availability at the lower trophic levels results in relatively low productivity at the higher trophic levels.

The combination of reservoir influences and many other related environmental factors has produced distinct macroinvertebrate communities in Poplar Creek and the Clinch. The benthic community below Melton Hill Dam is composed of the introduced Asiatic clam *(Corbicula)* and species of chironomids and tubificids (Meinert 1991). In addition, a few limited freshwater mussel stocks, possibly including a few specimens of listed endangered species, persist in sections of the Clinch River (TVA 1990a). The quality of the benthic community immediately below Melton Hill Dam is depressed, as it is at other inflow zones in the TVA system (Dycus and Meinert 1991). Benthic community quality of the Clinch River below Melton Hill Dam is characterized by TVA as generally poor to fair relative to inflow zones of mainstream Tennessee River reservoirs (Scott 1994). In both systems, the benthic communities suffer from a lack of species diversity; the systems suffer a lack of ephemeroptera, plecotera, tricoptera (EPT) taxa; long-lived species; and evenness of dominant taxa. Dominant taxa in the upper Clinch River (miles 10–23) are tubificid worms and *Chironomus* sp., whereas taxa in the lower section of the Clinch River (miles 0–9) are primarily *Chironomidae,* burrowing mayfly *(Hexagenia limbata),* fingernail clams *(Sphaeriidae),* and *Tubificidae.* In Poplar Creek, the benthic community quality is also rated as poor to fair. Deficiencies exist in long-lived taxa and EPT taxa. Abundance of benthic organisms is generally less than in the Clinch; *Tubificidae, Chironomidae,* and *Hexagenia* dominate, and very few EPT taxa are reported. The benthic community serves as a partial food supply for benthic detritivores such as gizzard shad, various sucker species, and some omnivorous species such as sunfish and small catfish.

The trophic relations of fish communities in the Clinch River and Poplar Creek are complex and highly dynamic on a temporal scale. Many species of fish are food generalists, and their feeding behavior can typically span several trophic levels (e.g., from periphyton grazing to piscivory) and can vary with season and with age. The standing crop and production of the fish communities in these two systems are dependent upon a host of environmental factors, the most important of which are water residence time, nutrient concentrations, the quality and quantity of food, and habitat availability. As noted earlier, reservoir operational practices (e.g., water level fluctuations, flow regimes) profoundly affect each of these factors, thus influencing the structure and function of fish communities.

The fish community in both the Clinch River and the Poplar Creek systems is characterized by relatively low species diversity, although representatives of all four main functional feeding groups are present. Planktivores include threadfin shad and primarily young-of-the-year fish of most other species. The principal detritivores are gizzard shad, carp, and suckers. Omnivores are dominated by water column feeders such as sunfish, and benthic consumers are primarily catfish. Resident piscivores are primarily largemouth bass. In both systems, striped bass and sauger are seasonal migratory species, and white bass use Poplar Creek as a major spawning area during the spring.

Analysis of fish community structure in the Clinch River by the TVA Reservoir Vital Signs Monitoring Program (Hickman et al. 1991) indicates that the dominant trophic group is planktivores/detritivores (shad), which comprise 50–60% of the abundance; furthermore, omnivores (carp, suckers) comprise 10–20%, insectivores (sunfish) comprise 10–15%, and piscivores (black bass, striped bass, and striped bass hybrids) comprise 5–10% of the abundance. The most abundant species in the Clinch River during the fall of 1993 and 1994 were found to be gizzard shad, bluegill minnows, *Cyprinidae*, and largemouth bass (listed in order of importance). In Poplar Creek during this period, the most dominant species were gizzard and threadfin shad, bluegill, largemouth bass, minnows, yellow bass, and carp (Bevelhimer and Adams, in press). The differences observed in the occurrence and relative abundance of species in Poplar Creek and the Clinch River are caused by the seasonal shifts in the distributions of various species as well as to the nature and range of habitat (quantity and quality) in these two areas. Fish communities surveys suggest that there may be a relationship between (1) species diversity and richness and (2) proximity to Poplar Creek. The number of species found in Poplar Creek and the Clinch River immediately downstream of the Poplar Creek mouth ranged from 26 to 28, whereas the number of species found at other sites in the Watts Bar system ranged from 30 to 37 (Bevelhimer and Adams, in press).

In general, fish communities in the Clinch River are in poorer condition than would be expected of communities in similar tributary streams in the Tennessee Valley (Hickman et al. 1991). Shad (gizzard and threadfin) dominate the fish community in the Clinch River and Poplar Creek. In late winter, sauger migrate upstream to spawn in Melton Hill tailwaters, while in late spring and early summer, striped bass migrate from other areas of Watts Bar to seek thermal refuges in the cooler waters of the upper Clinch. Poplar Creek and lower Poplar Creek, in particular, appear to be important spring spawning areas for both gizzard shad and white bass.

2.7.2 Terrestrial Resources

This section focuses on the terrestrial environment located adjacent to the Clinch River and Poplar Creek. Two zones will be discussed: (1) the river bank directly above the water level (i.e., supralittoral/epilittoral zone) and (2) a riparian zone that extends 100 m from the shoreline. Both zones will be discussed in relation to the vegetation, land use/land cover, wetlands, threatened or endangered plant/animal species, and wildlife present at the site.

The supralittoral/epilittoral zone of the Clinch River and Poplar Creek varies in size, riparian vegetation, and soil type (i.e., sand, clay, gravel). The river banks are predominantly covered with deciduous forests, grasslands, or emergent macrophytes, but some lack vegetative cover entirely. These exposed high vertical banks are prime burrow locations for various birds. The belted kingfisher (*Megaceryle alcyon*), rough-winged swallow (*Stelgidopteryz ruficollis*), and bank swallow (*Riparia riparia*) inhabit the burrows during their breeding seasons. These birds use the burrows from approximately April to June; laying and incubating a clutch of eggs and raising their young in the terminal chamber until they fledge. The burrows found along the banks of the Clinch River were ~1.88 m (6 ft, 2 in.) from the surface of the water and 0.61 m (2 ft) from the top of the embankment (Baron,

L. A., Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, Tenn., personal communication, June 1995). Through most of the year (excluding potential migratory habits), belted kingfishers and swallows within the Clinch River or Poplar Creek system have a piscivorous and insectivorous diet, respectively. Many other piscivorous avian species (e.g., great blue heron, black-crowned night heron, osprey) nest within the riparian vegetation or have platform nesting sites (specifically osprey) over the river or creek.

A variety of habitats border the Clinch River and Poplar Creek within the riparian zone. A land use/land cover map (Fig. 2.19) displays habitat types within 100 m of the shoreline. The habitat map was derived from LANDSAT Thematic Mapper and SPOT satellite images generated in September 1984. The map delineates nine land use/land cover categories: water, urban, agricultural, transitional areas, deciduous forest, mixed forest, evergreen forest, evergreen plantations, and barren land (Table 2.3). The percentage and area (hectares) of various land use and habitat types within each reach are found in Table 2.4. Habitat information was available for ~31 miles of the Clinch River, which included reaches 1 and 2, and subreach 4.01, and for all of the Poplar Creek Embayment (reach 3). Although the riparian zone of Poplar Creek is predominantly urban (43.8%), the riparian zone of the Clinch River is predominantly forest (45.5%) and transitional (33.4%).

Some wetlands exist along the Clinch River (subreachs 2.02 and 2.04), Poplar Creek (subreaches 3.04, 3.03, and 3.02), and upper McCoy Branch Embayment (subreach 7.02). These areas are considered wetlands based on the technical criteria of (1) hydrophytic vegetation, (2) hydric soils, and (3) wetland hydrology (U.S. Army Corps of Engineers 1987). Dominant and common plant species found in these areas are listed in Table A1.

Rare and endangered plant surveys have been performed for most of the ORR. Table A2 identifies all threatened, endangered, and special concern plant species that occur on the ORR, and in some cases they have been specifically located along the Clinch River (Pounds et al. 1993). Specific areas along the Clinch River are designated as natural areas and some have been state registered because of the presence of endangered or threatened plant species. These areas along the Clinch River include the Campbell Bend Bluff and Forest (mile ~11.8), Breeder Bluffs (mile 18), Raccoon Creek Golden Seal Area (mile ~19.9–20), Melton Dam Bluffs (mile ~23.6), Tower Shielding Bluffs (mile ~25.2–26), Health Physics Research Reactor Lake Bluffs (mile ~27.2), South Hickory Creek Bend Bluffs (mile ~28.4), North Hickory Creek Bend Bluffs (mile 29), Copper Ridge Outcrop (mile ~31.5), White Cedar Area (mile ~32.6), Bull Bluff (mile 36.9), and Rainy Knob Bluff, Freels Bend (mile ~40.5).

There is only one state-registered natural area along Poplar Creek (Poplar Creek Cliffs) upstream of the confluence with EFPC. Many species that are unusual for the ORR occur in small limestone cliffs near the stream. These species include hemlock (*Tsuga canadensis*), rhododendron (*Rhododendron maximum*), fringe tree (*Chionanthus virginicus*), spider lily (*Hymenocallis fulva*), and mock orange (*Philadelphis hirsutus*). The spreading false-foxglove (*Aureolaria patula*), which is a state-listed and federal-candidate species, occurs here (Pounds et al. 1993).

As would be expected given the diversity of plants and habitat types, the terrestrial environment adjacent to the Clinch River and Poplar Creek supports a wide variety of wildlife species. A list of amphibians, reptiles, birds, and mammals on the ORR has been compiled from various published sources and personal observations (Table A3). This list includes species that have been identified on the ORR as well as some animals that are not confirmed but are present in the Ridge and Valley region. These terrestrial species found on the ORR could use habitat in and adjacent to the riverain system. Table A3 also includes more than 40 species that are listed by either the Tennessee Wildlife Resources Agency

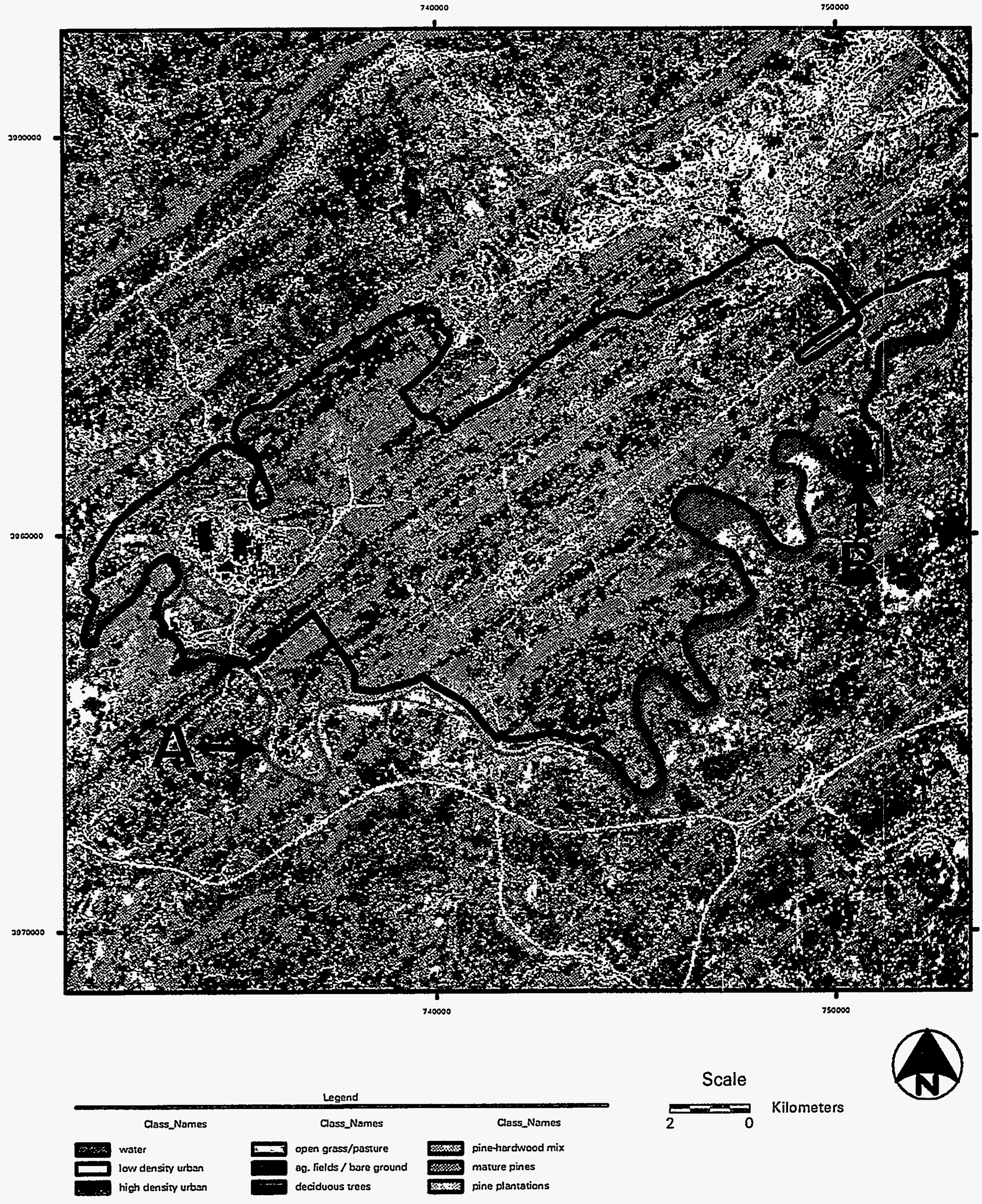


Fig. 2.19. Land use on and immediately surrounding the Oak Ridge Reservation. Site A was being developed at the time for the Tennessee Valley Authority's Clinch River Breeder Reactor and is not urban land as the legend suggests. Site B is the Freels Bend agricultural station.

Table 2.3. The land use/land cover classes used in habitat classification

Land use/land cover	Description
Urban land	Mixture of administrative buildings, laboratories, heavy commercial and industrial buildings, lawns, and clumped shade trees.
Deciduous forest land	Areas of hardwood forest types.
Mixed forest land	Areas of a mixture of hardwoods and pine trees.
Evergreen forest land	Areas dominated by mature pine forest type with trees generally older than 35 years (in 1994), with an uneven canopy.
Evergreen plantation	Areas of pine trees that are row planted, are of uniform age, and are generally younger than 35 years (in 1994).
Agricultural land	Fields of pasture grasses, grassland, row crops, and/or shrubland cover.
Transitional areas	Secondary early successional sites, usually grassland to grassland shrub mix; generally mowed along powerline corridors.
Barren land	Cropped fields, plowed or bare ground areas, or areas where vegetation has been removed, such as construction sites or quarries.

or the U.S. Fish and Wildlife Service as threatened, endangered, or in need of management. Each species is identified within a trophic category (i.e., aquatic, arboreal, or ground invertebrate feeder; flying insectivore; herbivore; omnivore; piscivore; predator) indicating potential sources of contaminant exposure from the animal's diet and life history. Several common terrestrial species present along the Clinch River and Poplar Creek systems, which may be exposed to site contaminants, will be addressed in detail in Chap. 6 (Baseline Ecological Risk Assessment).

These species include osprey, great blue heron, and mink (piscivorous wildlife; Sect. 6.4); rough-winged swallows, and little brown bats/gray bats (insectivorous wildlife; Sect. 6.5); and cottontail rabbits (herbivorous wildlife; Sect. 6.7).

2.8 SUMMARY OF SITE DESCRIPTION

The overall environmental setting of the ORR and CR/PC OU have been described. Subsequent chapters will characterize the nature and extent of contamination in the OU, assess the resulting risk to human and ecological receptors, and weigh the feasibility of implementing various remedial actions.

The volume of surface water in the OU is very large. Surface water is also very transient, with short average retention times.

Table 2.4. Land use/land cover categories, percentage and area (hectares) of coverage within each Clinch River and Poplar Creek reach (extending 100 m from shoreline on both sides of river and creek)

Category	Reach 1		Reach 2		Reach 3		Reach 4 (subreach 4.01 only)	
	Percentage	Area (ha)	Percentage	Area (ha)	Percentage	Area (ha)	Percentage	Area (ha)
Urban	9.6	100.88	15.1	95.13	43.8	99.44	8.3	13.81
Deciduous forest	9.1	95.44	7.3	46.00	11.3	25.56	26.7	44.63
Mixed forest	32.9	345.13	23.8	150.06	5.6	12.69	23.5	39.31
Evergreen forest	5.9	62.25	5.8	36.63	2.8	6.38	1.1	1.88
Evergreen plantation	2.6	26.75	4.7	29.70	1.7	3.81	0.78	1.31
Agricultural	6.9	72.44	11.8	74.69	3.1	7.00	3.4	5.63
Transitional areas	32.7	343.25	31.4	198.13	31.3	71.06	36.2	60.56
Barren	0.27	2.88	0.21	1.31	0.39	0.88	0.07	0.13

The greatest quantity of sediment-associated contaminants is expected to be found where large quantities of sediment occur. In the CR/PC OU (downstream of the primary contaminant sources), this would include the lower reaches of the Poplar Creek embayment and the Clinch River downstream of Poplar Creek. Even here, relatively little sediment is found in the near-shore and shallow areas, deposition being greatest in the deeper water along the river channel.

The CR/PC OU contains aquatic and terrestrial biological receptors, some of which would likely be exposed to contaminants that have been released from the ORR. Certain biota may serve as indirect sources of contamination to organisms higher on the food chain.

Finally, humans use the Clinch River and Poplar Creek as a source of drinking water, as a (limited) food source, and for recreation. Such persons would likely be exposed to contaminants that have been released from the ORR.

3. SITE HISTORY AND CURRENT CONDITIONS

This section characterizes the nature and extent of contamination in the CR/PC OU. Section 3.1 provides a more detailed description of the study area. Background information on ORR site operations and an overview of contaminant releases is provided in Sect. 3.2. The nature and extent of contamination is characterized in Sects. 3.3 (water), 3.4 (sediment), and 3.5 (biota). The results are summarized in Sect. 3.6. The quality of the CRRI Phase 2 data is summarized in Appendix J and evaluated in detail by Holladay et al. (1995).

3.1 SITE DESCRIPTION

To facilitate the conduct of the RI, CR/PC was divided into five study reaches and six reference reaches (Fig. 3.1). The more intensively studied reaches were divided further into a total of 14 subreaches. The boundaries of each reach and subreach are briefly described below. In general, boundaries were established to distinguish between (1) areas where natural environmental conditions are known to differ and (2) areas where effects are anticipated from plausible contaminant sources.

The reaches described below can be assigned to one of three categories: study reaches, negative reference reaches, and positive reference reaches. Study reaches are those areas within the CR/PC OU in which the nature and extent of contamination is being investigated. Negative reference reaches are those areas upstream of study areas but unaffected by releases from the ORR. A negative reference reach may, however, contain contaminants from sources other than the ORR. If contaminant releases from the ORR have resulted in off-site contamination, contaminant levels in the study reach should be higher than in the appropriate negative reference reach. A positive reference reach is also upstream of a study reach but has a known ORR contaminant source in its watershed (e.g., WOC, ORNL). Positive reference reaches are outside the CR/PC OU but generally are part of another OU on the ORR. Contaminant concentrations in a positive reference reach should be higher than in the study reaches. Contaminant data for positive reference reaches was collected by other environmental programs at the ORR and are presented here for perspective and a greater understanding of site conditions.

Reach 10—Norris Reservoir is a negative reference reach on the Clinch and Powell rivers. Norris Dam is located at CRM 79.8, almost 46 miles upstream of the ORR.

Reach 0—Upper Melton Hill Reservoir is a negative reference reach extending from CRM 49 upstream ~3 miles.

Reach 1—Lower Melton Hill Reservoir is the extent of Melton Hill Reservoir that borders the ORR. This portion of the reservoir extends from Melton Hill Dam at CRM 23.1 to CRM 49. The upstream boundary coincides with the location of the Elza Gate Formerly Utilized Sites Remedial Action Program site. Contaminant flux to reach 1 from DOE activities is believed to be small; the only documented DOE source is the Y-12 Plant through McCoy Branch.

Reach 7—McCoy Branch Embayment is a small embayment of Melton Hill Reservoir that has potentially been affected by the historical disposal of fly ash from the Y-12 Plant into McCoy Branch and Rogers Quarry, upstream. The reach is bisected by a road into two subreaches, described as follows.

Subreach 7.01—Upper McCoy Branch is the portion of the embayment upstream of Bull Bluff Road. The potential for impacts is greater in this subreach because the waters are poorly mixed with those of the reservoir and sediments accumulate here.

Subreach 7.02—Lower McCoy Branch is the part of McCoy Branch Embayment downstream of Bull Bluff Road. The hydraulic connection to upper McCoy Branch Embayment is by culvert.

Reach 8—Walker Branch Embayment is a negative reference reach for the McCoy Branch Embayment. No waste disposal has ever occurred in the Walker Branch Watershed.

Reach 2—The upper Clinch River arm of Watts Bar Reservoir is above the mouth of Poplar Creek (CRM 12.0–CRM 23.1). This reach is riverine in nature and has limited sediment accumulation in the main channel. White Oak Creek (WOC), the primary surface water source of contaminants from ORNL, enters the Clinch River in this reach. There are four subreaches, described as follows.

Subreach 2.01—Melton Hill Dam (CRM 23.1) to the mouth of WOC (CRM 20.8) is a very riverine subreach, the characteristics of which are strongly influenced by water releases from Melton Hill Dam. Although upstream of the contaminant sources on WOC, the potential for backflow exists when Melton Hill Dam is not discharging.

Subreach 2.02—The mouth of WOC (CRM 20.8) to the mouth of Pawpaw Creek (CRM 19.0) contains the least dilute contaminants from ORNL.

Subreach 2.03—The mouth of Pawpaw Creek (CRM 19.0) to the mouth of Grassy Creek (CRM 14.5) is riverine but has a zone of sediment accumulation at the downstream end of Jones Island. The K-25 Site potable water intake is at the lower boundary of this subreach.

Subreach 2.04—The mouth of Grassy Creek (CRM 14.5) to the mouth of Poplar Creek (CRM 12.0) is the last subreach upstream of contaminant sources on Poplar Creek.

Reach 13—Poplar Creek upstream of EFPC (above PCM 5.5) serves as the negative reference reach for Poplar Creek. This reach receives effluent from the town of Oliver Springs, located upstream.

Reach 3—The Poplar Creek Embayment of Watts Bar Reservoir is ecologically important as a spawning area for several fish species. The surrounding riparian forest supports piscivorous wildlife that feed on the aquatic life of the embayment and the Clinch River. The embayment is an area of sediment accumulation and has historically received contaminants from the Y-12 Plant and K-25 Site, as well as from the communities of Oak Ridge and Oliver Springs. Backflow from the Clinch River is common. This reach extends from the mouth of Poplar Creek upstream 5.5 miles to the mouth of EFPC. There are four subreaches, established primarily on the basis of potential contaminant sources.

Subreach 3.01—The mouth of EFPC (PCM 5.5) to the mouth of Mitchell Branch (PCM 4.6). Any effluent or runoff containing contaminants from the Y-12 Plant or the City of Oak Ridge enter Poplar Creek via EFPC.

Subreach 3.02—The mouth of Mitchell Branch (PCM 4.6) to the K-25 CNF outfall (PCM 3.4). Several K-25 waste outfalls have historically discharged effluent to Mitchell Branch.

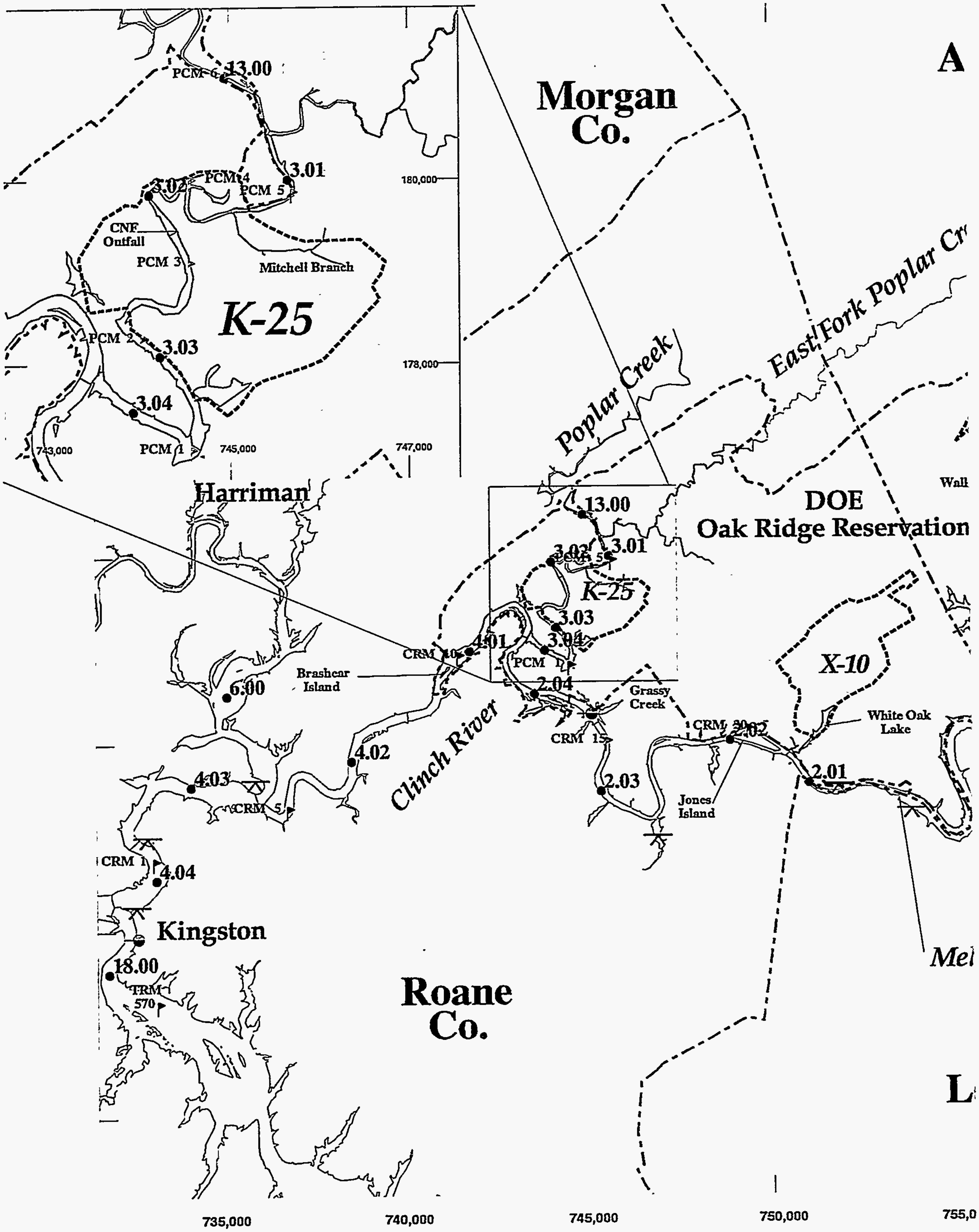
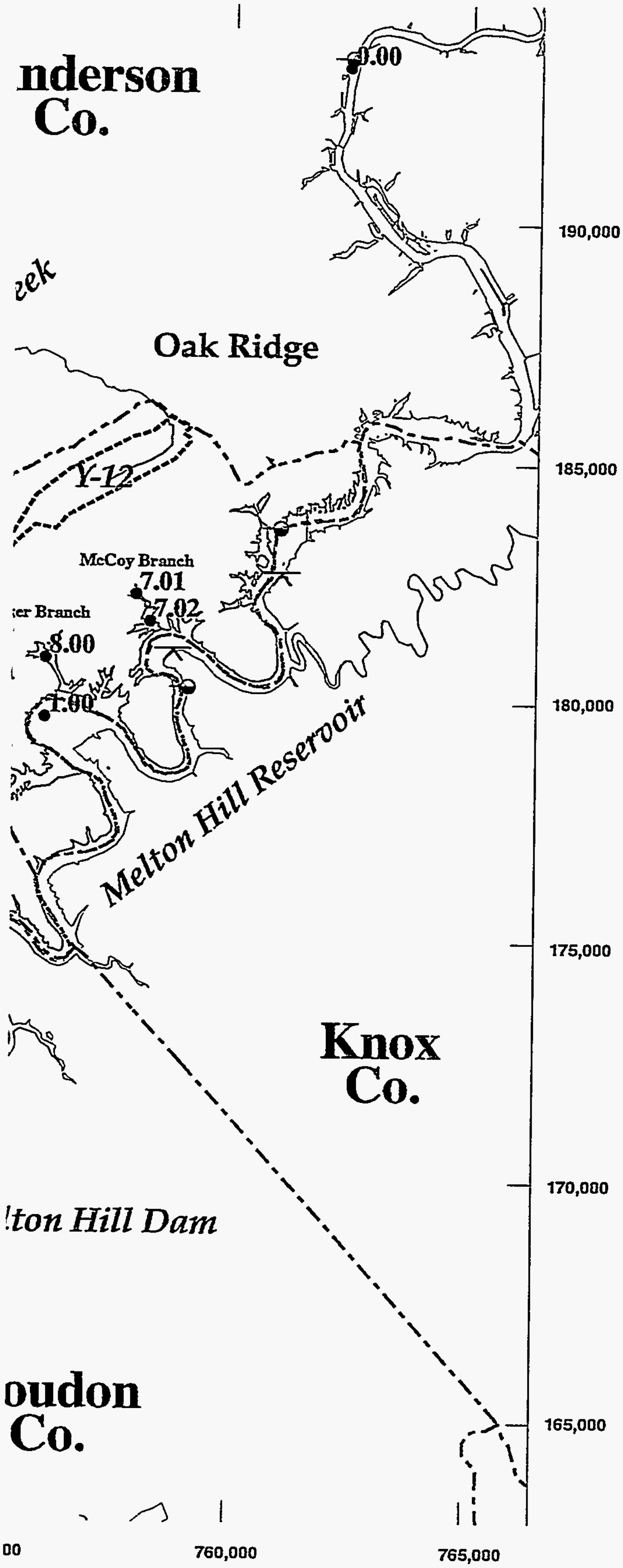


Fig. 3.1. Reaches and subreaches studied as part of the Clinch River/Poplar Creek Remedial Investigation.



Clinch River Environmental Restoration Program

Study Sites

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LEGEND

●	Subreaches
	River Mile Marker
	Water Intake
	Recreation Area
	DOE Boundary
	Plant Boundary
	County Boundary

Tennessee State Plane Coordinate System, NAD 83
Source: Lake boundary provided by TVA
Sample locations provided by
Environmental Science Division

Prepared by GeoSpatial Support (GSS)
Eironmental Information Management
Enviromental Restoration Division

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Subreach 3.03—The CNF outfall (PCM 3.4) to the Poplar Creek ash disposal area (PCM 1.0). Historically, several K-25 Site sources have discharged wastewaters to this subreach. The K-25 CNF currently discharges to this subreach but is being extended to allow discharge directly to the Clinch River.

Subreach 3.04—The ash disposal area (PCM 1.0) to the mouth of Poplar Creek (CRM 12.0). Fly ash from the K-25 steam plant was historically discharged to the creek as a means of disposal. This is a zone of sediment accumulation where particle-associated contaminants might be expected to accumulate.

Reach 4—The lower Clinch River arm of Watts Bar Reservoir extends from the mouth of Poplar Creek (CRM 12.0) downstream to the mouth of the Clinch River (at TRM 567.5). This reach integrates all ORR contaminant sources. Residential land use along the river is greatest in this reach. It is an area of increasingly lacustrine conditions and sediment deposition. There are four subreaches, described as follows.

Subreach 4.01—The mouth of Poplar Creek (PCM 12.0) to downstream of Brashear Island (CRM 8.5) makes up the Clinch River subreach with the least dilution of contaminants from Poplar Creek. Significant sediment accumulation occurs upstream and immediately downstream of Brashear Island.

Subreach 4.02—Brashear Island (CRM 8.5) to the mouth of the Emory River (CRM 4.4) is a zone of increasingly significant sediment accumulation and shoreline development.

Subreach 4.03—The mouth of the Emory River (CRM 4.4) to near Kingston City Park (CRM 1.5) is the subreach where dilution of ORR contaminants is expected because of flow from the Emory River. The Emory River receives effluent from TVA's Kingston Steam Plant. Urban areas within the Emory River watershed include Harriman and Crossville.

Subreach 4.04—Kingston City Park (CRM 1.5) to the mouth of the Clinch River is a lacustrine area with significant sediment accumulation. The shoreline is within the city of Kingston, and use is primarily residential and recreational.

Reach 18—The upper Tennessee River arm of Watts Bar Reservoir is located above the mouth of the Clinch River and serves as a negative reference reach for ORR contaminants.

Reach 5—Lower Watts Bar Reservoir is downstream of the mouth of the Clinch River. Initially investigated during Phase 1 of the CRRI, this reach was subsequently designated a separate OU, and a remedial decision was reached on the basis of Phase I data and other data (DOE 1995). Data from the Phase 1 investigation is presented here primarily for comparison.

Other reaches discussed to some extent in this report include WOC (reach 22), Mitchell Branch (reach 21), Bear Creek (reach 20.02), and EFPC (reach 20.01), all of which are outside the OU but which serve as positive reference reaches. Although no CRRI Phase 1 or Phase 2 data were collected from these reaches, ORR Environmental Monitoring Program (ORREMP) data are used to assist in the evaluation of downstream study reaches. In addition, fish from the Emory River (reach 6) were collected during the Phase 1 investigation and are discussed in Sect. 3.5.

3.2 OPERATIONAL INFORMATION AND RELEASE HISTORY

This section briefly describes the major releases of hazardous substances from each of the three main ORR facilities to the off-site environment. Little quantitative information regarding the release of hazardous substances is available for contaminants other than radionuclides, and even these estimates are relatively crude, particularly for the early years of operation.

The current or threatened release of hazardous substances from specific sites on the ORR is the focus of current CERCLA source control actions. The quantification of these releases is being accomplished at the source; similarly, remedies will also be effected at the source. The focus of the CRRI is to characterize ambient concentrations in Clinch River and Poplar Creek media.

3.2.1 The Y-12 Plant

The original mission of the Y-12 Plant (Fig. 3.2), completed in 1943, was the electromagnetic separation of ^{235}U from ^{238}U for use in the production of atomic weapons at Los Alamos National Laboratory. The production processes generated significant quantities of liquid waste, from which economically recoverable amounts of uranium were historically recycled on-site. However, the remaining wastes were discharged directly to the plant's storm sewer system and EFPC (Griffith 1957 in Bruce et al. 1993). An estimated 27 Ci (88,000 lbs) of uranium were discharged to EFPC by the time the electromagnetic enrichment process ended in 1947 (DOE 1988).

In the mid-1940s, the Y-12 Plant began producing enriched uranium weapons components. The associated large-scale processing (metal machining, chemical processing, uranium salvage, and recovery) operations again resulted in releases of uranium-containing liquid waste to the storm sewer system and to EFPC. Releases from these operations were greatest from 1959 through 1970, primarily reflecting increased productivity during this period (DOE 1988). Other elements used in the production of weapons components and that may have been released as contaminants to EFPC include Th, Li, Be, and Pb. In addition, the industrial-scale machining operations at the Y-12 Plant involved the use of large quantities of cutting oils, machine coolant, degreasers, and solvents, including a number of chlorinated solvents, all of which may have been released in small quantities to EFPC. For many years, cutting oils at the Y-12 Plant contained PCBs. Waste PCB oils were collected in storage tanks and disposed of in the Bear Creek Burial Grounds. These burial grounds are located about 2 miles west of the main plant on the southern slope of Pine Ridge. They were primarily used for the disposal of low-level radioactive waste (primarily uranium), but the disposal of wastes containing organics and metals occurred as well. Runoff and seepage from the burial grounds has contributed contaminants to Bear Creek, a tributary to EFPC (DOE 1994a). The use of cutting oils containing PCBs was discontinued in 1976. The manufacture of nuclear weapons components was discontinued at the Y-12 Plant in 1992. The Y-12 Plant is used today to store weapons-grade uranium from disassembled nuclear warheads.

In the early 1950s, the Y-12 Plant began the large-scale separation of lithium isotopes for use in the production of thermonuclear fusion weapons, or hydrogen bombs. The Colex process was developed to separate ^{6}Li from ^{7}Li. The name Colex referred to a "column exchange" process that used an aqueous lithium hydroxide solution and a solution of lithium in mercury to achieve the desired separation. Millions of pounds of mercury were required for the process, and large quantities were accidentally released to process buildings, soils, and surface water (Wilcox 1983). By the time the Colex process was no longer used in 1963, an estimated 239,000 lbs of mercury were released to EFPC, and another 428,000 lbs were released to the ground in the plant area (Wilcox 1983).

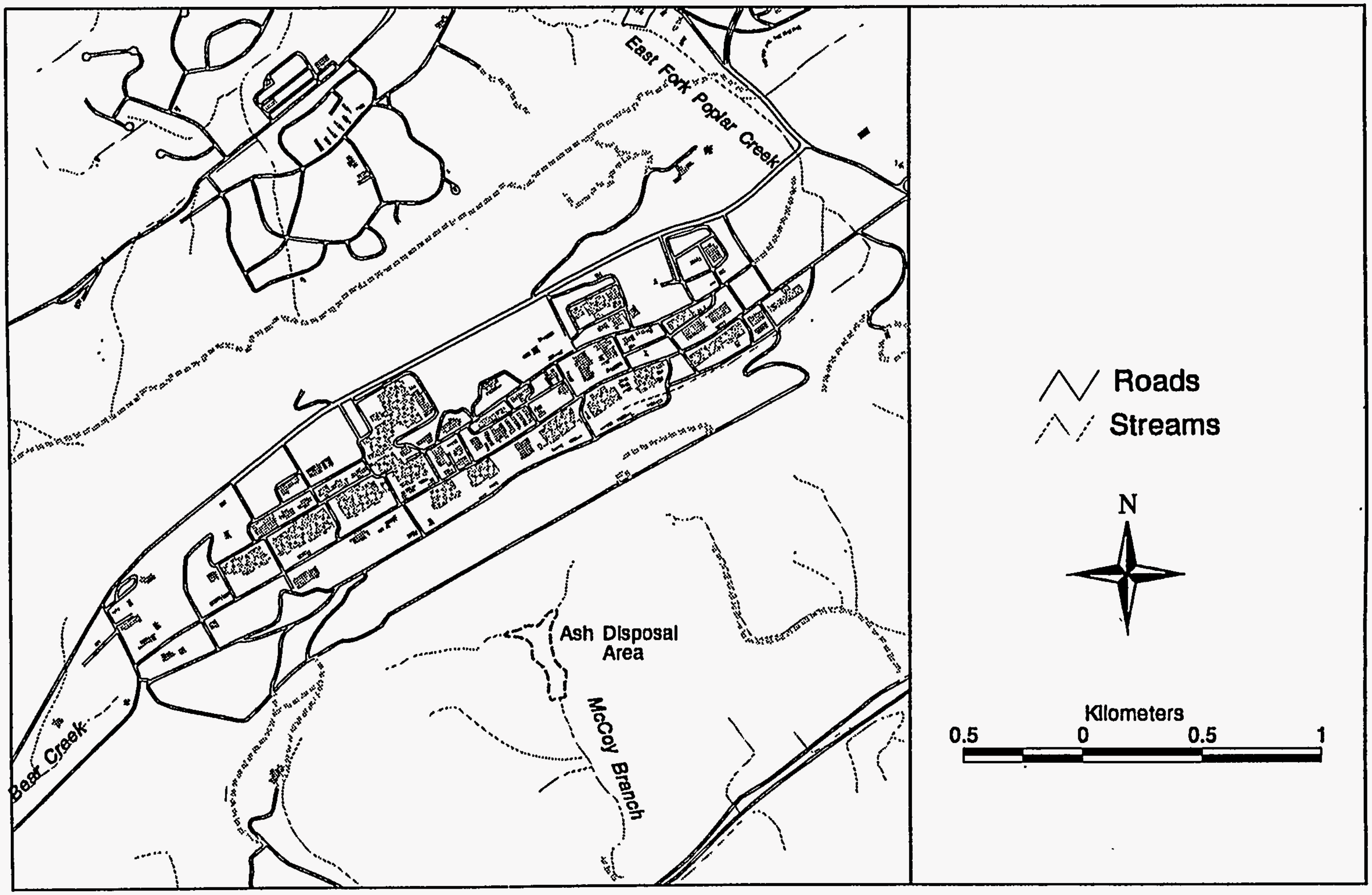


Fig. 3.2. The Oak Ridge Y-12 Plant.

Peak releases of mercury occurred in 1957 and 1958, when ~73,000 lbs and 64,000 lbs were released to EFPC, respectively (Fig. 3.3). Several hundred thousand pounds remained in buildings and process equipment at the Y-12 Plant. Mercury releases declined dramatically after 1958, but the contaminated storm sewer system, buildings, and soils at the Y-12 Plant continue to contribute several grams of mercury per day to EFPC. These areas are the subject of ongoing remedial actions and decontamination and decommissioning activities. Remedial action on EFPC floodplain soils is slated to be completed by the end of 1996.

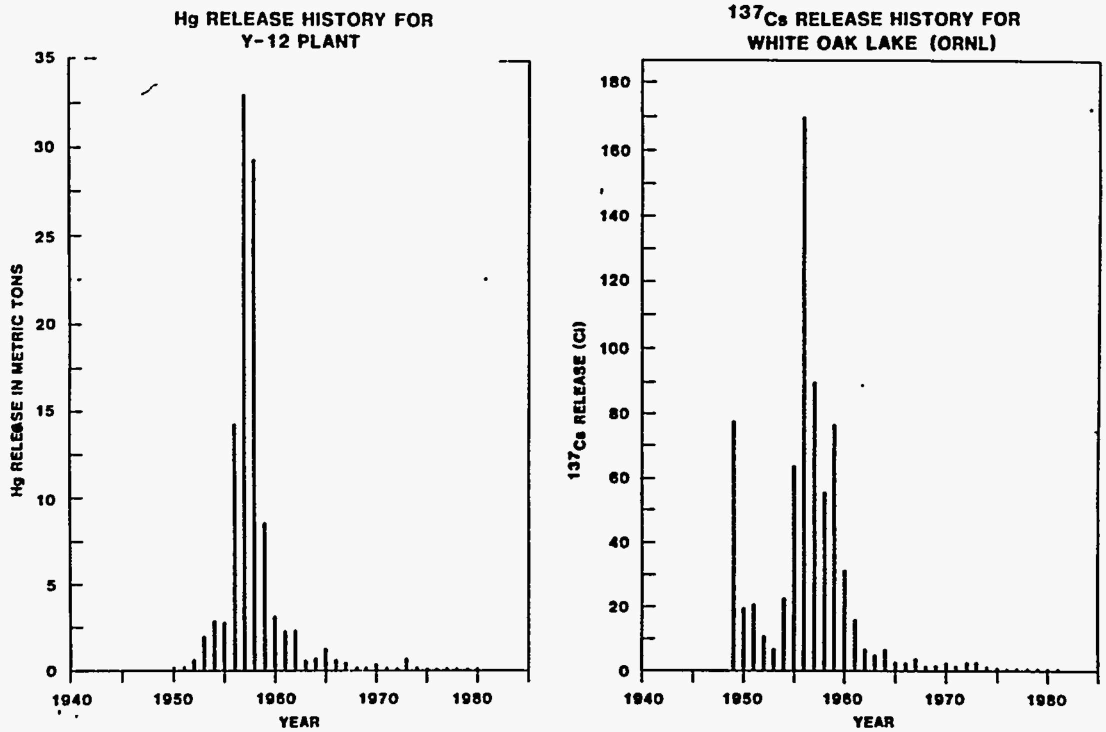


Fig. 3.3. Mercury release history from the Oak Ridge Y-12 Plant and ^{137}Cs release history from White Oak Lake (Oak Ridge National Laboratory).

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

The S-3 ponds were a series of four ponds constructed in 1953 at the west end of the plant. They provided 10 million gal of storage for liquid wastes containing low levels of radionuclides, such as uranium and transuranics. In addition, wastes such as nitric acid, other strong acids, and coolants were disposed of in the ponds. Seepage from the ponds flowed to Bear Creek, a tributary of EFPC. Groundwater in the area beneath the site of the former S-3 ponds is contaminated with nitrates, volatile organic contaminants, radionuclides, and trace metals (DOE 1994a). Discharge to the ponds ceased in 1984 (DOE 1988), and the S-3 ponds underwent RCRA closure in 1988 (DOE 1994a).

The Y-12 steam plant and associated operations also contributed contaminants to the aquatic environment. Until 1993, coal ash from the steam plant was discharged as a slurry into a fly ash pond near the headwaters of McCoy Branch. McCoy Branch drains into Rogers Quarry, which acted as a settling basin for the ash, and then into Melton Hill Reservoir (DOE 1994a).

3.2.2 The K-25 Site

The K-25 Site (Fig. 3.4) houses the Oak Ridge Gaseous Diffusion Plant (ORGDP), construction of which began in 1943. The primary mission of the plant was the enrichment of uranium through the gaseous diffusion process (DOE 1994a). This process produced enriched uranium by separating ^{235}U from ^{238}U by the diffusion of gaseous uranium hexafluoride across a long series of porous barriers. At each barrier in the diffusion cascade, the product became slightly enriched in ^{235}U as a result of its greater diffusion rate across the barrier in comparison with the slightly heavier ^{238}U. Before 1964, a long series of diffusion cascades was used to produce highly enriched uranium (96% ^{235}U) for use in nuclear weapons production. After 1964, several of the diffusion cascades were shut down, and less enriched uranium (10% ^{235}U) was produced for use as fuel in commercial nuclear power generating facilities. The gaseous diffusion process at ORGDP was placed on standby in August 1985 and permanently shut down in December 1987.

The ORGDP housed numerous support facilities. Building K-1420 was used for decontaminating equipment and recovering uranium from this and other waste streams. These operations released U, ^{99}Tc, and ^{237}Np as liquid waste (DOE 1988). The transuranic elements were introduced into the waste stream as a result of operations to recover uranium from spent nuclear reactor fuel shipped from the Savannah River and Hanford plants (Egli et al. 1985 in Bruce et al. 1993).

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

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

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

The K-1407B holding pond was a 1.3 acre, 1.5-million-gal, unlined settling basin that received waste for more than 40 years. The pond received waste organics and metal hydroxides neutralized in the K-1407A pit and functioned primarily to settle the precipitates formed during neutralization. The pond also received effluent directly from several plant buildings. Contaminants known to be discharged to the pond include U, transuranics, solvents and oils, PCBs, and metals such as Cd, Cr, Pb, and Ni. Effluent from the K-1407B pond flowed to Mitchell Branch, a tributary to Poplar Creek

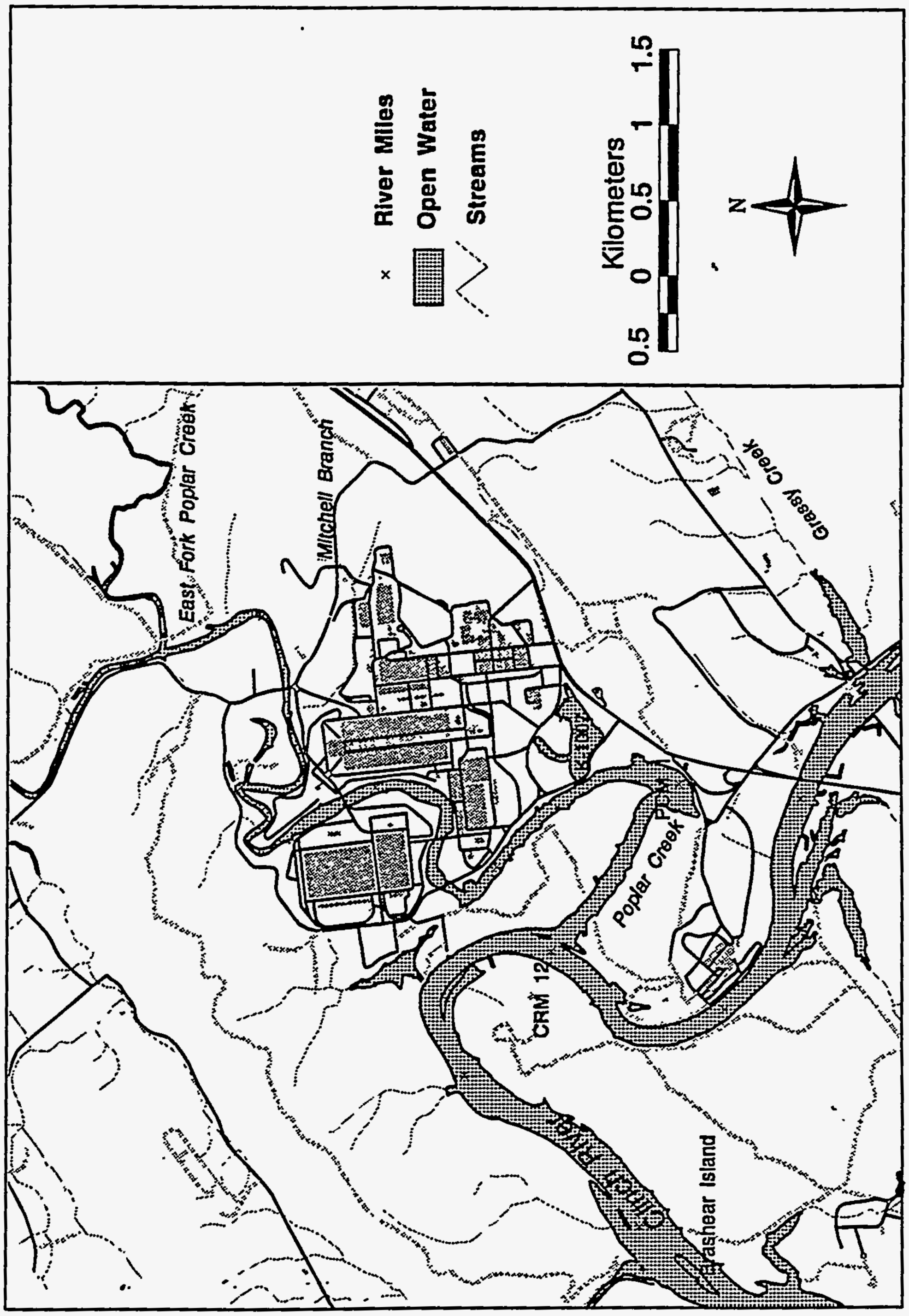


Fig. 3.4. The Oak Ridge K-25 Site.

(Goddard et al. 1991). In 1973, sludge from the pond was removed and disposed of in the K-1407C Retention Basin. A RCRA closure of the K-1407B pond was initiated in 1988. Sludges from the pond were placed in drums to which cement was added to solidify the wastes. Although the soil beneath the ponds was free of RCRA waste, remaining radionuclide contamination is the focus of a CERCLA remedial action.

The K-1407C retention basin was a 2.5-million-gal surface impoundment constructed in 1973 to

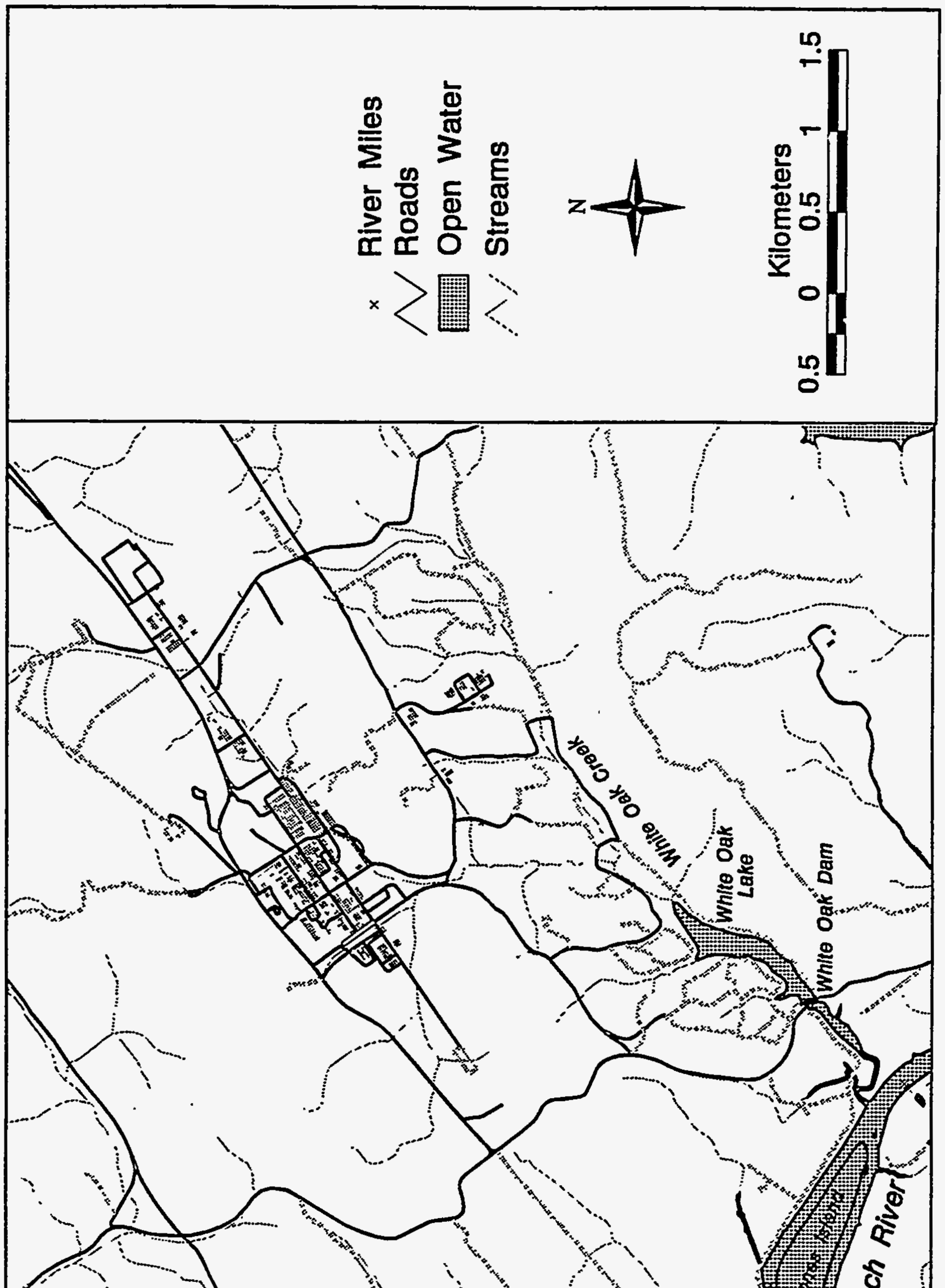


Fig. 3.5. Oak Ridge National Laboratory.

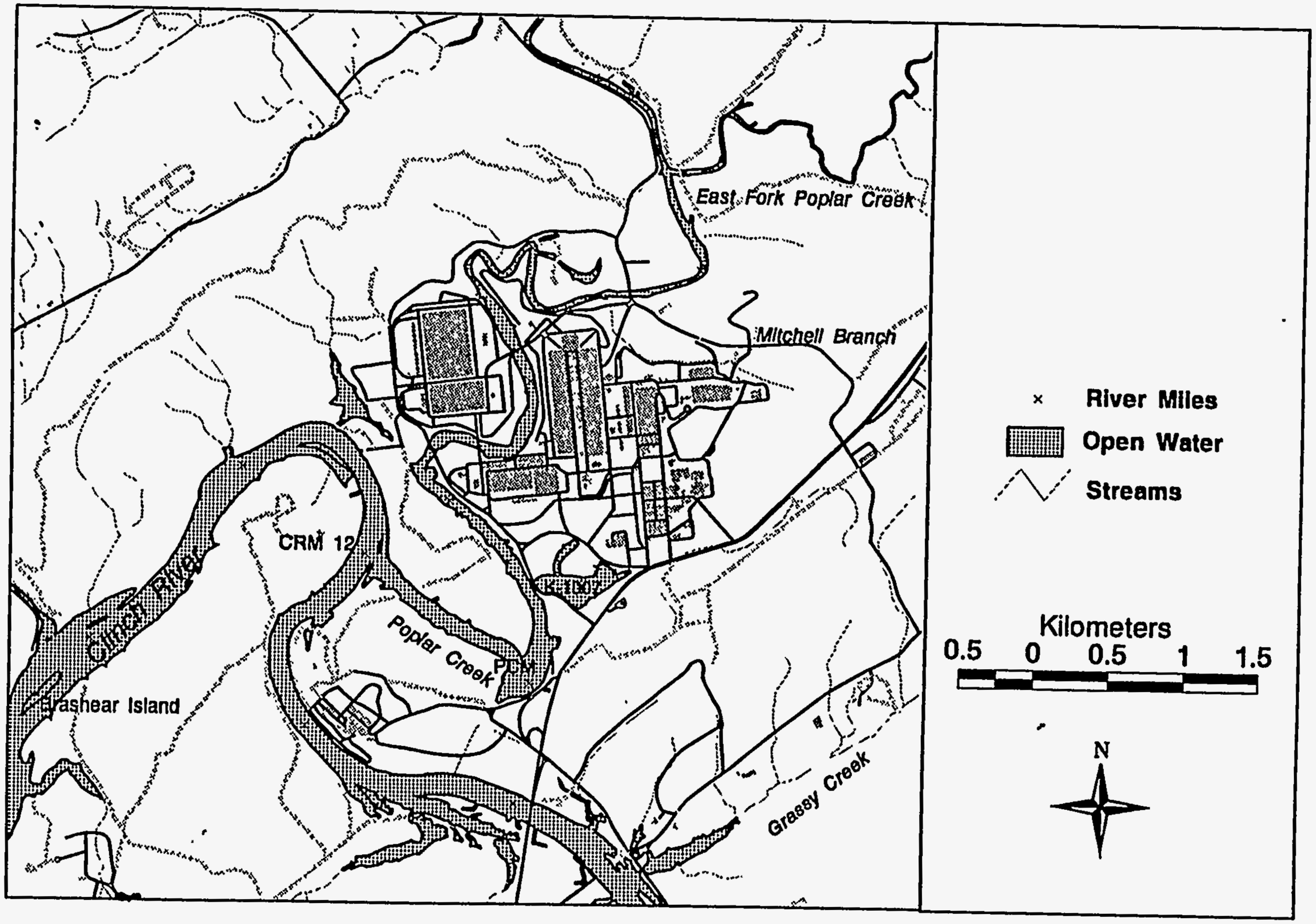


Fig. 3.4. The Oak Ridge K-25 Site.

(Goddard et al. 1991). In 1973, sludge from the pond was removed and disposed of in the K-1407C Retention Basin. A RCRA closure of the K-1407B pond was initiated in 1988. Sludges from the pond were placed in drums to which cement was added to solidify the wastes. Although the soil beneath the ponds was free of RCRA waste, remaining radionuclide contamination is the focus of a CERCLA remedial action.

The K-1407C retention basin was a 2.5-million-gal surface impoundment constructed in 1973 to store the sludge from the K-1407B pond. It was subsequently used to store potassium hydroxide scrubber sludge generated at the plant (DOE 1994a). Like the K-1407B pond, a RCRA closure was initiated in 1988, but the underlying soil was found to still be contaminated with metals and radioncuclides; this situation is being addressed under CERCLA. The sludges from the K-1407C pond were also collected and fixed in drums as above. The resulting 46,000 drums of sludge from these pond closures were stored outside at the K-1417 Drum Storage Area, where leaking drums were detected in 1989. Runoff from this area went directly to Mitchell Branch. The drums have been moved to indoor waste storage vaults at the K-25 Site (DOE 1994a).

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

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

The K-901A holding pond received wastes from the late 1950s until 1985 (Goddard et al. 1991). The pond was initially a marshy area but was dammed in 1965–66 to create the holding pond proper. In addition to the cooling system wastes described above, the pond was used to dispose of compressed gas cylinders containing unknown quantities of uranium hexafluoride, hydrogen fluoride, halides, and various fluoridated and chlorinated hydrocarbons (Goddard et al. 1991). The cylinders were reportedly breached before placement in the pond (Bruce et al. 1993). The K-901A holding pond discharges directly to the Clinch River.

The gaseous diffusion process required large amounts of electricity. Four on-site switchyards received power transmitted from off-site. PCB-containing transformers at the switchyard reduced and transformed this electricity for plant use. PCBs released from the switchyards (Goddard et al. 1991) have likely migrated to Poplar Creek and the Clinch River.

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

In addition to the storage ponds and pits described above, a number of land-based waste disposal operations were used to manage wastes at K-25; these operations could have contributed to contamination of on-site and off-site surface waters through surface runoff or groundwater flow. Burial grounds included the K-1070A, K-1070B, and K-1070C/D burial grounds (DOE 1994a). The K-1070 area operated from the late 1940s through 1975 for the disposal of low-level and mixed wastes, including thorium, lead, and uranium. The K-1070B area was used from the late 1950s to 1976 to dispose of classified equipment contaminated with Pb, U, Al, Cu, Be, bronze, and asbestos (Goddard et al. 1991). This area was created by filling a low marshy area adjacent to a small creek that flowed into Poplar Creek. The K-1070 C/D classified burial ground received wastes from 1972 to 1989. The area received ~9100 gal of waste solvents and 1600 lbs of chemicals (Goddard et al. 1991). From 1979 through 1985, an area known as the K-1070A landfarm was used to dispose of spent Fuller's Earth, a product used to remove impurities in the diffusion cascade oil (Goddard et al. 1991). Several other on-site areas were used for the open burning of waste solvents or the incineration of solid waste. Still other areas were used for waste storage, including the storage of radioactively contaminated scrap metals or of various liquid wastes in drums. The K-720 fly ash pile, used from the 1940s through the 1960s, received fly ash from the on-site coal fired steam plant. This site was located south of the plant near the Clinch River, and runoff from the waste pile was not controlled (Goddard et al. 1991).

3.2.3 Oak Ridge National Laboratory

The original mission of ORNL (Fig. 3.5), or X-10, as it was known then, was the pilot-scale production of plutonium for use in nuclear weapons research at Los Alamos (Johnson and Schaffer 1992). Construction at the X-10 site began in January 1943. Plutonium was produced in the Oak Ridge Graphite Reactor, which was on-line by October 1943. The plutonium was separated from the fission products in a neighboring chemical separation pilot plant. The first shipment of plutonium to Los Alamos occurred in February 1944. By the end of 1944, the use of the graphite reactor shifted from plutonium production to research and the production of other radionuclides. Having completed its original mission, ORNL became a center for the development and testing of nuclear reactors, for the chemical and physical separation of nuclear materials, and for the production of a wide array of radionuclides for worldwide use in research, medicine, and industry (DOE 1994a).

The most significant operations at ORNL that have released contaminants to off-site surface waters have been the management of liquid and solid wastes. The X-10 site was planned as a temporary pilot facility, and therefore waste production was anticipated to be small. A series of concrete gunite tanks was constructed to contain the wastes from the operations (Struxness et al. 1967); however, the mission of X-10 was almost immediately expanded, and the gunite tanks soon became inadequate for containing the volume of waste being generated. As a remedy, the waste in the tanks was treated to precipitate sludges and particle-reactive contaminants; the remaining liquids were released to WOC, along with large quantities of diluting water. In 1943, White Oak Dam was constructed across the creek to create a basin for the additional settling of any remaining solids. In June 1944, the 3513 pond was built to provide an additional settling basin for liquids pumped from the gunite tanks and to allow for the additional decay of short-lived radionuclides before discharge to WOC. From 1949 to 1954, an evaporator was used to concentrate and thereby further reduce the volume of liquid wastes stored in the gunite tanks.

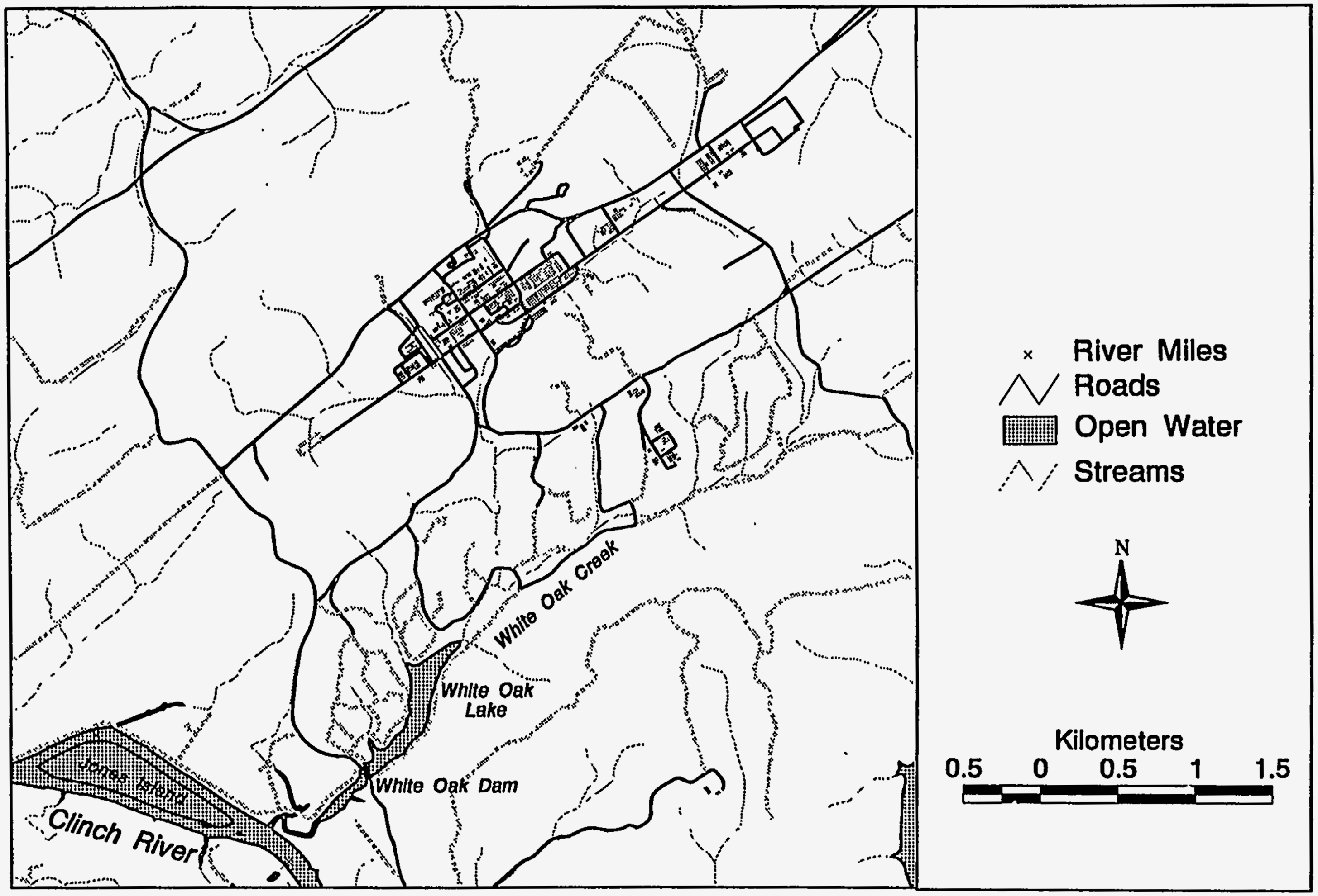


Fig. 3.5. Oak Ridge National Laboratory.

Waste volumes continued to increase, however, and in 1951 the use of earthen pits for the disposal of liquid wastes was initiated. Seven separate disposal pits were in use at various times from 1951 through 1976 (Spalding and Boegly 1985). In 1957, a process wastewater treatment plant was built to recover fission products from these (and other) liquid wastes before disposal. The effluent from this facility was only slightly radioactive compared to the low-level wastes that had previously been discharged to the pits. This waste treatment facility was replaced in 1976 with an upgraded facility (DOE 1994a).

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

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

The disposal of solid wastes at ORNL has occurred at six different solid waste storage areas (SWSAs) (DOE 1994a). The first three storage areas were located in Bethel Valley near ORNL. The sites were selected primarily on the basis of convenience and with little regard for the potential mobility of the wastes in the soil. SWSA 1 is a 1-acre site that was used from 1943 to 1944, SWSA 2 is a 4-acre site that was used from 1944 to 1946, and SWSA 3 is a 6-acre site that was used from 1946 to 1951. SWSAs 4 through 6 are located in Melton Valley, where the soil types are better suited for immobilization of radionuclides. For several years in the late 1950s and early 1960s, ORNL's SWSAs served as a regional burial ground for low-level wastes generated from a number of other federal facilities and private companies (Bates 1983). SWSA 4 is a 23-acre site that was used from 1951 to 1959, and SWSA 5 is a 50-acre site that was used from 1959 to 1973. SWSA 6 is a 68-acre site opened in 1969 and is still in use (DOE 1994a).

The radionuclides that have been released in the greatest quantities from ORNL directly to surface waters since 1944 are ^{3}H (166,300 Ci), ^{106}Ru (6,931 Ci), ^{90}Sr (1,197 Ci), unidentified beta-emitters (2,694 Ci), and the rare earth elements (1,295 Ci, excluding cerium) (DOE 1988). The radionuclides disposed of in greatest quantities since 1944 by being either buried on-site or discharged as liquid waste to pits and trenches include ^{137}Cs (1,174,709 Ci), ^{90}Sr (880,557 Ci), and unidentified beta emitters (1,152,686 Ci). However, of these totals, only 39% of the ^{137}Cs and 17% of the ^{90}Sr were disposed of in pits and trenches; almost 60% of the ^{137}Cs and 78% of the ^{90}Sr were disposed of in the hydrofracture facility. The remaining quantities were discharged as solid waste to the SWSAs (DOE 1988).

As monitored at White Oak Dam, the radionuclides released in the greatest quantities to the Clinch River are ^{3}H, ^{106}Ru, ^{137}Cs, and ^{90}Sr (Table 3.1). Tritium is an isotope of hydrogen with a 13-year half-life. It is readily incorporated into water and moves through the environment accordingly. Given the 5-day retention time of the Clinch River arm of WBR, all but the most recent releases of tritium have long since flushed through the reservoir. Peak ^{106}Ru discharges of 1400–2000 Ci/year occurred from 1960 to 1962, corresponding to the period of greatest seepage from Waste Pit 4. Releases have been estimated at zero since the late 1980s (DOE 1988). Ruthenium-106 is a water-soluble radionuclide with a half-life of 368 days and is not be expected to be found currently in the Clinch River. Cesium-137 is "particle-reactive," being readily adsorbed onto the surface of fine-grained sediment particles, particularly clays. Peak releases of ^{137}Cs occurred in 1956 (170 Ci), coinciding with the draining of White Oak Lake, when large quantities of sediment were eroded from the exposed lake bottom. An additional 266 Ci were released through 1961, after which releases continued to decline until 1975, when annual releases were <2 Ci; they have remained at that level since then (Table 3.1, Fig. 3.3). Total ^{137}Cs released to the Clinch River from ORNL is ~700 Ci (~8 g). Because ^{137}Cs has a half-life of 30 years, much of the material released from the ORR can still be found in the sediments of the Clinch River. Peak releases of ^{90}Sr occurred before 1961 but cannot be correlated with disposal data, which is nonexistent for those years (DOE 1988). Releases had declined to <4 Ci/year by the late 1980s. Although it has a half-life similar to ^{137}Cs, ^{90}Sr is more water-soluble and, therefore, has not accumulated in the Clinch River to the same extent (Struxness et al. 1967).

DOE (1988) estimated that 70 to 80% of the radioactive materials released to surface waters from ORNL originate as leakage from waste disposal areas. Radionuclides of concern are primarily ^{137}Cs and ^{90}Sr; ^{3}H and transuranics are of somewhat less significance. Another 10% of the releases originate from operating facilities at ORNL, such as reactors, laboratories, and processing plants. The remaining 10% originate as surface runoff from areas of contaminated soil in the vicinity of operating facilities that have been affected by spills or equipment leaks.

3.2.4 Summary of Significant Releases

Historically, operations at the ORR have resulted in the release of hazardous substances, including metals, organics, and radionuclides, to the off-site aquatic environment. Reliable estimates of the amount of these various contaminants are generally impossible to make because of a lack of quantitative monitoring information, particularly for the early years of operations. The Y-12 Plant is known to have released large quantities of mercury to EFPC. Large quantities of ^{238}U have been released from both the K-25 Site and the Y-12 Plant. Peak contaminant releases of mercury and uranium occurred before 1959 and have declined drastically since. Large quantities of fission products have been released from ORNL, including ^{3}H, ^{106}Ru, ^{137}Cs, and ^{90}Sr. Because of its particle-reactive nature, ^{137}Cs is the principal long-lived radionuclide (30-year half-life) expected to be found in the Clinch River, where it would be associated with sediment. The greater solubility of the other radionuclides results in their rapid dilution and flushing downstream. Other possible contaminants released from the ORR include various metals (such as As, Pb, Cr, Be, and Ni), various organic compounds (such as PCBs, chlorinated and nonchlorinated hydrocarbons, and various laboratory chemicals), and various radionuclides (such as transuranic radionuclides and fission products).

Table 3.1. Estimated[a] discharges (in curies) of radionuclides from White Oak Creek to the Clinch River, 1949–1992

Year	^{137}Cs	^{106}Ru	^{90}Sr	TRE[b]	^{144}Ce	^{95}Zr	^{131}I	^{60}Co[c]	^{3}H	TRU[d]
1949	77	110	150	77	18	180	77			0.04
1950	19	23	38	30		15	19			0.04
1951	20	18	29	11		5	18			0.08
1952	10	15	72	26	23	19	20			0.03
1953	6	26	130	110	7	8	2			0.08
1954	22	11	140	160	24	14	4			0.07
1955	63	31	93	150	85	5	7	7		0.25
1956	170	29	100	140	59	12	4	46		0.28
1957	89	60	83	110	13	23	1	5		0.15
1958	55	42	150	240	30	6	8	9		0.08
1959	76	520	60	94	48	27	1	77		0.68
1960	31	1900	28	48	27	38	5	72		0.19
1961	15	2000	22	24	4	20	4	31		0.07
1962	6	1400	9	11	1	2	0.4	14		0.06
1963	4	430	8	9	2	0.3	0.4	14		0.17
1964	6	190	7	13	0.3	0.2	0.3	15	1900	0.08
1965	2	69	3	6	0.1	0.3	0.2	12	1200	0.50
1966	2	29	3	5	0.1	0.7	0.2	7	3100	0.16
1967	3	17	5	9	0.2	0.5	0.9	3	13300	1.03
1968	1	5	3	4	0.03	0.3	0.3	1	9700	0.04
1969	1	2	3	5	0.02	0.2	0.5	1	12200	0.20
1970	2	1	4	5	0.06	0.02	0.3	1	9500	0.40
1971	1	0.5	3	3	0.05	0.01	0.2	1	8900	0.05
1972	2	0.5	6	5	0.03	0.01	0.3	1	10600	0.07
1973	2	0.7	7		0.02	0.05	0.5	1	15000	0.08
1974	1	0.2	6		0.02	0.02	0.2	0.6	8600	0.02
1975	0.6	0.3	7				0.3	0.5	11000	0.02
1976	0.2	0.2	5				0.03	0.9	7400	0.01
1977	0.2	0.2	3				0.03	0.4	6200	0.03
1978	0.3	0.2	2				0.04	0.4	6300	0.03
1979	0.2	0.1	2.4				0.04	0.4	7700	0.03
1980	0.6	0	1.5				0.04	0.4	4600	0.04
1981	0.2	0.1	1.5				0.04	0.7	2900	0.04
1982	1.5	0.2	2.7				0.06	1.0	5400	0.03
1983	1.2	0.2	2.1				0.004	0.3	5600	0.05
1984	0.6	0.2	2.6				0.05	0.2	6400	0.03
1985	0.4	0.007	3.0					0.6	3700	0.008
1986	1.0	0	1.8					0.54	2600	0.024
1987	0.6	0	1.2					0.12	2500	0.006
1988	0.4	0	1.1					<0.07	1700	
1989	1.2	0	2.9					0.13	4100	
1990	1.1	0	3.1					0.12	3100	
1991	1.7		2.7					0.12	2100	
1992	0.6		2.1					0.04	1900	
1993	0.5		2.1					0.04	1700	
1994	0.5		2.8					0.07	2200	
Total	700.6	6931.6	1214.6	1295	341.93	376.61	175.33	325.53	183100	5.248

Footnotes for Table 3.1
[a]All digits carried through to avoid rounding errors. Only first two are significant.
[b]Total rare earth elements, exclusive of cerium.
[c]Blank cells indicate no data reported.
[d]Transuranic radionuclides.

Sources:
B. G. Blaylock, M. L. Frank, L. A. Hook, F. O. Hoffman, and C. J. Ford. 1993. *White Oak Creek Embayment Site Characterization and Contaminant Screening Analysis.* ORNL/ER-81. Oak Ridge National Laboratory, Oak Ridge, Tenn.
Martin Marietta Energy Systems, Inc. 1992. *Oak Ridge Reservation Environmental Monitoring Report for 1991.* EH/ESH-22. Oak Ridge National Laboratory, Oak Ridge, Tenn.
Martin Marietta Energy Systems, Inc. 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31. Oak Ridge National Laboratory, Oak Ridge, Tenn.
U.S. Department of Energy. 1988. *Historical Releases from Current DOE Oak Ridge Operations Office Facilities.* OR-890. Oak Ridge Operations Office, Oak Ridge, Tenn.

Today, areas historically used for waste disposal and waste management continue to release contaminants from the ORR (primarily as releases to surface water). The principal streams draining the ORR thought to be contributing contaminants to the Clinch River and Poplar Creek include McCoy Branch, WOC, EFPC, and Mitchell Branch. The source streams identified here, as well as the upstream disposal areas, are the streams being addressed by the on-site components of the ORR environmental restoration program. The CRRI focuses on residual contamination in Clinch River and Poplar Creek sediment and seeks to assess the impact of this contamination, together with the cumulative impact of on-site releases, on the environment of these two streams.

3.3 WATER CHARACTERIZATION

This section characterizes the nature and extent of contamination in surface water of the CR/PC OU. First, data from historical studies and monitoring programs are summarized and used to create a site conceptual model (Sect. 3.3.1). Data from the CRRI Phase 1 and Phase 2 studies, as well as contemporaneous data from the ORREMP are then used to characterize the nature and extent of contamination and to ascertain compliance with ambient water quality criteria (AWQC) (Sect. 3.3.2). A number of Phase 2 supporting studies are described next (Sect. 3.3.3), and an overall summary of results is then provided (Sect. 3.3.4).

3.3.1 Historical Studies and Site Conceptual Model

This section summarizes the results of selected studies of surface water contamination in the Clinch River. The information from each of these studies is used to help formulate a site model of surface water contamination.

3.3.1.1 Clinch River Study

The comprehensive Clinch River Study was a multiagency, interdisciplinary, 5-year (1960–64) investigation into the effects of radionuclide releases from ORNL into the Clinch River. Organizations participating in the study were the U.S. Atomic Energy Commission, the U.S. Geological Survey, the U.S. Public Health Service, TVA, the Tennessee Department of Public Health, the Tennessee Stream Pollution Control Board, the Tennessee Game and Fish Commission, and ORNL. The objectives of the study were (1) to determine the fate of radioactive materials that were currently being discharged to the river, (2) to develop an understanding of the mechanisms of dispersion of radionuclides released to the river, (3) to evaluate the limitations of the river for receiving radioactive effluent safely, and (4) to suggest long-term monitoring procedures (Struxness et al. 1967).

The radionuclides of interest in the study (determined a priori on the basis of the quantities released, radioactive half-lives, and recommended maximum permissible concentrations in water) were ^{90}Sr, ^{60}Co, ^{106}Ru, and ^{137}Cs. A mass balance approach was used to inventory the radionuclides entering, leaving, remaining, or decaying in the Clinch and Tennessee rivers (Struxness et al. 1967) from WOC downstream to Chickamauga Dam (TRM 471).

The results of the study indicated that ^{106}Ru, ^{90}Sr, and ^{60}Co occurred principally in solution in WOC. Upon release to the Clinch, the concentrations of these radionuclides were rapidly diluted and almost the entire quantity transported downstream in the dissolved state (Churchill et al. 1965). For example, the calculated mass balance curve for ^{90}Sr is depicted in Fig. 3.6. Maximum total concentrations of ^{90}Sr in water during the study period ranged from 17,450 pCi/L at White Oak Dam to 14.1 pCi/L at Chickamauga Dam. The maximum concentration recorded for the Clinch River was 42.6 pCi/L at the Centers Ferry station (CRM 5.5). Flow-weighted mean concentrations for the study period were an order of magnitude less than maximum values at all stations. Average concentrations over time were not possible given the basic data (Churchill et al. 1965).

In contrast to the other radionuclides studied, ~70% of the ^{137}Cs in WOC waters was sorbed to suspended sediments. Upon release to the Clinch River, these suspended sediments were largely transported downstream to a point (around CRM 15.0) where decreased turbulence allowed significant sedimentation and concomitant accumulation of ^{137}Cs. Figure 3.7 depicts the calculated mass balance curve for ^{137}Cs. Maximum total concentrations of ^{137}Cs in water during the study period ranged from 6409 pCi/L at White Oak Dam to 6 pCi/L at Chickamauga Dam. The maximum value in the Clinch River was 35 pCi/L at the Centers Ferry Station. Flow-weighted means were not calculated because the maximum values listed were all substantially below the maximum permissible concentration standards in effect at that time (Churchill et al. 1965).

3.3.1.2 Annual Oak Ridge Reservation environmental monitoring reports

The ORREMP has collected and published data for a number of years on contaminants in surface water on and around the ORR. Table 3.2 summarizes the mean concentrations of the analytes most frequently detected at several ambient stations in the Clinch River and Poplar Creek over a recent 5-year period (1988–92). Average values during this period are within the range historically reported for these streams and typically do not exceed applicable water quality criteria for the protection of domestic water supplies or aquatic life. More recent ORREMP data have been used directly in the site characterization (Sect. 3.3.2)

3.3.1.3 CRRI Phase 1

The purpose of the Phase 1 water sampling was to verify, using rigorous quality assurance methods, historically reported water contaminant concentrations. A single sample was collected from each of numerous sites (Cook et al. 1992). Inorganic and organic analyte concentrations were within ranges reported in previous monitoring reports (e.g., Energy Systems 1991) and were below AWQC. Concentrations of radiological constituents were comparable with the mean values reported by Energy Systems (1991) and in previous investigations (Olsen et al. 1992).

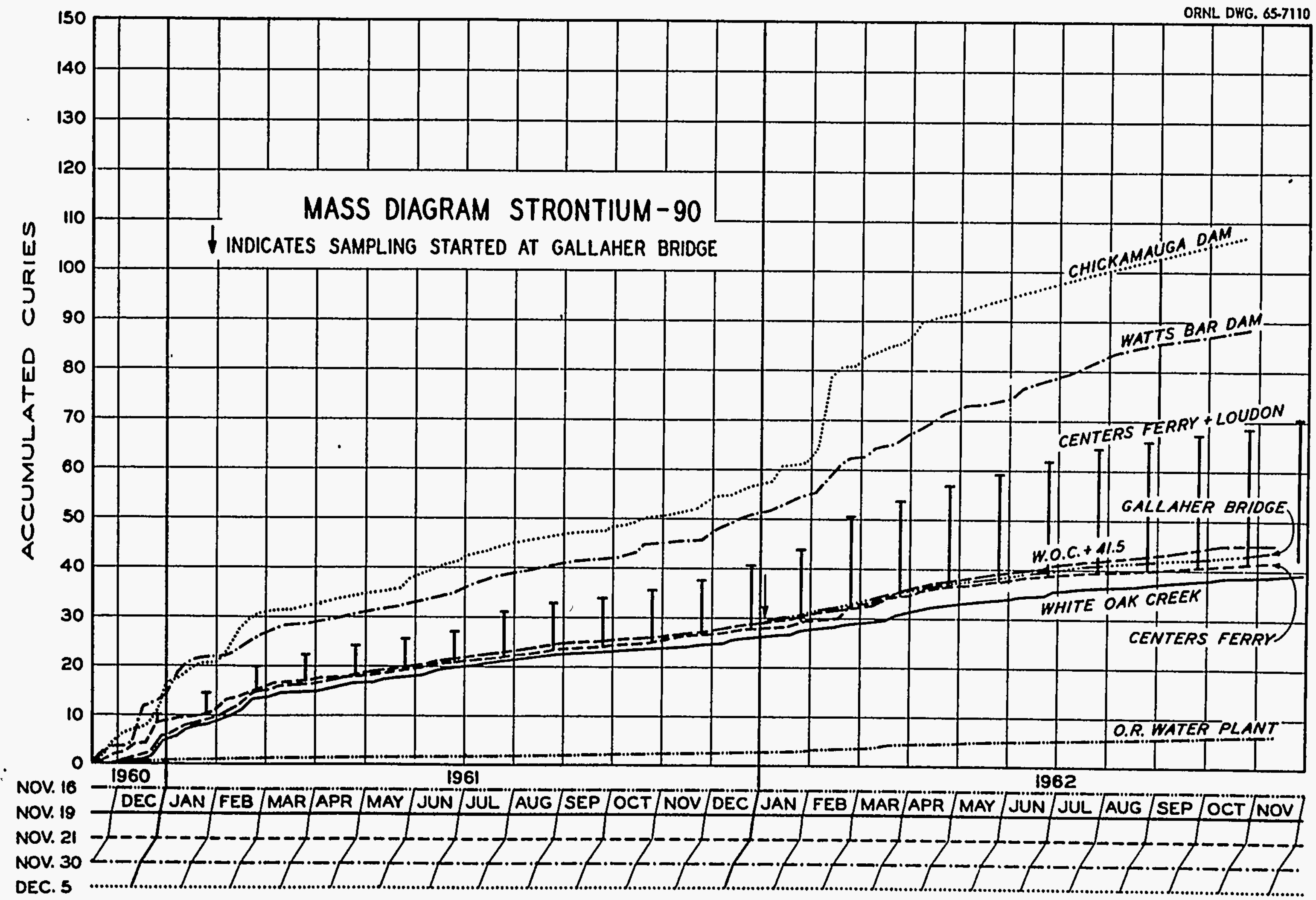


Fig. 3.6. Mass balance diagram for ^{90}Sr in surface water downstream of Oak Ridge National Laboratory. Note that the amount of accumulated curies is consistently greatest at downstream stations, indicating no loss of contaminant mass. *Source*: M. A. Churchill, J. S. Cragwall, Jr., R. W. Andrew, and S. L. Jones. 1965. *Concentrations, Total Stream Loads, and Mass Transport of Radionuclides in the Clinch and Tennessee Rivers.* ORNL-3721, Suppl. 1. Oak Ridge National Laboratory, Oak Ridge, Tenn.

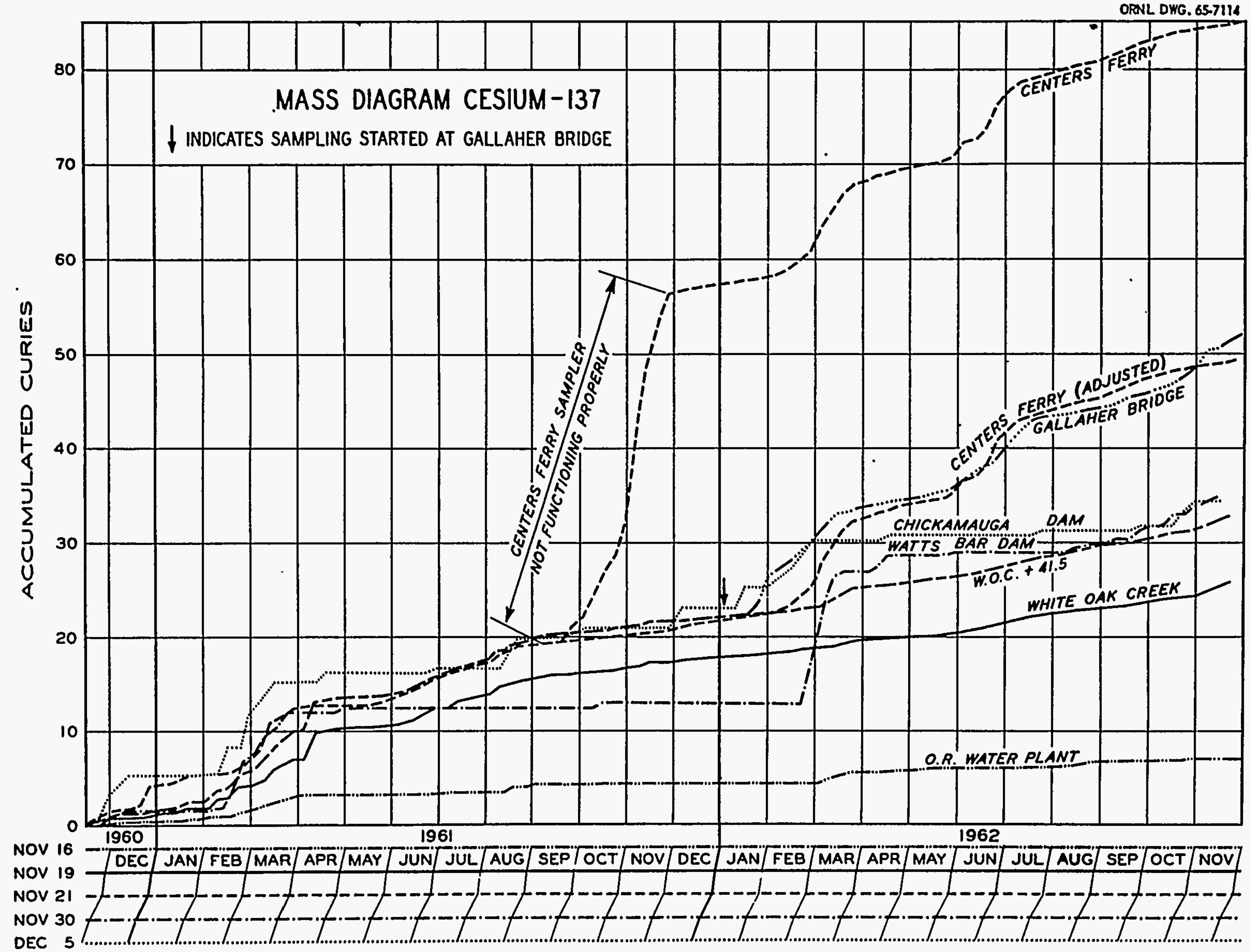


Fig. 3.7. Mass balance diagram for ^{137}Cs in surface water. Note that total accumulated curies is less at Watts Bar and Chickamauga dams than at Gallaher Bridge, indicating a net loss of contaminant mass (to sediments) in Watts Bar Reservoir. *Source*: M. A. Churchill, J. S. Cragwall, Jr., R. W. Andrew, and S. L. Jones. 1965. *Concentrations, Total Stream Loads, and Mass Transport of Radionuclides in the Clinch and Tennessee Rivers.* ORNL-3721, Suppl. 1. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Table 3.2. Mean concentrations (mg/L) of selected analytes in reference and downstream locations in the Clinch River and Poplar Creek, 1988–1992

only six samples had levels of ^3H above detection limits (200 to 230 pCi/L), and activities ranged from 300 to 853 pCi/L. The maximum value detected (at the Kingston water intake, TRM 568.4) was well below the maximum contaminant level (MCL) of 20,000 pCi/L. Tritium was also detected at three beach areas in WBR and two water intakes in the Clinch River (TVA's Kingston Steam Plant and the DOE intake at Grassy Creek).

3.3.1.6 Summary of historical data and discussion of site model

The historical data, considered together with the hydraulic conditions in the Clinch River (Chap. 2), point to the following site conceptual model for contaminants in water. Organic contaminants are infrequently detected and, when detected, are present at very low concentrations. Inorganic analytes are present at low levels and do not appear to exceed AWQC for domestic water supplies or protection of aquatic life at most locations, although detection limits are inadequate in many cases. Radionuclide concentrations are generally very low and do not exceed applicable AWQC. Primary sources of contaminant flux from the ORR to off-site waters appear to be via point-source discharges, tributary flow, or nonpoint-source runoff. No evidence of large-scale flux from baseflow directly to off-site waters has been documented. However, in McCoy Branch, surface water receives soluble arsenic that is released seasonally from contaminated sediment; maximum values exceeded human health screening criteria. In Poplar Creek, despite substantial arsenic contamination in the sediment, the migration of arsenic to surface water appears limited by sediment and hydraulic conditions. Therefore, it is expected that the CRRI would confirm certain spatial contaminant distribution patterns related to known contaminant sources on the ORR, would confirm the low levels of organic compounds throughout the system, would confirm that levels of radionuclides throughout the study area are generally acceptable, and would identify a limited number of metals that could exceed human health or ecological AWQC in certain locations.

3.3.2 Nature and Extent of Contamination in Surface Water

Data from the CRRI Phase 1 and Phase 2 investigations as well as contemporaneous data from the ORREMP (1993 and 1994) were used to describe the nature and extent of contamination in surface water. Data from each of these sources was combined into a single data set, and summary statistics were calculated on the combined data. This data set is also used in Chaps. 5 and 6 to assess human health and ecological risks. Table B1 summarizes the sampling effort that produced this data set. Sample locations are depicted in Fig. 3.8.

The CRRI Phase 1 sample plan is outlined above in Sect. 3.3.1. The ORREMP sample program is described in detail by DOE (1992). Data collected from 14 ORREMP sites during 1993 and 1994 (n = 9 to 10 at each site) are included in the comprehensive site characterization data set. The CRRI Phase 2 sample plan is described by DOE (1994b). Phase 2 samples from the Clinch River represent a compilation of samples from several discrete studies described in Sect. 3.3.3. In Poplar Creek, Phase 2 samples were collected systematically four times in each of three seasons (winter, spring, and summer) from eight sites. Because stratification was never observed in the water column, all samples were collected 1 m from the surface. Samples were analyzed for water quality parameters, metals, organics, and radionuclides (Table B1).

Consistent with the site model, a number of analytes (primarily organic compounds) were undetected at any location. Other analytes were detected in total water samples but not in filtered different purposes and at different times with several sample collection techniques and analytical methods. Therefore, these data may or may not be representative of site conditions, and the resulting summary statistics may be biased. Nonetheless, because the sample size in most subreaches is limited for most of the individual data sets, the combined summary statistics are considered the best available estimate of overall ambient conditions during the study period.

As expected, the positive reference reaches outside the OU (WOC, EFPC, Bear Creek, Mitchell Branch—all ORREMP data) generally exhibited the highest mean values for each of several contaminants. Mean concentrations of total Al, ^{137}Cs, chloride, Cr, Co, ^{60}Co, fluoride, gross alpha activity, gross beta activity, Fe, Hg, Ni, nitrate, Na, ^{90}Sr, sulfate, ^{99}Tc, trichloroethene, and total U were elevated in one or more of the positive reference reaches in comparison with the levels found in the study and negative reference reaches (Tables B3 and B4). In general, however, receiving streams rapidly diluted most of these contaminants such that no statistically significant increases in mean contaminant concentrations in receiving waters were found compared with negative reference reaches.

Exceptions to the general pattern above were total nitrite/nitrate in Poplar Creek downstream of the CNF outfall (Fig. 3.9), total mercury and total methyl mercury in Poplar Creek downstream of EFPC (Fig. 3.10), measures of total gross alpha and beta emission in winter samples collected downstream of Mitchell Branch (Fig. 3.11), and total and dissolved arsenic concentrations in upper McCoy Branch (Fig. 3.12). In addition, mean gross alpha activity, gross beta activity, and levels of ^{3}H and ^{90}Sr were elevated by an order of magnitude in subreach 2.02 (the Clinch River below WOC) over levels in lower Melton Hill Reservoir, but the variability of the data in the former prevents most of these differences from being statistically significant ($\alpha = 0.05$). Increased radionuclide levels below WOC are certainly consistent with the site model, and the variability in contaminant levels is probably a function of variable water flow below Melton Hill Dam.

In Poplar Creek, concentrations of several inorganic analytes varied seasonally, often being highest in summer, when baseflow typically contributes a greater proportion of total flow. It was expected that mercury flux in Poplar Creek would be primarily associated with periods of increased suspended sediment load, but the correlation of total mercury concentrations and total suspended particle values was weak ($R^2 < 0.2$). Moreover, contrary to expectations, the highest mean mercury concentrations were observed near the mouth of Poplar Creek rather than immediately below EFPC. One explanation may be the resuspension of contaminated sediment from the creek bed, although no data were collected to test this hypothesis.

For study and negative reference subreaches only, mean contaminant concentrations were evaluated for compliance with applicable ambient water quality criteria (AWQC) (TDEC 1995.) The State of Tennessee has not designated Poplar Creek or McCoy Branch for use as domestic water supplies; therefore, AWQC applicable to these waters are based on the designated uses of recreation (consumption of biota only) and the protection of aquatic life [chronic continuous criterion (CCC) was used]. The remaining study and negative reference reaches have criteria based on designated uses as domestic water supply (equivalent to MCLs under the Safe Drinking Water Act), recreation (consumption of water and biota), and protection of aquatic life (CCC). The mean concentration, 95% lower confidence limit concentration (LCL_{95}), and the 95% upper confidence limit concentration (UCL_{95}) were each compared with applicable criteria. This analysis is limited in that the standard error and bounds on the mean could only be calculated for reaches in which an analyte was detected more than once.

Results of this evaluation are summarized in Tables B5–B7. Those analytes which appear to exceed a criterion at one or more locations are summarized in Table 3.3. Analytes with LCL_{95} values greater than a criterion can be said with a high degree of confidence to exceed that criterion.The LCL_{95} for total arsenic exceeds the recreation-based criterion (fish consumption only) of 0.0014 mg/L in subreach 7.01 (upper McCoy Branch). The LCL_{95} for total mercury exceeds the criterion continuous concentration (CCC) of 0.000012 mg/L in reaches 0, 3.01, 3.02, 3.03, 3.04, 4.01, 4.02, and 4.04.

Analytes with an LCL_{95} below a specific criterion but with mean concentrations which exceed that criterion are possibly in compliance but are more likely to exceed the criterion. The mean concentration of total mercury exceeds the recreation criterion (fish consumption only) of 0.00015 mg/L in subreach 3.04. The mean arsenic concentration exceeds the recreation criterion (fish consumption only) in reach 13.

Despite a mean value that is less than a specific criterion, analytes with a UCL_{95} greater than that criterion cannot be said with a high degree of confidence to be in compliance. The only such instance is in subreach 3.03, where the UCL_{95} concentration of total arsenic slightly exceeds the recreation criterion (fish consumption only).

Table B5 also indicates that LCL_{95} values of arsenic exceed the recreation-based criterion (consumption of water and fish) in reach 1 and subreaches 4.01 and 4.04. However, this criterion is below the minimum detection limit used in this study and LCL_{95}, and therefore is constrained by the detection limit. The low frequency of detection in each reach above suggests that this criterion is probably not being exceeded. Table B5 also indicates that mean concentrations of arsenic exceed the recreation criteria (consumption of water and biota) in reach 0 and subreaches 2.02, 2.04, and 4.02. In each case, however, arsenic was detected in only one sample. Similarly, the mean concentration of thallium exceeds the drinking water and recreation criteria (consumption of water and biota) in reach 1 and exceeds the recreation criterion (fish consumption only) in subreach 3.04. However, the thallium values are also detected in only one sample in each reach. For these reasons, the analytes discussed in this paragraph can only be said to possibly exceed these criteria (Table 3.3).

Man-made radionuclides were conservatively evaluated against the domestic water supply criterion by summing the dose from individual radionuclides through the use of UCL_{95} values where available, maximum values where UCL_{95} values were lacking, or maximum detection limit values if a radionuclide was undetected (Table B3). Values were summed for ^{137}Cs, ^{3}H, ^{90}Sr, ^{60}Co, and total U. DOE (1990) derived concentration guide values were used to calculate potential dose bywater ingestion. The summed dose did not exceed the domestic supply criterion of 4 mrem/year in any study reach.

The remaining analytes were undetected, were detected only once in a subreach with the product limit estimator mean value less than all applicable criteria, or had UCL_{95} values that were below all applicable criteria across all subreaches. In the latter case, it can be said with a high degree of certainty that mean concentrations are below applicable criteria. In the former cases, inadequate detection limits for some analytes in some locations do not allow a conclusive determination, particularly in the case of lipophilic organic compounds (e.g., PCBs) with recreation-based criteria below current analytical capabilities.

3.3.3 Supporting CRRI Phase 2 Studies—Water Characterization

The CRRI Phase 2 surface water investigation involved 6 tasks (DOE 1994b), four of which are described below. An originally planned task, the characterization of contaminant distribution downstream of point sources (DOE 1993b), was discontinued after a few initial data were collected as a result of reassessment of the sample plan (DOE 1994). These data are not discussed in this section but are summarized in Tables L1–L3. In addition, the characterization of ambient conditions in Poplar Creek was

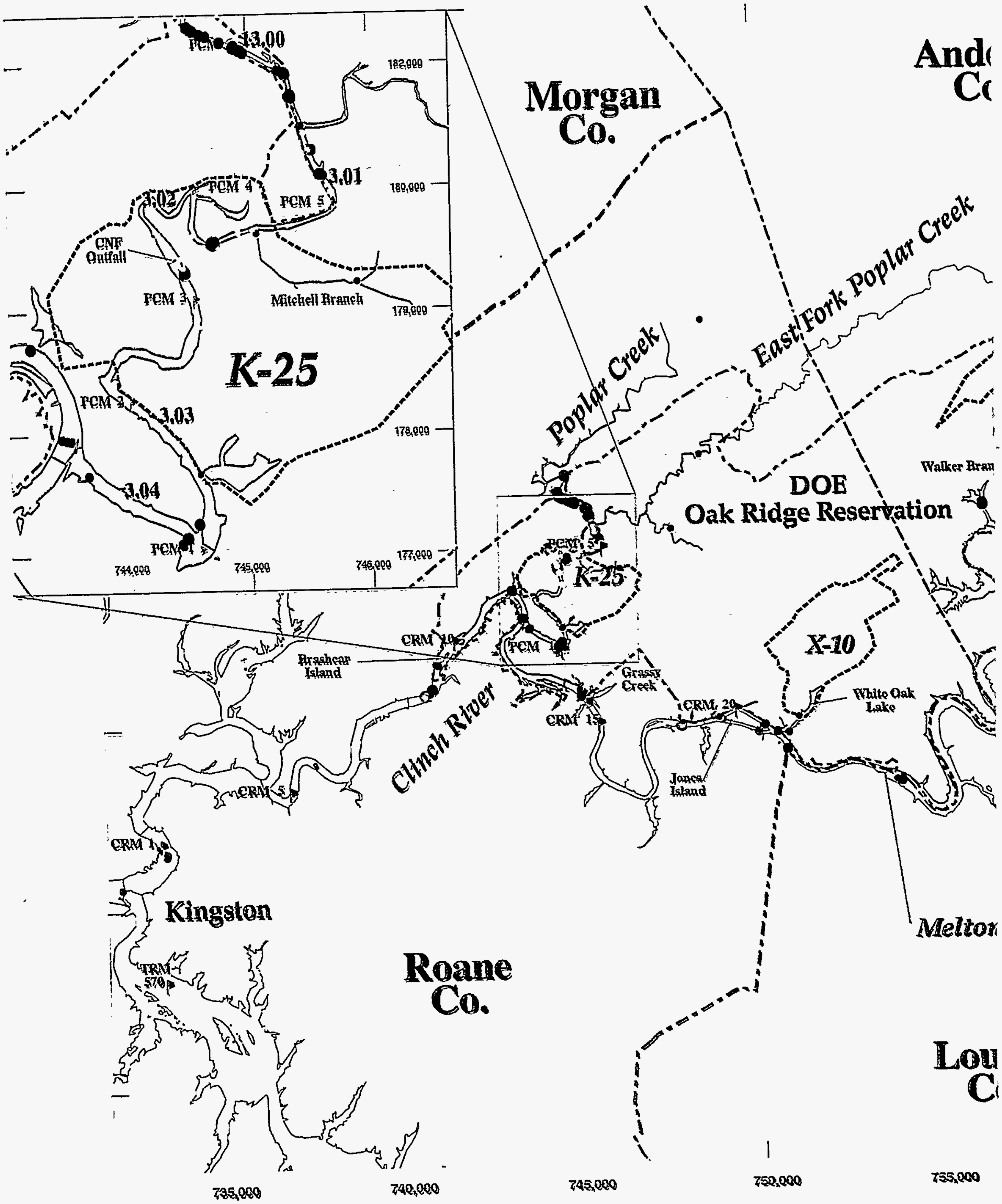
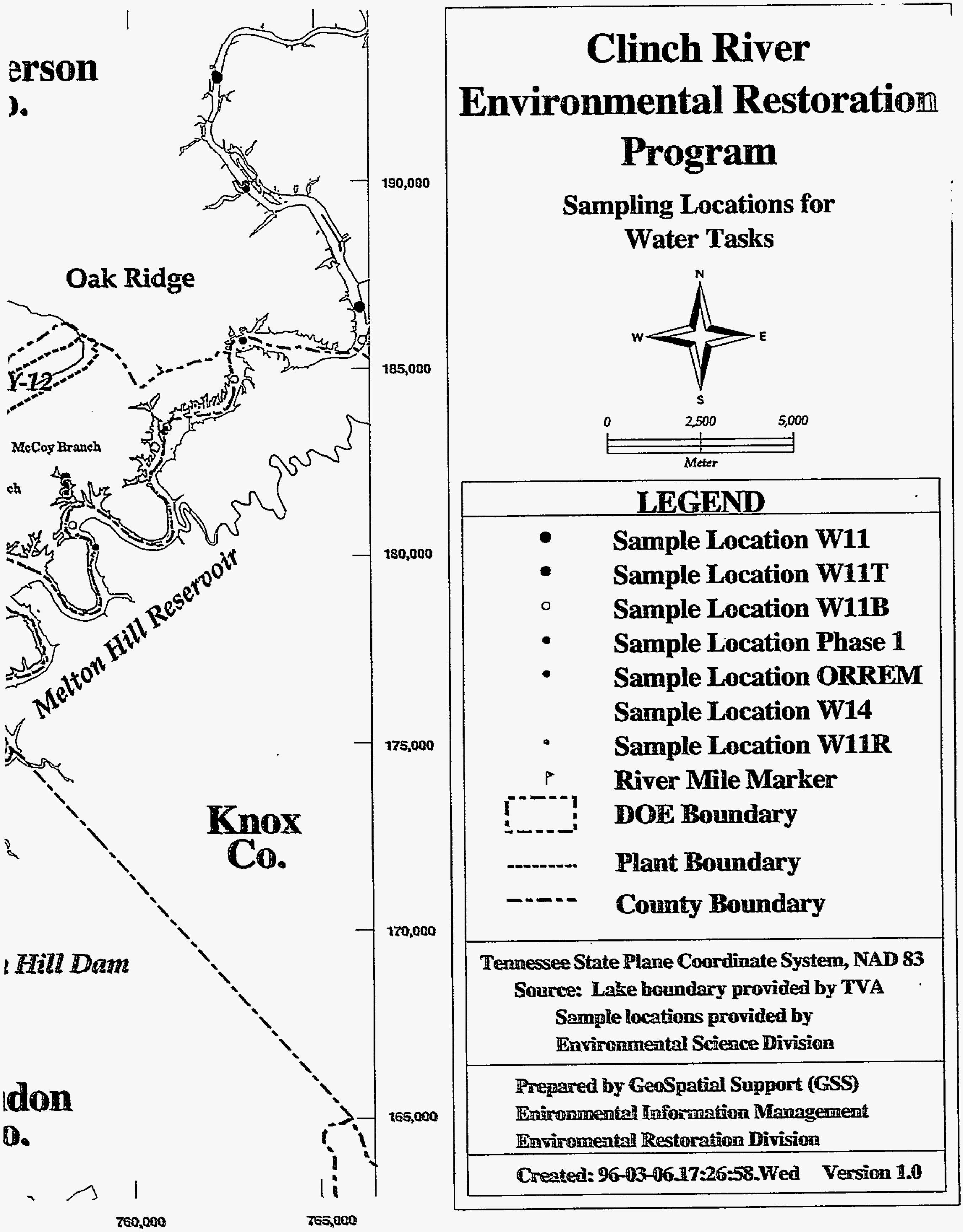


Fig. 3.8. Surface water sampling locations.

Data from individual studies were combined into a comprehensive database for overall site characterization purposes. Individual sample locations are identified for studies that characterize ambient site conditions (W11), biased hydrologic event conditions (W11B), surface water toxicity (W11T), and the remobilization of contaminants from sediment (W11R). Other data are from the Clinch River Remedial Investigation Phase 1 and Oak Ridge Reservation Environmental Monitoring Program.



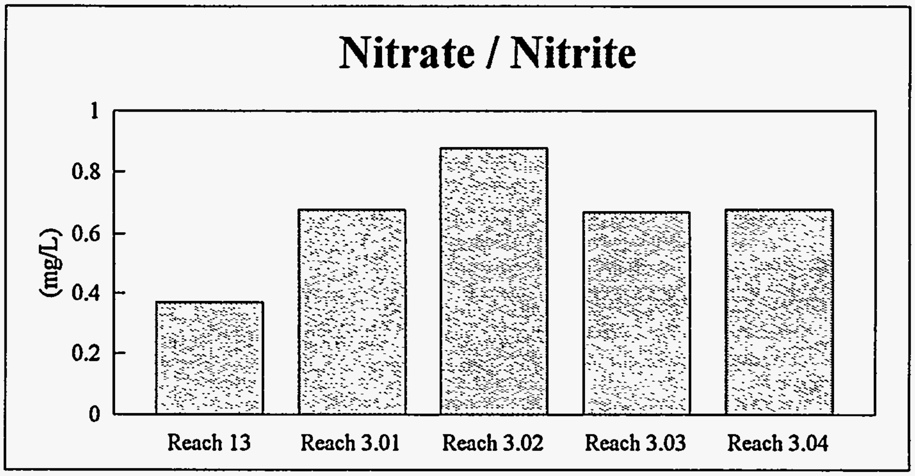


Fig. 3.9. Mean dissolved nitrate/nitrite concentrations in Poplar Creek by reach.

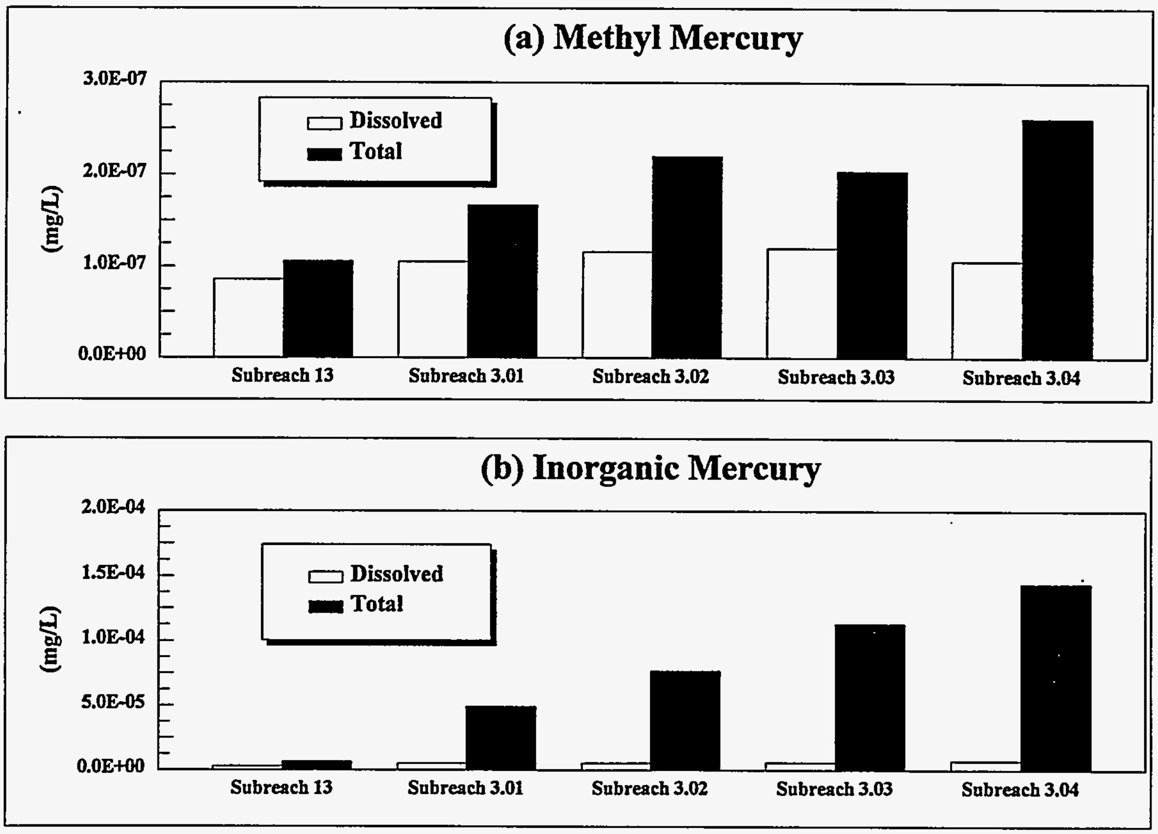


Fig. 3.10. Mean mercury concentrations by reach in Poplar Creek (a) methyl mercury and (b) inorganic mercury.

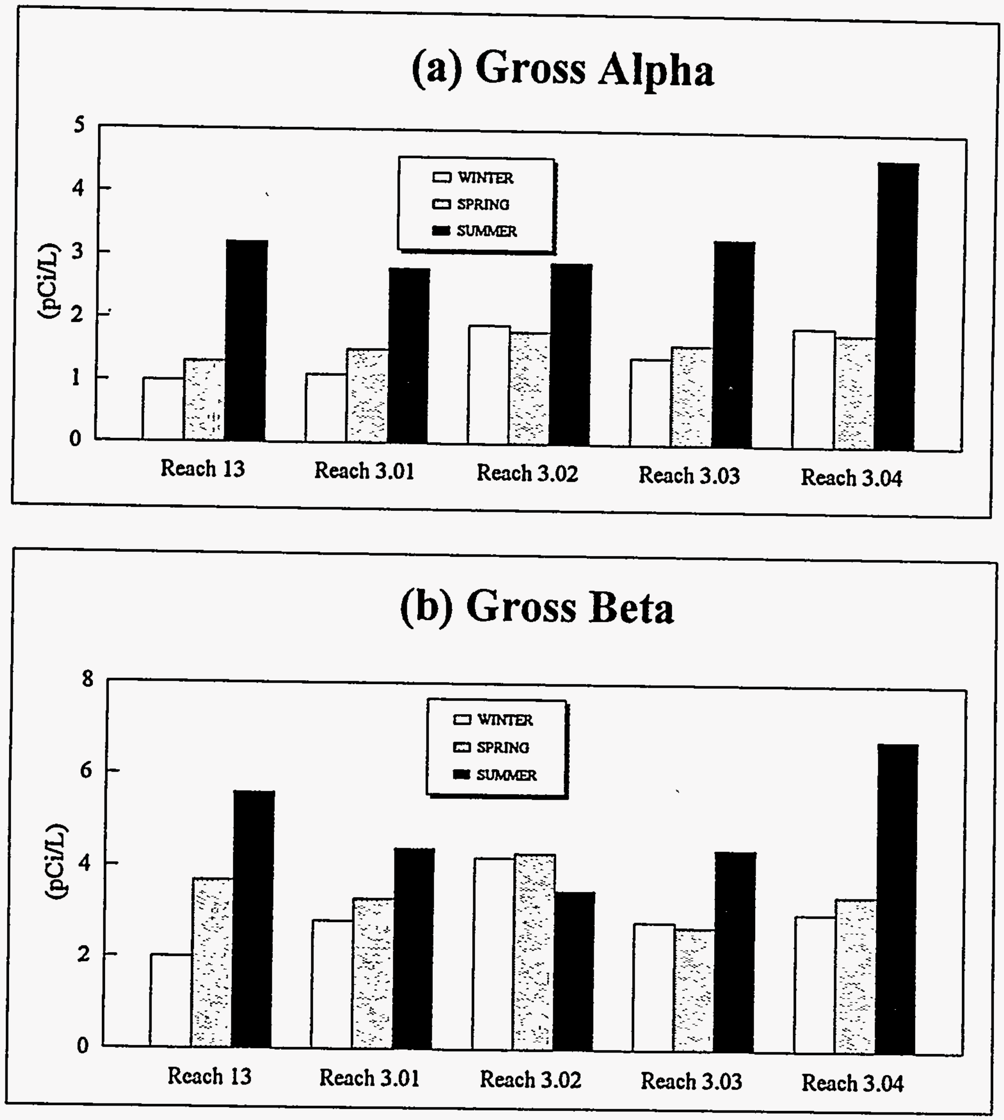


Fig. 3.11. Mean radioactivity concentrations by season within each reach for Poplar Creek (a) gross alpha and (b) gross beta (Phase 2 data only).

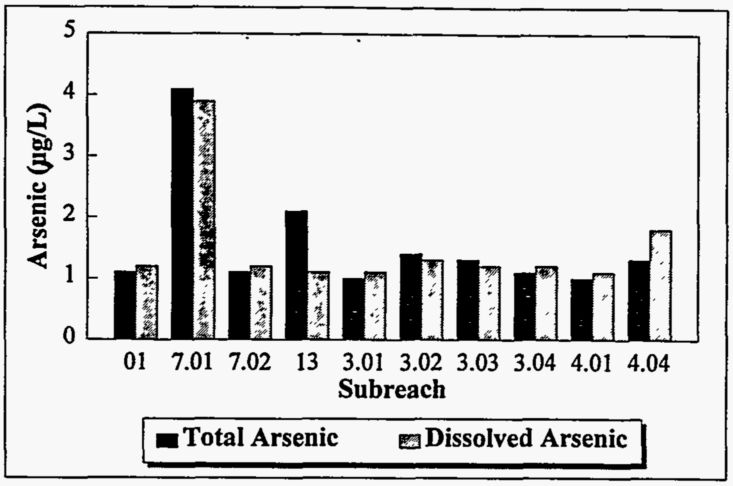


Fig. 3.12. Mean total and dissolved arsenic concentrations by subreach.

Table 3.3. Analytes which potentially exceed ambient water quality criteria in the Clinch River/Poplar Creek Operable Unit

Analyte	Criterion	Reach/subreach		
		Likely exceed[a]	Probably exceed[b]	Possibly exceed[c]
Arsenic	Recreation	1, 4.01, 4.04, 7.01	0, 2.02, 2.04, 4.02, 13	3.03
Mercury	Aquatic life (chronic continuous criterion)	0, 3.01–3.04, 4.01–4.04[d]		
	Recreation		3.04	

[a]The 95% lower confidence limit is greater than criterion.
[b]Mean is greater than criterion, but 95% lower confidence limit is less than criterion.
[c]Mean is less than criterion, but 95% upper confidence limit is greater than criterion.
[d]No data for subreach 4.03.

described by DOE (1994b) as a discrete task, but these data have been combined with CRRI Phase 1 and ORREMP data and discussed above; no further discussion occurs here.

3.3.3.1 Characterization of contaminant concentrations during extreme flow events

This task was designed to characterize the maximum contaminant concentrations in the Clinch River resulting from uncontrolled releases from the ORR. Three hydrologic conditions were considered to bound this worst-case scenario: (1) high baseflow/high runoff (spring rain), maximizing contaminant flux from both groundwater and runoff but allowing high dilution; (2) low baseflow/high runoff (summer rain), minimizing contaminant flux from groundwater while maximizing contaminant flux from surface runoff; and (3) low baseflow/low runoff (summer dry), minimizing contaminant flux from both sources but also minimizing dilution (DOE 1994b). Initially a fourth condition, high baseflow/low surface runoff was to be sampled, but this hydrologic condition was not observed during the study period. The latter flow condition would have focused on contaminants transported primarily by groundwater. The failure to collect these data is not critical in that (1) the site model does not predict the large-scale flux of contaminants via groundwater flow in the saturated zone and (2) groundwater contamination is being addressed separately within the ORR Environmental Restoration Program (DOE 1994a).

Because the maximum Clinch River concentrations would be highest at the mouth of streams draining known contaminant sources on the ORR, sample collection focused on three such streams: McCoy Branch, WOC, and Poplar Creek (Sect. 3.2). The 1994 "spring rain" event was sampled on April 19 and 20, the "summer rain" event July 28 and 29, and the "summer dry" event August 29 and 30 (Figs. L1 and L2). Ideally, flow in the Clinch River would be minimal during sampling and dilution would thus be minimal; however, because of extensive rainfall throughout the region in the early spring of 1994 (Sect. 2.5.3), low flow was not observed during the high baseflow/high runoff condition.

Samples were collected from five locations (Fig. 3.8) at each of the three streams as follows: (1) mid-channel in the mouth of the source stream; (2) along a three-site transect in the Clinch River (at 25%, 50%, and 75% of the channel), 0.2 to 0.6 miles downstream of the source stream; and (3) one mid-channel site in the Clinch River located far enough downstream to ensure uniform mixing of contaminants. Four reference sites were also sampled. Each of the 19 sites was sampled once per flow event. Information regarding analytical classes, sampling locations, and frequency of collection for these data is listed in Table L4. Water quality parameters for all sites reflected seasonally expected norms (Sect. 2.5.3).

Results for each source stream are presented in Tables L5–L8. With the exception of ^{3}H at the WOC source location and arsenic in the McCoy Branch source location, no analyte concentrations exceeded AWQC during any event. Other than the exceptions noted above, analyte concentrations at reference locations were similar to the source and near-source locations. Therefore, no large-scale flux of contaminants leading to increased contaminant concentrations was evident during any of the presumed worst-case hydrologic events.

Interestingly, arsenic concentrations at CRM 1.0 appeared higher than at any study site except upper McCoy Branch. These data reinforce the evidence of Ford et al. (1995) that a significant arsenic source exists downstream of the ORR. Another interesting finding was that, during the "summer rain" event, dissolved and total mercury concentrations, although low at all locations, appeared higher in the Clinch River (CRM 11.8) than in the mouth of Poplar Creek. This finding is contrary to the decrease in concentrations expected from dilution of Poplar Creek water, but the data were too scant to draw any conclusions.

3.3.3.2 Contaminant remobilization from sediment

This task evaluated the potential for sediments to serve as a contaminant source to Clinch River surface waters through mechanical or chemical means (DOE 1994b). Samples from 1 m above the sediment surface were collected at four locations (Fig. 3.8) three times per season for two seasons and analyzed for metals, radionuclides, and organic contaminants (Table L4). Results are presented in Tables L9–L15. These data were compared to near-surface data for the same locations for evidence of contaminant remobilization at depth. Although the data are limited and the near-surface to near-sediment comparisons are severely hampered by the fact that the data were not collected concurrently, no evidence of migration of contaminants from sediment to the water column was found at any location.

3.3.3.3 Characterization of additional sources of contaminants to the CR/PC OU

In August and September 1994, TVA (1995) collected sediment and surface water samples from seven locations in the Clinch River and its tributaries near the ORR (Table L28). Six sites were sampled because of the presence of suspected or known contaminant sources; one site (Walker Branch) was a reference. Only one sample was collected for each media at each location; therefore, no statistical

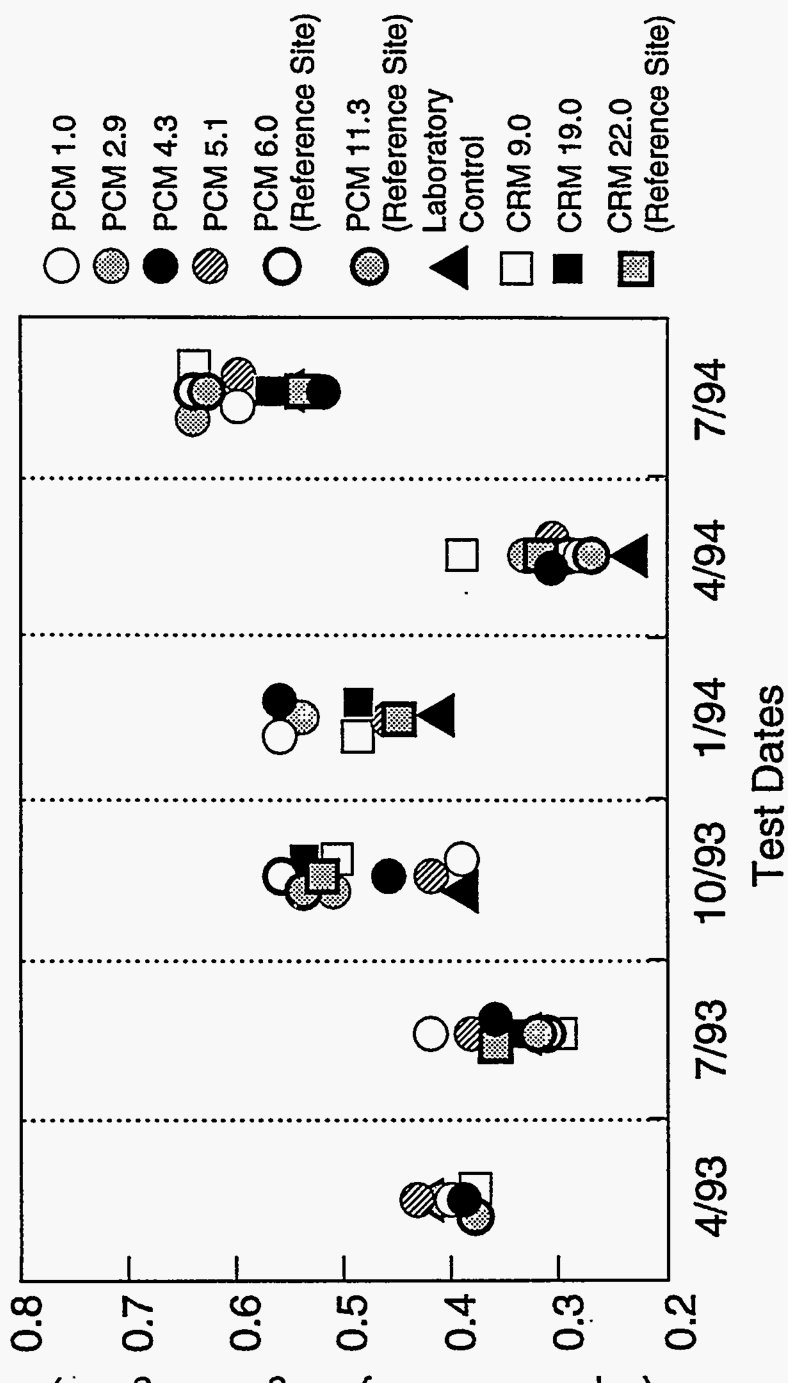


Fig. 3.14. Fathead minnow growth in water samples from Poplar Creek and Clinch River test sites by test date.

period. The latter flow condition would have focused on contaminants transported primarily by groundwater. The failure to collect these data is not critical in that (1) the site model does not predict the large-scale flux of contaminants via groundwater flow in the saturated zone and (2) groundwater contamination is being addressed separately within the ORR Environmental Restoration Program (DOE 1994a).

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Most source area sites did not appear to differ from the reference site with respect to contaminants in water or sediment. However, dissolved chromium in surface water at CRM 50.1 (Oak Ridge Marina) and CRM 47.9 (Bull Run Steam Plant) and total copper at CRM 47.9 were all more than four times the reference concentrations. In sediment, arsenic was three times higher at CRM 37.5 (McCoy Branch) than at the reference site. PCBs (Aroclors 1260 and 1254) were detected (up to 100 μg/kg) only at CRM 47.9 and at CRM 41 (Aroclor 1260 only). Chlordane was detected (up to 26 μg/kg) in sediment only at the upper Clinch River sites (Oak Ridge Marina, Bull Run, and Scarboro Branch).

3.3.3.4 Aqueous toxicity tests

Ambient water from the CR/PC OU was evaluated for toxicity to aquatic organisms, and the results were used in an ecological risk assessment (Chap. 6). Toxicity was evaluated as measures of (1) survival and growth of fathead minnow (*Pimephales promelas*) larvae; (2) fecundity and survival of the cladoceran *Ceriodaphnia dubia*; (3) hatching success, frequency of abnormal development, and survival of Japanese medaka (*Oryzias latipes*) embryos and larvae; and (4) genotoxicity as measured by microbial assay. Study sites (Fig. 3.9) in Melton Hill Reservoir were sampled twice for each type of test above; sites in the Clinch River below Melton Hill Dam and in Poplar Creek were sampled six times each for the cladoceran and fathead minnow tests and three times each for the medaka and genotoxicity tests. To facilitate interpretation of test results, the collection of aqueous samples for toxicity testing was coordinated with samples collected for contaminant analysis.

In the fathead minnow and *Ceriodaphnia* tests, data from the Clinch River were analyzed separately from data from Poplar Creek. Standard test protocols for fathead minnow and *Ceriodaphnia* tests are described by Kszos et al. (1989). Medaka test procedures can be found in the standard operating procedure "Japanese Medaka Embryo-Larval Toxicity Tests" (Energy Systems 1993a).

Fathead minnow test results. An analysis of variance revealed no significant differences in fathead minnow survival or growth among either the Poplar Creek or the Clinch River ambient test sites during each of the six tests conducted, with one exception. In the April 1993 test, minnow survival in subreach 3.01 (PCM 5.1) was significantly lower than survival in subreach 3.04 (PCM 1.0) (Fig. 3.13). Fathead minnow survival data are summarized in Tables L16–L17. Large among-replicate variation in survival was encountered in nearly all of the tests of ambient water. Generally, minnow survival was high (90–100%) during the first few days of the test and then declined thereafter.

Fathead minnow growth in the ambient water was generally greater than minnow growth in control water (Tables L18–L19). However, growth in all test sites and the control was lower in the April 1994 test (Fig. 3.14), apparently a result of the younger, more sensitive organisms that were used in this test.

In addition to the statistical analyses above, reductions in survival or growth of 20% or greater between study sites and appropriate reference sites were noted. Several Poplar Creek study sites exhibited reduced survival or growth in one or more tests as compared to at least one of the reference locations (Fig. 3.15); however, the two Poplar Creek reference sites also differed from each other by

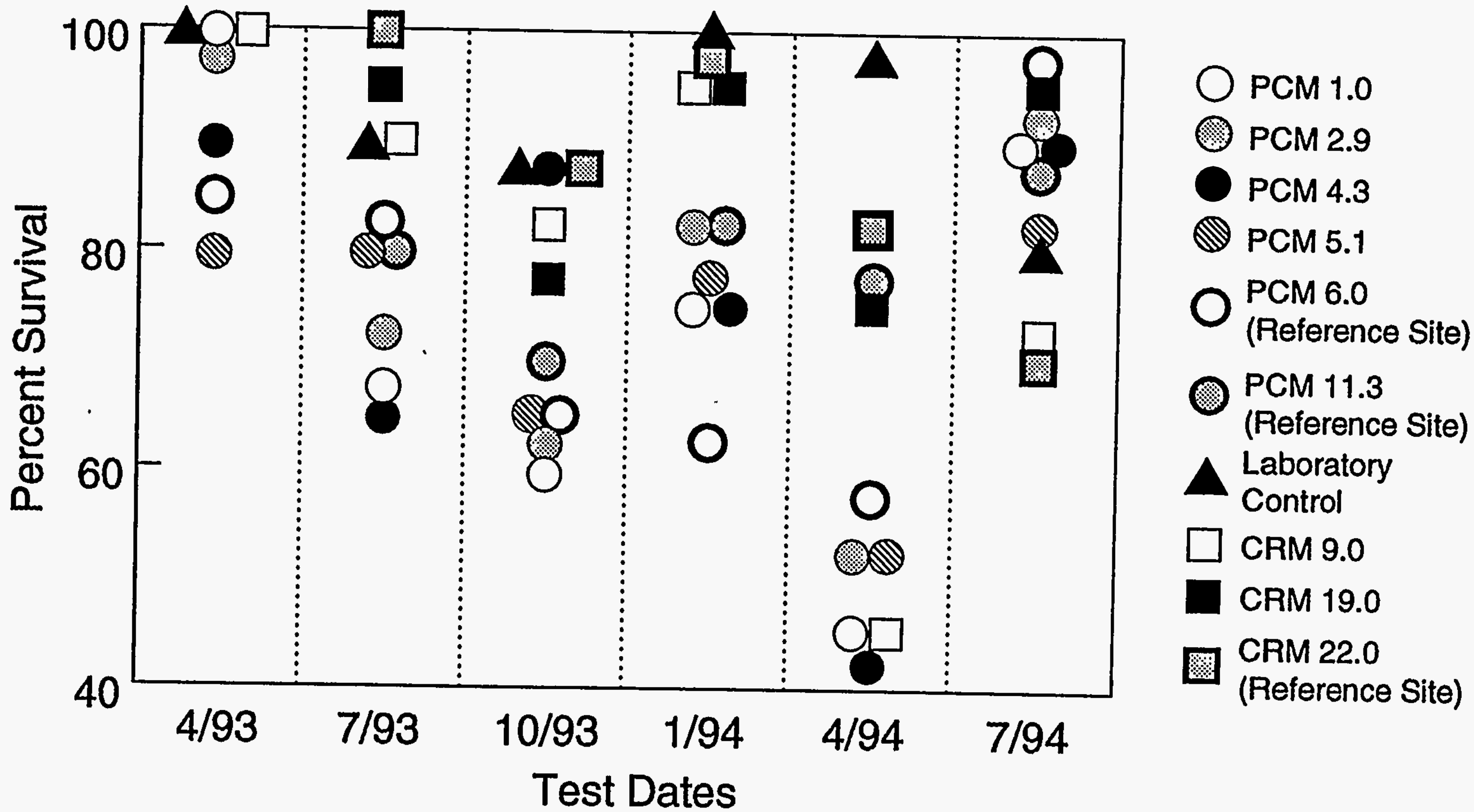

Fig. 3.13. Fathead minnow survival in water samples from Poplar Creek and Clinch River sites by test date.

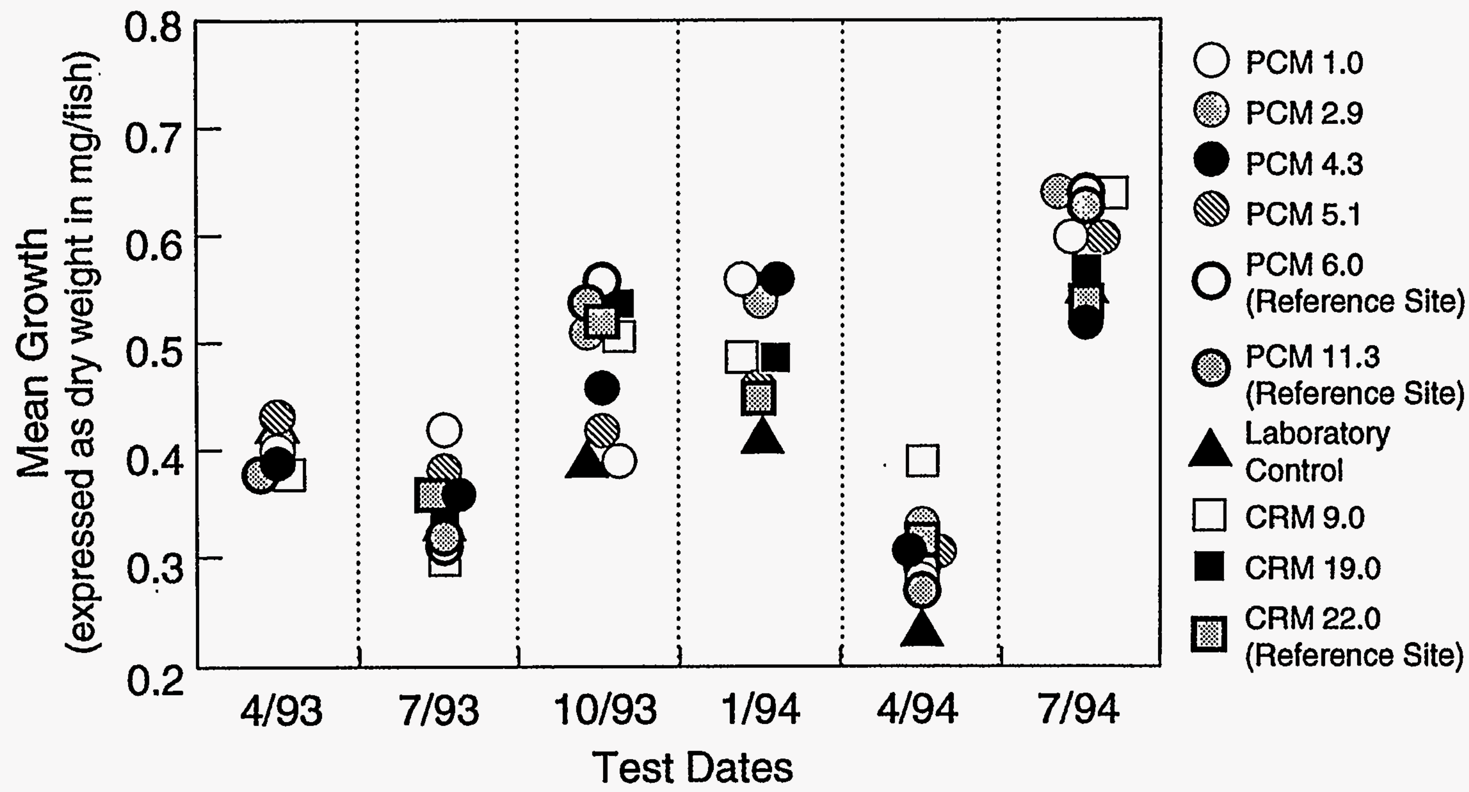

Fig. 3.14. Fathead minnow growth in water samples from Poplar Creek and Clinch River test sites by test date.

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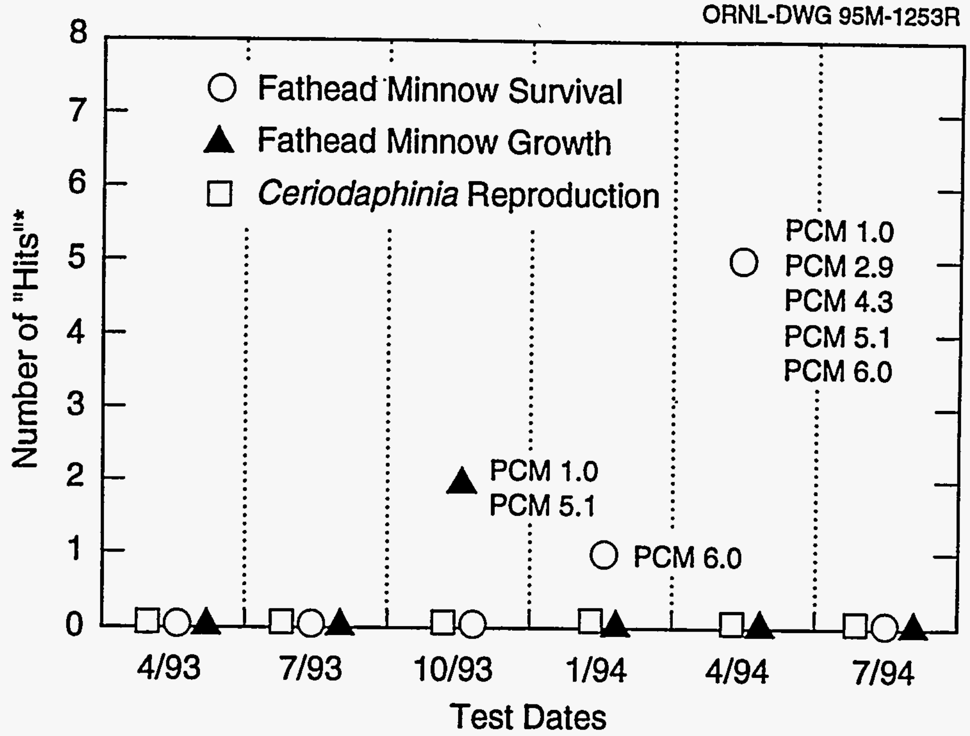

*Differences of 20% or greater; comparisons made to reference site PCM 11.3

Fig. 3.15(a). Comparisons made to reference site Poplar Creek mile 11.3 of fathead minnow survival and growth and *Ceriodaphnia dubia* reproduction; number of hits by test date.

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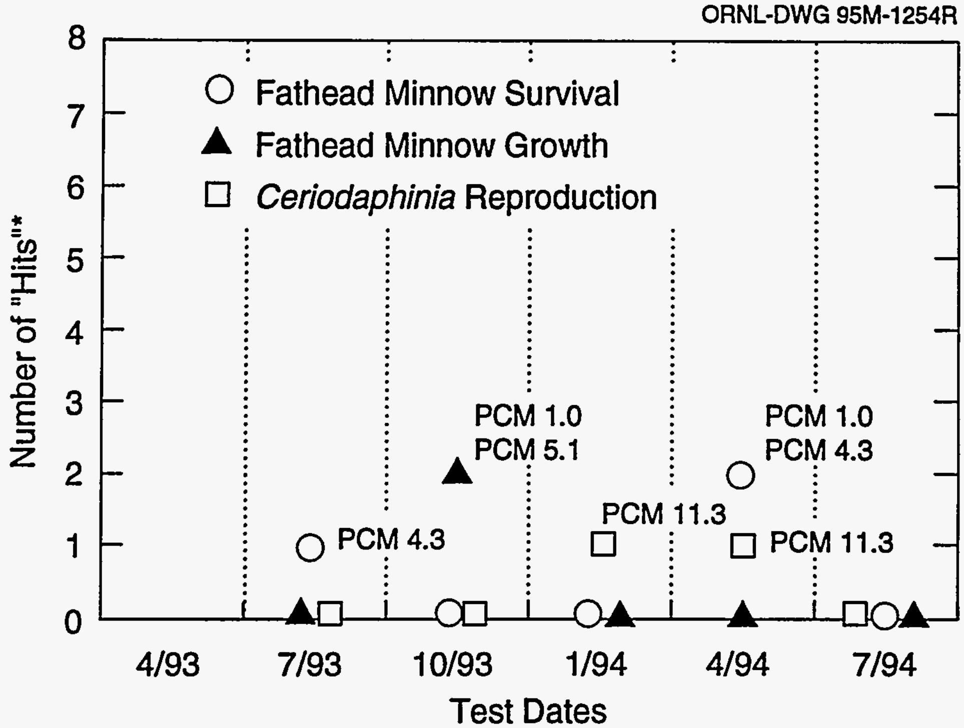


*Differences of 20% or greater; comparisons made to reference site PCM 6.0

Fig. 3.15(b). Comparisons made to reference site Poplar Creek mile 6.0 of fathead minnow survival and growth and *Ceriodaphnia dubia* reproduction; number of hits by test date.

20% during several tests. In only one test (April 1994) did any site exhibit a 20% reduction in survival. No reductions of 20% in growth were observed for any site during any test.

***Ceriodaphnia dubia* test results.** No spatial or temporal patterns in *Ceriodaphnia* survival were observed; survival was uniformly high (>80%) in all tests (Tables L20–L21). An analysis of variance revealed no significant spatial effects in *Ceriodaphnia* reproduction among Clinch River sites, and no significant differences in *Ceriodaphnia* reproduction were found among Poplar Creek sites in four of the six test periods. However, reproduction in Walker Branch Embayment (Walker Branch mile 0.4, reach 8) was significantly greater than reproduction in upper McCoy Branch Embayment (Melton Branch mile 0.4, subreach 7.02) during both test periods.

Although significant temporal differences existed, the organisms tended to produce more offspring in ambient water than in laboratory control water. Ambient waters may provide additional nutrients such as algae, bacteria, and detritus that contribute to an increase in fecundity (Kszos et al. 1992). *Ceriodaphnia* fecundity data are summarized in Tables L20–L21.

The only (statistically nonsignificant) reductions in fecundity of 20% or more between sites were reductions in Poplar Creek reference sites compared with each other [Fig. 3.15 (a) and (b)].

Medaka embryo-larval test results. Five medaka embryo-larval tests were conducted for the CR-ERP Phase 2 Investigation on surface water samples from six sites in Poplar Creek and three sites in the Clinch River (DOE 1994b) at quarterly intervals from July 1993 through July 1994. After these tests, two additional medaka embryo-larval tests were conducted on many of the same sites for the K-25 Site Biological Monitoring and Abatement Program (BMAP). Two tests were also conducted in the summer of 1994 on two sites in lower McCoy Branch and a site in Walker Branch as a reference. In all tests, survival, hatching success, and the incidence of various developmental abnormalities were scored for individual medaka embryos exposed throughout embryonic development to water samples from the study sites.

Largemouth bass (*Micropterus salmoides*) and redbreast sunfish (*Lepomis auratus*) embryos were obtained by artificially spawning sexually mature adults, and they were used in a limited series of tests to directly examine the toxicity to gamefish of surface water from many of the sites tested with medaka embryos. Methods for these one-time tests were adapted from the medaka embryo-larval test.

Results of embryo-larval tests were statistically examined by Chi-square analysis, with significant differences ($\alpha = 0.05$) between sites determined by comparison of sample proportions as described in Daniel (1987).

Hatching success was similar to survival in all tests. Apparently as a result of a relatively long prehatching interval (10–12 days average), the medaka embryo appears most sensitive to adverse water conditions in the period before hatch. The discussion below thus focuses only on the survival and developmental abnormality statistics of these tests.

Survival in water from three sites in the Clinch River (CRM 22, CRM 19, and CRM 9) was excellent in four of the five Phase 2 tests (Table L22) and in both of the K-25 Site BMAP tests (Table L23). The only exceptions occurred in the final quarterly Phase 2 test initiated in July of 1994, when survival at the two upstream sites in the river (CRM 22 and CRM 19) plummeted to 36% and 16%, respectively. Survival in control solutions was uniformly high (≥80%) in all tests.

Survival of medaka embryos and fry in water from sites in Poplar Creek was more variable. For example, survival in water from a reference site (PCM 11.0) located well upstream of any influences from the ORR differed significantly from control survival in four of seven tests (Tables L22 and L23). Nonetheless, the general pattern throughout Poplar Creek was a decrease in mean survival in comparison with the reference site (Fig. 3.16), with greatest impacts observed in subreach 3.02 (PCM 4.3) and subreach 3.04 (PCM 1.0).

The incidence of developmental abnormalities observed in medaka embryos exposed to water from the Clinch River differed significantly from controls only in the July 1994 test (Tables L22 and L23). Certain abnormalities were seen more frequently at the two downstream sites (subreach 2.02 at CRM 19 and subreach 4.01 at CRM 9) in comparison with the upstream reference site (subreach 2.01 at CRM 22) (Fig. 3.16), but overall there was little significant pattern to the occurrences of developmental abnormalities in medaka exposed to water from any sites in the Clinch River.

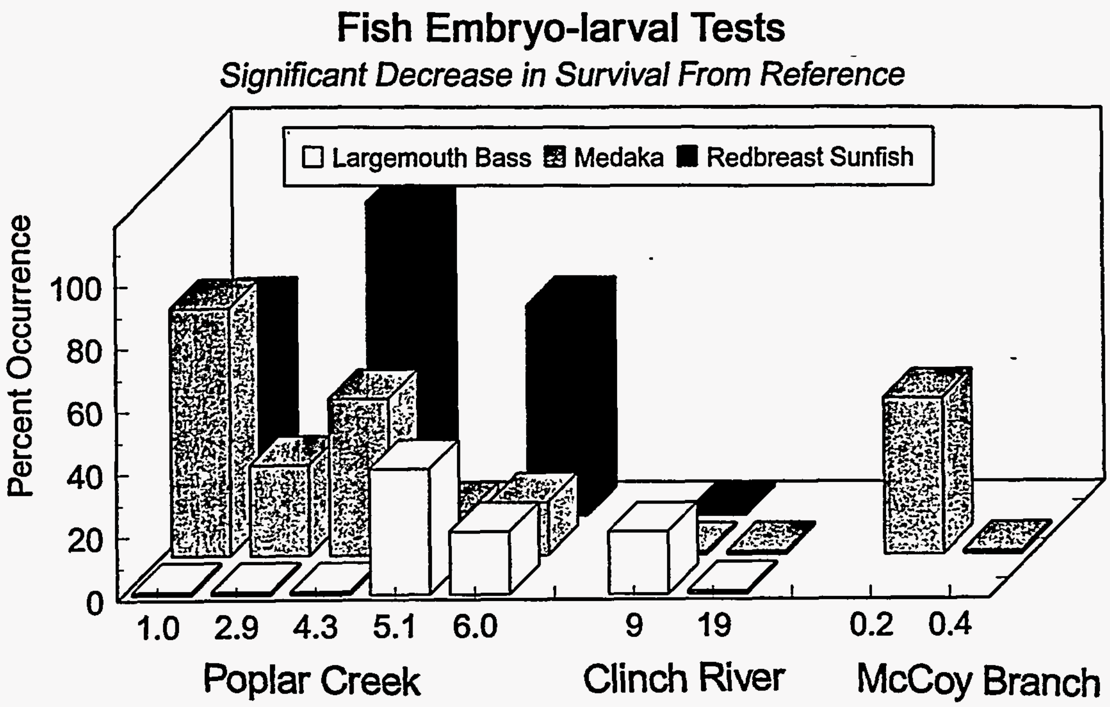


Fig. 3.16. Mean frequency of occurrence of significant decreases in fish embryo-larval survival in water from the various study sites as compared with the appropriate reference. For medaka, the values are based on a series of five to eight different tests of water from Poplar Creek and Clinch River sites and two tests for McCoy Branch sites. Values for largemouth bass and redbreast sunfish are based on tests of single water samples against embryos derived from multiple female and male in vitro pairings (totals of five and three, respectively).

In contrast, developmental abnormalities were much more common in embryos exposed to water from all sites in Poplar Creek, particularly the sites adjacent to the ORR (Tables L22 and L23). For example, overall incidence of developmental abnormalities significantly higher than those in controls were noted in six of seven medaka embryo-larval tests conducted with water samples from subreach 3.02 (PCM 4.3). The incidence of many specific abnormalities were also greatest in embryos exposed to water from sites in lower Poplar Creek adjacent to the ORR (Fig. 3.17). The most frequent abnormalities noted for these lower Poplar Creek sites were associated with the chorionic layer, which encloses the embryo until hatching. In fact, the poor survival of medaka embryos in water from the lower Poplar Creek sites often appears to be the result of problems with the proper maintenance of chorionic integrity; such problems lead to premature and largely unsuccessful attempts to hatch.

Survival of medaka embryos in two tests of water from sites in lower McCoy Branch (subreach 7.02) and Walker Branch (subreach 8; reference for McCoy Branch) was generally very high (Table L24). Although survival was significantly less at Melton Branch mile 0.2 compared with the Walker Branch reference site (Walker Branch mile 0.4) in one of the two tests (Fig. 3.20), it was still at a very acceptable 84% level. Very few developmental abnormalities were observed in embryos exposed to any of these water samples (Table L24 and Fig. 3.17).

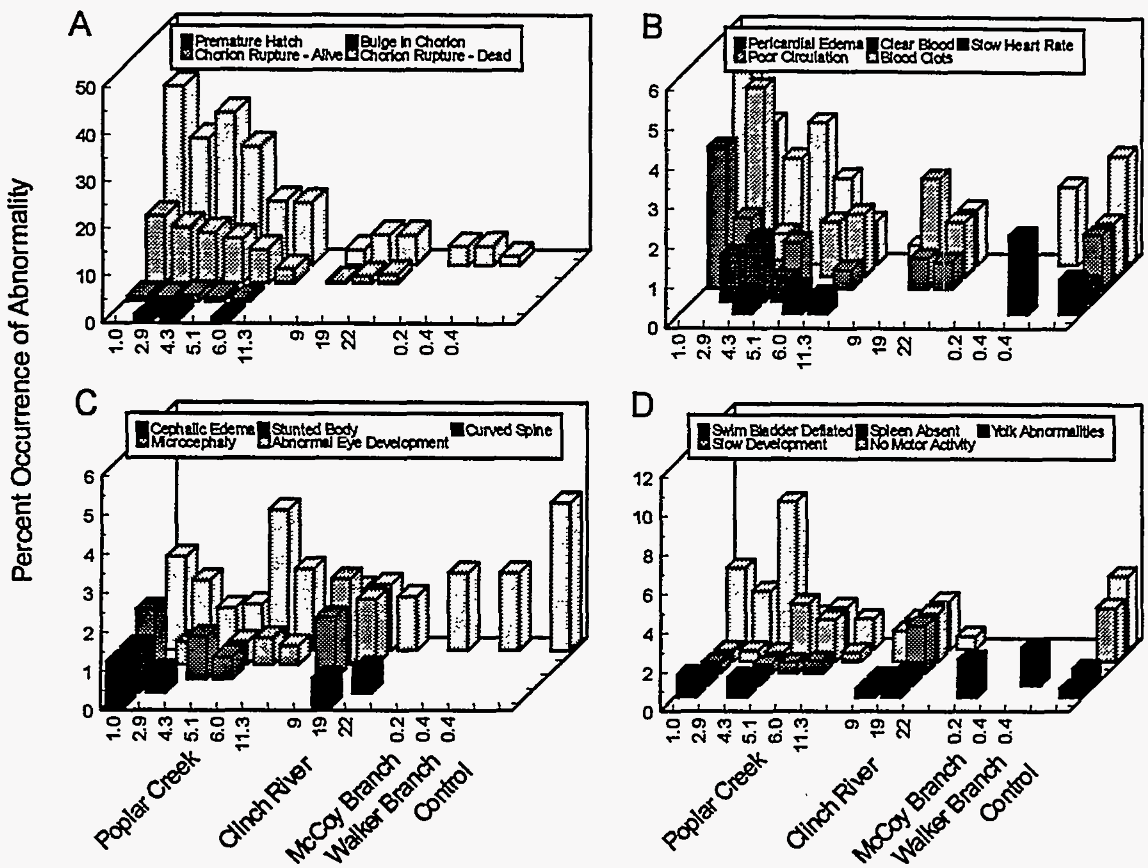


Fig. 3.17. Percent occurrence of developmental abnormalities during medaka embryo-larval tests. Abnormalities are grouped according to the following major classifications: (a) chorion-associated abnormalities, (b) circulation-associated abnormalities, (c) skeletal-muscular and head region abnormalities, and (d) other abnormalities.

Redbreast sunfish toxicity tests. A set of fish embryo-larval tests was initiated in July 1994 to compare the responses of medaka and redbreast sunfish embryos and fry with water from select sites in Poplar Creek and the Clinch River. Redbreast sunfish tests were conducted with eggs from females collected from a reference site and fertilized in vitro. The normal control was not included in this series of tests because of uncertainties about the suitability of using the millipore-filtered distilled water control in testing redbreast sunfish. Instead, all comparisons within tests were made to the appropriate reference site.

In this comparison test, survival was again very high (80% for embryos and 85% for fry), and the incidence of developmental abnormalities relatively low (10% for embryos and 20% for fry) for medaka embryos and fry exposed to water from both CRM 22.0, the reference site on the Clinch River, and CRM 9.0 (Table L25). Survival was only slightly less (70%) and the incidence of medaka embryos with developmental abnormalities even lower (5%) at PCM 11.3, the reference site on Poplar Creek. Survival plummeted (to 11% in both instances) and the incidence of embryos with abnormalities increased significantly (to 50% and 58%, respectively) following exposure of medaka embryos to water from PCM 4.3 and PCM 1.0.

Similarly, the survival of redbreast sunfish embryos and fry was uniformly high in water from both the Clinch River sites, as well from the upstream reference site on Poplar Creek (Table L25; Fig. 3.16). Furthermore, survival of redbreast sunfish embryos and fry was also adversely affected by exposure to water from downstream study sites on Poplar Creek, although the results were not as consistent nor as pronounced as in tests involving medaka embryos. Few developmental abnormalities were observed during the course of these redbreast sunfish toxicity tests.

Largemouth bass toxicity tests. Embryo-larval tests were also conducted during April 1994 on largemouth bass embryos and fry exposed during early development to water from several sites in the Clinch River and Poplar Creek and from one site in the Tennessee River. Because of time constraints caused by difficulties in collecting female largemouth bass that possessed mature but not overripe, fertilizable eggs from reference sites, no attempt was made to conduct a medaka embryo-larval test concurrent with these largemouth bass tests. Largemouth bass embryo-larval test procedures were similar to those employed in tests with redbreast sunfish and medaka embryos, except that hatching success was not scored in the bass tests.

Survival statistics for embryos and fry derived through in vitro fertilization techniques from each of five different largemouth bass pairings are presented in Table L26 and summarized in Fig. 3.16. Survival was uniformly high in these tests; however, small but statistically significant decreases in survival in comparison with that of controls were observed in two of five groups of embryos tested against water from CRM 9.0. In addition, minor, but statistically significant, decreases occurred in survival in water from several downstream sites in Poplar Creek. As with the redbreast sunfish embryo-larval tests, obvious developmental abnormalities (other than mortality) were rarely encountered in these largemouth bass embryo-larval tests (Table L27).

Summary of medaka tests. The results of the medaka embryo-larval tests clearly demonstrate that water from lower Poplar Creek has an intermittent toxicity to fish embryos. This toxicity does not appear to extend into the Clinch River. Redbreast sunfish embryos appear to respond much like medaka embryos to water from both systems, although to a lesser degree. Largemouth bass embryos and fry appear to be the least sensitive of the fish species tested, although the absence of concurrent medaka test data weakens this comparison.

Genotoxicity test results. Surface water from the Clinch River and Poplar Creek was evaluated for its potential to cause genotoxic or toxic effects in DNA or the cell, respectively, with the commercially available SOS-Chromotest kit. No significant genotoxic or toxic effects were observed in the Clinch River and Poplar Creek samples tested. Isolated samples exhibited slight genotoxicity, but no trends were established.

3.3.4 Water Characterization—Summary of Findings

The site model, based on historical data, indicates that organic compounds have been detected infrequently. Those that are detected are relatively ubiquitous in nature and have been present only at very low concentrations. This model also indicates that inorganic analytes and radionuclides have generally been present at low concentrations and typically do not exceed the AWQC for protection of domestic water supplies or aquatic life; however, detection limits are inadequate in some cases. The concentrations of certain inorganic analytes and radionuclides have historically been greatest immediately downstream of streams that drain contaminant source areas on the ORR (i.e., EFPC, Mitchell Branch, WOC, McCoy Branch).

The CRRI Phase 2 surface water investigation consisted of a number of discrete studies, most of which did not by themselves fully characterize a defined subreach or reach of the OU (except for Poplar Creek). Therefore, the data were compiled across these various tasks, augmented with CRRI Phase 1 data and concurrent ORREMP data (1993–94), and this combined data set used to characterize the nature and extent of contamination.

Mean concentrations of a number of analytes were elevated in source streams (WOC, EFPC, and Mitchell Branch). However, the only significantly elevated mean values within the OU were (1) total nitrate/nitrite in Poplar Creek below the CNF outfall, (2) total mercury and total methyl mercury in Poplar Creek below EFPC, (3) gross alpha and beta counts in Poplar Creek downstream of Mitchell Branch (winter samples only), and (4) total and dissolved arsenic in the upper McCoy Branch Embayment of Melton Hill Reservoir. Concentrations of organic compounds were expectedly low throughout the system; many compounds were undetected anywhere.

Several analytes potentially exceed AWQC in one or more locations. Mean arsenic levels likely (LCL_{95} > criterion) exceed recreation-based criteria in upper McCoy Branch (subreach 7.01); mean concentrations here are the highest of any subreach. Mean mercury levels likely exceed the CCC for protection of aquatic life in upper Melton Hill Reservoir (reach 0), in Poplar Creek (subreaches 3.01–3.04), and in the lower Clinch River (subreaches 4.01, 4.03, and 4.04). Mercury concentrations are highest in Poplar Creek.

Mean arsenic levels likely (LCL_{95} < criterion < mean) exceed the recreation-based criterion in upper Poplar Creek (reach 13). The mean mercury concentration likely exceeds the recreation criterion in lower Poplar Creek (subreach 3.04). Finally, the mean arsenic concentration possibly (mean < criterion < UCL_{95}) exceeds the recreation criteria in subreach 3.03 and throughout the Clinch River, including Melton Hill Reservoir.

Some Phase 2 studies answer specific contaminant-related questions. A study designed to characterize contaminant concentrations in the Clinch River during presumed worst-case contaminant flux conditions found no evidence of large-scale contaminant migration that originated from known source streams and that resulted in increased ambient concentrations in the Clinch River. Maximum concentrations of contaminants in source streams were generally within the range of historically reported values. In a study that evaluates the potential for mobilization of contaminants from sediment, no evidence was found of significant contaminant remobilization from sediment to surface water. Finally, the toxicity of surface water from a number of sites in the OU was evaluated through the use of several test organisms and measures of toxicity. No site-related patterns of toxicity were observed in the majority of the toxicity tests. The exception was the Japanese medaka embryo-larval series of tests, which demonstrated intermittent toxicity to water from Poplar Creek. The medaka results are corroborated by similar results with redbreast sunfish, a species native to the Clinch River–Poplar

Creek area. Toxic effects in Medaka and to redbreast sunfish did not extend into the Clinch River downstream of Poplar Creek.

3.4 SEDIMENT CHARACTERIZATION

This section describes the nature and extent of sediment contamination in the CR/PC OU. Findings from previous investigations are first summarized and used to build a site conceptual model (Sect. 3.4.1). The CRRI Phase 1 and Phase 2 data are then used to describe the current distribution of contaminants in the system (Sect. 3.4.2). Data from CRRI Phase 2 supporting studies are presented in Sect. 3.4.3, and include an evaluation of sediment toxicity (Sect. 3.4.3.1) and modeled predictions of future contaminant distributions in sediment (Sect. 3.4.3.2). Results of the site characterization are summarized in Sect. 3.4.4.

3.4.1 Historical Studies and Site Conceptual Model

Clinch River sediment has been studied since the early 1950s. Most of the early studies focused on radiological contamination, particularly gamma-emitters. However, more recent studies have included analyses for uranium isotopes and metals, particularly mercury, and to a lesser extent, organic chemicals. The following discussion describes the general findings of these studies.

3.4.1.1 Annual surveys of gamma activity in Clinch River and Tennessee River bottom sediment

From 1951 to 1966, various researchers (Garner and Kochitzky 1956; Cottrell 1960; ORNL 1961, 1962, 1963, 1964, 1965, 1966, 1967) measured total gamma radiation at the surface of the sediment with a submersible Geiger-Mueller counter (called a "flounder"). In the Clinch River, surveys were conducted from CRM 27.5 to the mouth.

Transects were established across the river at various intervals, and gamma readings were taken annually at points along each transect. Readings (gamma counts per second) were averaged across the transect. Beginning in 1954, composite surface grab samples were also collected along each transect for radionuclide analysis (Cottrell 1960). The principal radionuclides contributing to gamma activity in most years were ^{137}Cs, ^{60}Co, ^{144}Ce, total trivalent rare earths (TRE), and ^{90}Sr (Fig. 3.18). The distribution of activity along a transect was generally proportional to the depth of the water (e.g., Fig. 3.19); most of the activity was found in the main channel.

In general, gamma count rates in the Clinch River gradually increased with distance downstream from the mouth of WOC. The highest count rates found throughout the entire Clinch and Tennessee river system typically occurred between CRM 11.0 and CRM 8.0, although annual variations exist (Fig. 3.20). Count rates remained relatively constant from CRM 8.0 to the mouth of the river, except for low counts at scour points at CRM 4.7 and CRM 2.6.

An increase in activity in 1952 (Fig. 3.20) was attributed to the release of an unspecified short-lived radionuclide, possibly barium, which had mostly decayed by 1953 (Garner and Kochitzky 1956). The large increase in activity during the period 1956–60 was primarily caused by the draining of White Oak Lake and the attendant scouring of contaminated sediment from the lake bottom. The increased activity in the 1961 survey probably resulted from increased releases of ^{106}Ru in 1960. The gamma surveys demonstrated that the general distribution of radioactive sediments has remained much the same in the Clinch River over time. Annual variations in surface activities typically reflected the quantity of particle-associated radionuclides released at White Oak Dam.

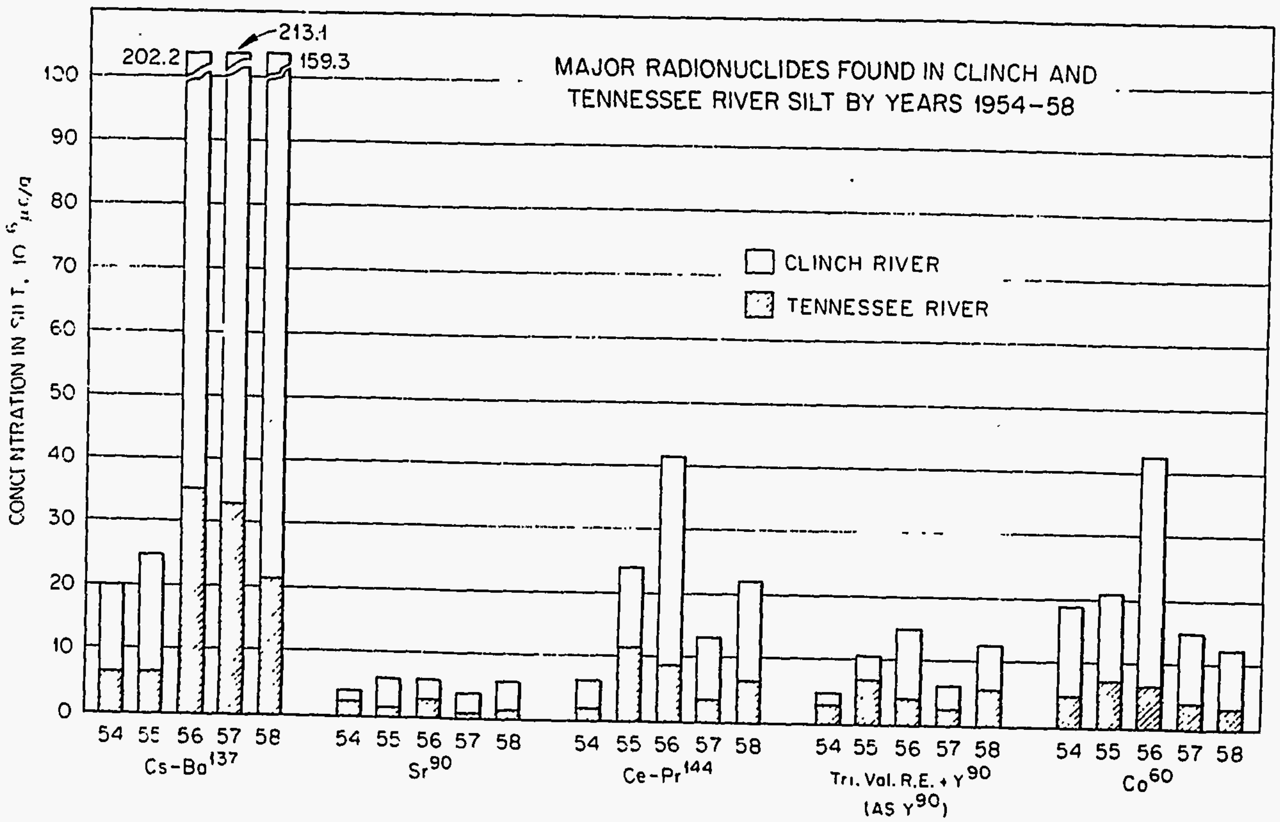


Fig. 3.18. Major radionuclides found in Clinch River and Tennessee River silt by years 1954 through 1958. *Source*: W. D. Cottrell 1960. *Radioactivity in Silt of the Clinch and Tennessee Rivers.* ORNL-2847. Oak Ridge National Laboratory, Oak Ridge, Tenn.

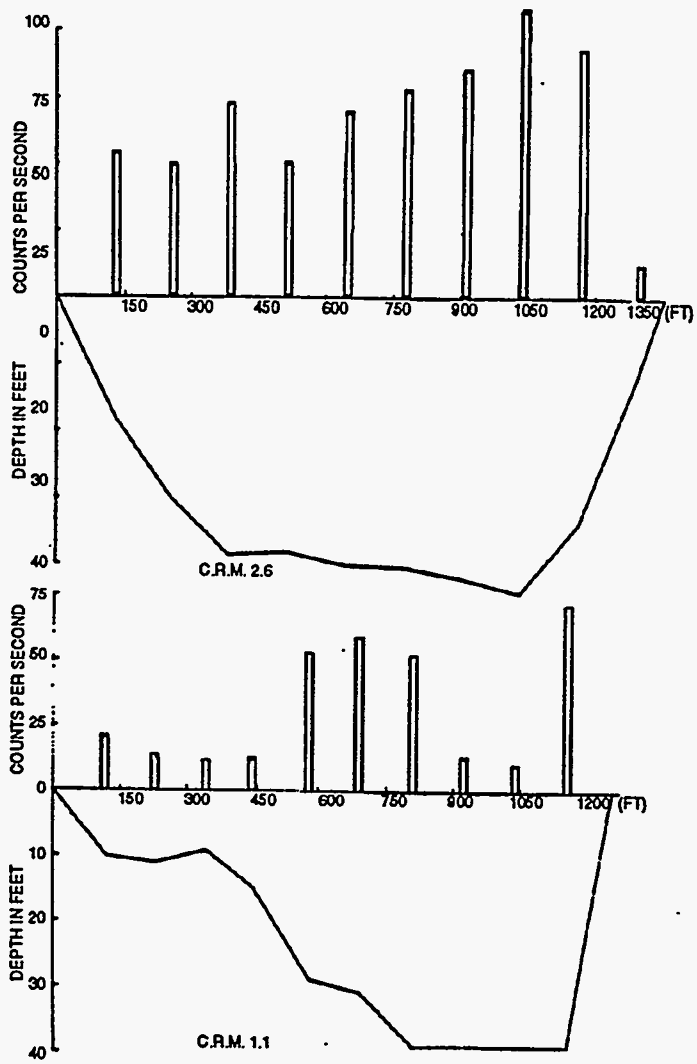


Fig. 3.19. Gamma activity vs depth profile for selected locations in the Clinch River. *Source:* W. D. Cottrell. 1960. *Radioactivity in Silt of the Clinch and Tennessee Rivers.* ORNL-2847. Oak Ridge National Laboratory, Oak Ridge, Tenn.

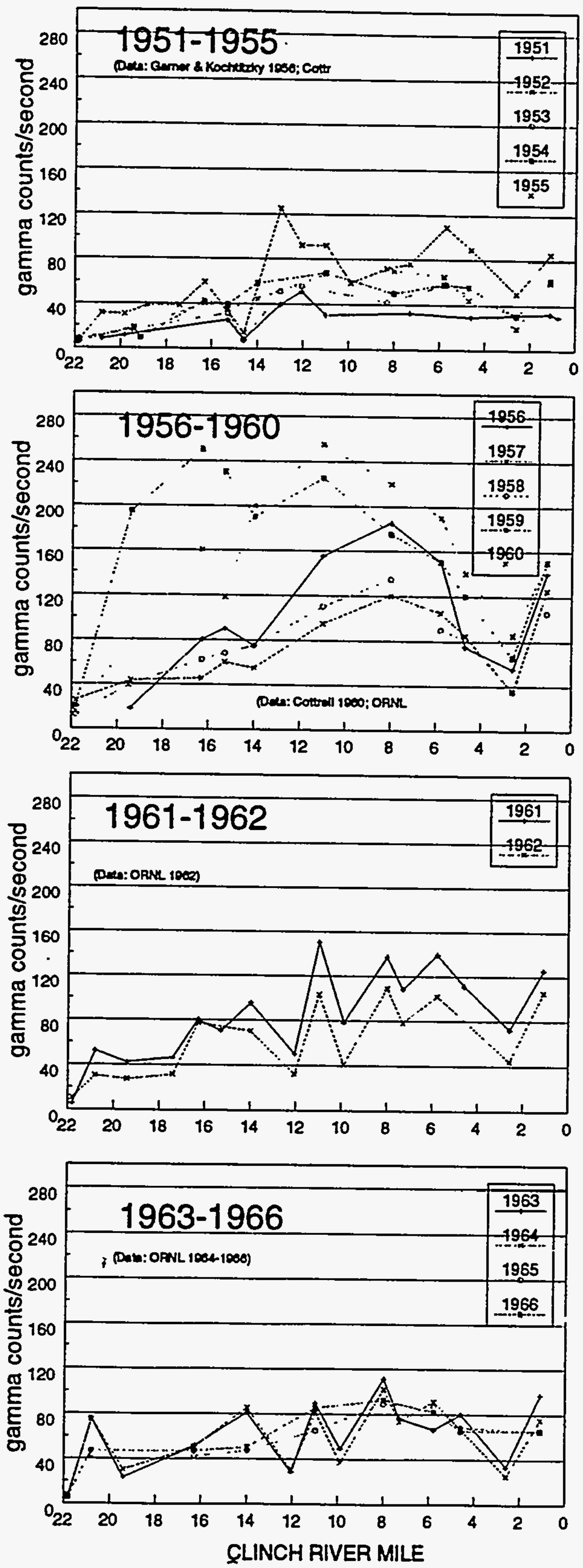


Fig. 3.20. Results of gamma surveys in the Clinch River, 1951 through 1966.

3.4.1.2 Clinch River Study

The purpose and scope of the Clinch River Study are outlined in Sect. 3.3.1.1. The following text briefly summarizes salient findings with respect to contaminated sediment.

In 1962, 113 cores were collected from 14 locations in the Clinch River and were used to estimate the radionuclide inventory in sediment. A total inventory of about 200 Ci was estimated for the river as of July 1, 1962 (Table 3.4). The bulk of this inventory was ^{137}Cs (~150 Ci, or 75% of total). The remaining inventory was composed of an estimated 18 Ci (9%) of ^{60}Co, 16 Ci (8%) of ^{106}Ru, 10 Ci (5%) of TRE elements, and 2.9 Ci (1.5%) of ^{90}Sr. Approximately 95% of the total inventory was downstream of CRM 15, and at least 60% was downstream of CRM 9. The inventories listed here represent ~1.5% of the total activity released from ORNL to that time, including ~21% of the ^{137}Cs, 9% of the ^{60}Co, 0.4% of the ^{106}Ru, and 0.2% of the ^{90}Sr (Carrigan and Pickering 1967).

Table 3.4. Estimated radionuclide inventory of the Clinch River below the Oak Ridge Reservation, as of July 1, 1962

Location of subreach		Total identified activity (Ci)		Volume of radioactive sediment (acre-ft)	
From Clinch River mile	To Clinch River mile	In subreach	Cumulative	In subreach	Cumulative
0	2.80	22	22	340	340
2.8	5.9	42	64	380	720
5.9	8.95	54	118	480	1200
8.95	10.95	46	164	430	1630
10.95	12.00	9.3	173	93	1720
12.00	13.05	10	183	85	1810
13.05	15.00	6.8	191	38	1850
15.00	16.75	2.2	193	33	1880
16.75	18.35	4.7	198	39	1920
18.35	19.85	0.1	198	4.7	1920
19.85	20.65	2.3	200	5.9	1930
20.65	20.90	0.1	200	1.4	1930
20.90	21.00	0.2	200	0.3	1930

Source: P. H. Carrigan, Jr. and R. J. Pickering. 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part B, Inventory of Radionuclides in Undisturbed Cores.* ORNL-372. Oak Ridge National Laboratory, Oak Ridge, Tenn.

In general, the distribution of radionuclides in the Clinch River is controlled by the mechanics of sedimentation. The longitudinal distribution among individual radionuclides was similar (Fig. 3.21), and each exhibited the same general pattern (e.g., Fig. 3.22) identified in the gamma surveys (Sect. 3.4.1.1). Although areas of greater and lesser deposition are evident, the thickness of radioactive sediment generally increased linearly from CRM 21 to the mouth of the Emory River (Fig. 3.23) (Carrigan and Pickering 1967). Laterally, radiation levels are lower in sloughs of the Clinch River than in sediments of the main channel, and radionuclide levels were less in areas exposed during winter drawdown than in areas that were continuously inundated.

Carrigan and Pickering (1967) found that peak concentrations of gamma activity (almost entirely from ^{137}Cs) in core samples occurred at depth; the greater the sediment accumulation rate in an area, the greater the depth of the ^{137}Cs peak in the core (Fig. 3.24). The ^{137}Cs profile in a core was found to correlate with the release history from ORNL (Fig. 3.25).

Carrigan and Pickering (1967) found that, in general, radioactive bottom sediment in the Clinch River could be classified as clayey silt, composed of ~35% mica and other clays and 65% quartz. The potential for desorption of radionuclides was investigated by using contaminated sediments from WOC (Morton 1965). Only ^{90}Sr was found to be held by simple ion exchange and easily removed by circumneutral salt solutions. Strongly acidic solutions desorbed 80–90% of the ^{90}Sr and 65–80% of the ^{60}Co, whereas strongly alkaline solutions were more effective in desorbing ^{106}Ru. Regardless of treatment, <6% of the ^{137}Cs was desorbed, indicating a strong affinity for particles. This affinity is greatest for certain clay minerals.

3.4.1.3 1977 Clinch River sediment survey

Sediments in the vicinity of the proposed Clinch River Breeder Reactor site (CRM 20.8) were sampled in 1977 to determine the fate and distribution of radionuclides in the Clinch River (Oakes et al. 1982). Specific objectives were to analyze selected cores for transuranic radionuclide activity (previous studies had not reported alpha emitters) and to examine the effect that the altered flow regime, which was a result of operations at Melton Hill Dam, had on the distribution of fission products in the sediment. A total of 250 cores were collected along the length of the Clinch River, from the mouth of WOC to the confluence with the Tennessee River; cores were also collected from the Tennessee River above and below the mouth of the Clinch River. An attempt was made to sample areas of elevated gamma activity by first identifying those areas through use of the flounder. However, subsequent analysis of the cores revealed that the flounder had been more strongly influenced by background radiation than by radionuclides discharged from ORNL. An additional limitation affecting the representativeness of this sampling effort was that the sampling device could only be used effectively in water depths of 15 ft or less. This restriction limited sampling to the extreme near-shore environment and overbank areas.

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

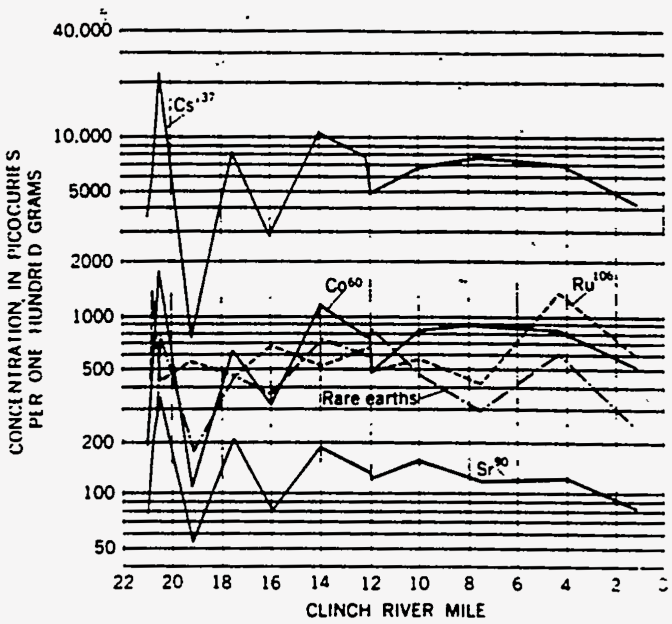


Fig. 3.21. Longitudinal distribution of radionuclides in the Clinch River downstream of the Oak Ridge Reservation. *Source*: P. H. Carrigan, Jr. and R. J. Pickering. 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part B, Inventory of Radionuclides in Undisturbed Cores.* ORNL-372. Oak Ridge National Laboratory, Oak Ridge, Tenn.

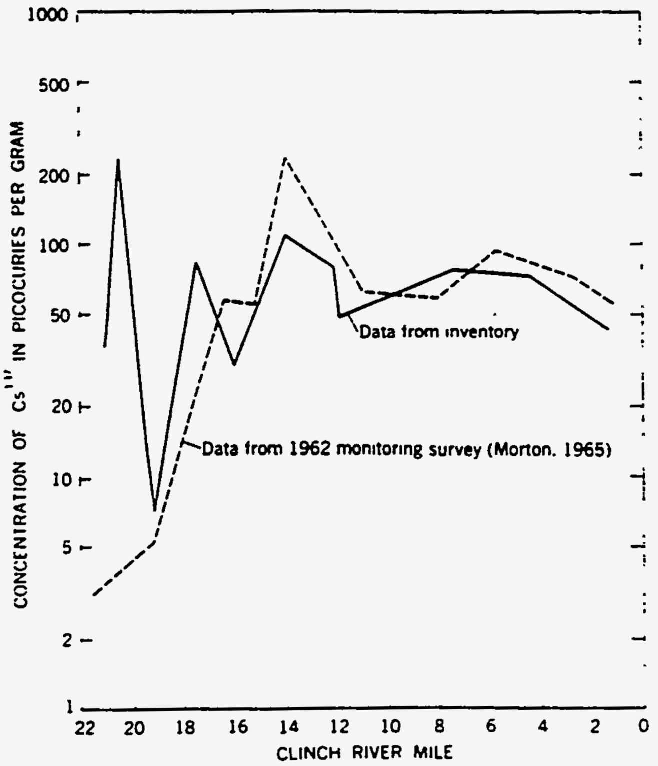


Fig. 3.22. Comparison of the longitudinal distribution of ^{137}Cs with gross gamma activity in the Clinch River downstream of the Oak Ridge Reservation. *Source*: P. H. Carrigan, Jr. and R. J. Pickering. 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part B, Inventory of Radionuclides in Undisturbed Cores.* ORNL-372. Oak Ridge National Laboratory, Oak Ridge, Tenn.

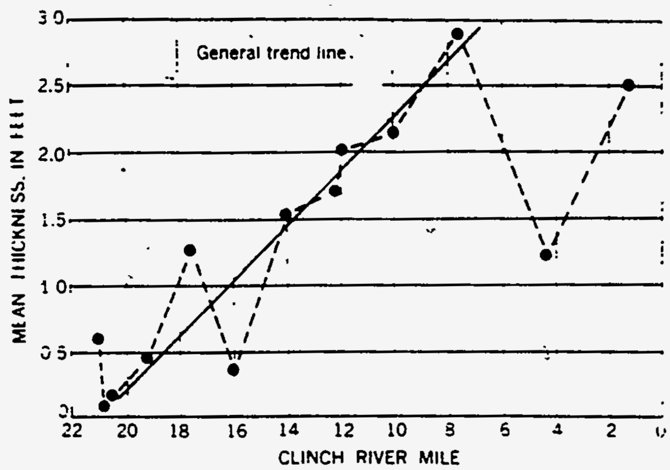


Fig. 3.23. The thickness of radioactive sediment in the Clinch River was found to increase between the mouth of White Oak Creek and Clinch River mile 8, after which it declined sharply. *Source*: P. H. Carrigan, Jr. and R. J. Pickering. 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part B, Inventory of Radionuclides in Undisturbed Cores.* ORNL-372. Oak Ridge National Laboratory, Oak Ridge, Tenn.

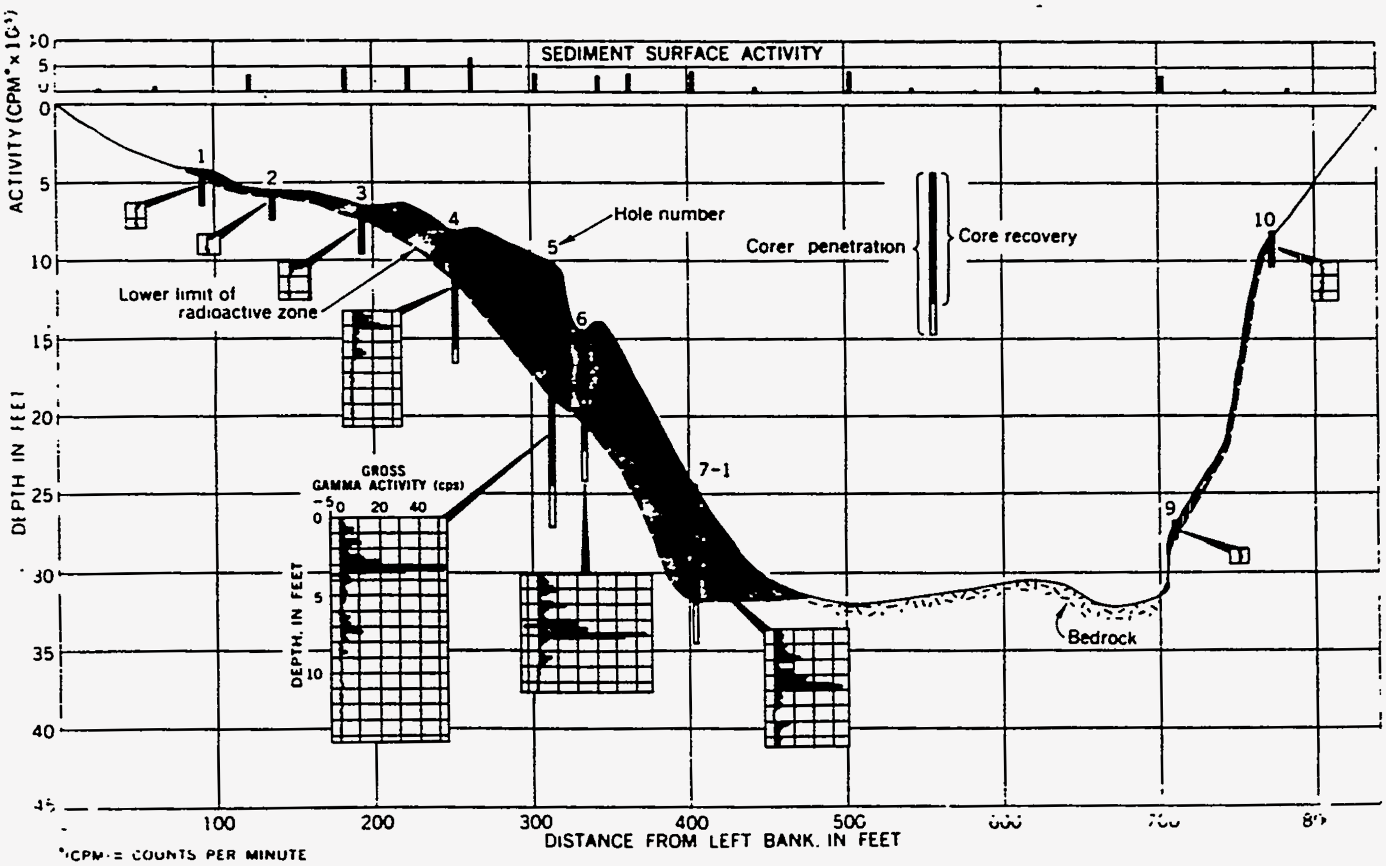


Fig. 3.24. Cross section of the Clinch River at mile 7.5, showing lateral distribution of sediment and vertical distribution of gross gamma activity. *Source*: P. H. Carrigan, Jr. and R. J. Pickering. 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part B, Inventory of Radionuclides in Undisturbed Cores.* ORNL-372. Oak Ridge National Laboratory, Oak Ridge, Tenn.

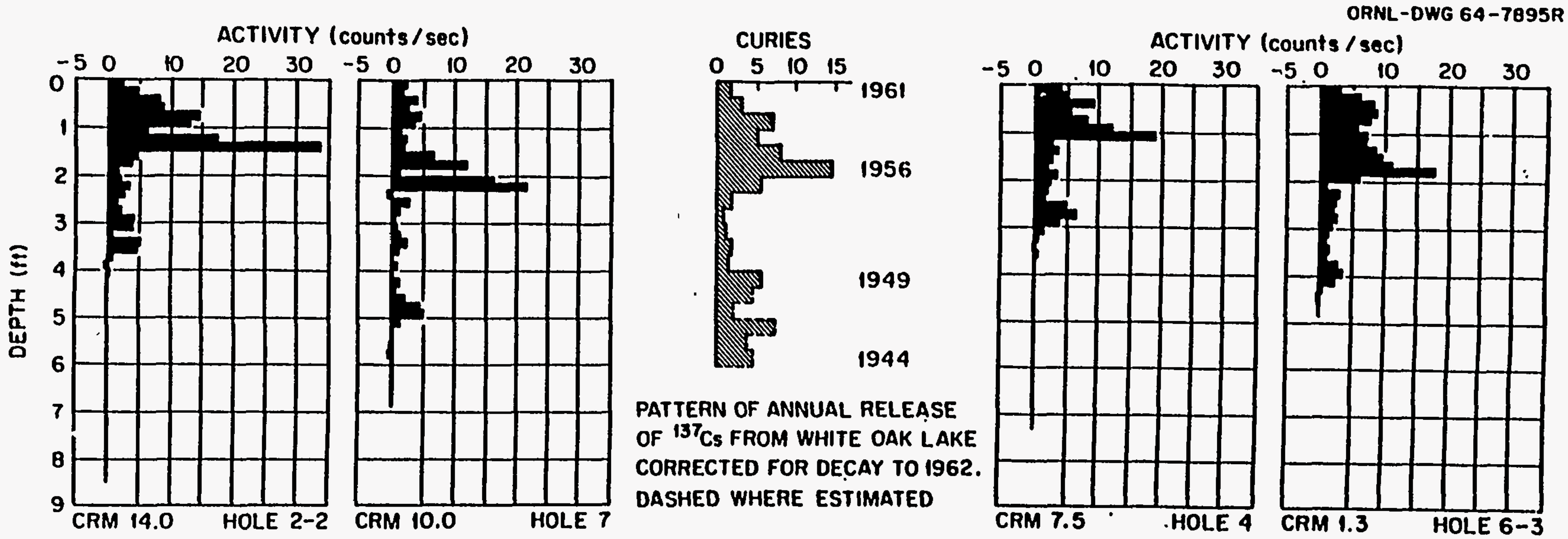


Fig. 3.25. Comparison of patterns of variation with depth of gross gamma radioactivity in four bottom sediment cores to variations in annual releases of ^{137}Cs to Clinch River. *Source*: P. H. Carrigan, Jr. and R. J. Pickering. 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part B, Inventory of Radionuclides in Undisturbed Cores.* ORNL-372. Oak Ridge National Laboratory, Oak Ridge, Tenn.

The report of Oakes et al. (1982) is a data summary report, and as such, it offers only limited discussion and interpretation of the data. As in the Clinch River Study, however, peak concentrations of radionuclides were found to occur at varying depths within cores. Peak concentrations recorded in the study are presented in Table 3.5.

Table 3.5. Maximum concentrations of radionuclides reported in the 1977 Clinch River Sediment Survey

Radionuclide	Maximum concentration (pCi/g)	Location (Clinch River mile)	Depth in core (cm)
^{137}Cs	606	14.25	28
^{60}Co	12.2	14.25	28
^{90}Sr	11.7	14.25	31
$^{238,239}Pu$	6.5	14.0	23

Source: T. W. Oakes, W .F. Ohnesorge, J. S. Eldridge, T. G. Scott, D. W. Parsons, H. M. Hubbard, O. M. Sealand, K. E. Shank, and L. D. Eyman. 1982. *Technical Background Information for the Environmental and Safety Report, Vol. 5: The 1977 Clinch River Sediment Survey—Data Presentation.* ORNL-5878. Oak Ridge National Laboratory, Oak Ridge, Tenn.

3.4.1.4 Environmental fate of mercury and ^{137}Cs discharged from ORR facilities

Turner et al. (1984) collected sediment cores from the Clinch and Tennessee rivers in 1983 to independently confirm the release histories of mercury from the ORR facilities and to discover the fate of mercury discharged into area streams. Sediment cores were collected at CRM 6.8 and CRM 1.0 and in the Tennessee River in both Watts Bar and Chickamauga reservoirs. Sample collection was intentionally biased for areas of stable sediment deposition. All cores were sectioned into 1-, 2-, 3-, or 5-cm intervals for analysis. The core from CRM 1.0 was analyzed for total mercury, gamma emitters, total uranium, and selected metals. The core from CRM 6.8 was analyzed for total mercury only.

The results are illustrated in Fig. 3.26. The strong correlation between mercury and ^{137}Cs concentrations was observed throughout WBR, where peak concentrations were typically located 60 cm or more below the core surface. The peak uranium concentration (7.8 mg/kg) in the CRM 1.0 core also coincided with the mercury and ^{137}Cs peaks.

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

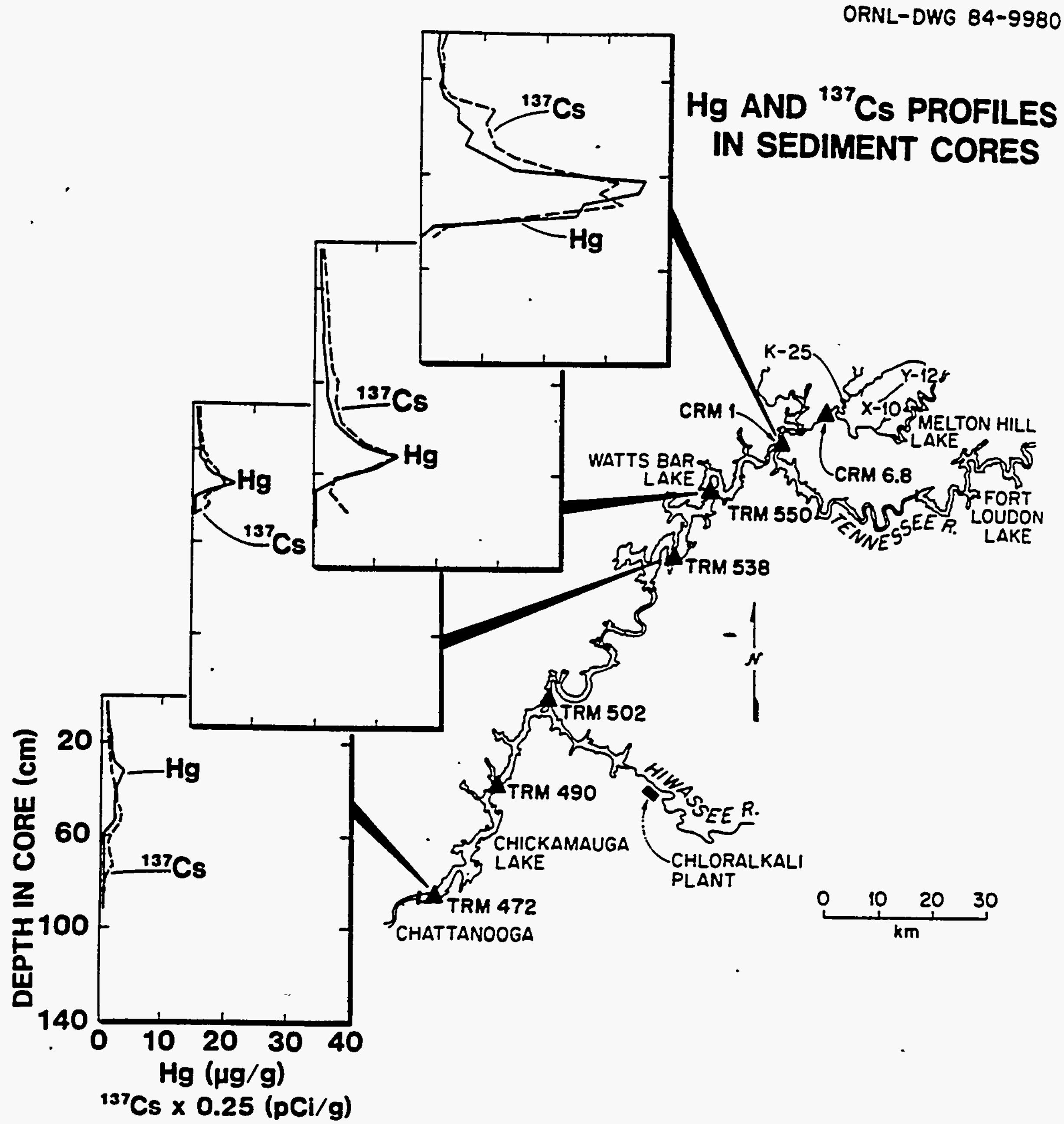


Fig. 3.26. The vertical distributions of mercury and ^{137}Cs in selected sediment cores obtained from Watts Bar and Chickamauga reservoirs. *Source*: R. R. Turner, C. R. Olsen, and W. J. Wilcox, Jr. 1984. "Environmental Fate of Hg and ^{137}Cs Discharged from Oak Ridge Facilities." In D. D. Hemphill (ed.), *Trace Substances in Environmental Health-XVIII, 1984, a Symposium.* University of Missouri, Columbia.

3.4.1.5 TVA Instream Contaminant Study

TVA conducted the Instream Contaminant Study for the Oak Ridge Task Force, a multiagency body investigating the mercury releases from the Y-12 Plant (TVA 1986b). Eight fine-particle surface sediment samples (the upper 10–15 cm of sediment) were collected from four locations in the Clinch River arm of WBR (CRM 3.7, 10.0, 15.6, and 18.3). A total of four reference samples were collected from locations in Melton Hill Reservoir (CRM 24.0) and Norris Reservoir (CRM 85.3 and 94.1 and Powell River mile 6.0). All samples were analyzed for radiological contaminants, base/neutral organic priority pollutants, priority pollutant metals, cyanide, phenols, and PCBs (TVA 1985a).

None of the 53 organic contaminants for which analyses were conducted was detected in any of the Clinch River surface sediment samples. Total phenols were detected at levels (6 μg/kg) comparable to those at the Norris and Melton Hill reservoirs reference areas (5 μg/kg). Results of the metals analyses are presented in Table K1. Values for selected radionuclides are presented in Table K2.

In addition, cores from eight locations in the Clinch River downstream of Melton Hill Dam were collected and analyzed for mercury; limited radiological analyses were also performed. Five of these cores were collected on Jones and Grubb islands rather than from instream locations, and all contained less than detectable levels of mercury (<0.1 mg/kg). The core samples were analyzed in two groups on the basis of sediment particle size fraction: (1) the fine sands and smaller fraction (containing particles <500μm) and (2) the silt and clay fraction only (containing particles <62μm but >500μm). Fine sands, silts, and clays collectively accounted for 90% or more of the sample volume at all sites (TVA 1985a). The Clinch River core data for mercury are presented in Fig. 3.27.

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

3.4.1.6 ORGDP sediment survey

Ashwood et al. (1986) collected ~180 surface sediment samples from streams around the K-25 Site. Samples were collected from the Clinch River, Poplar Creek, and tributaries draining the ORR. Samples were analyzed for radionuclides, metals, and organic contaminants. In addition, three core samples were collected, including one from the Clinch River near Kingston. The purpose of the study was to identify sites where pollutants from the K-25 Sitehad historically entered or were currently entering the surface water. Samples were collected in January and February 1985. The study focused on contaminants that Hoffman et al. (1984) identified, on the basis of data from the TVA Instream Contaminant Study (TVA 1986b), as warranting further study on the ORR.

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

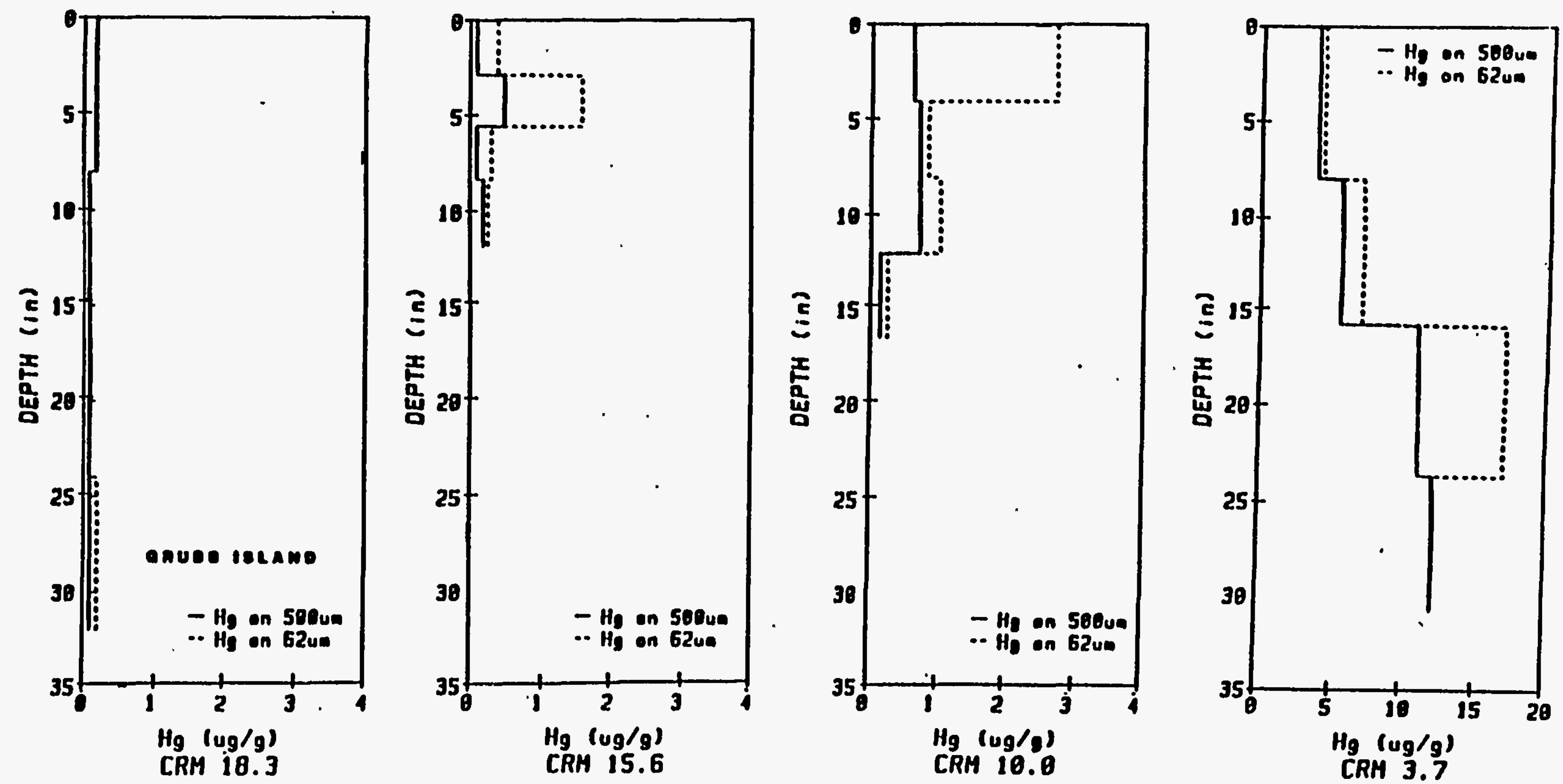


Fig. 3.27. Mercury concentrations vs depth and particle size for Clinch River core sampling. *Source*: Tennessee Valley Authority. 1985. *Instream Contaminant Study—Task 2, Sediment Characterization.* Vol. I. Office of Natural Resources and Economic Development, Knoxville, Tenn.

3.4.1.7 TVA recreation area and water intake sampling

The overall scope and purpose of this sampling are described in Sect. 3.3.1. Sediment samples were analyzed for radionuclides, metals, and organic compounds. Sediment sampling at the beach sites consisted of collecting five 12-in. cores from each site; the cores were composited for analysis. Analyses for volatile organic compounds were conducted on three cores from each beach site without compositing. Sediment sampling at the intake locations consisted of a single core sample from each site (TVA 1991a). Results of the sediment analyses are presented in Table K3.

No radionuclides other than ^{137}Cs were detected at any location during the study. Mercury levels downstream of Poplar Creek were above maximum reference concentrations (Table K3). No other metals were found at levels that exceeded upstream reference values. No organic compounds other than PAHs and phthalates were detected in any Clinch River samples; these compounds were generally present at reference sites at comparable levels.

3.4.1.8 Clinch River Remedial Investigation—Phase 1

The purpose and scope of Phase 1 of the CRRI are discussed in Sect 3.3.1. Sediment cores were collected at nine locations in the CR/PC OU (Fig. 3.28). Cores were sectioned vertically into 6-cm increments in the upper 18 cm of core and into 4-cm increments in the remaining length of core. Each core segment was analyzed for inorganic and organic contaminants and radionuclides. Cook et al. (1992) summarize the Phase 1 data. Results indicated that elevated levels of As, Be, Cd, Cr, Cu, and Hg were present in Poplar Creek and in the Clinch River immediately downstream of the mouth of Poplar Creek. Organic compounds were detected infrequently in sediment, consisting primarily of PAHs and phthalates at low concentrations. Most alpha-emitting radionuclides were present at or near background concentrations at all sites, but concentrations of ^{234}U, ^{238}U and ^{235}U were elevated in Poplar Creek. Gamma-emitting radionuclides exhibited expected spatial patterns, generally found at background levels above WOC (the exception being ^{60}Co in Melton Hill Reservoir below Braden Branch, a known source of ^{60}Co) and elevated below WOC. Peak mean concentrations of ^{137}Cs (63.64 pCi/g) were found downstream at CRM 9.5, the first significant sediment accumulation zone sampled downstream of WOC.

During the Phase 1 sampling and analysis, high concentrations of ^{137}Cs (>40,000 pCi/g dry wt) were found in surface sediment of WOC Embayment (Blaylock et al. 1993a). These sediments were typically exposed as mudflats during periods of low flow in the Clinch River. An action was immediately initiated to prevent erosion of these sediments into the Clinch River and to prevent direct exposure to sediment during periods of low flow. By the end of April 1992, a coffer dam was constructed across the mouth of WOC. This dam was designed to minimize the constant washing effect of water level fluctuations resulting from reservoir operations at Melton Hill Dam and to keep the sediment in the embayment inundated.

3.4.1.9 Near-shore surface sediment characterization

Concentrations of ^{137}Cs and ^{60}Co in near-shore surface sediments of the Clinch River, Poplar Creek, Tennessee River of WBR, and upstream reference locations were evaluated as part of the CRRI. The near-shore area, where potential human exposure to contaminants is highest, had not been an area of focus in earlier contaminant studies. Between 1991 and 1994, 926 surface sediment samples, 300 from the Clinch River arm of WBR, were collected for analysis. Results are summarized by Levine et al. (1994).

The spatial patterns of contaminant accumulation are depicted in Figs. 3.29 and 3.30. In general, mean near-shore surface sediment concentrations are greatest in reach 4. The slightly lower concentrations immediately below WOC (reach 2) are attributed to less sediment accumulation in this reach. The variability in mean concentrations observed throughout the Clinch River is thought to be a function of equally variable sediment deposition. As expected, concentrations declined sharply at the mouth of the Clinch River as a result of dilution by clean sediment from the Tennessee River. Concentrations of ^{137}Cs and ^{60}Co in the Emory and Tennessee rivers near their confluences with the Clinch River suggest some contamination caused by reverse flows in the system. Reference concentrations were at background levels, except for the expected increase in ^{60}Co downstream of Braden Branch.

3.4.1.10 Summary of historical studies and discussion of site model

Early investigations demonstrated that many contaminants of concern in the Clinch River system have an affinity for particulates. Sorption onto particles is the primary mechanism for removing such contaminants from the water phase, and sedimentation is the principal mechanism for the accumulation of these contaminants over time. Therefore, the site model recognizes that depositional patterns are a critical element in determining the current nature and extent of sediment contamination in Poplar Creek and the Clinch River.

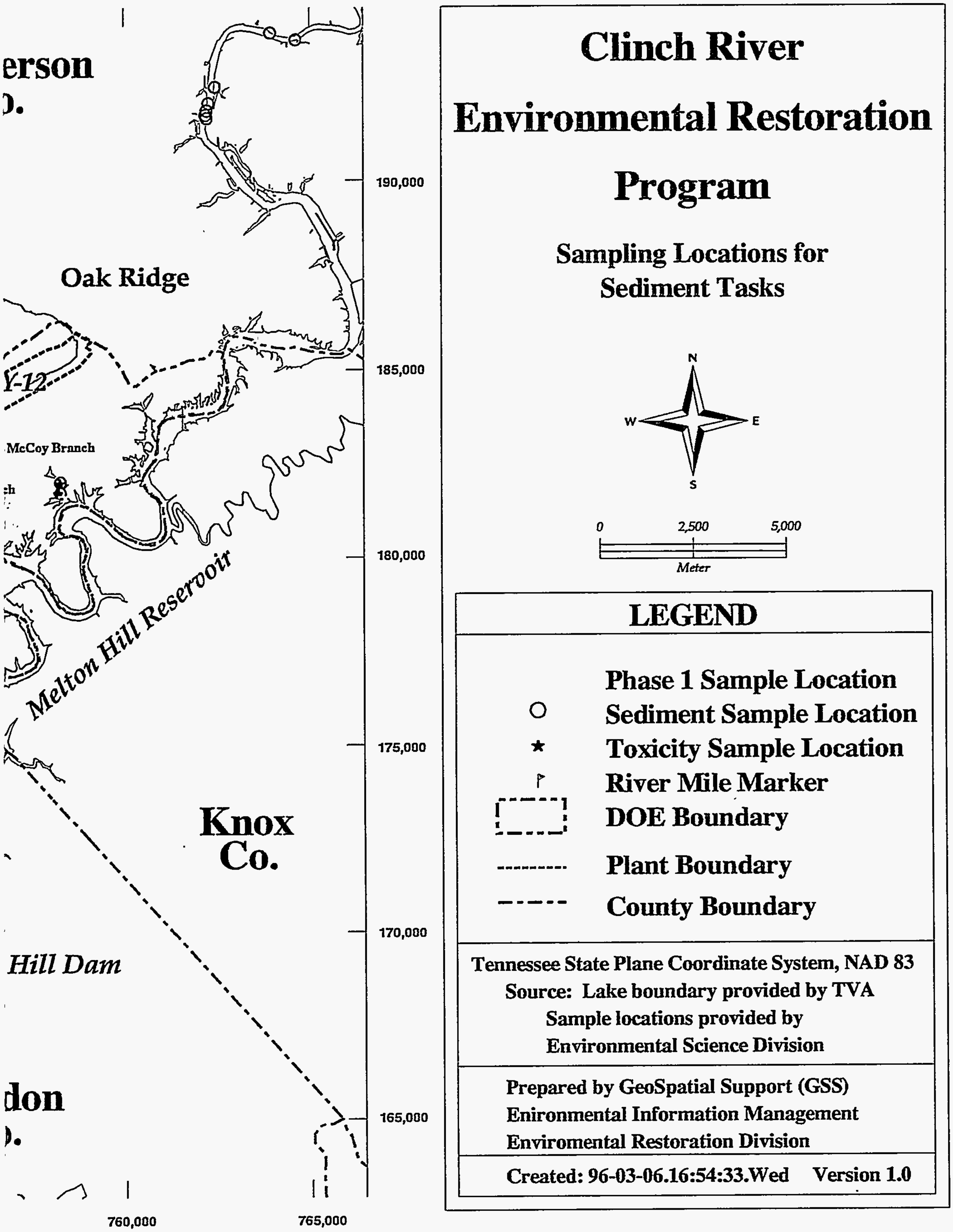
The early sediment studies focused on radionuclides, identifying several areas in the Clinch River where sediment deposition, and, therefore, sediment-associated contaminant accumulation, was greatest. Subsequent studies have demonstrated that since the period of peak releases, many of these radionuclides have decayed, been transported down river, or have been buried under layers of cleaner sediment. Therefore, the release history of sorbed contaminants also strongly influences current distribution patterns.

Mercury is known to be present at elevated concentrations in Poplar Creek and in the Clinch River below the mouth of Poplar Creek. Because of their concurrent release histories, peak mercury concentrations are coincident with peak ^{137}Cs concentrations in sediment profiles collected downstream of Poplar Creek. The distribution of other metals in Clinch River and Poplar Creek sediments is not well characterized in these studies but according to the site model concentrations, are expected to be highest in areas of sediment deposition. The areas of greatest potential for contaminant accumulation in the Clinch River appear to be depositional areas immediately downstream of Grassy Creek (CRM 14.5), downstream of Poplar Creek (between CRM 8 and CRM 11), and near CRM 4.5 and CRM 0.5. In Poplar Creek, sediment accumulation is greatest in the lower portions of the embayment, where the influence of impoundment is greatest. However, sediment is found throughout the creek within the study area.

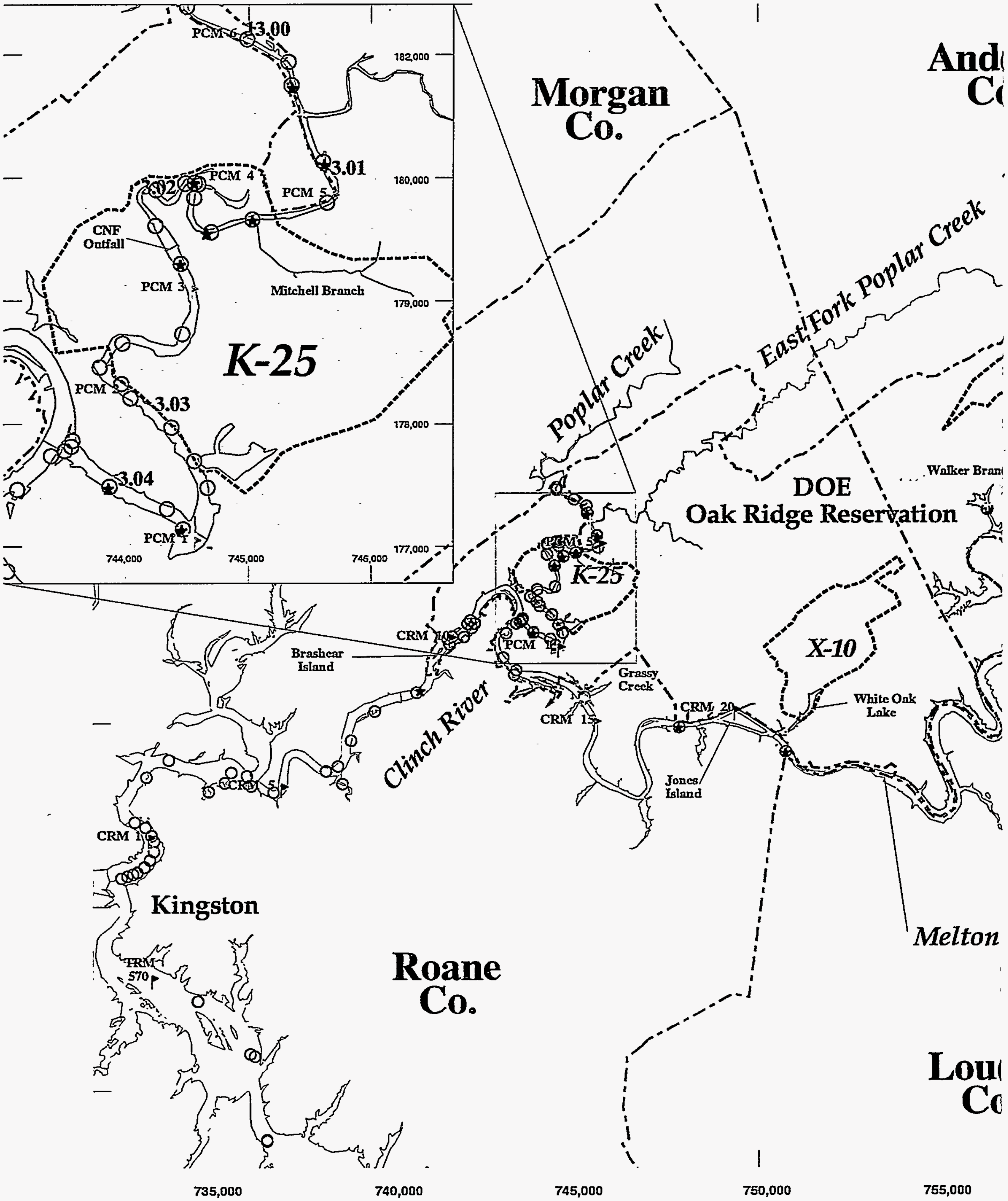
Although limited in scope, historical data on organic pollutants in sediment indicate low levels of a relatively few compounds, most of which (phthalate esters and PAHs) are ubiquitous in sediments near urban or industrial areas or downstream of them.

3.4.2 Nature and Extent of Contamination in Sediment

The spatial characterization of contaminants in sediment focused on areas of sediment deposition, primarily in Poplar Creek and in the Clinch River downstream of Melton Hill Dam. As discussed in the site model above, these areas were indicated by the release histories and historical data as the areas of greatest potential concern.



:ations for collection of Clinch River Remedial Investigation Phase 1 and Phase 2 sediment samples.



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746,000
Poplar Creek
East Fork Poplar Creek
DOE
Oak Ridge Reservation
Walker Bran
PCM 5
K-25
X-10
CRM 10
PCM 1
Brashear Island
Grassy Creek
CRM 15
CRM 20
White Oak Lake
Clinch River
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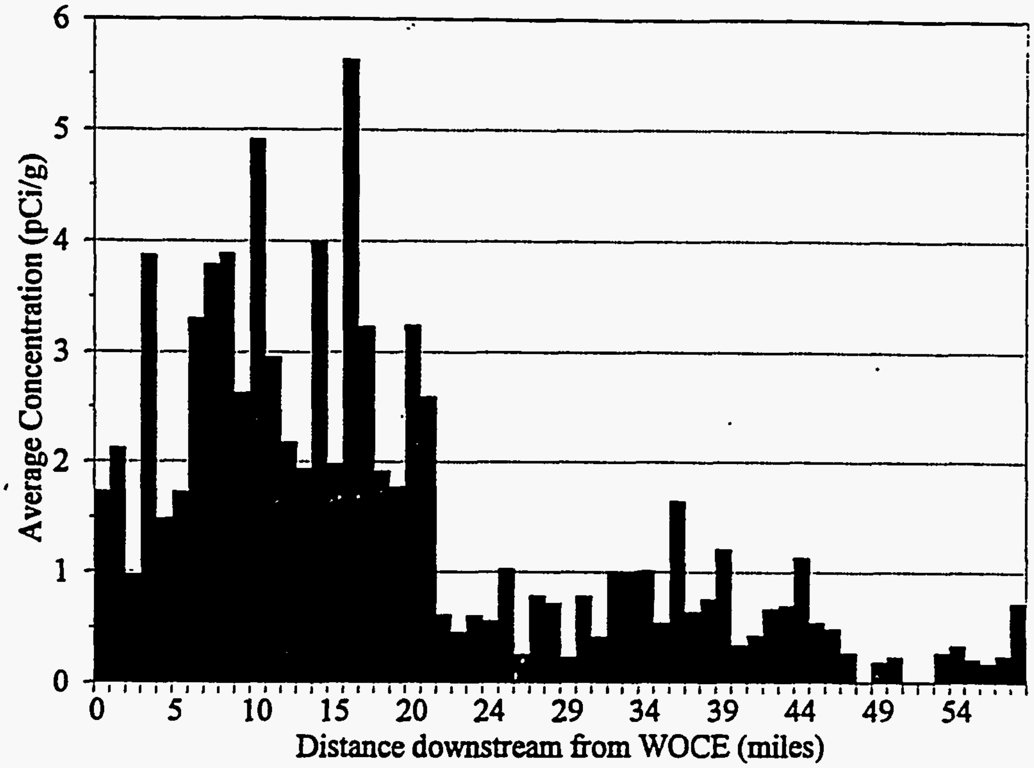


Fig. 3.29. Mean ^{137}Cs concentrations relative to distance downstream from White Oak Creek Embayment. The confluence of the Clinch and Tennessee rivers is 22 miles downstream from the mouth of Poplar Creek.

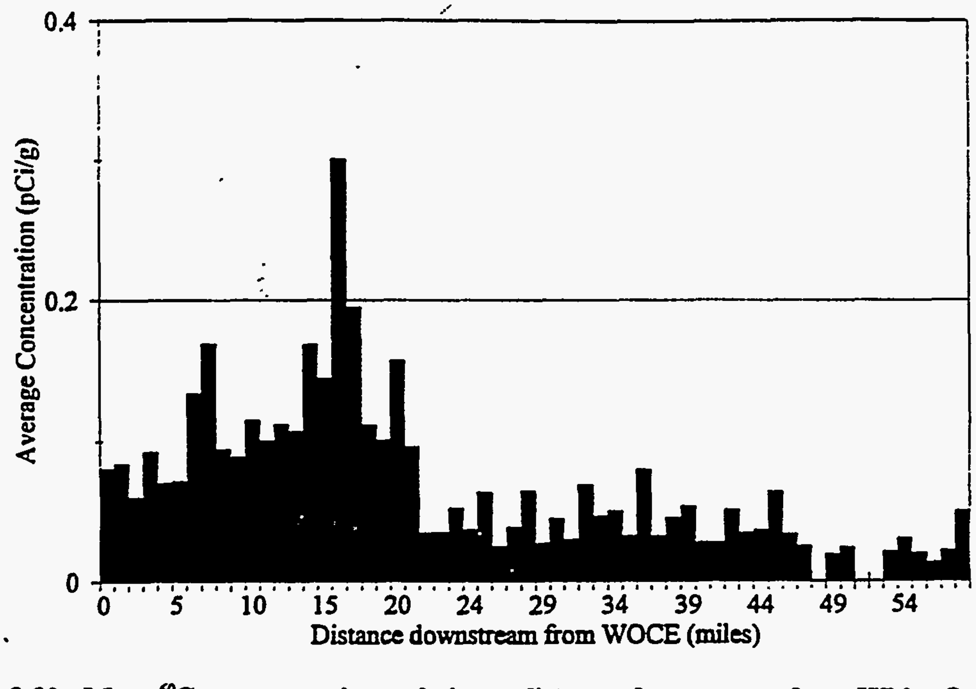


Fig. 3.30. Mean ^{60}Co concentrations relative to distance downstream from White Oak Creek Embayment. The confluence of the Clinch and Tennessee rivers is 22 miles downstream from the mouth of Poplar Creek.

For purposes of site characterization, CRRI Phase 2 data are augmented with the Phase 1 data and the near-shore sediment data of Levine et al. (1994) described previously. All of these data were combined into one data set, which is also used later in human health and ecological risk assessment (Chaps. 5 and 6).

Data were collected on contaminant concentrations in surface sediment, in sediment cores, and in sediment pore water. The following discussion attempts to identify meaningful spatial patterns of contamination in these media and to interpret these patterns both in relation to known or suspected contaminant sources and in relation to the dynamics of the river system.

Sample locations are shown in Fig. 3.28. Locations were selected on the basis of knowledge of depositional areas (Sect. 2.6) and the presence of known or suspected contaminant sources. Sample sizes for each reach and subreach are tabulated in Table C1. Analytes that were undetected in one or more of the sediment media are identified in Table C2. Data for surface sediment, core samples, and pore water are summarized in Tables C3, C4, and C5, respectively. In addition, near-shore surface sediment data are summarized in Table C6. However, the latter data are a special subset of the overall surface sediment data of Table C3 and are used in human health risk assessment only; they are not discussed here.

In addition to contaminant data, data on several noncontaminant-related sediment parameters important in characterizing ecological health were collected for use in the ecological baseline risk assessment (Chap. 6). Whole sediment parameters measured included particle size, total organic carbon content (TOC), pH, and total Kjeldahl nitrogen (TKN). Pore water analysis included TOC, TKN, ammonia nitrogen, and total hardness. As expected, most surface sediment consisted of fine material, although spatial variability was high. Results for the other parameters are illustrated in Fig. 3.31.

To identify trends in contamination, mean contaminant concentrations in a study reach or subreach were compared with those from the appropriate reference reach and upstream study reaches. This discussion focuses on contaminants that are elevated above reference values in at least two of the three sediment media sampled (surface sediment, deep sediment, and pore water). For this analyses, "elevated" means the concentrations that are statistically elevated (at the 0.05 level) as measured by one-way analysis of variance (ANOVA) and subsequent pairwise comparisons. In addition, those concentrations that are elevated by a factor of two or more in comparison with reference or upstream values, even though not statistically significant, are considered elevated for the purpose of this discussion.

In this discussion, elevated surface contaminant values are considered evidence of recent deposition and, therefore, some upstream source of contamination. Elevated contaminant concentrations in core samples generally indicate areas of stable sediment deposition and (where surface sediment values are low) are considered indicative of historical contamination. Sediment pore water values are influenced by complex physical and chemical factors in the sediment, pore water, and surface water; the solubility of the analyte itself; and the amount of contaminant present in surface sediment available for release to pore water. Pore water concentrations often do not correlate with surface sediment values from the same location.

Mean concentrations of contaminants in surface sediment, core samples, and pore water, by subreach, are illustrated in Figs. 3.32–3.34 (inorganics), Figs. 3.35–3.36 (organics), and Figs. 3.37–3.39 (radionuclides).

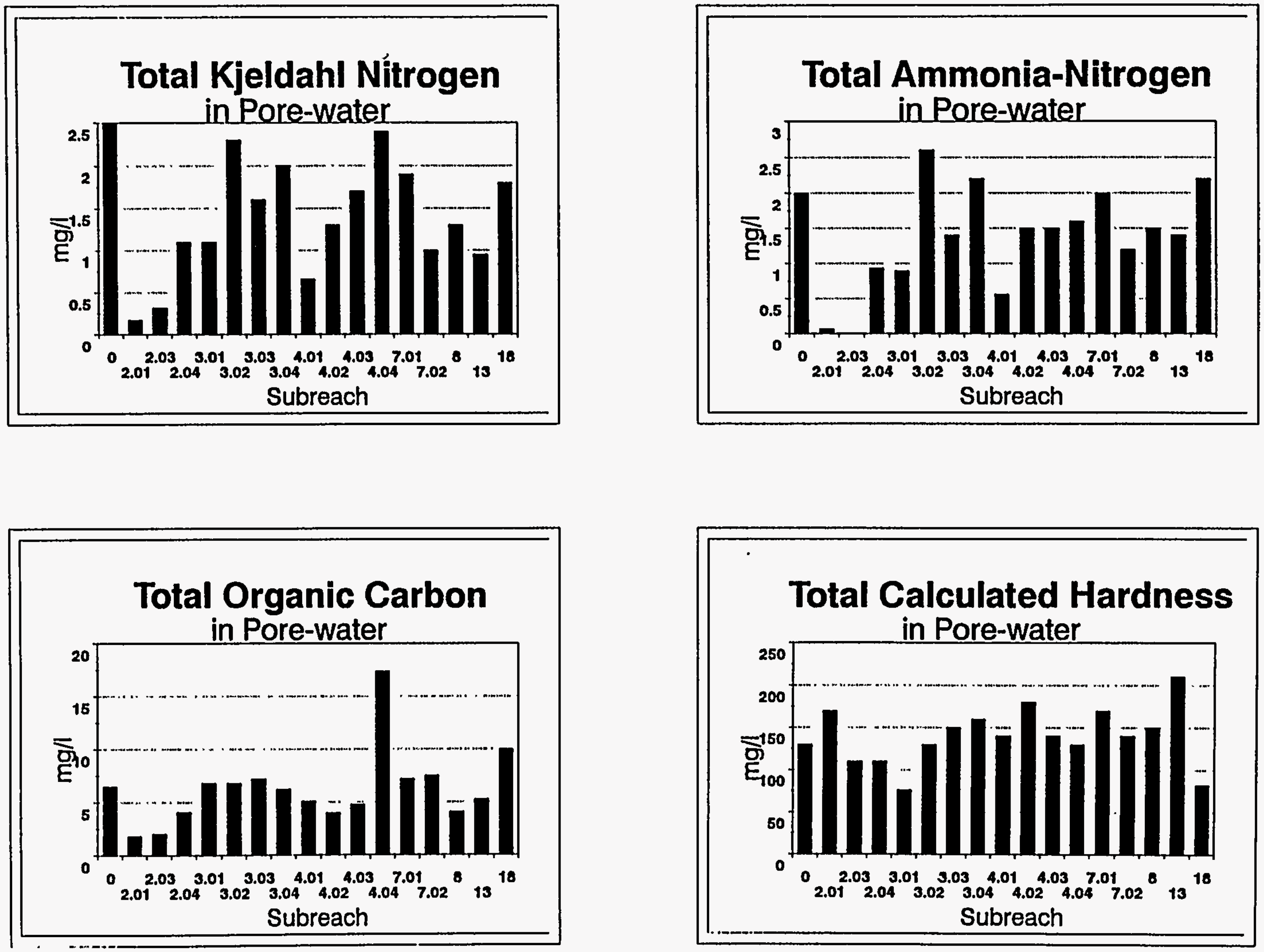
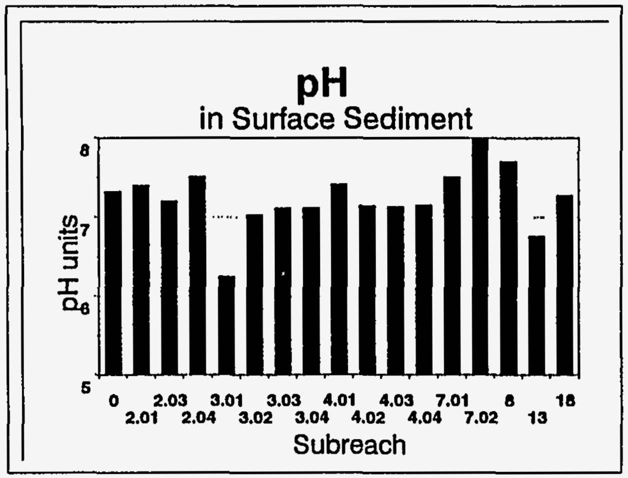
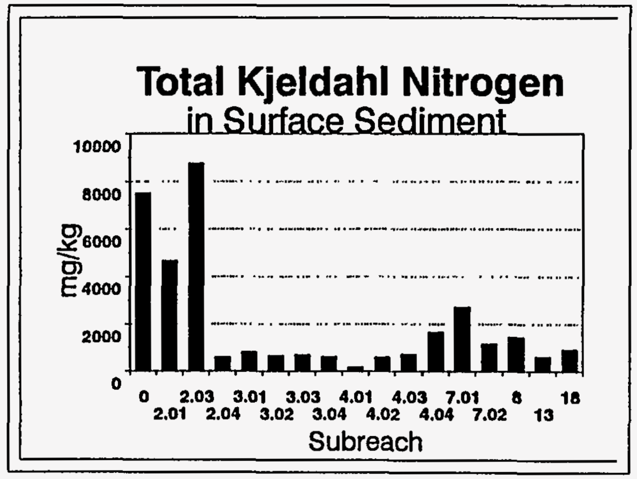
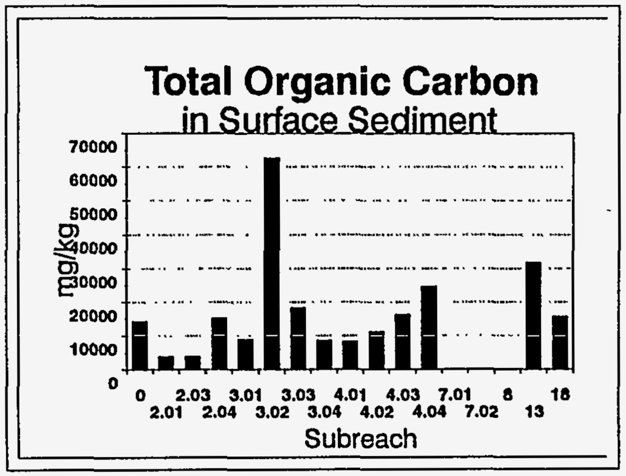


Fig. 3.31. Mean concentrations of selected noncontaminant parameters relevant to the ecological health of benthic communities.







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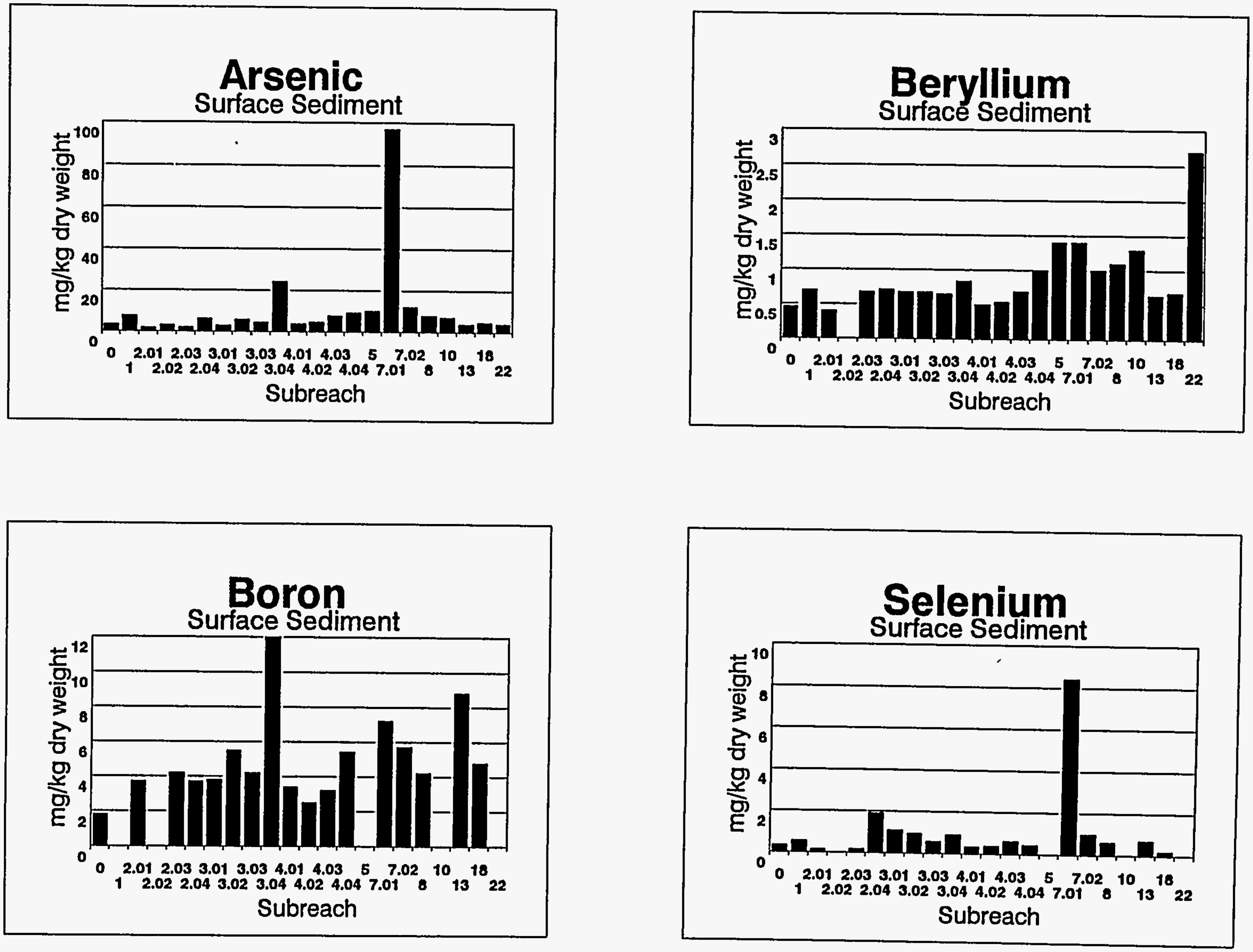
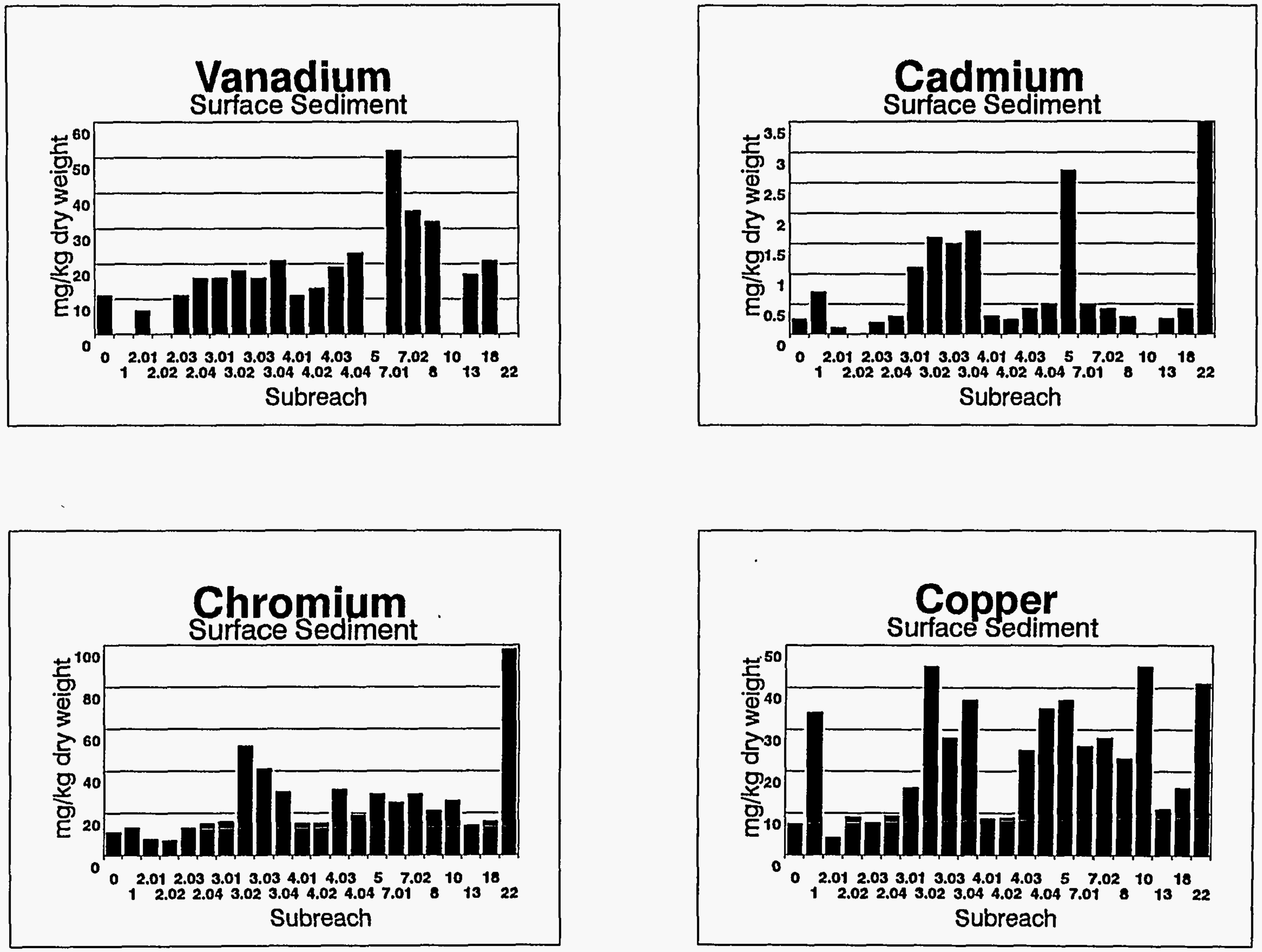


Fig. 3.32. Surface sediment mean concentrations of inorganic contaminants, by subreach.



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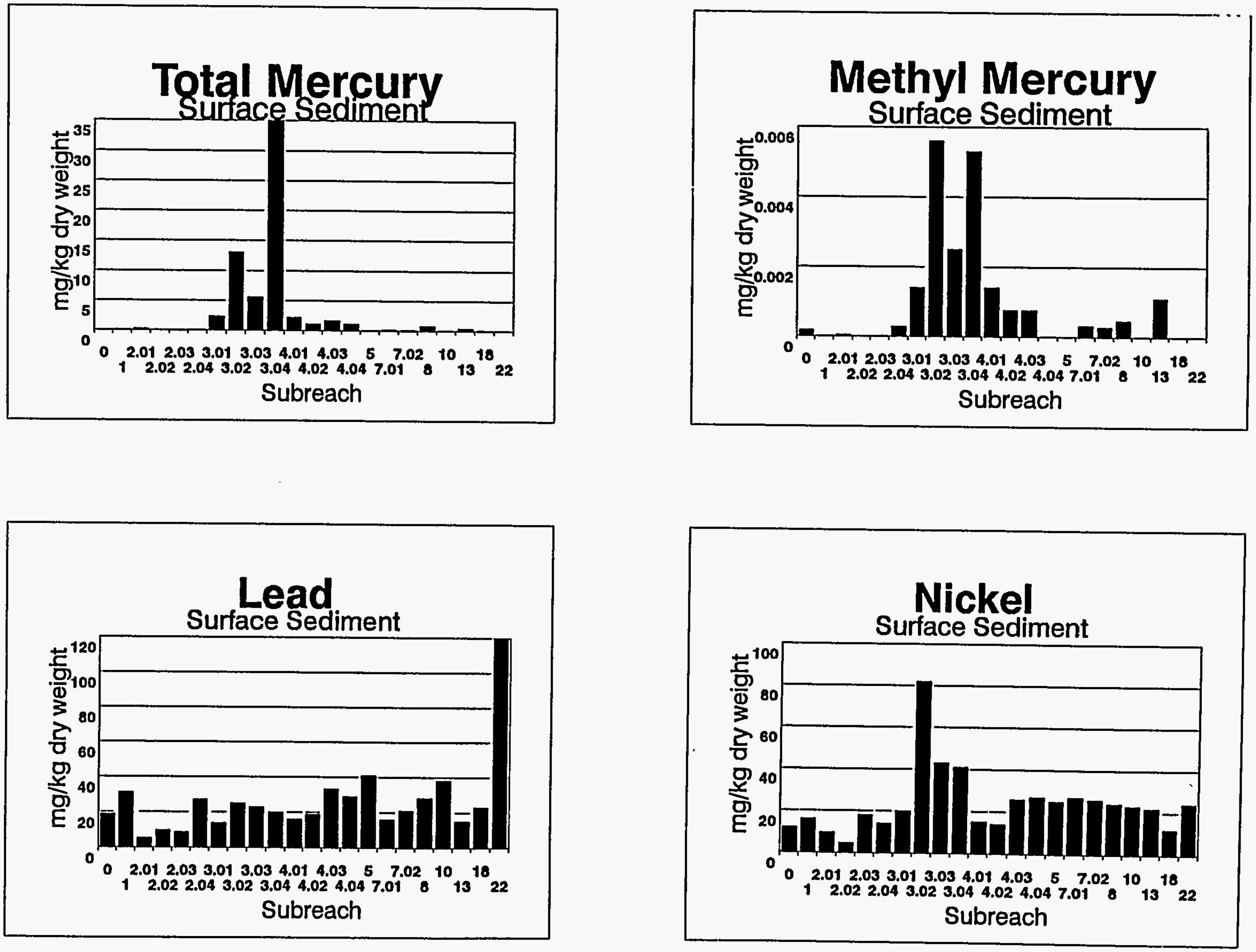

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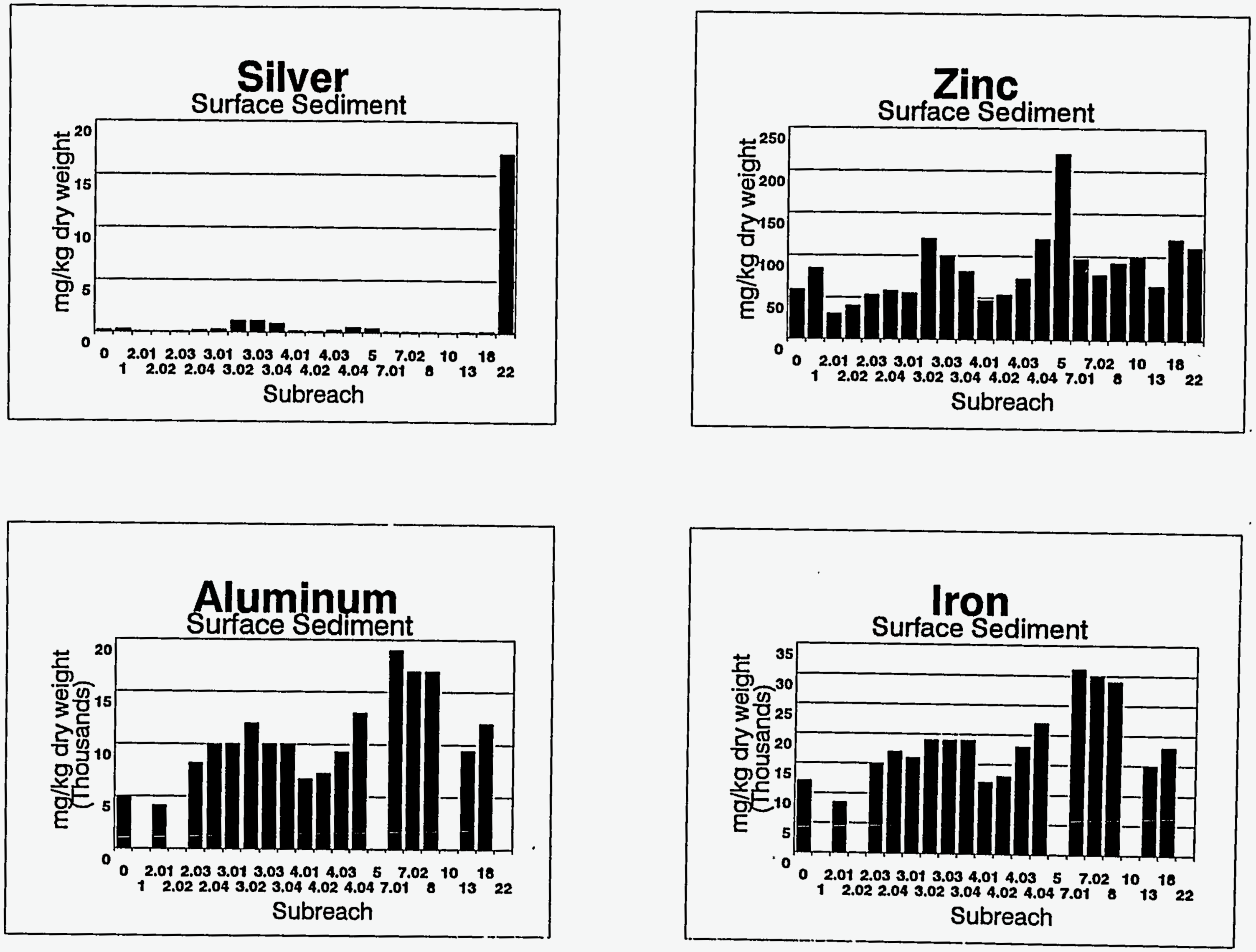

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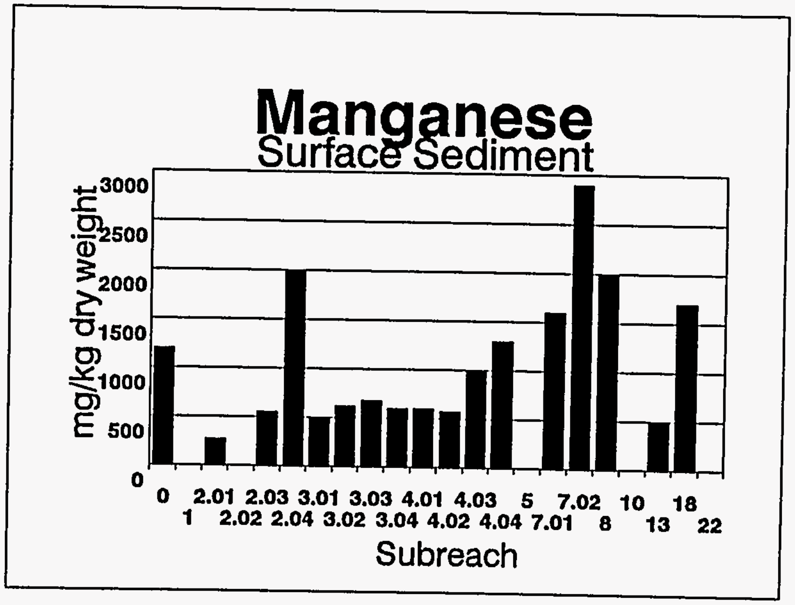

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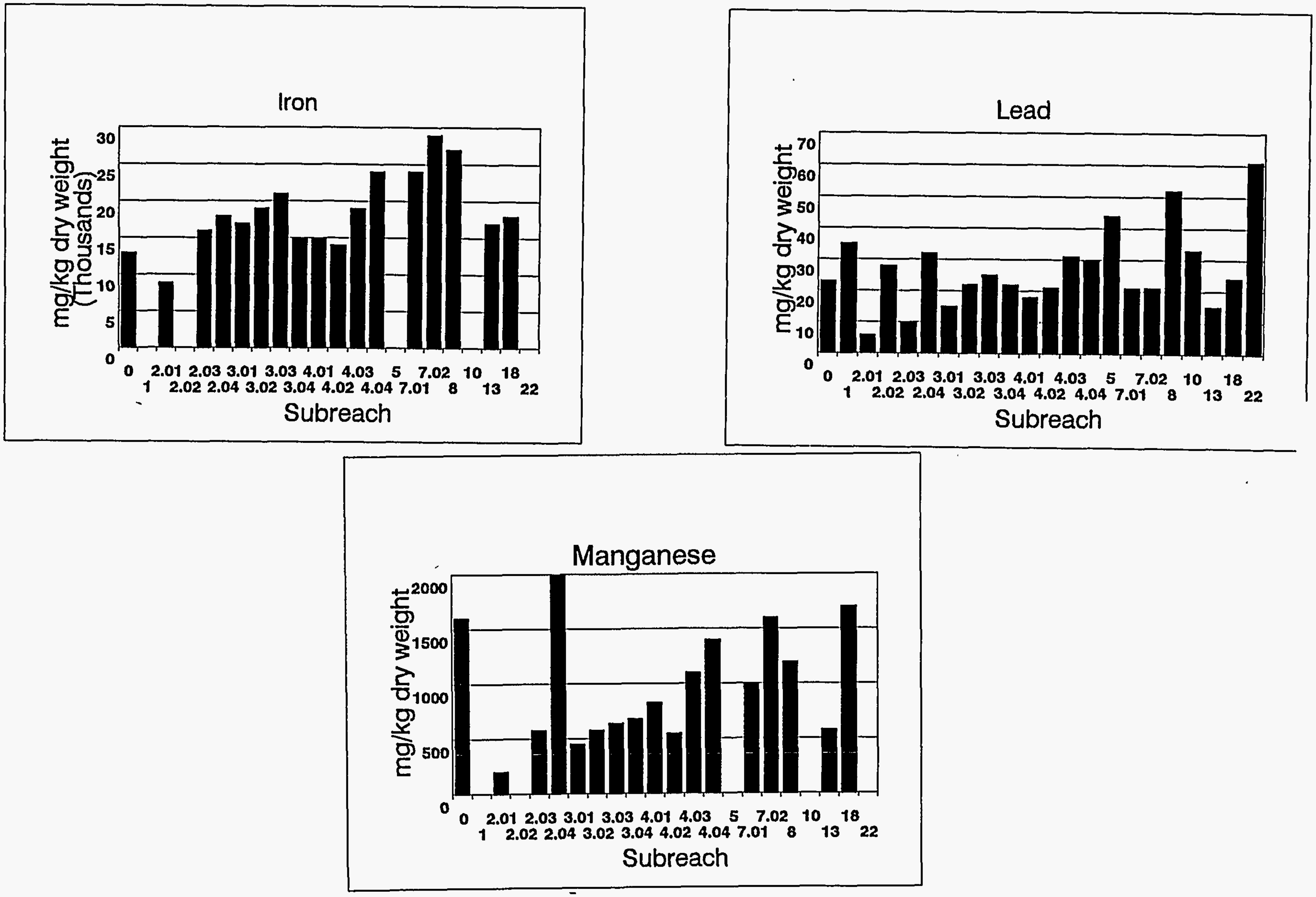
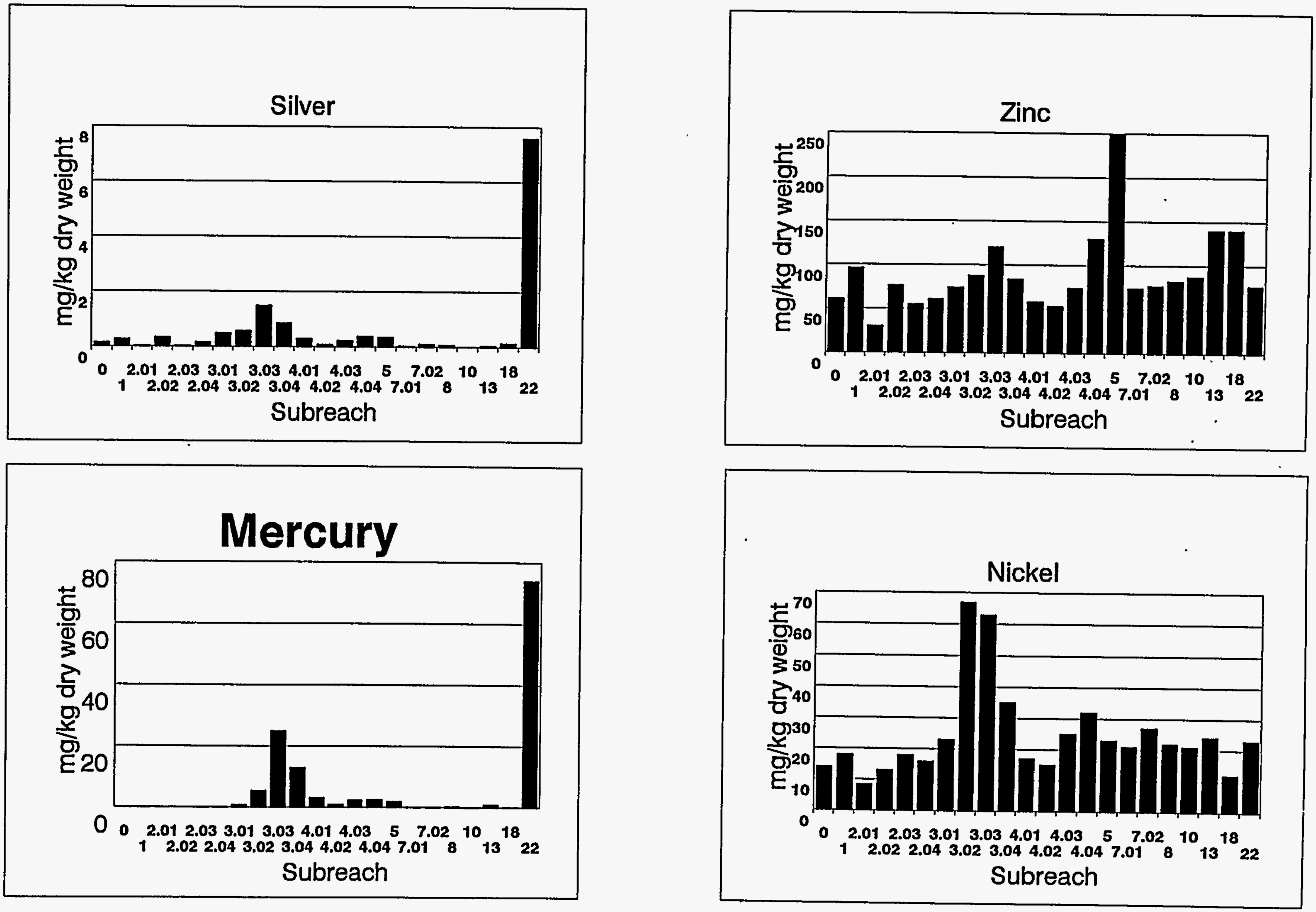


Fig. 3.33. Core-weighted mean concentrations of inorganic contaminants, by subreach.



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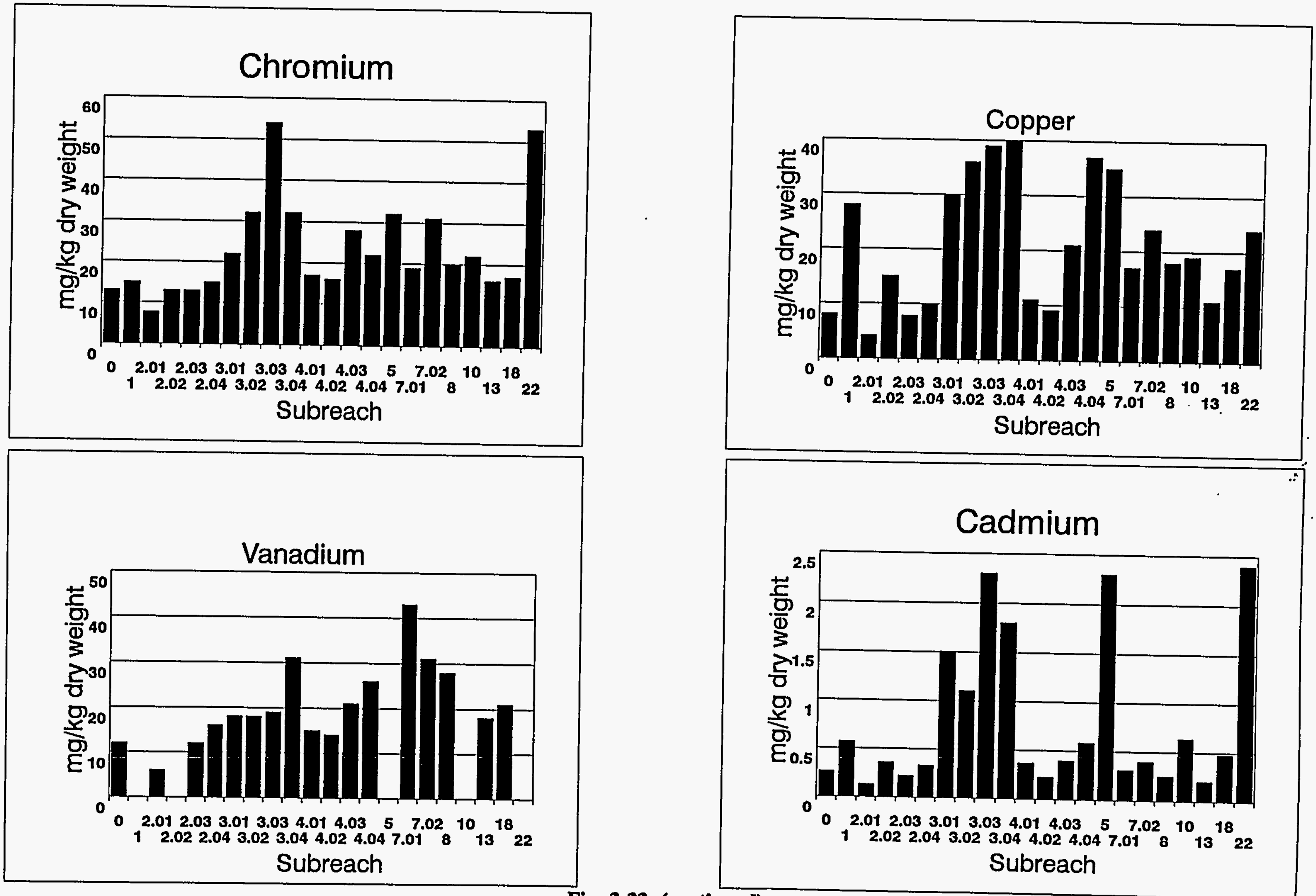

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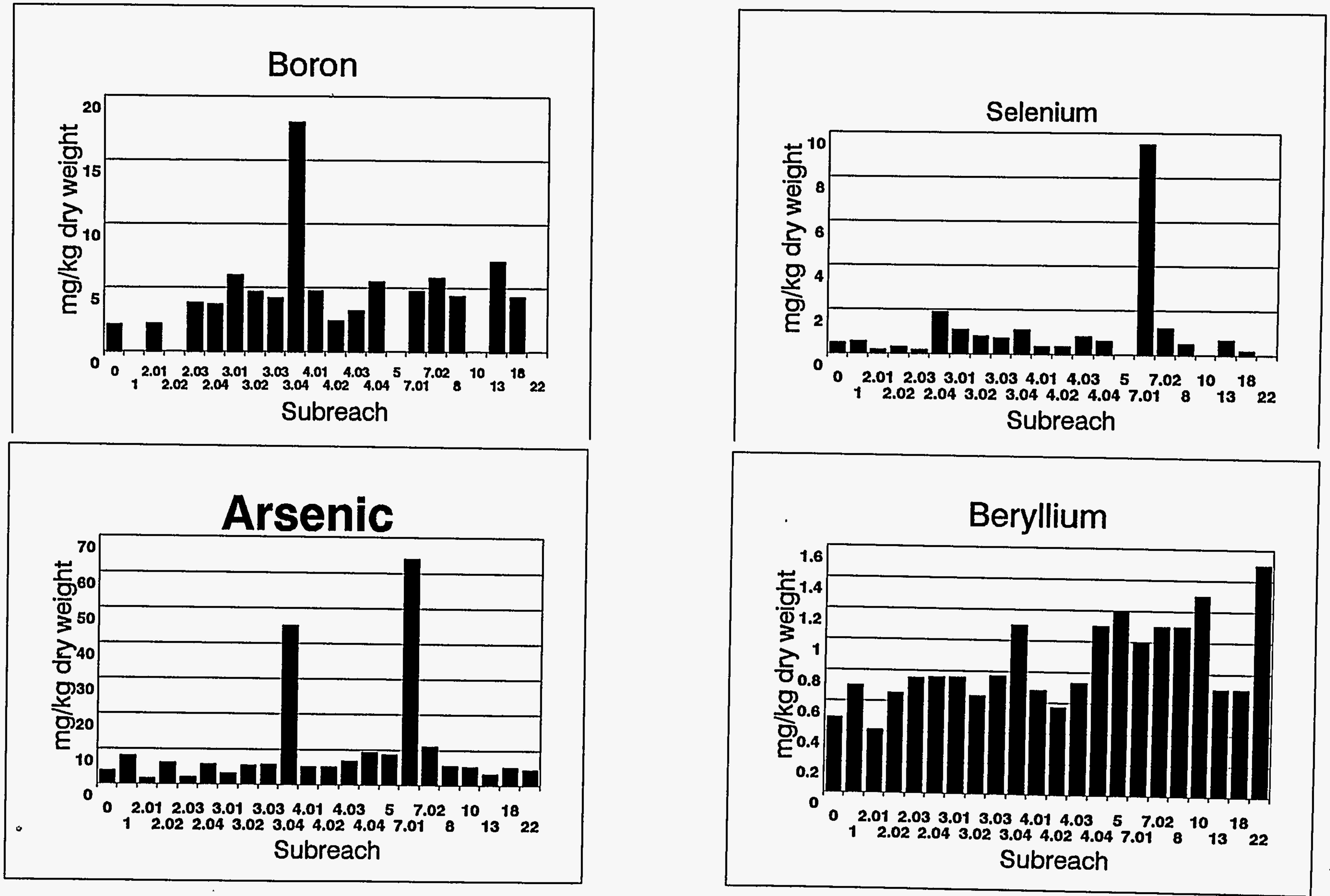

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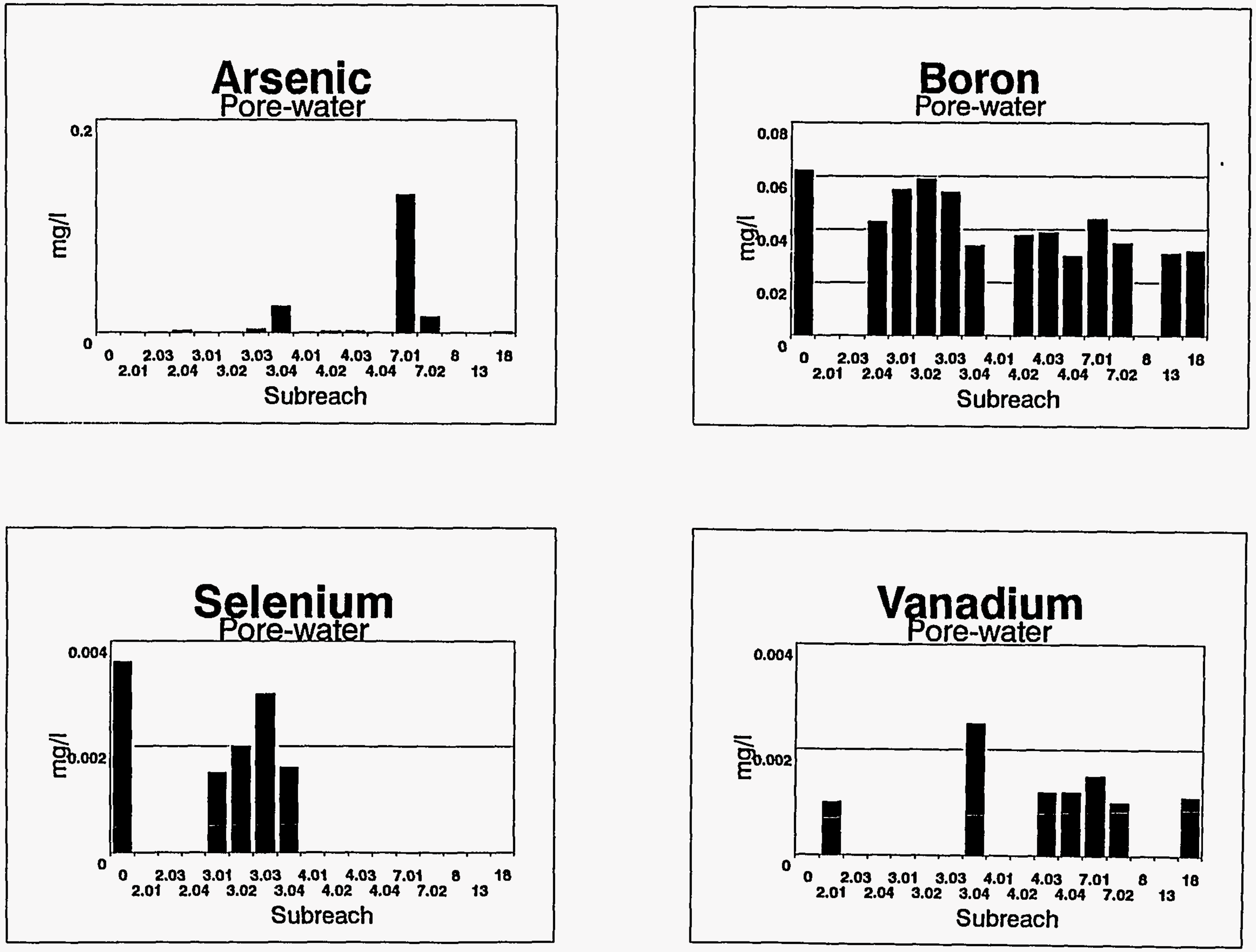
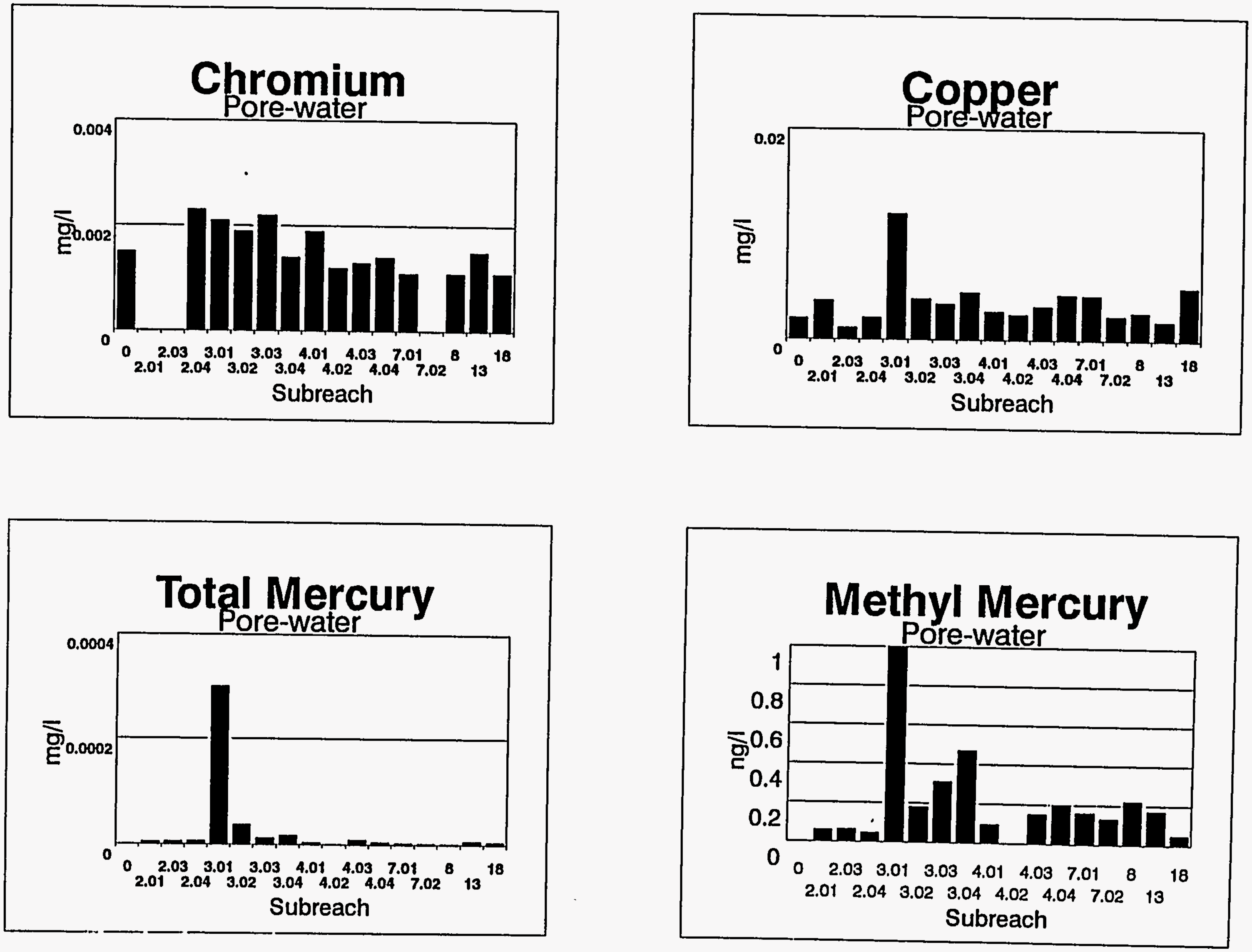


Figure 3.34. Mean pore water concentrations of inorganic contaminants, by subreach.



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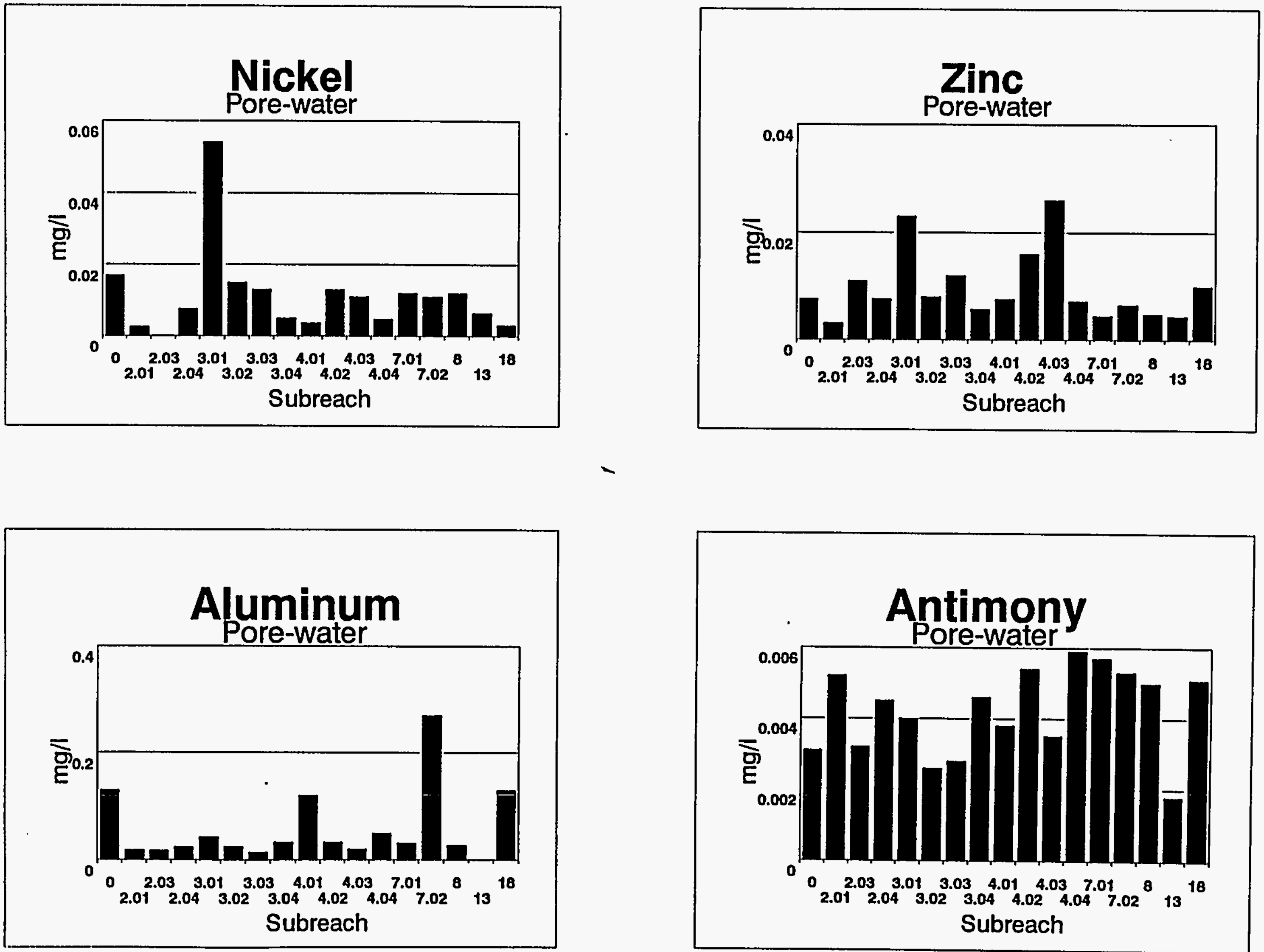

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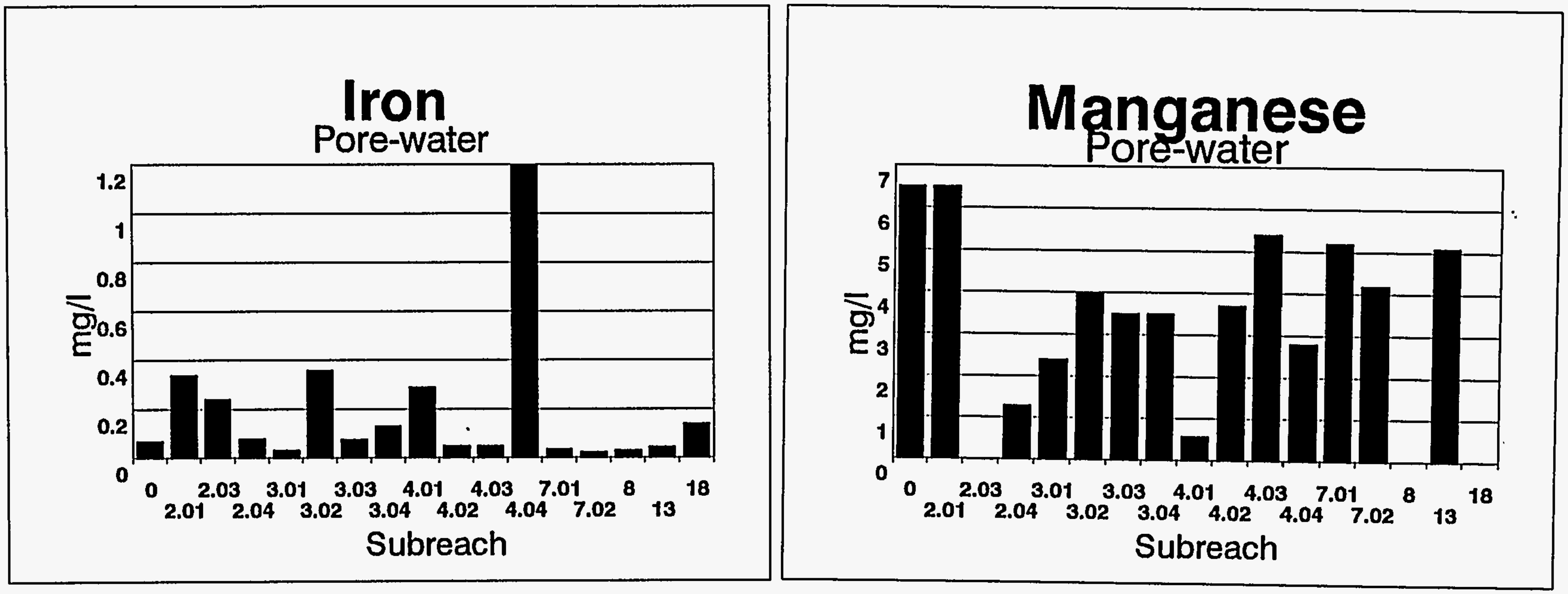

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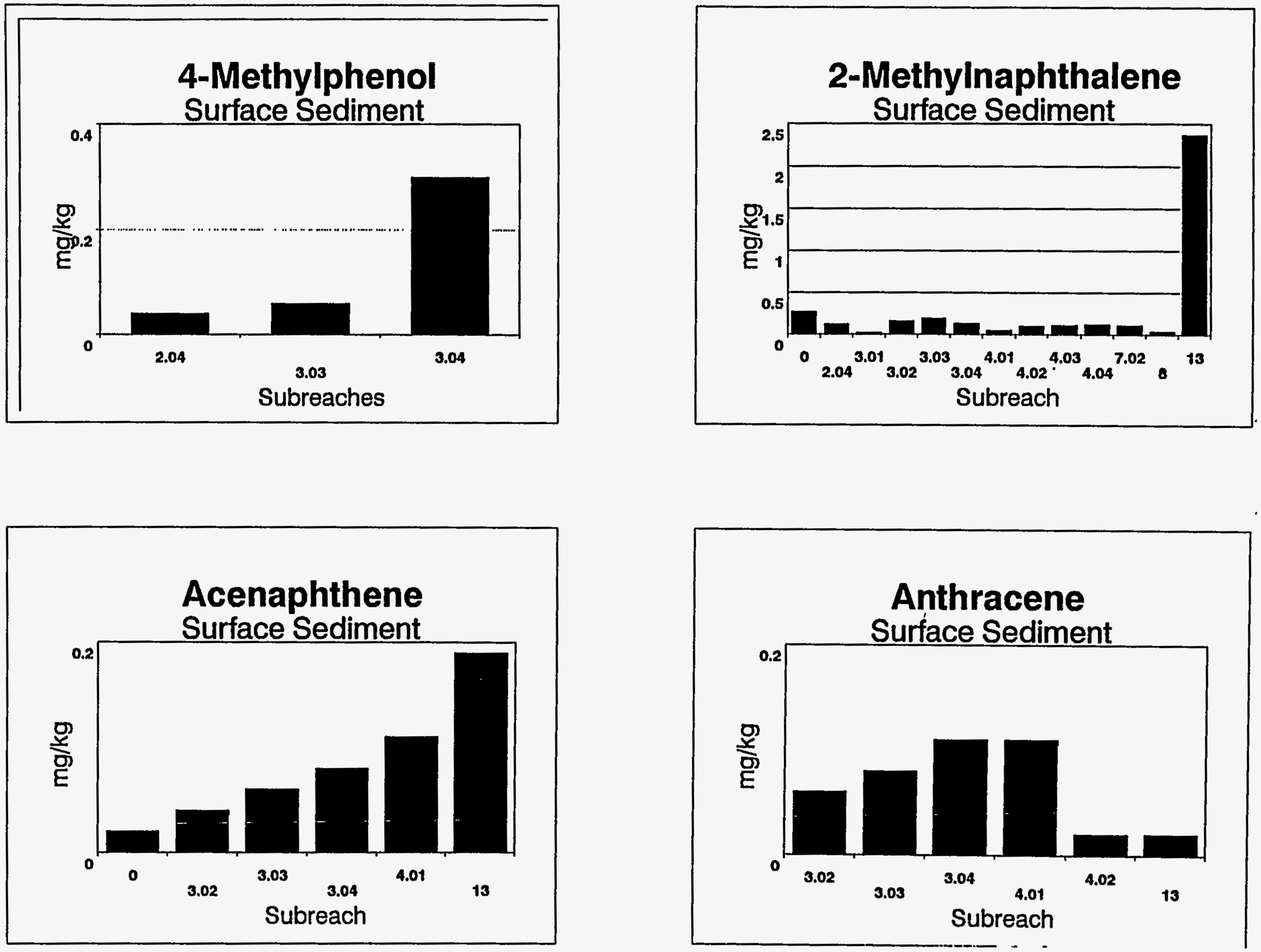
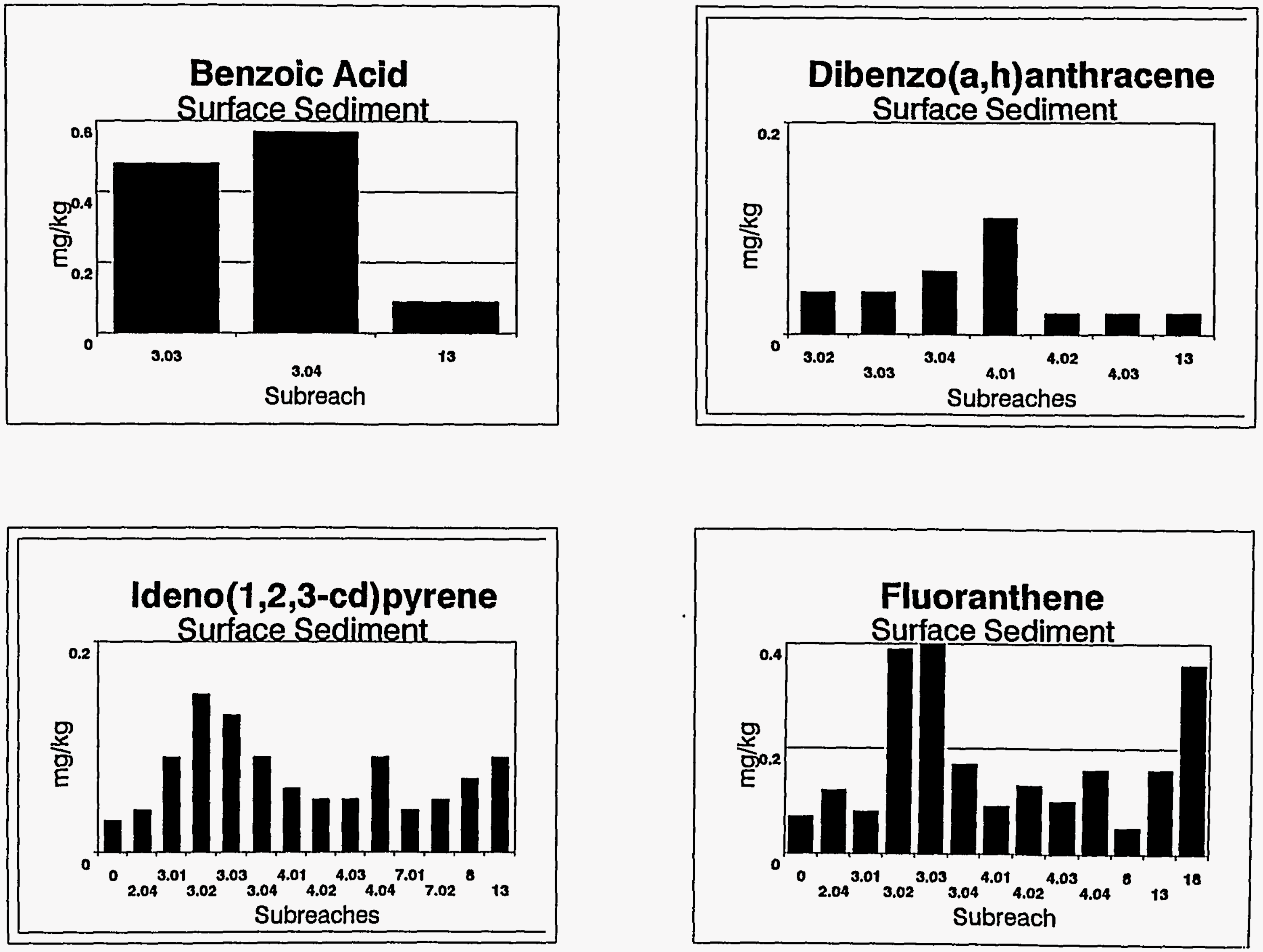


Fig. 3.35. Surface sediment mean concentrations of organic contaminants, by subreach.



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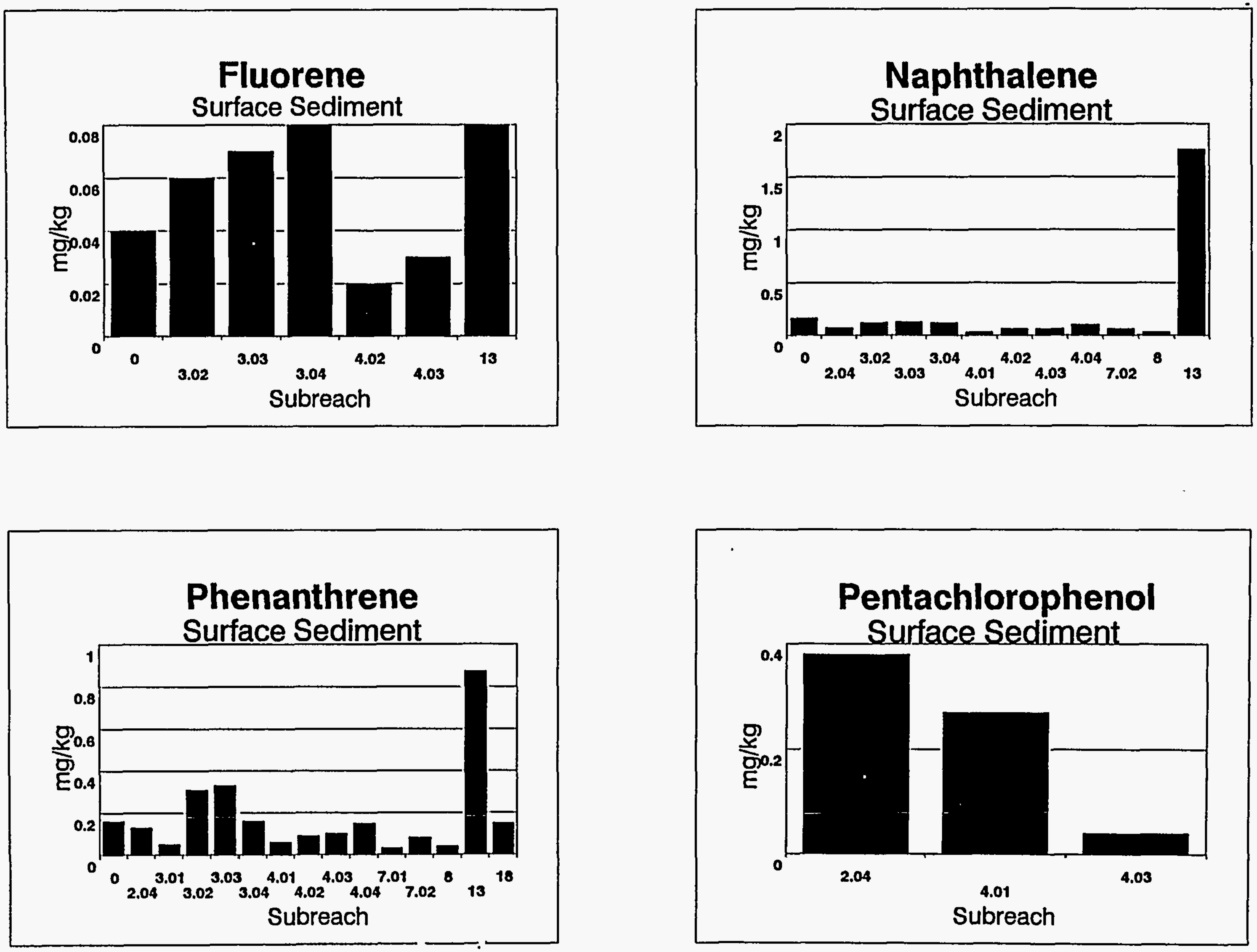

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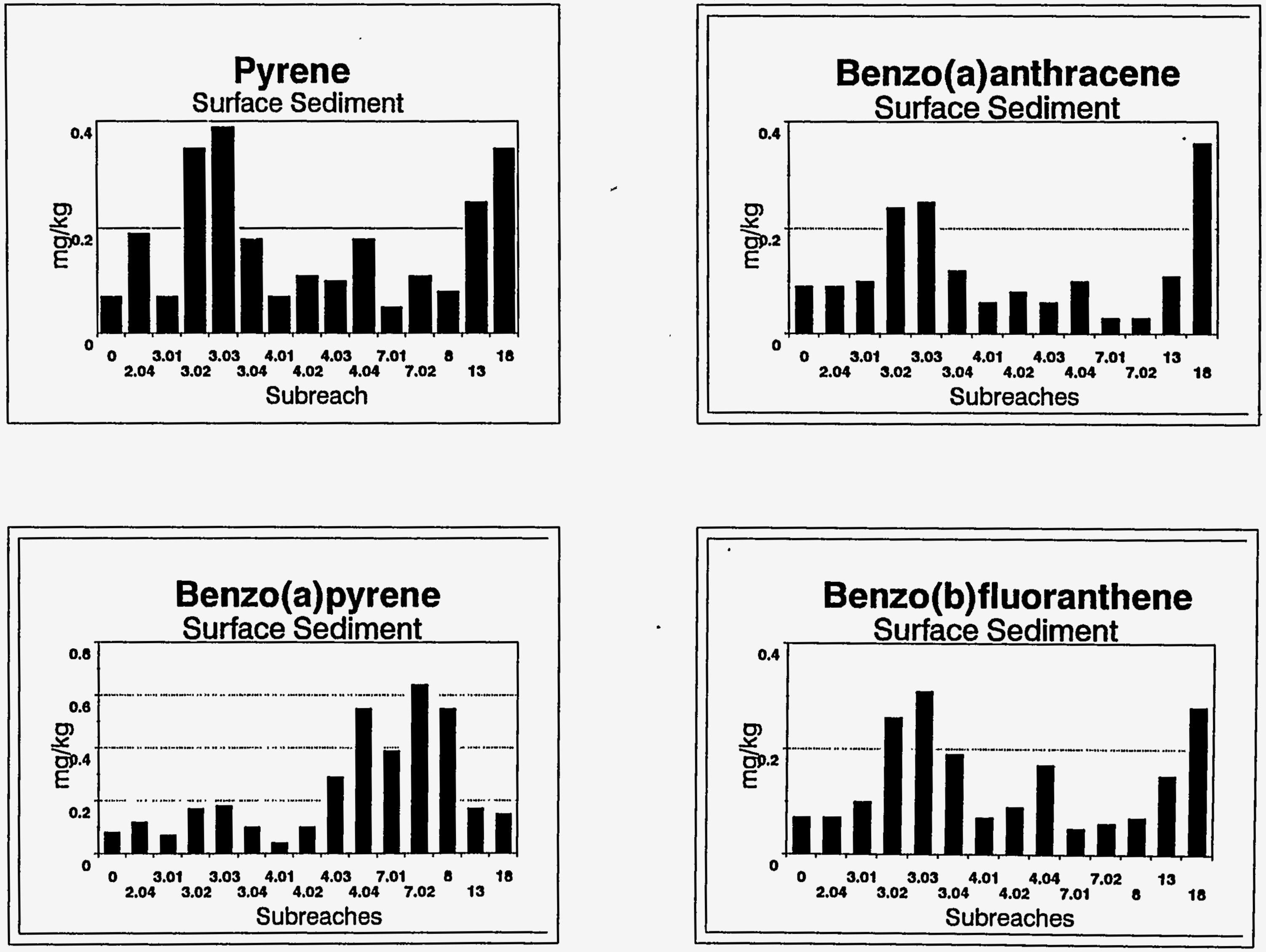

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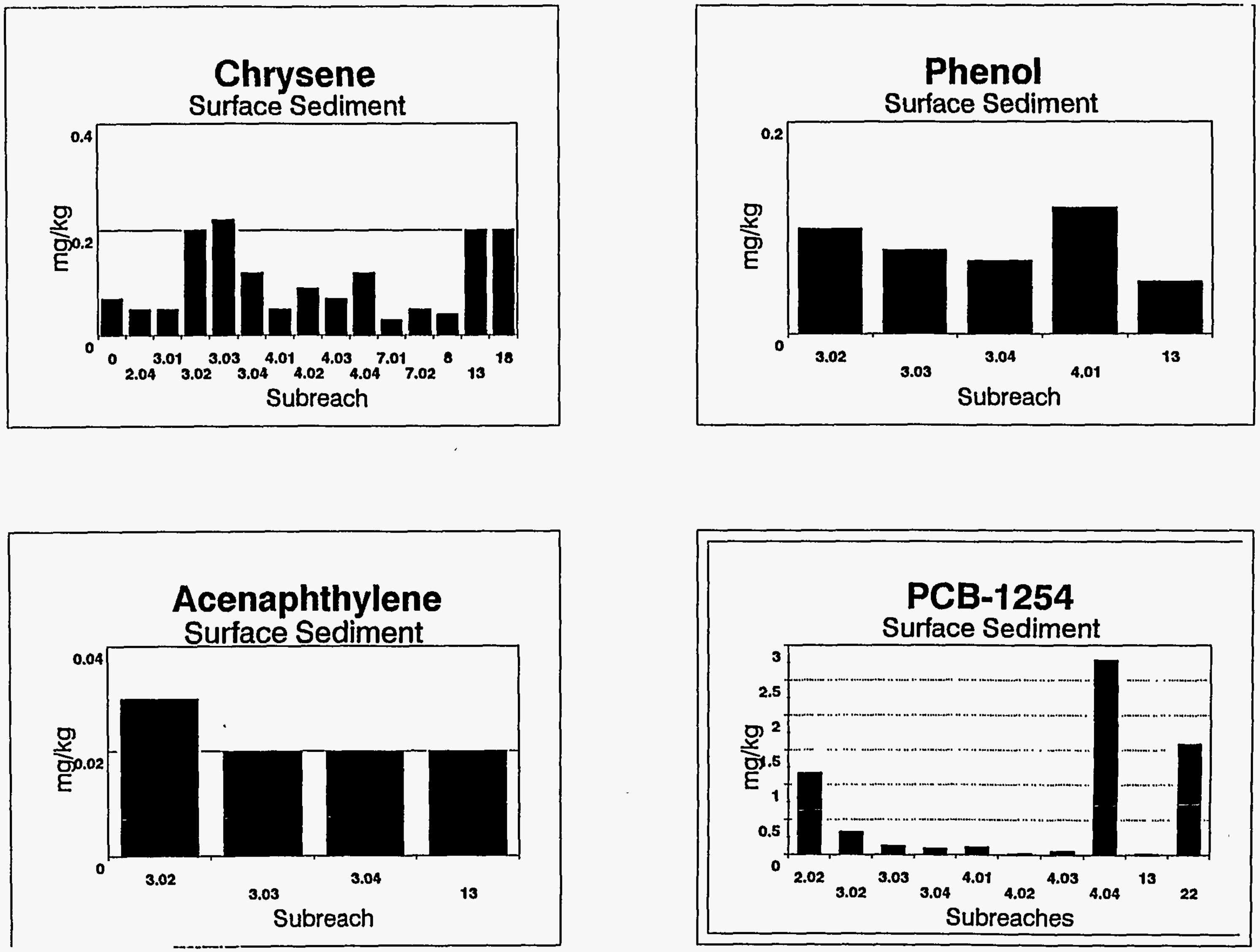

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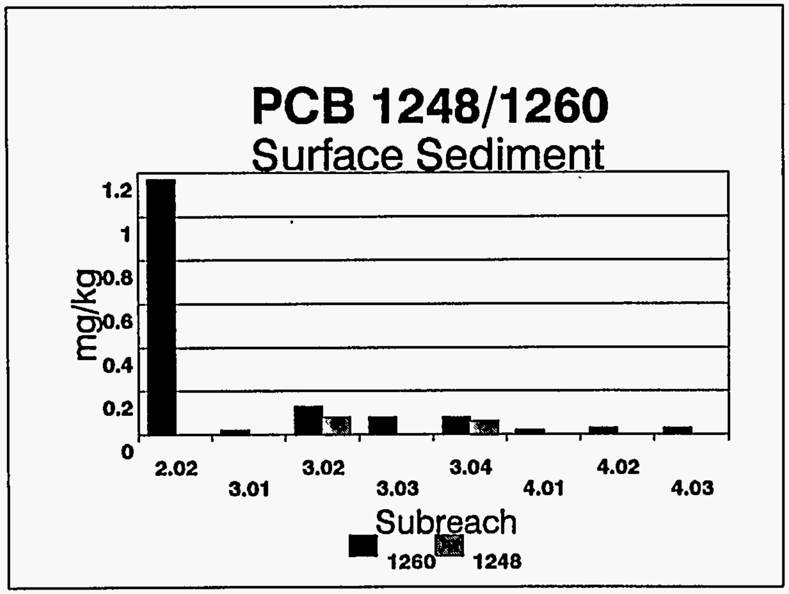

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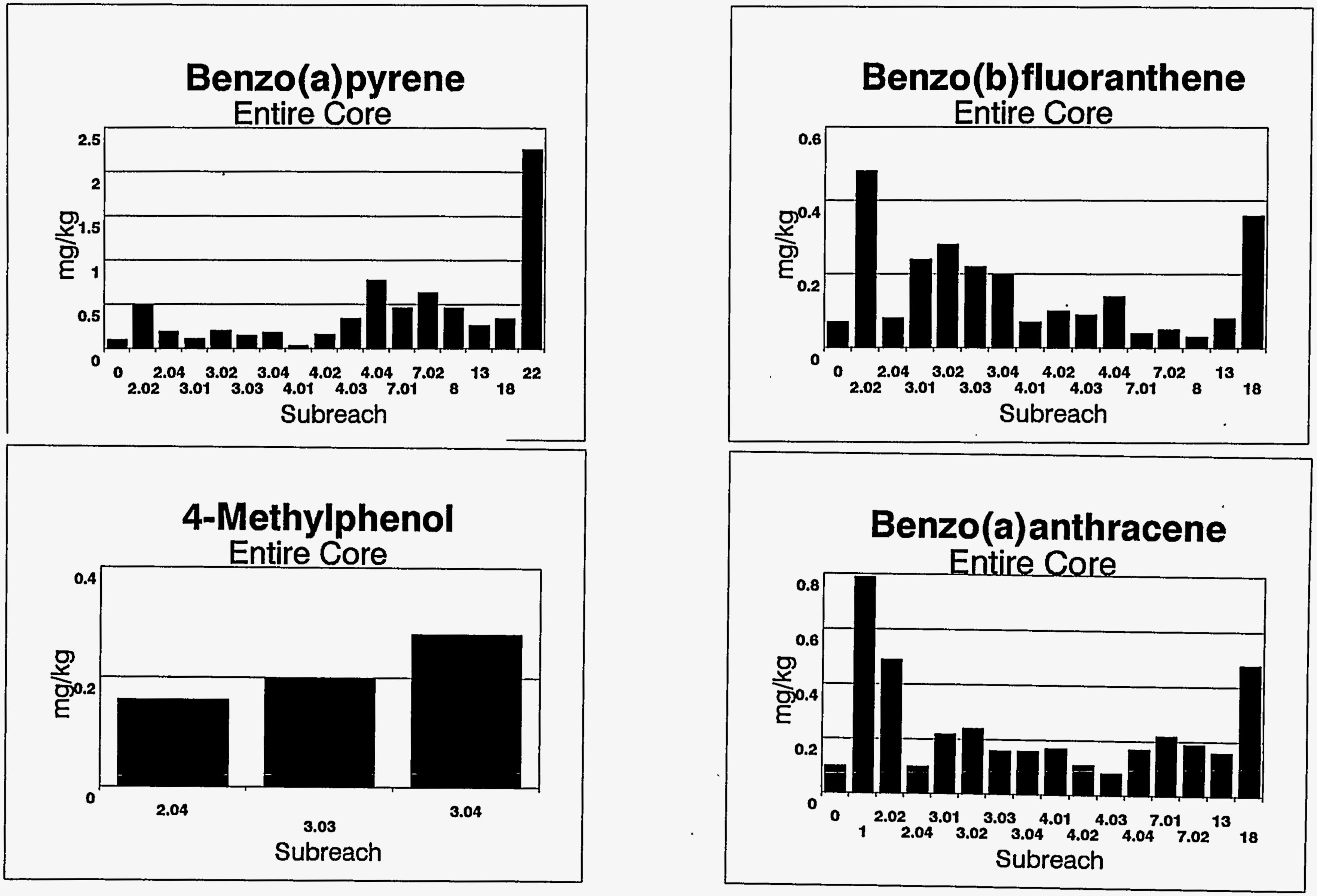
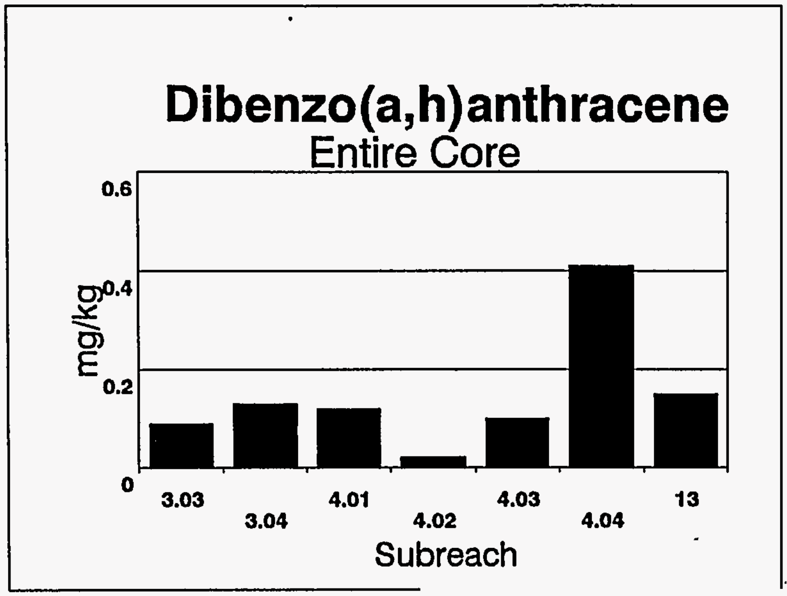
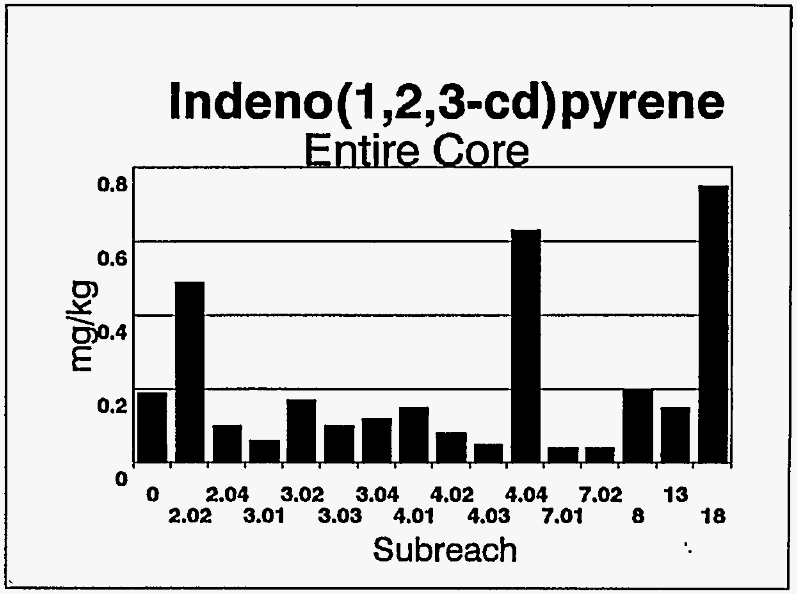
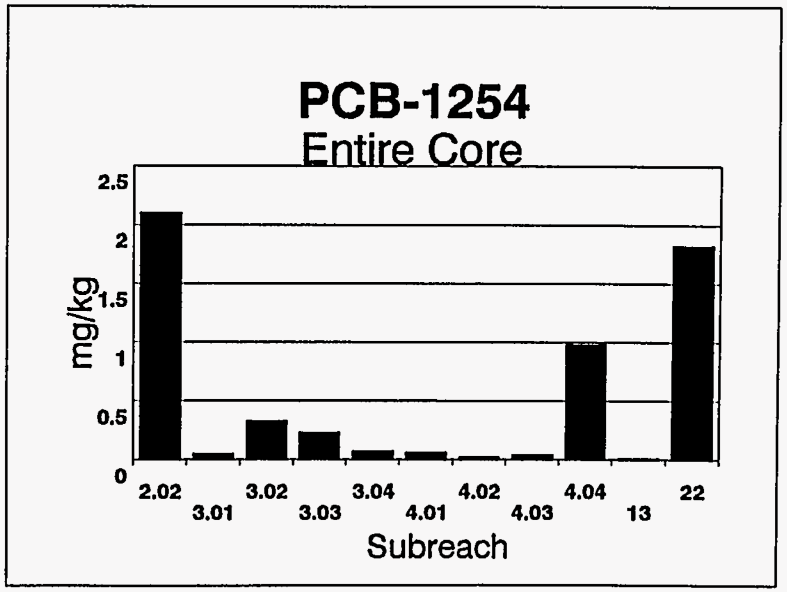


Fig. 3.36. Core-weighted mean concentrations of organic contaminants, by subreach.







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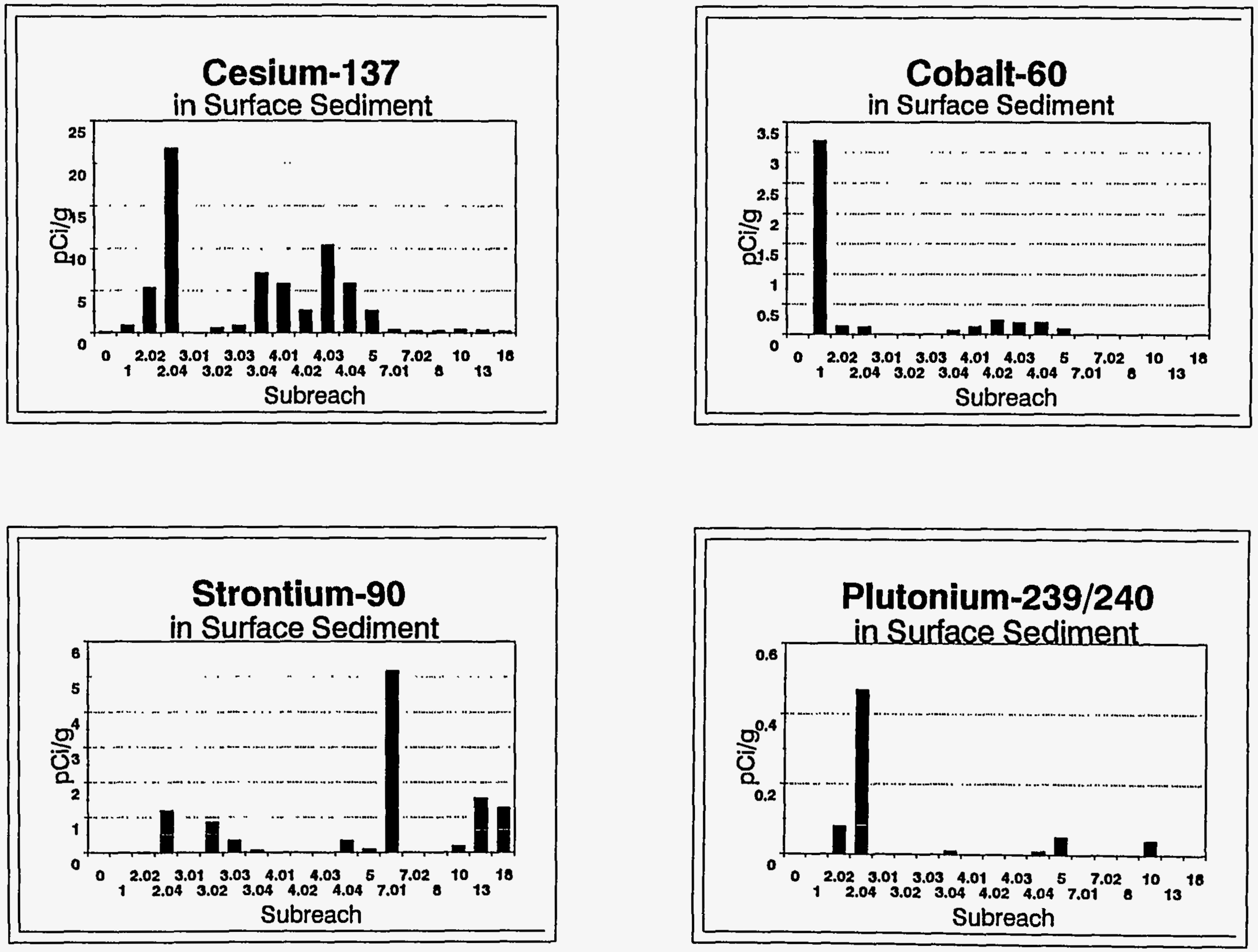
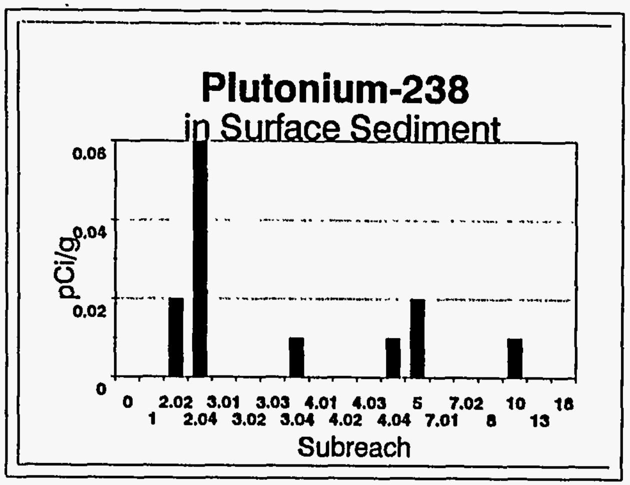
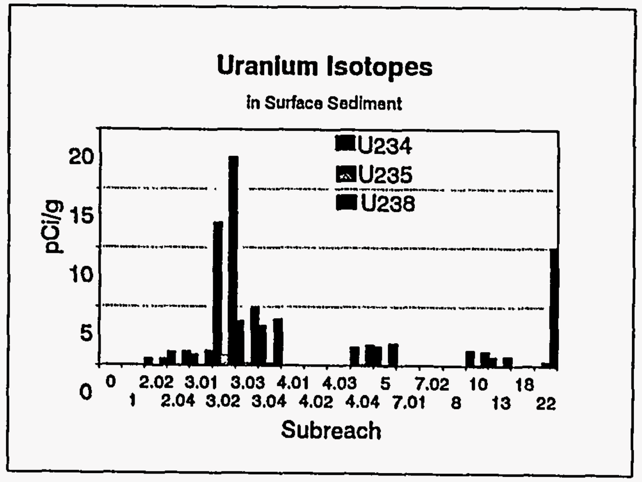


Fig. 3.37. Mean concentrations of radionuclides in surface sediment, by subreach.





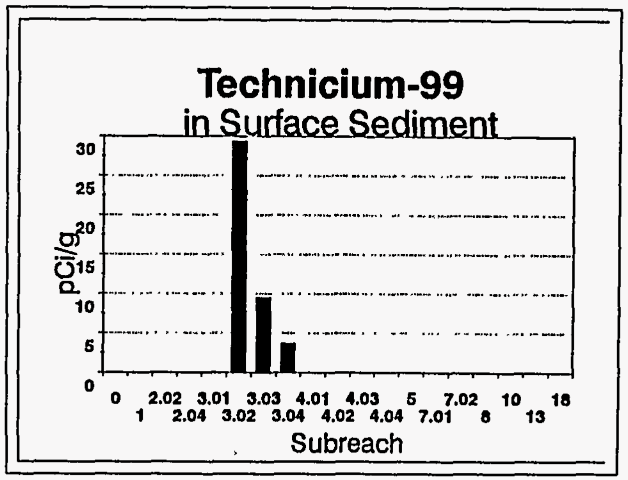


Fig. 3.37 (continued)

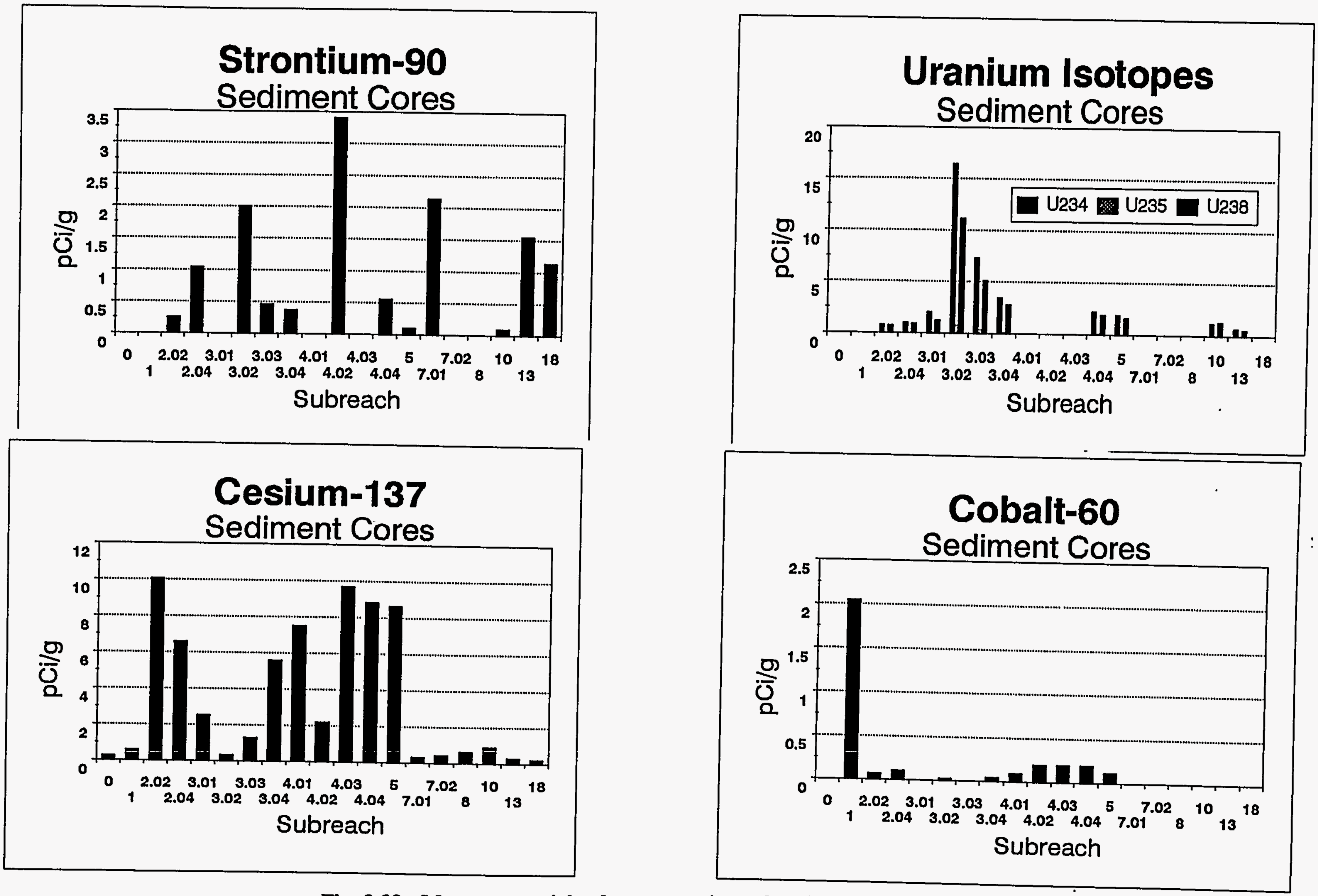


Fig. 3.38. Mean core-weighted concentrations of radionuclides, by subreach.

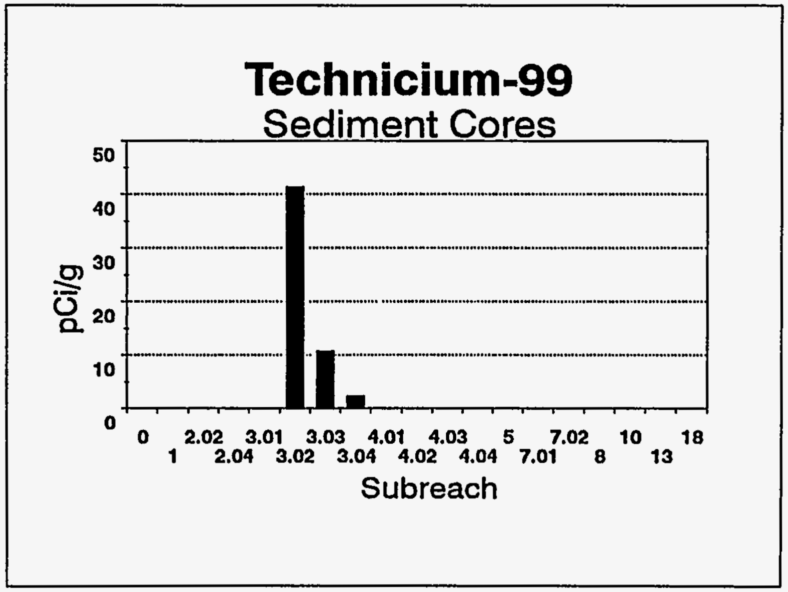
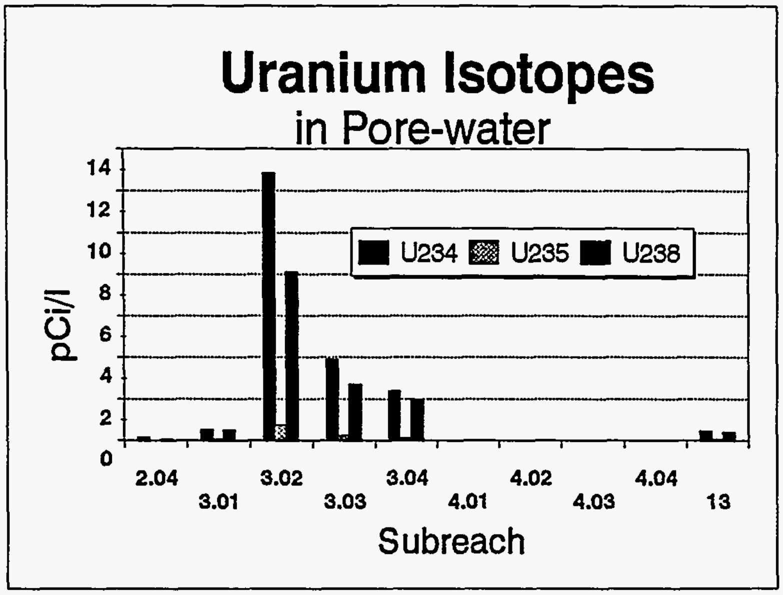


Fig. 3.38 (continued)



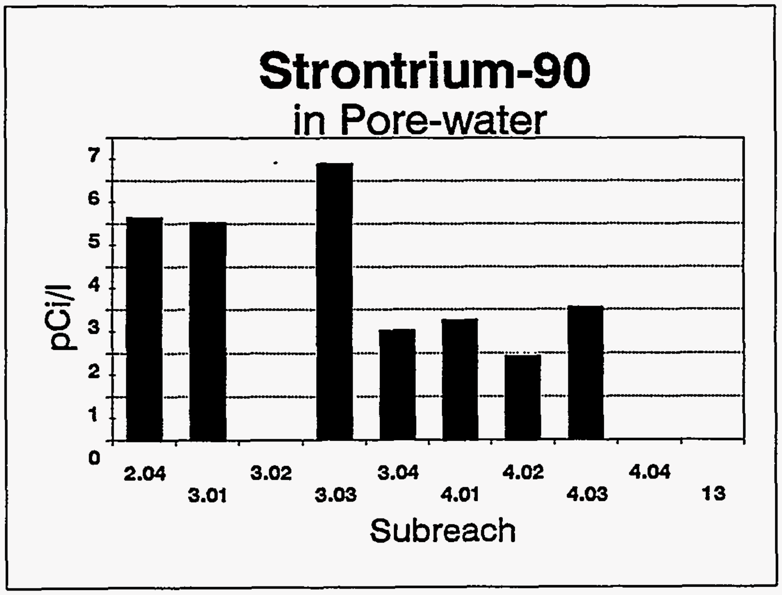


Fig. 3.39. Mean concentrations of radionuclides in pore water, by subreach.

3.4.2.1 Contaminant distributions in Poplar Creek

Subreach 3.01—below EFPC. Interpretation of data in this subreach is limited by sample size ($n = 2$), but mean concentrations of Hg, Cd, Cr, and Cu are elevated above reference values (reach 13) in at least two sediment media downstream of EFPC (Figs. 3.32–3.34). As noted in Sect. 3.1, EFPC is a known source of mercury to Poplar Creek. Ashwood et al. (1986) also identified EFPC as a potential source of each of the metals listed. However, none of these analytes exhibits its peak core or surface sediment concentrations in subreach 3.01, probably because of a relative lack of long-term, large-scale sediment deposition in this subreach. Peak pore water concentrations of mercury, methyl mercury, and copper (as well as several other metals) were observed in this subreach (Fig. 3.34).

Subreach 3.02—below Mitchell Branch. Mean concentrations of Hg, Cd, Cu, and Cr remain elevated and are generally greater than concentrations in subreach 3.01, probably partly a result of increased sediment deposition (Figs. 3.32–3.34). However, concentrations of chromium and copper in surface sediment peak here, are significantly elevated in comparison with concentrations in subreach 3.01, and they likely indicate that Mitchell Branch is a significant source of these analytes. Analytes that are significantly elevated in two or more sediment media for the first time are Ni, Ag, several isotopes of U, ^{99}Tc, and benzo(b)fluoranthene, indicating sources of these contaminants in Mitchell Branch as well. Aroclor-1254 is also elevated in this subreach for the first time, but the samples in which this PCB mixture was found appear to be associated with a source other than Mitchell Branch. Each of the analytes listed, except benzo(b)fluoranthene and silver, exhibits peak core and surface sediment values in this subreach; silver exhibits peak mean surface sediment concentrations here. The sediment of Mitchell Branch has previously been identified as containing elevated concentrations of Ni (Fig. 3.40), Cr, Cu, Ag, Zn, U isotopes (Fig. 3.41), and ^{99}Tc (Ashwood et al. 1986).

Subreach 3.03—below CNF. Mean concentrations of each of these analytes remain elevated in at least two sediment media in subreach 3.03. Peak values of Hg, Ag, Cd, and Cr in core samples occur here (Fig. 3.33). Concentrations of nickel, copper, and benzo-(b)fluoranthene remain elevated in core samples (Fig. 3.33) but drop significantly in surface sediments (Fig. 3.34). Levels of Aroclor-1254, isotopes of U, and ^{99}Tc decline in both core and surface sediment samples. The organic compound 4-methylphenol is elevated in both surface and core sediment samples for the first time in this reach.

Subreach 3.04—below fly ash disposal area. Mean concentrations of Hg, Ag, Cr, Cu, Cd, 4-methylphenol, Aroclor-1254, U, and ^{99}Tc remain elevated in both surface and core sediment samples in relation to reference values; nickel and copper remain elevated in core samples. Most of these analytes, however, have decreased from peak upstream concentrations. Notable exceptions are total and methyl mercury; peak surface sediment concentrations are found in this subreach. Peak surface sediment and core values for As, B, and V occur here (Figs. 3.32 and 3.33) and are significantly elevated in both media for the first time. Increased core values are attributed to the former disposal of fly ash in this subreach (Sect. 3.1). Surface sediment concentrations of boron and vanadium, although elevated here in relation to subreach 3.03, are also relatively high in the reference reach. Concentrations of ^{137}Cs and ^{60}Co are also elevated in core and surface sediment in this subreach, most likely as a result of backflow from the Clinch River or the discharge of cooling water from the K-1770 steam plant (cooling water for the steam plant was drawn from the Clinch River and released to Poplar Creek).

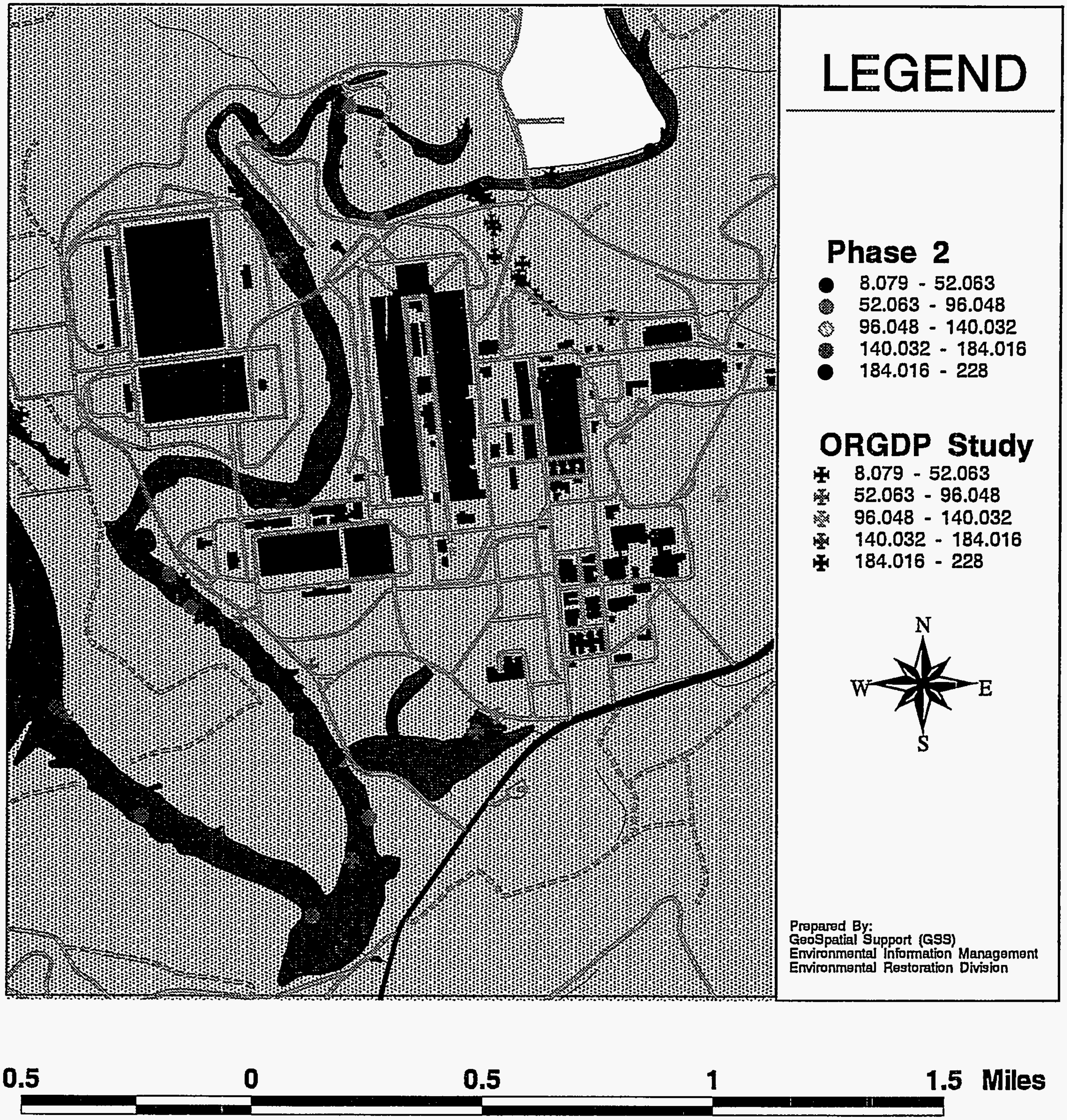


Fig. 3.40. Nickel concentrations (mg/kg) in surface sediment in Mitchell Branch and vicinity, according to the Oak Ridge Gaseous Diffusion Plant study and Clinch River Remedial Investigation Phase 2. *Source*: T. L. Ashwood, C. R. Olsen, I. L. Larsen, and P. D. Lowry. 1986. *Sediment Contamination in Streams Surrounding the Oak Ridge Gaseous Diffusion Plant.* ORNL/TM-9791. Oak Ridge National Laboratory, Oak Ridge, Tenn.

3.4.2.2 Contaminant distributions in the Clinch River

This discussion considers spatial trends in the distribution of contaminants, beginning in upper Melton Hill Reservoir (reach 0) and extending downstream through subreach 4.04 at the mouth of the Clinch River. Reach 0 concentrations are considered reference values for the purpose of this discussion. As expected from the historical data, organic contaminants were detected only infrequently (Table C2), consisted mostly of relatively ubiquitous PAHs and phenols, were generally present at low concentrations (<1 mg/kg dry sediment), and were often present in the reference reach (Tables C3–C5). No clear pattern of organic contaminant distribution in the Clinch River is evident (Figs. 3.35 and 3.36). Therefore, the following discussion will not consider organic compounds other than PCBs.

Reach 1—lower Melton Hill Reservoir. Mean concentrations of ^{60}Co were elevated in both surface and deep sediment (Figs. 3.37–3.39). This increase is attributed to known releases from the American Nuclear Corporation site on Braden Branch (see Sect. 3.1) and not from the ORR. Concentrations of ^{60}Co remain elevated in both surface and deep sediment in relation to reference values throughout the Clinch River, but they are much lower than those observed in reach 1. Copper concentrations are also elevated in both surface sediment and core samples in reach 1; the source is unclear.

Reach 2—Clinch River between Melton Hill Dam and Poplar Creek. The upper subreaches of reach 2 are not significant zones of sediment deposition and, therefore, were sampled only sparsely in this study. As a result, detecting trends in reach 2 with confidence is difficult. Among the most apparent trends are the expected increase in ^{137}Cs concentrations in both surface sediment and core samples collected downstream of WOC (Figs. 3.37 and 3.38). Peak values of each occur within reach 2 but remain elevated downstream throughout reach 4. Uranium-234 and ^{238}U, undetected in reaches 0 and 1, are detected at low concentrations throughout subreaches 2.02–2.04. Aroclor 1254 concentrations appear elevated in both surface and deep sediment in subreach 2.02 (Figs. 3.35 and 3.36). Although the data are derived from a single core sample, the spatial pattern is consistent with known PCB contamination in sediment of WOC (subreach 22), which is located immediately upstream. Selenium and manganese concentrations increased significantly in both surface sediment and core samples collected from subreach 2.04, downstream of Grassy Creek (Figs. 3.32 and 3.33). Subreach 2.04 is adjacent to the K-1770 steam plant area of the K-25 Site; increased levels of these analytes may be related to operations at this site or the nearby K-700 scrap metal pile.

Reach 4—Clinch River downstream of Poplar Creek. As expected, mean concentrations of mercury in both surface and deep sediment are elevated downstream of Poplar Creek (Figs. 3.32 and 3.33) but at levels much reduced from those within Poplar Creek itself. Mean mercury concentrations are relatively constant throughout reach 4. As noted earlier, ^{60}Co remains elevated throughout reach 4, but mean concentrations are low (<1 pCi/g). In addition, concentrations of ^{137}Cs remain elevated throughout reach 4, in which core values are generally highest (Fig. 3.38); surface concentrations appear slightly higher in reach 2 (Fig. 3.37).

The mean core concentration of several analytes appears to decrease in subreach 4.02, leading to an apparent increase in contaminant concentrations in subreaches 4.03 and 4.04. However, because this pattern is evident for contaminants whose sole or primary source is the ORR (e.g., ^{137}Cs, mercury), this is thought to be primarily a function of sediment transport dynamics and is not thought to indicate a significant source of contaminant flux via the Emory River.

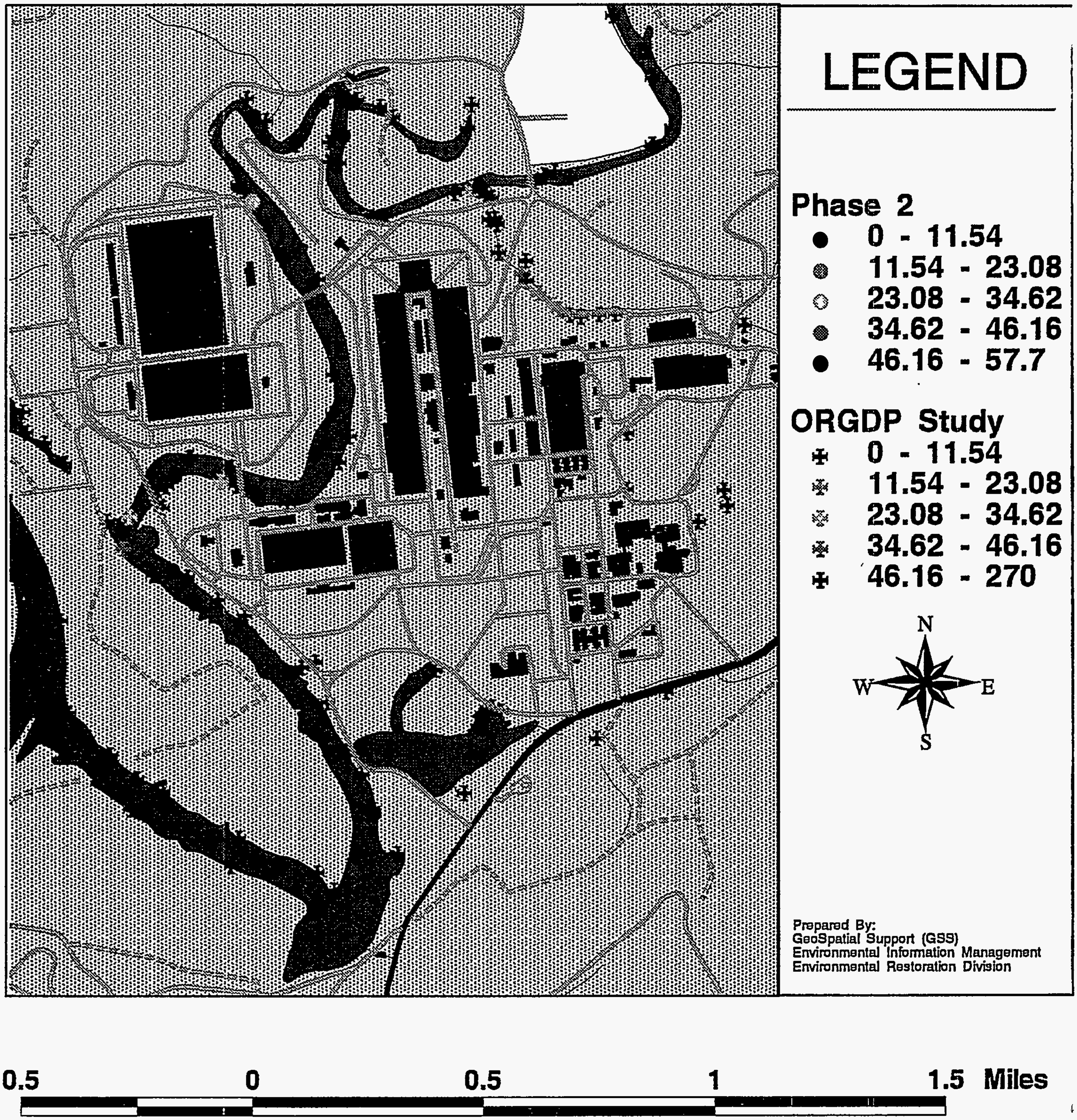


Fig. 3.41. Uranium-238 concentrations (pCi/g) in surface sediment in Mitchell Branch and vicinity, according to the Oak Ridge Gaseous Diffusion Plant study and Clinch River Remedial Investigation Phase 2. *Source*: T. L. Ashwood, C. R. Olsen, I. L. Larsen, and P. D. Lowry. 1986. *Sediment Contamination in Streams Surrounding the Oak Ridge Gaseous Diffusion Plant.* ORNL/TM-9791. Oak Ridge National Laboratory, Oak Ridge, Tenn.

A possible exception is copper, mean concentrations of which increased by a factor of two or more in both surface sediment and core samples collected downstream of the Emory River (subreach 4.03; Figs. 3.32 and 3.33). Aroclor 1254 concentrations were also elevated below the Emory River; although the greatest mean core concentrations were found in subreach 4.03, the greatest mean surface sediment concentration was found in subreach 4.04. This pattern may suggest separate releases of Aroclor 1254 from more than one source.

Reach 7—McCoy Branch. Sampling was limited to only one core sample each from subreaches 7.01 and 7.02 and from reach 8, the reference reach at Walker Branch. Therefore, no statistical data analysis was possible. Nonetheless, the data did show the expected elevated levels of arsenic in surface sediment, deep sediment, and pore water, particularly in subreach 7.01 (Figs. 3.32–3.34). Concentrations of selenium were elevated in surface and deep sediment in subreaches 7.01 and 7.02. Vanadium concentrations appeared elevated in deep sediment and pore water and to a lesser degree in surface sediment. These three metals are associated with coal ash, and their presence is presumed to result from the historical disposal of fly ash from the Y-12 Plant in McCoy Branch and Rogers Quarry.

3.4.3 Supporting Studies—CRRI Phase 2 Sediment Characterization

The primary supporting studies conducted during the CRRI Phase 2 investigation were sediment toxicity tests and sediment and contaminant transport modeling. The findings of both efforts are summarized below.

3.4.3.1 Toxicity studies of whole sediment and pore water

Laboratory sediment toxicity tests were conducted to assess the sediment quality of Poplar Creek and the Clinch River. Results of the sediment toxicity tests are used in the baseline ecological risk assessment (Chap. 6).

Three exposure routes were evaluated: exposure to particles (whole sediment), exposure to interstitial pore water, and exposure to overlying water (elutriate). Toxicity from exposure to whole sediment was evaluated by using *Hyalella azteca* (survival endpoint) and *Anodonta imbecillis* (survival endpoint). Toxicity of pore water was evaluated with the *Ceriodaphnia dubia* acute test (survival endpoint) and the liquid-phase Microtox® test. The toxicity of elutriate was evaluated by using the crustacean *Daphnia magna* (survival and reproduction endpoints). All test protocols are described by Kszoz et al. (1989).

Toxicity was initially evaluated during nine sampling events between July 1993 and September 1994. A total of 304 tests were conducted on 78 samples from 15 sites in 6 reaches (Fig. 3.28); not all sites were sampled during each sampling event.

The interpretation of sediment toxicity test results is made within the context of two important test issues. First, the sites were not sampled in replicate; rather, the samples were divided into "replicate" subsamples for statistical analysis. This sampling approach precludes an analysis of spatial variability of sediment toxicity. Second, the study did not have a true control sediment against which to assess site results. Comparisons therefore focus on study sites in relation to reference sites.

Either of two criteria was used to identify a significant toxic response: (1) if the study site had a response that was significantly ($p < 0.05$) more toxic that at the reference site or (2) if the study site had a toxic response that was 20% greater than that of the reference site.

Toxicity of whole sediment. Initially, 62 individual samples were tested for toxicity to *H. azteca*, and from them 46 site-to-reference comparisons were made. In 85% of the *H. azteca* samples, survival was ≥70%. Only one sample (PCM 4.3 in event 1) showed a statistically significant reduction in survival in comparison with the reference site. In addition, only 11 of the 46 comparisons (23.9%) showed a ≥20% reduction in survival in relation to survival at the reference sites (Table M1). No significant reductions in survival were observed in any McCoy Branch or Walker Branch tests (Table M2).

In December 1995, additional samples were collected from those locations in Poplar Creek that the CRRI Phase 2 data indicated had the most heavily contaminated surface sediment. Toxicity was evaluatedby using *H. azteca* to determine if the lack of toxicity demonstrated in the previous tests could be attributed to sampling of relatively clean areas within Poplar Creek. Again, survival was ≥70% in all tests and none of the study sites differed significantly from the test site (Table M1).

Seventy-five percent of all *A. imbecillis* tests resulted in ≥75% survival. A total of 24 individual tests were conducted, and 18 site-to-reference comparisons were made. Only 1 test was statistically significant and only 4 of the 24 tests (16.7%) showed a ≥20% reduction in survival in relation to reference sites (Table M3). Actual survival may have been even greater than indicated by these results, as survival in split-sample tests conducted by TVA's Aquatic Research Laboratory was consistently higher than in CRRI tests, and organism recovery in the latter was often below 100%.

Toxicity of pore water. None of the Microtox® pore water tests showed toxicity, and of the 43 *C. dubia* pore water tests, only one site on one event showed a marginally significant reduction in survival (Tables M4 and M5). Overall survival was ≥90% in 96% of the tests. It should be noted that a major limitation of pore water evaluations is that the chemistry (and therefore, toxicity) of the pore water sample may change during sample extraction and manipulation.

Toxicity of elutriate. Toxicity was not suggested by either the *D. magna* survival data or the toxicity data. Overall survival was high; ~85% of the 54 tests resulted in ≥80% survival. Only 5 of the 40 site-to-reference comparisons (12.5%) showed a ≥20% reduction in survival in relation to reference sites (Tables M6 and M7). None of these comparisons was statistically significant.

Of the 40 site-to-reference reproduction comparisons, only 7 (17.5%) showed a ≥20% reduction in reproduction (Tables M8 and M9). Of these, only two were statistically significant. All reductions in reproduction occurred in one of two sampling events in the first half of 1994.

Summary of sediment toxicity test results. Overall, very little toxicity was evident. No striking spatial trends were observed, but the higher percentage of positive responses were found at PCM 4.3, CRM 9.0, and CRM 19.0. Temporal trends are difficult to determine given the minimal response detected, but the greatest probability of observing a significant response appears to occur in winter through spring. No historical sediment toxicity data were available for the study area.

Phase 2 data indicated that the most sensitive tests were the *H. azteca* and the A. imbecillis whole sediment tests, in which direct exposure to most contaminants is presumed greatest. The least sensitive tests were the two pore water tests (the *C. dubia* acute toxicity test and the Microtox® liquid-phase test).

3.4.3.2 Modeling of sediment transport in the Clinch River

One dimensional modeling of sediment and ^{137}Cs transport. The fate of historically released ^{137}Cs (1949–91) was simulated through the use of three separate one-dimensional water and sediment transport models: HEC-6-R (U.S. Army Corps of Engineers 1993), CHARIMA (Holly et al. 1990), and TODAM (Onishi et al. 1981). The models were implemented independently by ORNL, by TVA in cooperation with the Iowa Institute of Hydraulic Research, and by Pacific Northwest Laboratory (PNL). This multiple modeling approach was used to foster confidence in model predictions characterizing future scenarios (Rose et al. 1993).

The HEC-6-R (implemented by ORNL) and CHARIMA (implemented by TVA) models simulated hydrodynamic conditions, sediment deposition, erosion and transport, and contaminant fate. The TODAM model simulated sediment and contaminant fate processes but received hydrodynamic information from the hydrodynamic module in CHARIMA. The three models were configured similarly for the total Clinch River–WBR system, but they differed in numerical solution techniques and in algorithms for calculating sediment transport and contaminant fate.

All models were calibrated with TVA data on historical sedimentation rates and validated with existing data on ^{137}Cs distributions (Cottrell 1960; Morton 1965; Struxness et al. 1967; Oakes et al. 1982; Olsen et al. 1992). Each model predicted sediment accumulation that generally matched the measured accumulation, even though the latter varied significantly between time periods. (e.g., TVA silt range data indicate that only 25 acre-ft of sediment accumulated in the Clinch River during the 30-year period ending 1991). TODAM tended to overpredict sediment accumulation for the periods 1951–56 and 1961–91.

The "known" ^{137}Cs inventory in the Clinch River in the summer of 1977 was extrapolated through a Voronoi tesselation in GRASS of the measured ^{137}Cs core data of Oakes et al. (1982). The predicted 1977 inventories were calculated by each of the modeling groups and compared to this known inventory for several segments of the Clinch River (Fig. 3.42). The extrapolated core data of Oakes et al. (1982) indicate a total of 42.5 Ci of ^{137}Cs in the river. The TODAM model predicts four-fold that amount, CHARIMA predicts a total of 40 Ci, and HEC-6-R predicts 57 Ci. The much larger predictions of the TODAM model result from the inclusion of a greater proportion of coarse sediments entering the Clinch River than was assumed in the other models. This is significant in that coarser sediments would be deposited more quickly and closer to the contaminant source than finer sediments. Therefore, the TODAM model is considered the least accurate in that it fails to simulate the known erosion and subsequent movement downstream of contaminated sediment deposited in the Clinch River during the period of peak ^{137}Cs flux from WOC in the late 1950s.

Each of these models predicts deposition over relatively large reaches of the Clinch River and therefore does not attempt to predict small-scale patterns of ^{137}Cs distribution. A strong spatial heterogeneity in erosion and deposition in Clinch River sediment exists at a much finer scale than in the modeled reaches. For example, only about 25% of the >200 cores sampled by Oakes had significant ^{137}Cs levels, and many samples collected very near each other differed significantly in ^{137}Cs inventory. In addition, the bathymetry of the Clinch River was represented by only five silt ranges. This resulted in some necessary additional bathymetry assumptions to obtain successful hydrodynamic simulations, which might have adversely affected the pattern of sediment deposition. Nonetheless, CHARIMA and HEC-6-R predicted total ^{137}Cs inventory very well.

Each model made predictions on the basis of three future scenarios: (1) no major storm events or releases of ^{137}Cs, (2) a 100-year storm event localized in the WOC watershed, and (3) a system-wide 100-year storm event. The scenarios begin on April 30, 1991, and each modeling group begins with the ^{137}Cs inventories predicted for that date by their respective models. All simulations evaluated both short-term (2-week, 1-month, and 6-month) and long-term (1-year and 10-year) responses. The increased flux

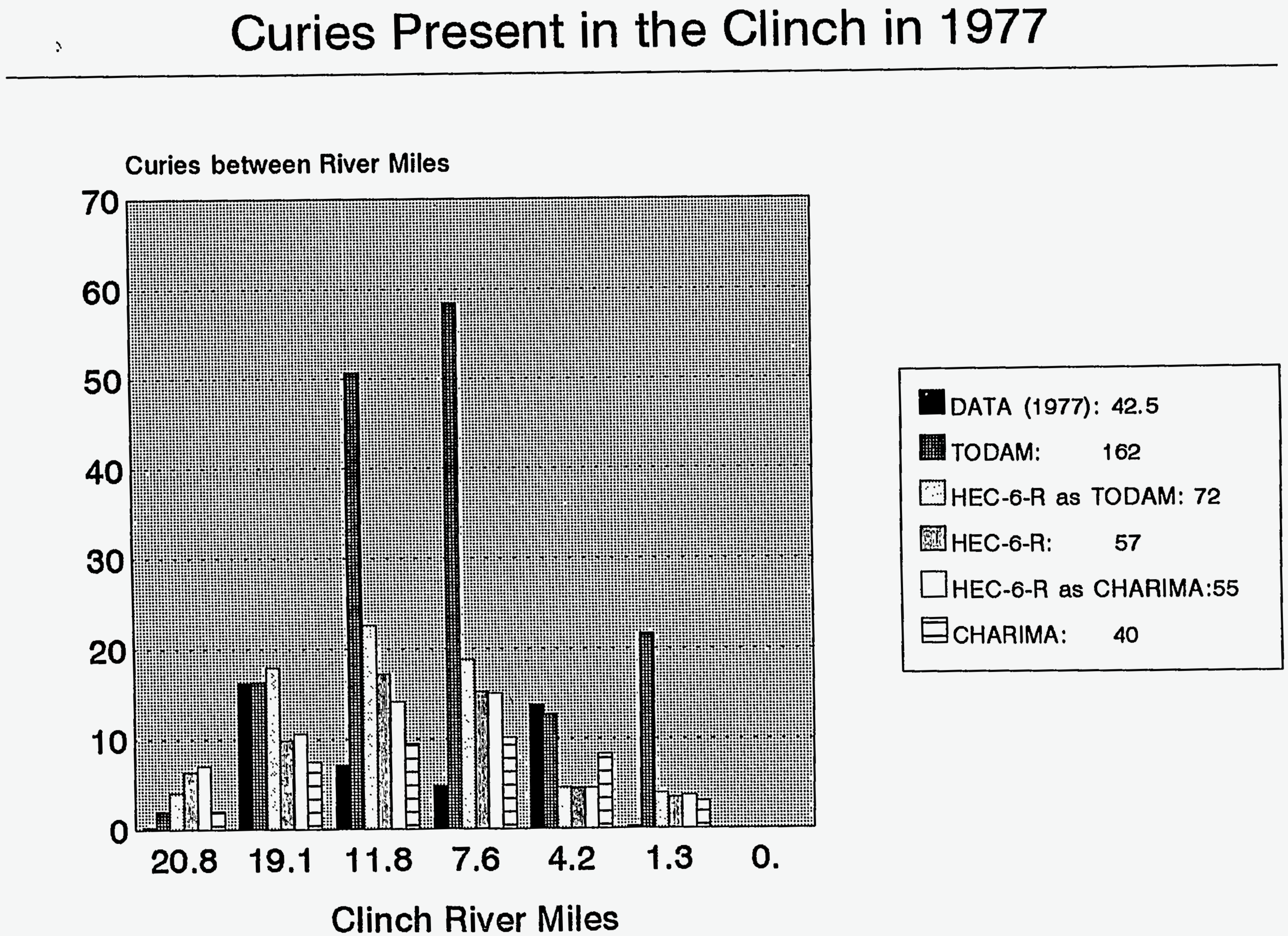


Fig. 3.42. Results of model simulations for estimating total ^{137}Cs inventories (curies) within reaches of the Clinch River for 1977.

of ^{137}Cs from WOC was based on preliminary predictions by the HSPF watershed model (Bicknell et al. 1993) applied to the WOC watershed. Results are summarized in Table 3.6.

Table 3.6. Clinch River storm responses with regard to ^{137}Cs in bed sediments

Model and scenario	Onset	2 weeks	1 month	6 months	1 year	10 years
HEC-6-R						
No storm	39.9	39.9	39.8	37.5	37.2	30.5
Local storm	39.9	46.6	44.4	37.8	37.5	30.7
System storm	39.9	34.0	33.3	31.9	31.6	26.3
CHARIMA						
No storm	30.7	30.7	30.7	28.9	28.7	25.7
Local storm	30.7	33.2	33.2	28.0	27.9	24.7
System storm	30.7	25.7	24.1	22.9	22.8	20.2
TODAM						
No storm	120.8	120.7	120.6	119.1	118.1	98.7
Local storm	120.8	139.9	139.7	128.4	127.0	105.4
System storm	120.8	127.0	125.3	124.1	122.9	102.5

HEC-6-R model results. When no specific storm occurs in the WOC watershed and therefore no increased release of ^{137}Cs takes place, the dominant process for loss of radioactivity is radioactive decay. Through erosion, this loss is somewhat enhanced in the 1- to 6-month period. When a local 100-year-return-period storm is simulated in the WOC watershed, 23.5 Ci are released, and 6.7 Ci accumulate in the Clinch River within 2 weeks after the onset of the storm. However, within 6 months, the level of activity in the Clinch River decreases to a level lower than that before the storm (i.e., the curies deposited in the Clinch River reside there for a relatively short period). During the system-wide storm, in spite of increased ^{137}Cs flux from WOC, a net loss of 5.9 Ci occurs in the Clinch River through erosion. Thus, 1 and 10 years after either the local storm or the no-storm scenario, the ^{137}Cs inventory in the river declines, primarily through radioactive decay. Following a system-wide storm, contaminant levels decline in the short term as a result of sediment erosion and in the long term through radioactive decay.

CHARIMA model results. Results of the CHARIMA simulations are similar to those of HEC-6-R. When no specific storm occurs, radioactive decay is the dominant process in the Clinch River. Again, some erosion occurs in the short term. CHARIMA predicts less of an increase (i.e., only 3 Ci) in contaminant levels than HEC-6-R as a result of the local storm, but again the increase is only temporary. As with HEC-6-R, the system-wide storm results in erosion that lowers the ^{137}Cs inventory by 5 Ci. CHARIMA's prediction of historic ^{137}Cs deposition (i.e., conditions at the onset of the simulations) was lower than that of HEC-6-R, and its storm scenario response was, therefore, similarly muted.

TODAM model results. The TODAM scenario results differ from those of HEC-6-R and CHARIMA in a number of significant ways. To begin, the 1991 ^{137}Cs inventory in the river predicted by TODAM is three- and four-fold of that predicted by HEC-6-R and CHARIMA, respectively. In the no-storm scenario, both HEC-6-R and CHARIMA show roughly 5% reduction in ^{137}Cs levels between 1 and 6 months. TODAM predicts only a 1% reduction. In other words, in comparison with HEC-6-R and CHARIMA, TODAM assumes that bed sediments are less erodible, which is in keeping with the assumption of generally coarser bed particle-size distributions used in the TODAM model. The local storm scenario results differ in that TODAM predicts a larger increase of ^{137}Cs in the Clinch River (+19 Ci) than HEC-6-R (+6.7 Ci) and CHARIMA (+2.5 Ci) do. TODAM, like the other models, predicts that most of the recently deposited ^{137}Cs resides there only temporarily (<6 months), although a net increase persists longer. For the system-wide storm scenario, TODAM predicts a large increase (+6.2 Ci) in ^{137}Cs levels, whereas both HEC-6-R and CHARIMA showed an immediate decrease as a result of erosion.

Summary of one-dimensional models. Each model predicts that, in the absence of a significant (i.e., 100-year) storm event, the existing inventory of ^{137}Cs in the Clinch River will continue to decline, primarily through radioactive decay. The HEC-6-R and CHARIMA models predict that despite increased contaminant flux from WOC following a system-wide, 100-year storm, contaminant levels in the Clinch River will decrease in the short-term as a result of erosion of existing sediment. Conversely, TODAM predicts an increased inventory under this scenario. The HEC-6-R and CHARIMA models predict that a sudden flux of contaminants from the WOC watershed during a local storm would result in a temporary (<6 month) increase in contaminant levels in Clinch River sediment; TODAM predicts a much larger and longer-lasting increase. The TODAM model results apparently differ because a critical underlying assumption differs; TODAM assumes that a greater proportion of coarse sediment is transported and thus generally predicts quicker sediment deposition following storm events and less erosion subsequent to deposition. However, the coffer dam at the mouth of WOC will prevent the coarsest sediment particles from entering the Clinch River, and discharges from Melton Hill Dam can easily cause resuspension and entrainment of freshly deposited sediments. Indeed, TODAM was the least calibrated of the three models.

Sediment transport in the Clinch River with a two-dimensional, finite-element model. A two-dimensional model of the Clinch River was used to simulate horizontal flow circulation and sediment erosion, deposition, and transport. The model is used to better understand and quantify the short-term fate of an accidental release of a sediment-reactive contaminant under baseline and high flow conditions.

The RMA2 hydrodynamic module and the STUDH sediment transport module (both from TABS-2, Norton et al. 1973; Thomas and McAnally 1991) were applied to the Clinch River. A two-dimensional model permits lateral (across-river) resolution of predictions and has the advantages of being able to model unsteady flow (time-dependent); reverse and lateral flows; and complex flow patterns related to the power plant, channel islands, weirs, and tributaries. A two-dimensional finite-element grid network was created to represent the bathymetry of the Clinch River system, on the basis of data collected by TVA, ORNL, and the U.S. Army Corps of Engineers. Grids are formed by connecting the nodes into triangular or quadrilateral elements. Grid generation was performed with an automated algorithm and selected points from observed data. Cross-sectional areas based on generated grids differed by <5% from those estimated from measured silt range data. The grid network is composed of 4137 nodes and 1313 elements. The system receives water released at Melton Hill Dam on the upstream boundary and inflows from the tributaries including WOC, Poplar Creek, and the Emory River. The downstream boundary of the system is located at the Clinch River mouth (the confluence with the Tennessee River).

The RMA2 hydrodynamic module was calibrated with November 1993 flows and water surface elevation data. Because RMA2 uses observed water surface elevation at the downstream boundary and inflow values from each tributary as boundary conditions, the largest difference between simulated and observed water surface elevations generally occurs at the upstream-most boundary (Melton Hill Dam). The regression relationship between simulated and measured stage heights at the Melton Hill Dam was

$$Y = 17.4593 + 0.92286X,$$

where Y is the observed stage and X is the simulated stage. The regression (R^2) was 0.989[2], implying good correspondence between simulated and observed values. The simulation results also showed that reverse flow occurred several times around the Kingston Steam Plant in November 1993.

The STUDH sediment transport module was coupled to the calibrated RMA2 hydrodynamics module to simulate deposition, resuspension, and transport of sediments (cohesive and noncohesive material). Predicted flow velocities and water depths at each node in the system from RMA2 were input to STUDH. STUDH simulates one sediment type (grain size class) at a time to simplify numerical solution. Two classes of fine-grained sediments (clay and silt) were used as a surrogate for contaminated sediments. Fine-grained sediments are a good surrogate for ^{137}Cs provided that the ^{137}Cs entering from WOC is in equilibrium between the dissolved and adsorbed phases and undergoes negligible decay during model simulations. Field measured data indicated that ~90% of ^{137}Cs released was adsorbed to suspended sediments (Churchill et al. 1965). Simulated sediment transport was deemed realistic because predicted locations of clay, silt, and sand deposition were similar to qualitative patterns of these sediment classes observed in the Clinch River (Levine et al. 1994).

Six simulationswith the coupled hydrodynamic and sediment transport modules were performed. The six simulations involved three flow conditions (baseline, a local storm in WOC, and a system-wide storm), each with a sudden 3-day release of clay or silt in WOC. The baseline simulation imposed the 3-day release under average flow conditions. The local storm scenario imposed the 3-day release during 3 days of high tributary inflows in the WOC basin only. The system-wide storm scenario imposed the 3-day release during 3 days of high tributary inflows for WOC and the Clinch River. The movement and distribution of the contaminated sediments suspended in the water column and newly deposited on the river bottom after the storm release were tracked for 30 days. The river bottom was assumed initially to have no sediments, and no erosion was allowed to prevent resuspension. Thus, only deposition was permitted and all deposited sediments came from the 3-day release event.

Model predictions are expressed as the percentage of the total amount of introduced sediment that is deposited in the WOC Embayment (from the WOC dam to the coffer dam), upper Clinch River region (between WOC Embayment and Poplar Creek), and lower Clinch River region (between Poplar Creek and the confluence with the Tennessee River), and the percentage exported out of the Clinch River. Because of different tributary inflows, more sediment was introduced to the system under the local and system-wide storm scenarios (325,202 kg) than during the baseline scenario (20,689 kg).

Summary of two-dimensional model results. Predicted percentages deposited or exported out of the system for the six simulations after 30 days are presented in Table 3.7. The following conclusions can be drawn from the simulation results:

1. As expected, under the same flow conditions, clay was transported further downstream than silt.

2. A higher percentage of sediment was retained in WOC Embayment under baseline conditions in comparison with the local and system-wide storm scenarios; however, >70% of the clay was exported out of the system under baseline conditions.
3. No significant deposition occurred in the lower Clinch River in any of the simulations.
4. Significant deposition occurred only in the upper Clinch River region under the local storm scenario.
5. Simulations with clay and the system-wide storm with silt resulted in significant export (>27%) of sediments out of the Clinch River and into lower WBR.

Table 3.7. Predicted percentage of clay or silt deposited in regions of the Clinch River and exported out of the Clinch River 30 days after a 3-day release in White Oak Creek during various storm conditions

	Clay (%)			Silt (%)		
	Baseline	Local storm	System storm	Baseline	Local storm	System storm
White Oak Creek Embayment	29.36	4.27	4.63	85.38	39.77	46.06
Upper Clinch River	0.00	68.17	0.53	0.00	48.50	25.97
Lower Clinch River	0.00	0.00	0.00	0.00	0.00	0.00
Out of Clinch River	70.38	27.57	94.65	14.62	11.73	27.92

3.4.4 Summary of Sediment Characterization

The CRRI Phase 2 data confirm much of what was known about sediment contamination in the Clinch River and Poplar Creek from earlier investigations. In general, organic contaminants are found infrequently and at very low concentrations throughout the system, and they consist almost exclusively of compounds that are relatively ubiquitous at low concentrations in the environment.

In Melton Hill Reservoir, contaminant levels are generally at or near background levels except for ^{60}Co, which originates from a non-DOE source in Braden Branch. The historical practice of disposing of fly ash from the Y-12 Plant in McCoy Branch has resulted in elevated concentrations of several metals (As, Se, and V) in the sediment of McCoy Branch Embayment of Melton Hill Reservoir.

The primary radionuclide found at elevated levels in the Clinch River downstream of WOC is ^{137}Cs. Peak concentrations are found in areas of significant sediment accumulation, primarily downstream of Poplar Creek, and are buried under as much as a foot of cleaner sediment. Near-shore areas, as a result of less sediment accumulation, have lower average ^{137}Cs concentrations than sediment deposition zones in and along the river channel. Mercury is also elevated in the Clinch River downstream of Poplar Creek, where peak concentrations of mercury coincide with those of ^{137}Cs as a result of concurrent release histories.

Several metals and radionuclides are elevated in Poplar Creek sediment. Mercury, copper, chromium, and cadmium are elevated below the mouth of EFPC. Mitchell Branch is an apparent source of several contaminants, as evidenced by increased levels of Ni, Ag, Cr, Cu, U, and ^{99}Tc downstream of its mouth. In lower Poplar Creek, the historical disposal of fly ash from the K-25 Site steam plant has resulted in increased levels of arsenic, boron, and vanadium in both surface and deep sediment. Levels of ^{137}Cs and ^{60}Co are also slightly elevated here in comparison with upper subreaches of Poplar Creek, possibly as a result of backflow from the Clinch River or the former discharge of cooling water from the K-770 steam plant.

Tests were conducted to evaluate the toxicity of whole sediment, sediment elutriate, and sediment pore water to several different test organisms. Overall, toxicity was very low in all tests and across all sites. Few tests indicated statistically significant toxic effects in comparison with the appropriate reference tests, and no meaningful pattern of toxicity could be discerned.

One-dimensional sediment transport modeling was conducted to predict future contaminant (^{137}Cs) fate and distribution under a variety of hydrological conditions. In general, barring any significant additional release to the Clinch River, existing levels of ^{137}Cs will continue to decline, primarily through radioactive decay. In the event of a local 100-year storm in the WOC watershed, sediment and ^{137}Cs flux would result in temporarily increased levels in the Clinch River, but these recently deposited sediments would be scoured and transported downstream within 6 months. In a regional 100-year storm existing contamination would immediately be scoured and transported downstream, and their levels in the Clinch River would decline immediately as a direct result of the storm. Previous modeling (DOE 1995) indicated that much of the scoured sediment would be deposited in lower Watts Bar Reservoir at concentrations that would not pose an unacceptable risk to human health or the environment.

3.5 CHARACTERIZATION OF BIOTA

The characterization of contaminants in biota focuses on the nature and extent of contamination in fish flesh (i.e., fillets). Section 3.5.1 summarizes the historical data regarding contaminants in fish from the Clinch River and Poplar Creek, and it presents a site conceptual model based on this information. Section 3.5.2 discusses the current nature and extent of contamination in fish. A number of supporting studies were conducted as part of the biota characterization task; these are outlined and discussed in Sect. 3.5.3. The overall characterization of biota is summarized in Sect. 3.5.4.

3.5.1 Historical Studies of Contaminants in Fish and Presentation of Site Model

The following sections briefly outline the results of previous studies of contaminants in fish from off-site waters in the vicinity of the ORR. The studies provide an overview of conditions anticipated during the CRRI.

3.5.1.1 Clinch River Study

Fish were collected from the Clinch River downstream of ORNL during the period 1960–63 and analyzed for several radionuclides for the purpose of estimating potential risk to humans from fish consumption (Cowser and Snyder 1966). Several species of bottom feeders (carp, carpsucker, buffalo, and catfish) and several species of "sight feeders" (white crappie, bluegill, white bass, largemouth bass, sauger, and drum) were collected.

Radionuclide analyses on fish were conducted in two ways: (1) by analyzing only the flesh of the fish and (2) by analyzing the whole fish, which includes the flesh and the bones. Results are presented for each bottom feeder species (Table K4). Analyses of sight feeders used only the flesh; because of limited sample size, no interspecific or year-to-year results were presented.

The greatest radionuclide intake was calculated to result from the consumption of whole (i.e., bones and flesh) bottom feeders. It was estimated that, as a result primarily of ^{90}Sr in the bones of such fish, an individual could receive up to 7% of the maximum permissible intake allowed at that time (Cowser and Snyder 1966). The authors noted that ~45% of the ^{90}Sr and 20% of the ^{137}Cs measured in fish tissues were attributable to atmospheric fallout from nuclear tests.

3.5.1.2 TVA Instream Contaminant Study

The overall purpose of this study is outlined in Sect. 3.4.1.5. In preparing for this study, TVA (1983) conducted an extensive review of existing data. These data led TVA to expect levels of Hg, Cr, Ni, and PCBs in Clinch River and Poplar Creek fish that were above background levels in at least one species (Tables K5 and K6). In both the Clinch River and in Poplar Creek, existing data indicated that mean mercury concentrations, although elevated, were below the Food and Drug Administration (FDA) action level, 1 mg/kg.

During the study, TVA (1985d) collected fish (primarily bass and bluegill) from several sites in the Clinch River, from one site in Poplar Creek, from several streams on the ORR, and from reference areas. Mean fillet concentrations of metals other than mercury were within the range of background levels. Mean mercury concentrations in the Clinch River and Poplar Creek ranged from 0.03 mg/kg (in bluegill at CRM 2.0) to 0.93 mg/kg (in smallmouth buffalo at PCM 0.2). With all species combined, mean mercury concentrations in fish from Poplar Creek and the Clinch River were elevated above background levels but were below the FDA action level, 1.0 mg/kg; this confirmed the historical data.

Organic contaminants other than PCBs were detected rarely during the study. Mean PCB concentrations in channel catfish (fillets) were greatest at WOC mile 0.2 (3.1 mg/kg). Concentrations in Clinch River catfish decreased with distance downstream of WOC, but appeared elevated in relation to catfish from Melton Hill Reservoir. Mean concentrations (<1.0 mg/kg) in catfish from the Clinch River were below the FDA action level, 2.0 mg/kg (Table K7).

Of the several radiological parameters measured, only gross beta and ^{137}Cs appeared elevated in Clinch River fish in comparison with historical values for fish from the Tennessee River (Table K8).

3.5.1.3 CRRI Phase 1

Three fish species (bluegill, channel catfish, and largemouth bass) were collected from Watts Bar, Melton Hill, and Norris reservoirs in 1989 and analyzed for metals, organic compounds, and radionuclides (Cook et al. 1992). Seven potential human-health contaminants of concern were identified —As, Be, Hg, Se, PCBs (Aroclors 1254 and 1260), and total chlordane (Cook et al. 1992). No additional contaminants of concern were identified by a screening-level ecological risk assessment. Compared with data from the Norris Reservoir reference site, the concentrations of several analytes were significantly elevated at one or more Poplar Creek or Clinch River stations. However, only mercury and ^{137}Cs exhibited a spatial pattern that could clearly be attributed to releases from the ORR. Mean mercury and PCB values in Clinch River and Poplar Creek fish were again below FDA action levels.

The CRRI Phase 1 data are used directly in this site characterization (Sect. 3.5.2) and in human health and ecological risk assessment (Chaps. 5 and 6).

3.5.1.4 TVA fish tissue studies

TVA monitors contaminant concentrations in fish tissue throughout the Tennessee River valley. Since the late 1980s, TVA sampling in the vicinity of the ORR has focused on elevated levels of PCBs in fish from Fort Loudon, Watts Bar, and Melton Hill reservoirs (Dycus and Hickman 1988; Dycus 1990; Meinert 1991; Meinert and Fehring 1992; Fehring and Meinert 1993). TDEC (1992) has issued fish consumption advisories for these waters, including a "no consumption" advisory for Melton Hill catfish and "limited consumption" advisories for catfish and sauger in the Clinch River below Melton Hill Dam. Recent TVA data collection efforts have focused on PCB levels in catfish. The TVA data, augmented with ORNL data, are summarized in Figs. 3.43–3.44.

3.5.1.5 Site conceptual model

The historical and CRRI Phase 1 data indicate that levels of mercury and certain radionuclides are elevated in the Clinch River and Poplar Creek downstream of the ORR. PCB levels are generally elevated as well, and WOC is an apparent source of PCBs to the Clinch River. However, PCBs were widely used in a variety of industrial settings in the past and are common in fish collected upstream of the ORR in Melton Hill Reservoir. Mean concentrations in species for which data are available have been below the FDA action level, 2.0 mg/kg. Organic contaminants other than PCBs and chlordane are generally very low throughout the system. Mercury concentrations, although known to be elevated downstream of EFPC, have typically been below FDA action levels in both streams. Despite significantly elevated levels of ^{137}Cs in fish downstream of WOC, levels have historically been low enough that they do not pose a threat to human health.

3.5.2 Nature and Extent of Contamination in Fish

The nature and extent of contamination in fish was evaluated for four species of sportfish. These four species represent a variety of ecological and trophic levels and differ in their preference among anglers. Although whole fish data were also collected for purposes of ecological risk assessment, this characterization relies on the more extensive fillet data. Data from the CRRI Phase 1 and Phase 2 studies are combined here for purposes of site characterization.

3.5.2.1 Methods and approach

Sample locations are identified in Fig. 3.45. In Table D1, the sample size per reach and analytical class is summarized by species. The analytes evaluated in Phase 1 and Phase 2 differ; Phase 2 analyses focused on contaminants of concern identified by Cook et al. (1992) in the Phase 1 study.

Bluegill sunfish (*Lepomis macrochirus*). These small invertivores are a popular food fish. Because they have a relatively limited home range, the contaminant burden of these fish should be representative of conditions at the site from which they were collected. Phase 1 analyses were restricted to metals and gamma-emitting radionuclides. Phase 2 sampling was initially restricted to analytes of concern identified in Phase 1—As, Hg, Be, Cu, and Se. However, a limited number of bluegill were collected from reach 4 during Phase 2 for PCB (Aroclor) analyses for purposes of the human health risk assessment.

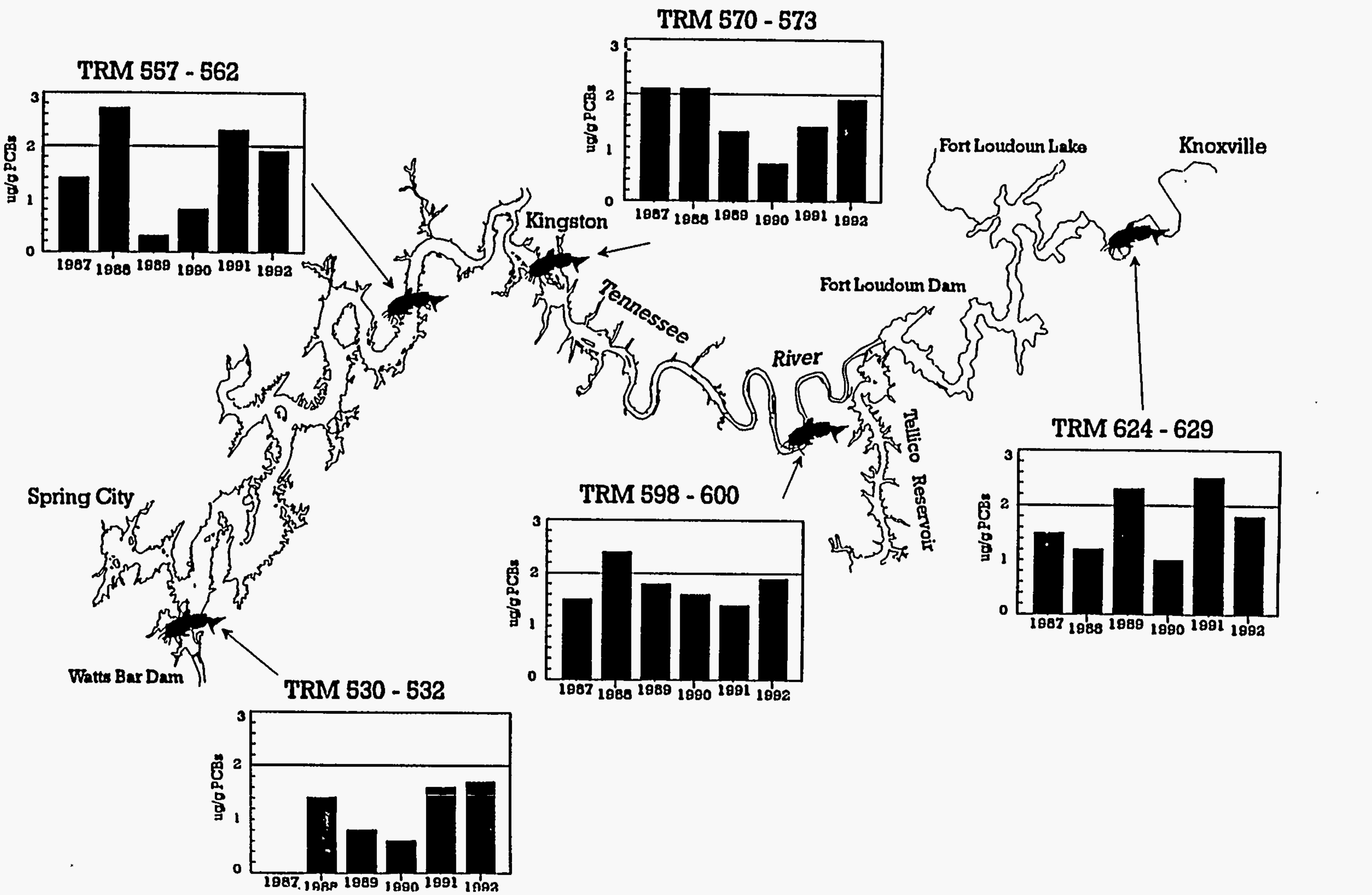


Fig. 3.43. Mean total PCB concentrations (mg/kg) in catfish at various sites throughout the Tennessee River section of Watts Bar Reservoir from 1987 to 1992.

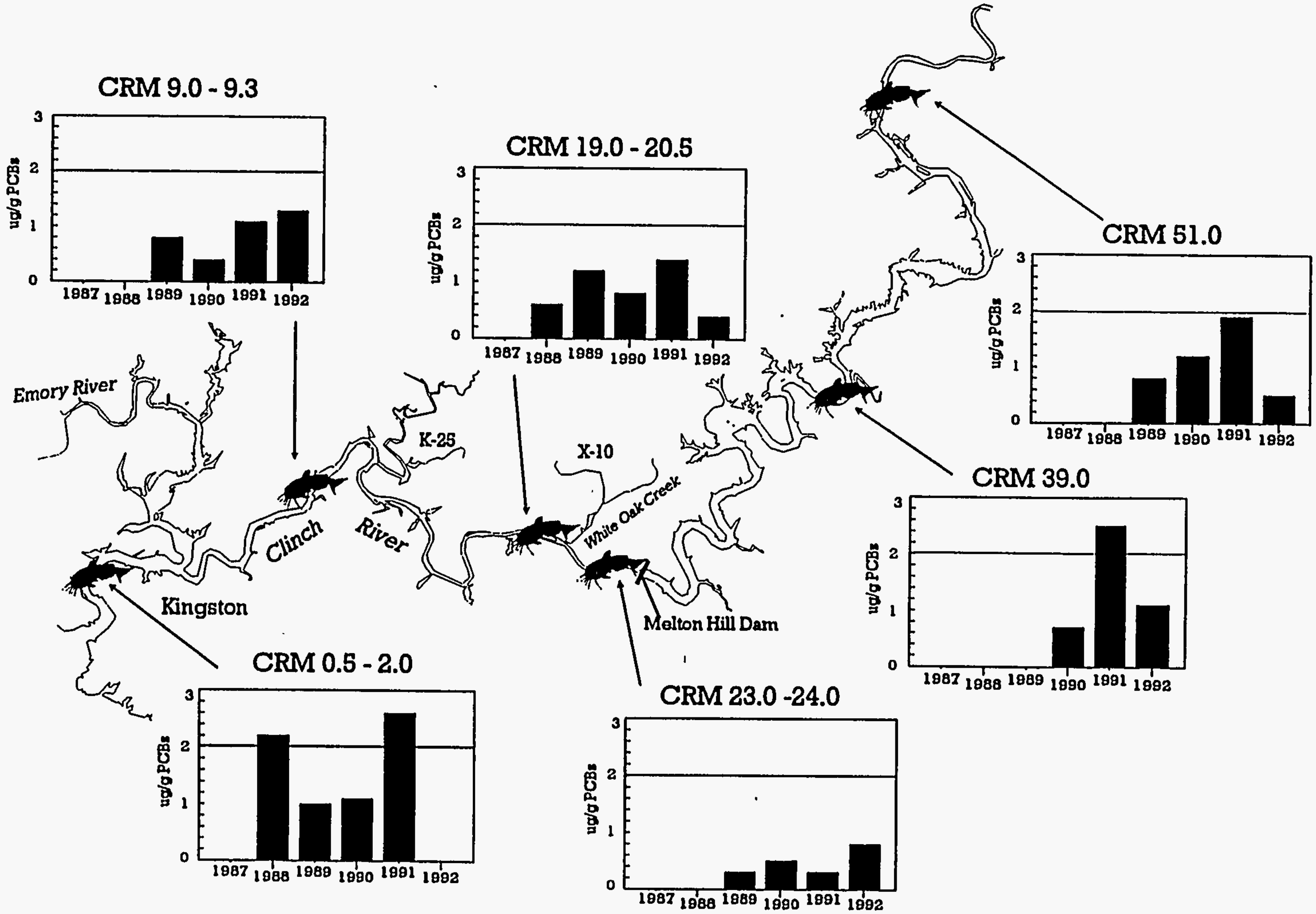


Fig. 3.44. Mean total PCB concentrations (mg/kg) in catfish at various sites throughout the Clinch River arm of Watts Bar Reservoir and Poplar Creek embayment from 1988 to 1992.

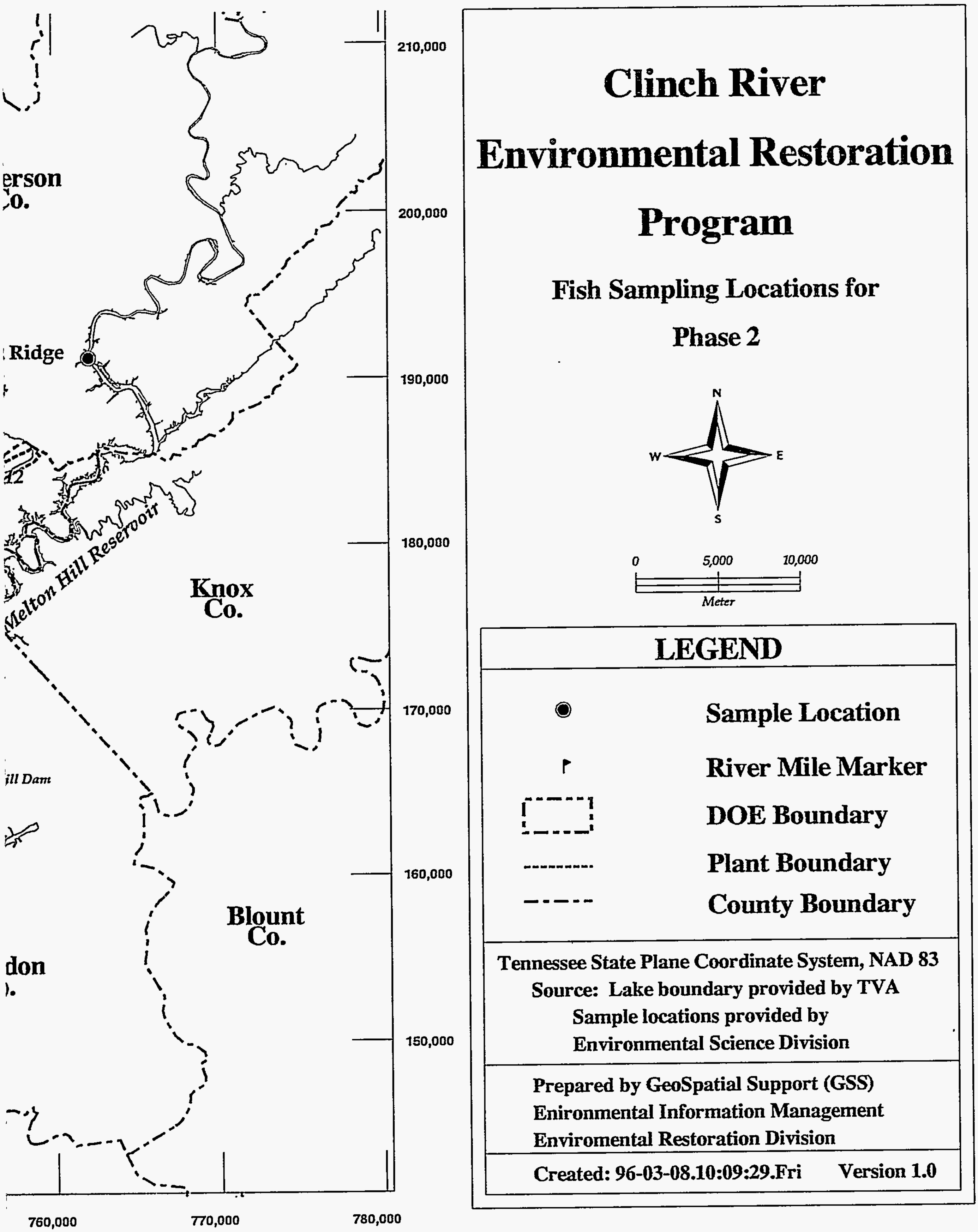
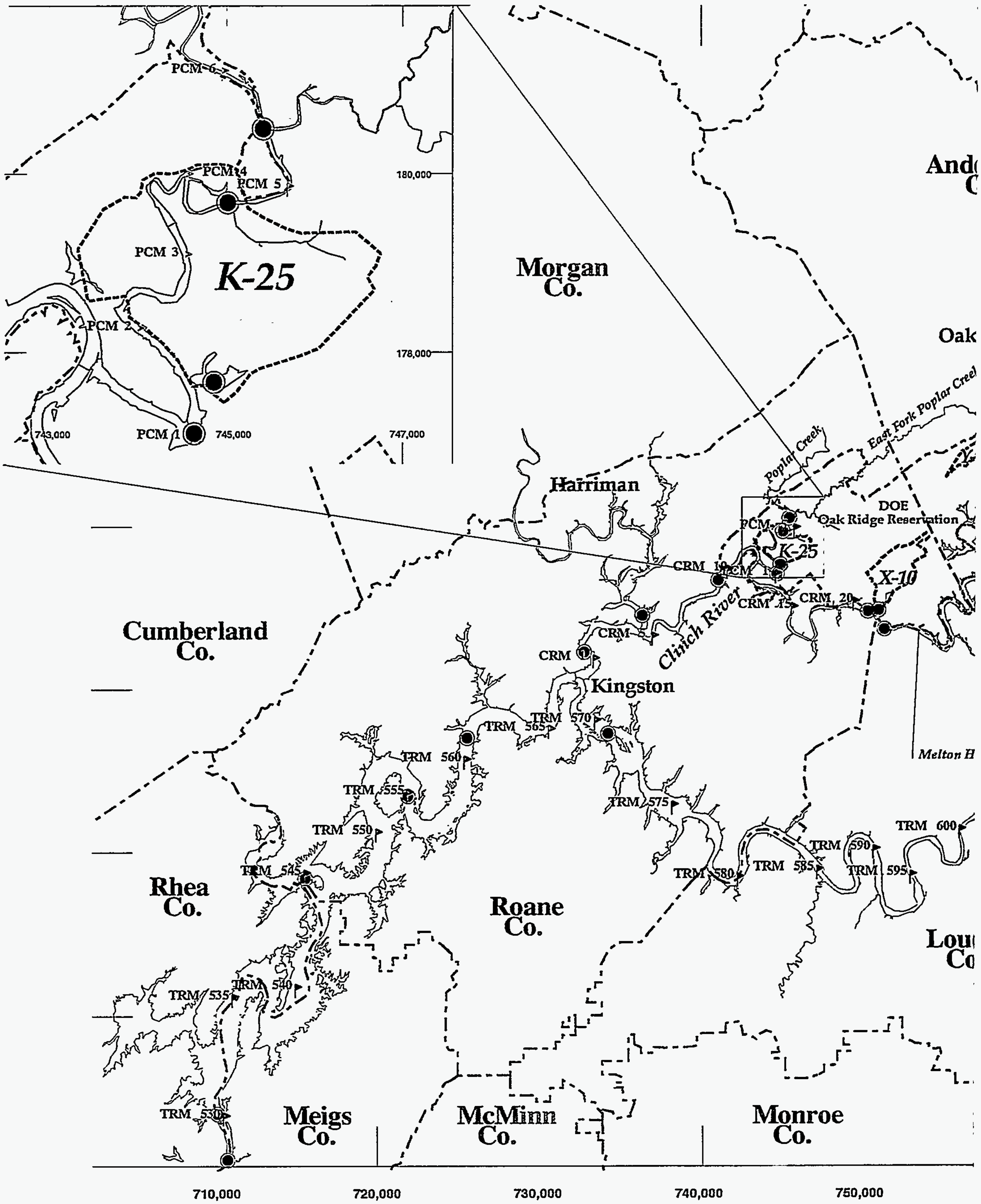


Fig. 3.45. Sample locations (Clinch River Remedial Investigation Phase 2) for characterizing the nature and extent of contaminants in fish.


PCM 6
PCM 4
PCM 5
180,000
PCM 3
K-25
PCM 2
178,000
PCM 1
743,000
745,000
747,000
Morgan Co.
And
Oak
East Fork Poplar Creek
Poplar Creek
Harriman
DOE
Oak Ridge Reservation
PCM
K-25
CRM 10
CRM 15
CRM 20
X-10
CRM 5
CRM
Clinch River
Kingston
Cumberland Co.
Melton H
TRM 565
TRM 570
TRM 560
TRM 555
TRM 575
TRM 550
TRM 545
TRM 580
TRM 585
TRM 590
TRM 595
TRM 600
Rhea Co.
Roane Co.
Lou Co
TRM 535
TRM 540
TRM 530
Meigs Co.
McMinn Co.
Monroe Co.
710,000
720,000
730,000
740,000
750,000

Largemouth bass (*Micropterus salmoides*). These mid-sized piscivores have a somewhat limited home range. They are an extremely popular sport fish. Phase 1 analyses were restricted to metals and gamma-emitting radionuclides. Phase 2 analyses focused on the metals of concern identified in Phase 1 (As, Hg, Be, Cu, and Se) and on PCBs (congeners and Aroclors) and pesticides (primarily chlordane).

Catfish (channel, *Ictalurus punctatus*, and blue, *I. furcatus*). These omnivorous, bottom-dwelling fish have a relatively high lipid content. They are a popular food fish, particularly for subsistence fishers. Fish with a high lipid content, such as catfish, generally accumulate higher concentrations of hydrophobic contaminants (e.g., PCBs) than other fish. For this reason, in addition to measuring gamma-emitting radionuclides, the Phase 1 study included analyses for a number of pesticides, PCBs, and semivolatile organic compounds in catfish. The Phase 1 study also measured levels of ^{90}Sr in catfish vertebrae. The Phase 2 study focused on pesticides (primarily chlordane) and PCBs (congeners and Aroclors). In addition, because much of the semivolatile data collected during the Phase 1 study were rejected for quality assurance/quality control reasons, a limited number of catfish were also analyzed for these compounds in Phase 2 to confirm their general absence in fish tissue. The only metals data collected for catfish during the Phase 2 investigation were in reach 4.

Striped bass (including the striped bass X white bass hybrid *Morone saxatilis* X *M. chrysops*). These large piscivores are known to range throughout Watts Bar Reservoir on a seasonal basis (Cheek et al. 1985). They are a popular sportfish. Because of their wide range, they are expected to integrate contaminant exposure over a large area. Striped bass were not collected during the Phase 1 study, and Phase 2 analyses were restricted to PCBs (congeners and Aroclors) and pesticides (chlordane).

Other species that were sampled sparingly include carp (*Cyprinus carpio*) and white crappie (*Pomoxis annularis*). Because sample sizes were limited, neither species is evaluated here.

Fish were usually analyzed as individual fillet samples, but in some cases, five fish were combined into a single composite sample. Catfish were skinned, whereas other species were scaled and analyzed with the skin on. The size of the fish collected were within the range typically harvested by fishermen. Bluegill and largemouth bass were collected in the spring; the other species were, with a few exceptions, collected in the fall. Samples were analyzed by standard analytical methods identified in the Phase 2 sampling plan (DOE 1993b).

The statistical analysis of contaminant data used nonparametric methods (SAS LIFETEST procedure) to estimate the mean analyte concentration. These methods take into account nondetects by testing for the equality of the analyte distribution function from the different sites. For each species, not all analyte-site-year combinations were represented; as a result it was very difficult to evaluate any temporal effect. Typically, were sufficient data available, including the variable "year" as a factor in the ANOVA would have yielded more power in discriminating between sites. However, because of the large number of empty cells that would otherwise be created, this variable was not included in the analysis. By ignoring the temporal factor, the analysis conducted here is valid only when this factor is not significant, and it may fail to detect a site effect that would have otherwise been detected. On the other hand, one can be reasonably confident that some type of site effect or temporal effect exists when detected. Where a significant site effect ($p > 0.05$) was found, individual pairwise comparisons (z-test of independent means) were made between sites of a priori interest (e.g., study sites, upstream reference sites) by averaging the means over the available sampling periods to determine if these mean values were significantly different. The results of these comparisons should be interpreted cautiously, because the means may be averaged over different sampling periods.

3.5.2.2 Results

A number of analytes, primarily the semivolatile organic compounds, were undetected during the study (Table D2). Other compounds, primarily specific PCB congeners, were undetected in one or more species at any location (Table D2). Results of the statistical anlysis for the remaining analytes are summarized in Tables D3 (bluegill sunfish), D4 (largemouth bass), D5 (catfish), and D8 (striped bass and other species) and are discussed in the following sections.

Bluegill sunfish. Of the five metals that were identified in the Phase 1 study as potential contaminants of concern, site-related patterns were evident for Hg, Be, and As. Mercury levels in bluegill were significantly elevated at all Poplar Creek stations downstream of EFPC and in the Clinch River immediately below Poplar Creek (subreach 4.01). The mean concentration was greatest in subreach 3.01 (0.49 mg/kg), and concentrations declined as a gradient from that point downstream (Fig. 3.48). In the lower Clinch River (subreach 4.04), levels were no longer significantly different from reference values. Mean mercury levels did not exceed the FDA action level (1.0 mg/kg) at any location, and the UCL_{95} and maximum concentration were less than this value at all sites (Table D3).

Mean beryllium concentrations appear to be highest in subreaches 3.02 and 3.03 of Poplar Creek (Fig. 3.46), but drop to reference levels in subreach 3.04. Beryllium was undetected at the Poplar Creek reference site (reach 13); however, only three fish were collected from this site, all during Phase 1.

Arsenic concentrations were significantly elevated in bluegill from Poplar Creek (subreaches 3.01, 3.03, and 3.04) in comparison with concentrations at reach 13 (Fig. 3.47). Mean concentrations were greater in subreaches 3.01 (0.096 mg/kg) and 3.04 (0.085 mg/kg) than at any other location sampled. The lowest concentrations were found in bluegill from reach 18 (0.021 mg/kg).

Copper was detected infrequently in bluegill; >60% nondetects occurred at all sites. Interestingly, the highest concentrations of selenium were found in fish from Melton Hill Reservoir (reaches 0 and 1), upstream of almost all potential ORR sources. Mean levels (0.75–0.86 mg/kg) in these reaches were significantly greater than any of the reference values.

The Phase 1 data demonstrated that ^{137}Cs activity was significantly elevated in bluegill collected in the Clinch River below the mouth of WOC (reach 2). Mean concentrations (2 pCi/g) were roughly two orders of magnitude greater than those at upstream reference sites (Fig. 3.49). No other gamma-emitting radionuclides exhibited a site-related pattern.

Largemouth bass. The overall pattern of metals contamination in largemouth bass resembles that of bluegill. Mercury concentrations in bass were greatest in Poplar Creek and immediately downstream of its mouth in the Clinch River (Fig. 3.51). Mean concentrations were significantly greater in all Poplar Creek subreaches than in Melton Hill Reservoir or in the Clinch River below Melton Hill (reach 2). Except for subreach 3.01, where a limited sample size resulted in a high standard error, mean concentrations in bass at all Poplar Creek subreaches were significantly higher than the Norris Reservoir reference concentration as well. The UCL_{95} concentration in bass was below the FDA action level (1.0 mg/kg) in all reaches (Table D4), but seven fish from Poplar Creek had mercury levels in excess of this amount. Neither selenium nor copper showed ORR site-related patterns. Copper was again detected infrequently. The greatest rate of detection for copper was 67% in subreach 4.04; in all other subreaches, nondetects accounted for at least 85% of the data. Selenium concentrations were again highest in Melton Hill Reservoir. Beryllium concentrations were highest in bass from Poplar Creek (subreaches 3.01 and 3.02).

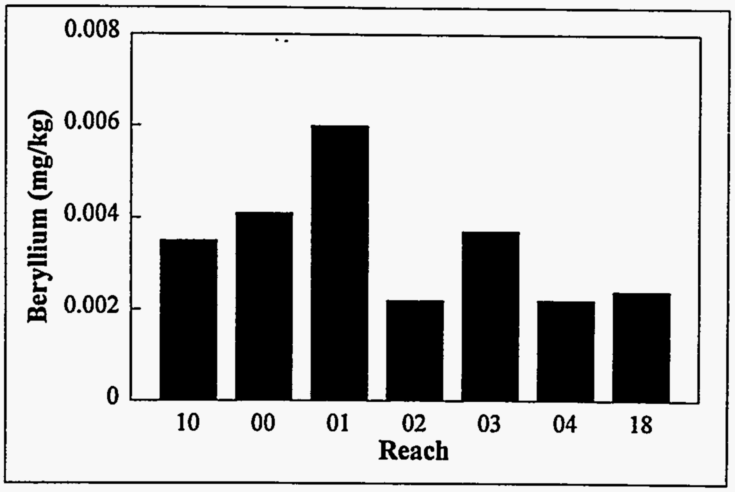


Fig. 3.46. Mean beryllium concentrations in bluegill sunfish fillets.

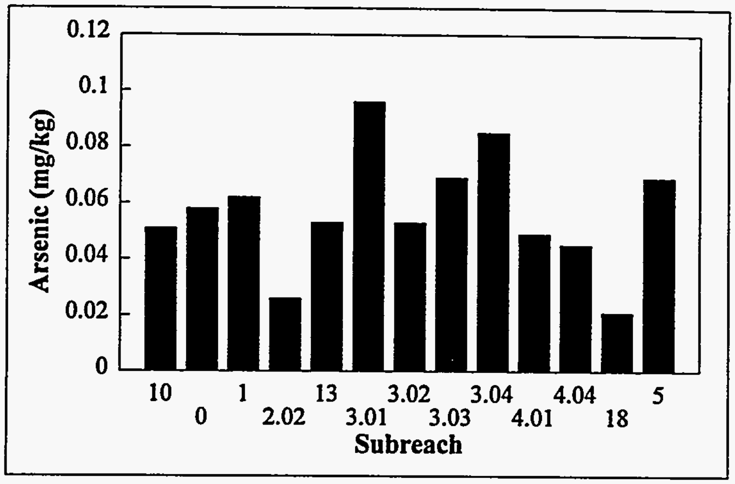


Fig. 3.47. Mean arsenic concentrations in bluegill sunfish fillets.

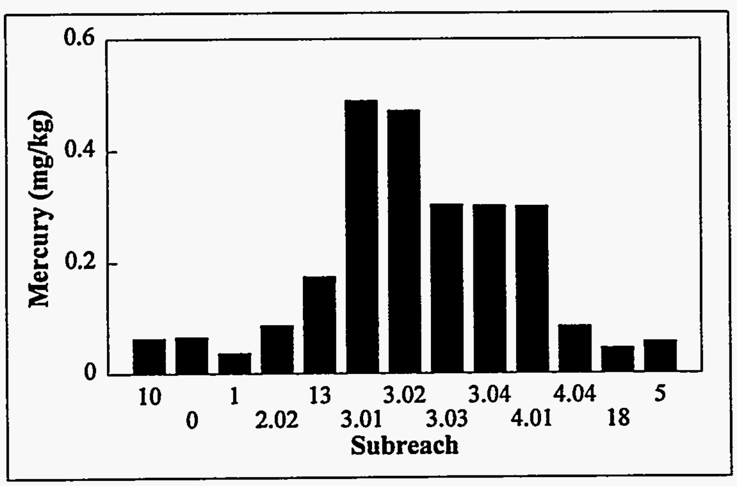


Fig. 3.48. Mean mercury concentrations in bluegill sunfish fillets.

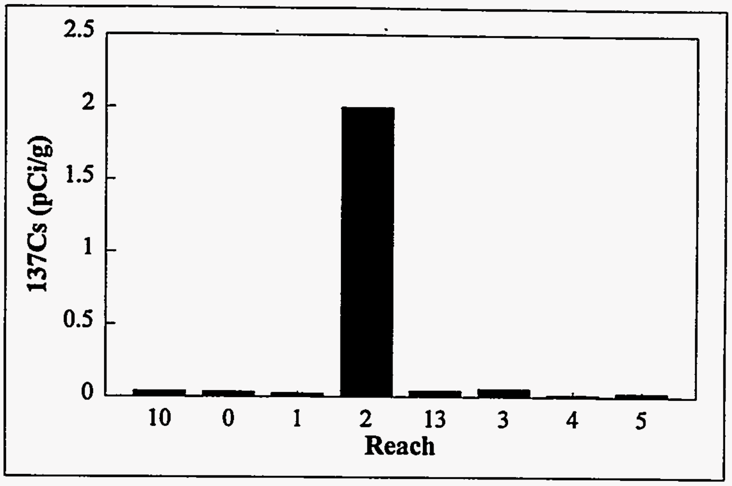


Fig. 3.49. Mean ^{137}Cs activity in bluegill sunfish.

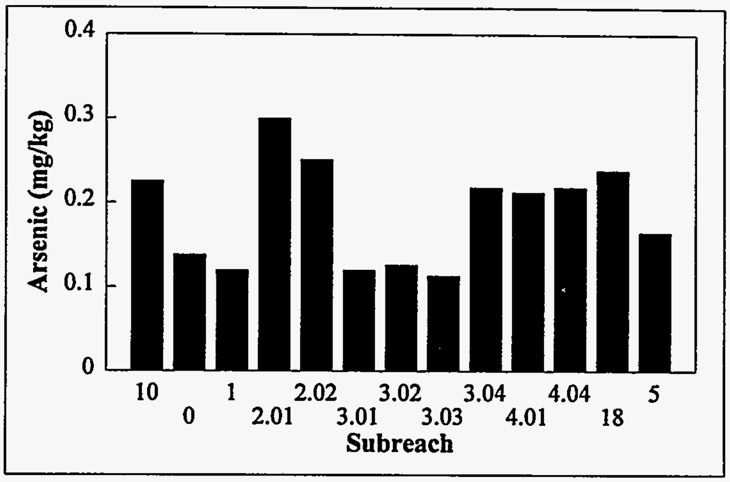


Fig. 3.50. Mean arsenic concentrations in largemouth bass.

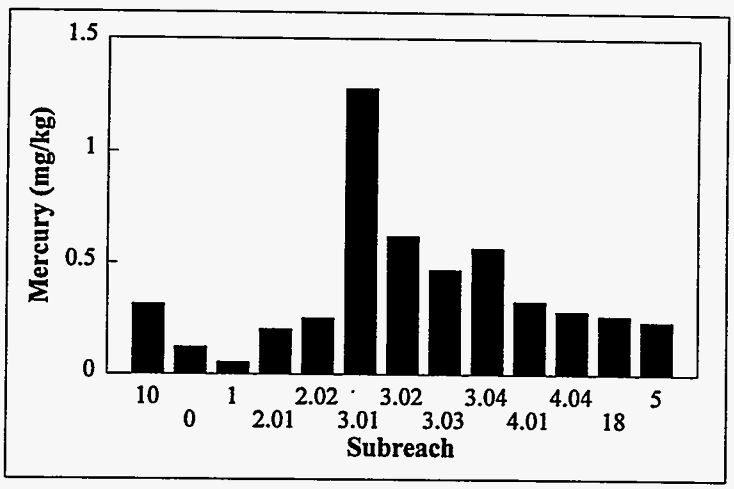


Fig. 3.51. Mean mercury concentrations in largemouth bass.

Contrary to the pattern in bluegill sunfish, mean arsenic concentrations in bass were highest in the upper Clinch River (reach 2), whereas some of the lowest mean concentrations were observed in Poplar Creek (Fig. 3.50). Mean concentrations in bass from reach 2 were significantly elevated in comparison with concentrations in bass from Melton Hill Reservoir but not with concentrations in bass from Norris Reservoir or from the reference arm of Watts Bar Reservoir (reach 18).

Again, as with bluegill, the mean concentration of ^{137}Cs (5 pCi/g) in bass from reach 2 was two orders of magnitude higher than that in any of the upstream reference reaches (Fig. 3.52).

Of the seven PCB Aroclors for which analyses were conducted, only Aroclors 1254 and 1260 were detected in largemouth bass samples (see Table D2 for nondetects). The highest mean concentrations of both Aroclors are found in Poplar Creek; whereas the lowest are found in Norris Reservoir (Fig. 3.53). The mean concentration of Aroclor 1254 in reach 3 (0.31 mg/kg) was significantly higher than in any other reach. The mean concentration of Aroclor 1260 in reach 3 (0.29 mg/kg), although higher than in most reaches, was not statistically different from mean values in either

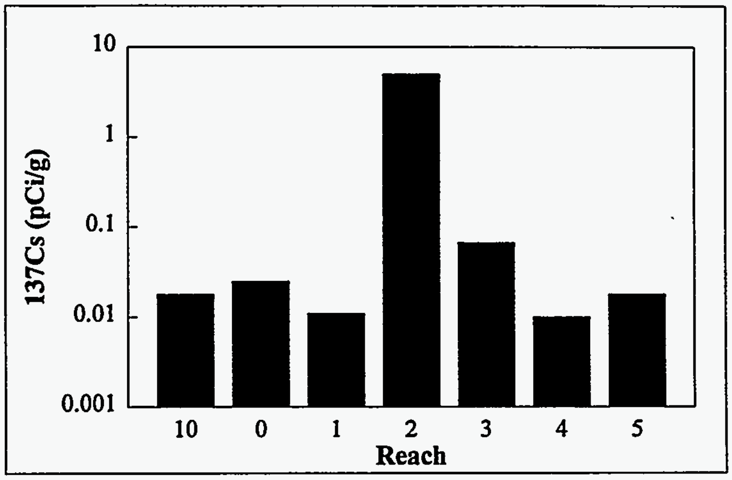


Fig. 3.52. Mean ^{137}Cs activity in largemouth bass.

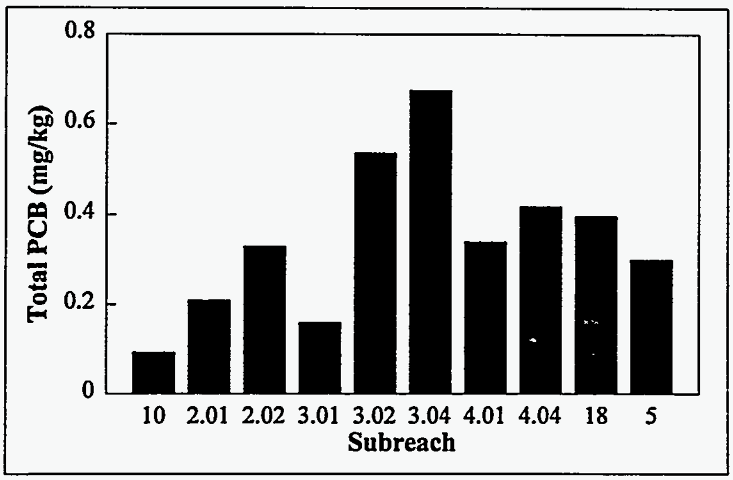


Fig. 3.53. Mean total PCB concentrations in largemouth bass.

the lower Clinch River (reach 4) or the reference arm of WBR (reach 18). Both the UCL_{95} and the maximum values for total PCBs (sum of Aroclor 1260 and 1254) are below the FDA action level (2.0 mg/kg) at all locations (Table D4). The chlordane data did not reveal any site-related patterns (Fig. 3.54). The UCL_{95} concentration for total chlordane in bass was less than the FDA action level (1.0 mg/kg) at all locations.

Catfish. Consistent with both the bluegill and largemouth bass data, mean ^{137}Cs activity in catfish was approximately an order of magnitude greater in the Clinch River below WOC (subreach 2.02) than in upstream reference reaches (Fig. 3.55). Mean ^{90}Sr activity in vertebrae of catfish from subreach 2.02 was ~3 times greater than in fish from Norris Reservoir.

No site-related spatial pattern was evident for chlordane in catfish (Fig. 3.56). Mean concentrations were highest at lower Melton Hill, WOC Embayment, and upper Clinch River (subreach 2.02) sites and did not differ significantly among these sites.

Only three PCB Aroclors were detected in catfish during the study, one of which (Aroclor 1016) was detected only once in 277 samples. Total PCB concentrations (the sum of the remaining Aroclors —1254 and 1260) were lowest in Norris and Melton Hill reservoirs and in upper Poplar Creek (Fig.3.57). The highest concentrations were found in the WOC embayment. However, this site was sampled during Phase 1 only and has since been the focus of a CERCLA removal action during which a weir was constructed across the mouth of the embayment to restrict movement of contaminated sediment into the Clinch River (see Chap. 2).The WOC embayment is no longer included in the CR/PC OU. Catfish from the upper Clinch River immediately downstream of WOC (subreach 2.02) had the second highest total PCB concentrations. Concentrations decline downstream in the Clinch River but increase again in subreach 4.04, where levels are comparable to those in the lower Emory River (reach 6) and the Tennessee River (reaches 18 and 5). Concentrations of both Aroclor 1254 and 1260 in one or more subreaches of Poplar Creek were significantly higher than in reach 13. Mean concentrations of total PCBs generally increased in the downstream direction in Poplar Creek, except for a slight drop in concentration in subreach 3.03. Maximum total PCB

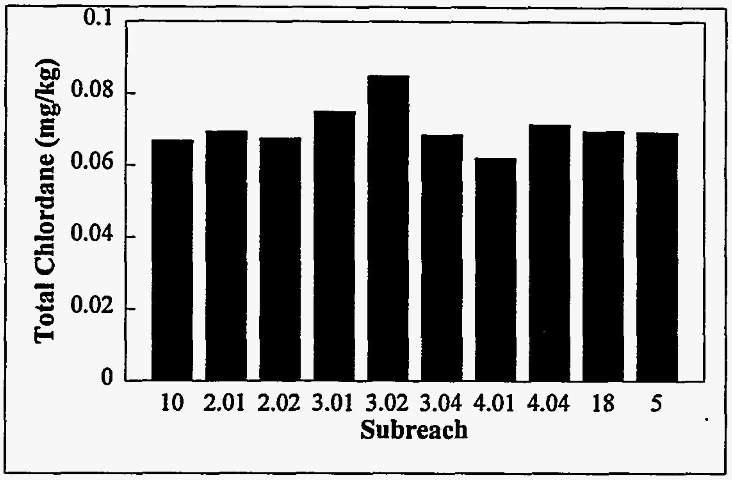


Fig. 3.54. Mean total chlordane concentrations in largemouth bass.

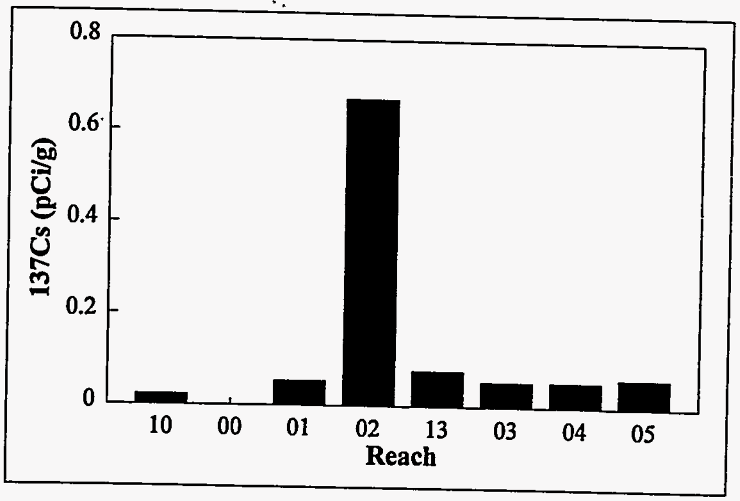


Fig. 3.55. Mean ^{137}Cs activity in catfish.

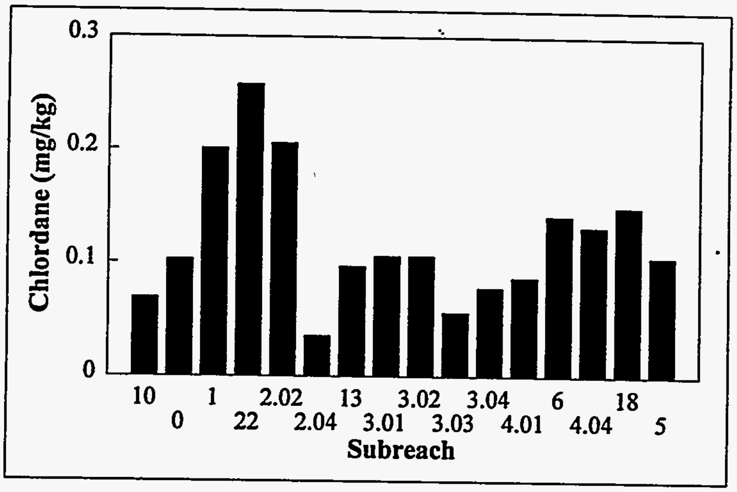


Fig. 3.56. Mean total chlordane concentrations in catfish.

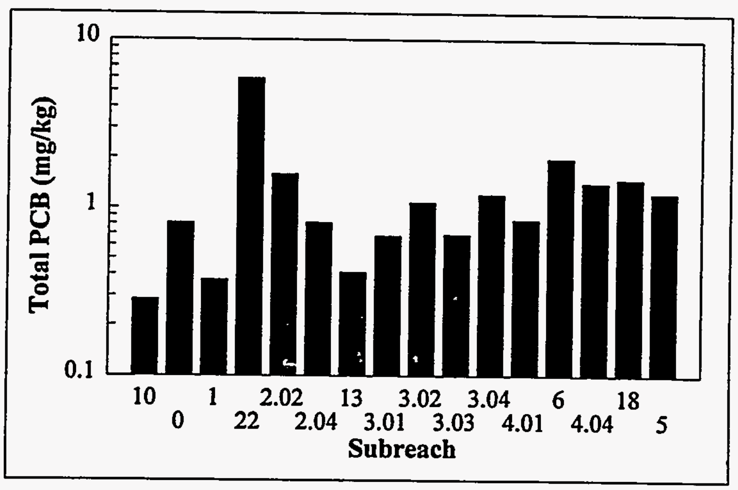


Fig. 3.57. Mean total PCB concentrations in catfish.

concentrations in individual fish exceed the FDA action limit (2.0 mg/kg) in reaches 22, 2, 3, 4, 18, and 5. The mean concentration exceeded this criterion only in the WOC embayment (Table D5).

As noted above, ten catfish from one location (Poplar Creek) were analyzed for semivolatile organic compounds to verify the low concentrations observed in the Phase 1 data, which for reasons of quality assurance needed to be confirmed. Most analytes were undetected (Table D2). Several phthalate esters and the compound napthalene were detected at low concentrations in several fish (Table D5).

Striped bass. Striped bass were collected from four reaches: Norris Reservoir (reach 10), reaches 2 and 4 in the Clinch River, and in lower Watts Bar Reservoir (reach 5). Mean concentrations of chlordane, Aroclor 1254, and Aroclor 1260 were lowest in Norris Reservoir and substantially higher in the Clinch River and lower Watts Bar Reservoir (Fig. 3.58). None of the data indicate a strong site-related effect, but the absence of data from several of the study reaches (particularly Melton Hill Reservoir) limits interpretation of the data. Mean and UCL_{95} concentrations did not exceed the FDA action level for total PCBs of 2.0 mg/kg in any reach. Only one fish (from reach 5) had a concentration in excess of this value.

3.5.2.3 Other characterization considerations

Contaminant levels in fish are known to vary widely from year to year and from one species to another at the same site. An ANOVA of the Phase 2 data was conducted to evaluate the effects of sampling location, year, and species on contaminant concentrations. Because this analysis was was based on an unbalanced design with missing cells, the conclusions should be regarded with caution. Where a significant main effect ($\alpha = 0.05$) was observed, Scheffe's multiple comparisons test was used to identify significant differences between individual means (sites or species).

Figure 3.59 summarizes the Phase 2 mercury data. As was the case with the single species analysis (Sect. 3.5.2.2) for bluegill and largemouth bass, sites in Poplar Creek had the highest mercury concentrations. Among species, mean concentrations were greatest in largemouth bass and smallest in bluegill. Significant interactions between the year and species effects indicate that concentrations do not vary in the same fashion among species from year to year.

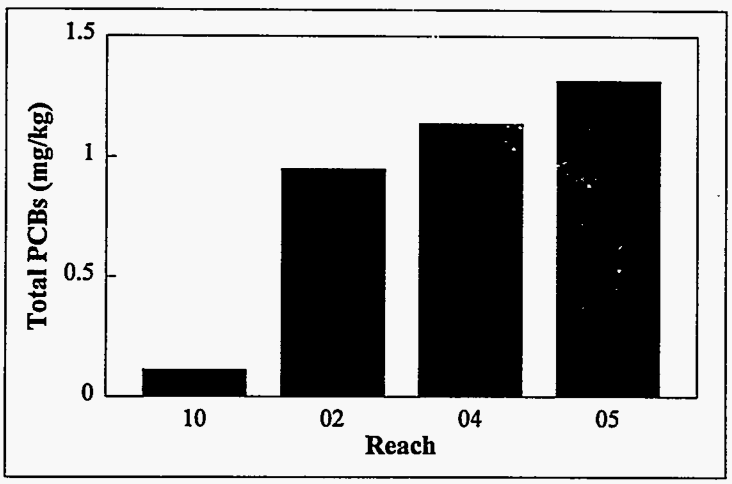


Fig. 3.58. Mean total PCB concentrations in striped bass.

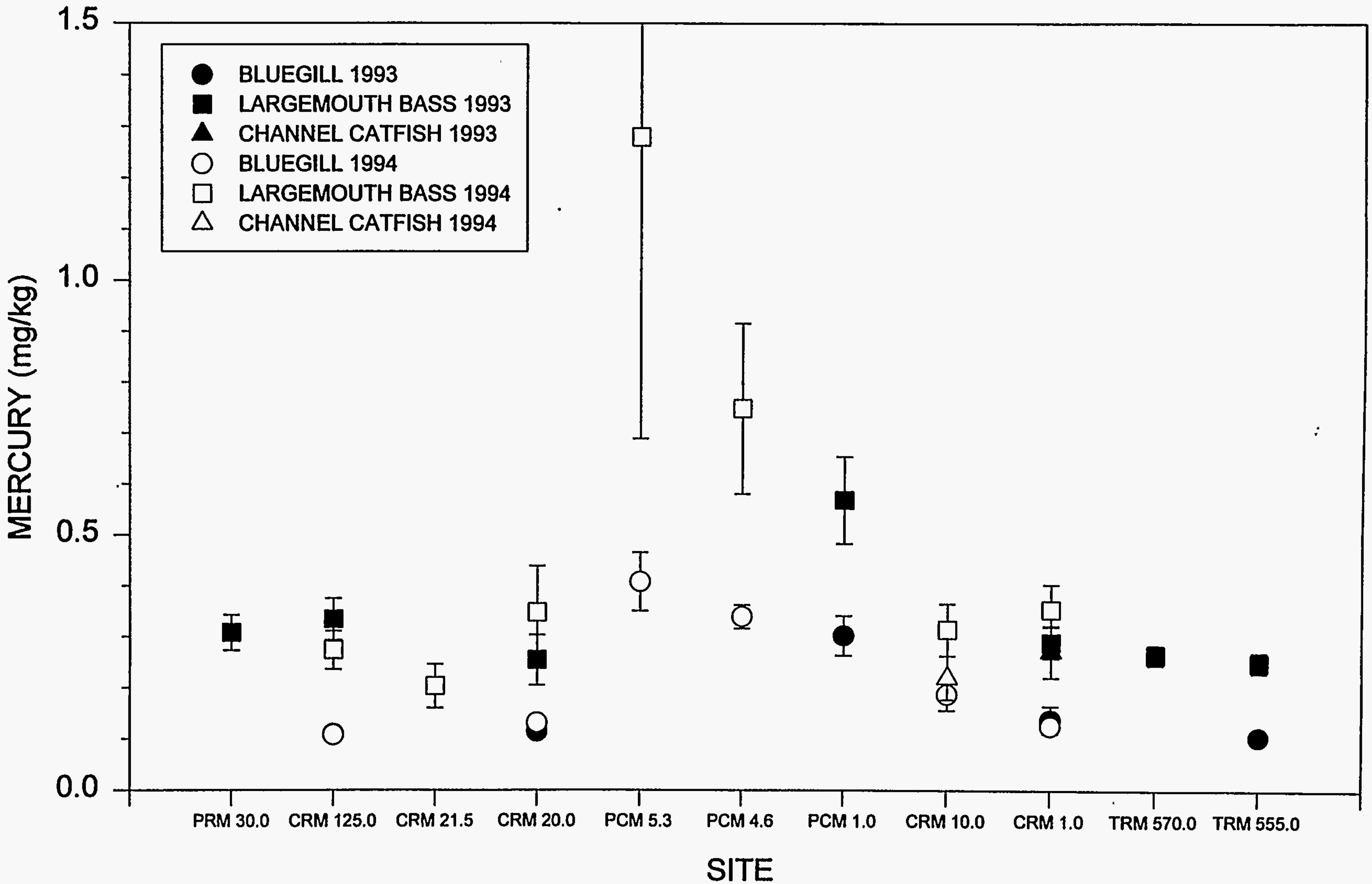


Fig. 3.59. Mercury concentrations in largemouth bass, bluegill, and catfish fillets, Clinch River Remedial Investigation Phase 2 data (mean + SE).

The ANOVA of the remaining metals data (As, Be, Se, and Cu) indicated significant site, year, and species effects, but in each case, significant interactions between main effects variables indicate that the mean concentrations of the analytes did not consistently vary among species-site-year combinations.

The total PCB data are summarized in Figs. 3.60 (catfish) and 3.61 (remaining species). The ANOVA indicated significant site and species effects. In general, the spatial patterns for all species were similar. The lowest concentrations for all species were found in Norris Reservoir and tended to increase downstream, with generally increased levels in the lower Clinch River and the Tennessee River. Each species for which data are available exhibited a relatively sharp increase in mean concentrations in the Clinch River immediately below WOC in comparison with upstream values. As expected, catfish had the highest mean concentrations of PCBs among the species sampled. Concentrations in striped bass averaged 78% that of catfish collected at the same location, and concentrations in largemouth averaged 37% that of catfish. Concentrations in composite samples of carp, bluegill, and white crappie collected at CRM 1.0 were also a fraction (45%, 38%, and 17%, respectively) of that found in catfish from the same location. No year effect was found even when the analysis was restricted to the catfish data alone. Although substantial variation exists at any site from year to year, the trend is not consistent among sites. These local temporal effects may be the result of local short-term changes in PCB loading to the system, remobilization of PCB-contaminated sediment, or changes in the aquatic community and food chain dynamics.

ANOVA for total chlordane indicated significant effects for each of the main variables and also significant interaction between each variable. As with the single species analysis, although there were spatial effects, the spatial pattern did not appear strongly related to the ORR. Overall, concentrations were highest in catfish and striped bass, whereas those in largemouth bass were lower.

3.5.3 Supporting Studies

A number of studies involving biota were conducted in support of the CRRI Phase 2 investigation. Most were designed to gain insight into specific contaminant issues relevant to the ecological or human health risk assessment. These data are presented here and discussed briefly, but the interpretation and full use of these data is deferred until Chap. 5 (human health risk assessment) and Chap. 6 (ecological risk assessment). Supporting studies include the analysis of contaminants in the food web (Sect. 3.5.3.1), the extent and effects of contaminant loading in great blue heron (Sect. 3.5.3.2), the effects of contaminant loading in mink (Sect. 3.5.3.3), the analysis of bioindicators in fish (Sect. 3.5.3.4), the analysis of reproductive indicators in fish (Sect. 3.5.3.5), a spatial and temporal analysis of PCB congeners in catfish (Sect. 3.5.3.6), and the nature and extent of contamination in geese (Sect. 3.5.3.7).

3.5.3.1 Contaminants in the food web

The goal of this task was to identify important sources and pathways of contaminant transport, movement, and bioaccumulation within the Clinch River–Poplar Creek food web. The study consisted of collecting mayflies and shad from several locations for contaminant analysis. The resulting data are used in the ecological risk assessment to directly assess risk to selected insectivores and piscivores.

Adult emergent mayflies (*Hexagenia* sp.) were collected from three locations: PCM 2.0 in 1993 and 1994; CRM 1.0 in 1993 and 1994; and TRM 570 in 1993. Several dozen mayflies were collected and composited into three samples at each site and analyzed for Aroclors 1260 and 1254, pesticides, and five metals (As, Pb, Hg, Ag, and Zn).

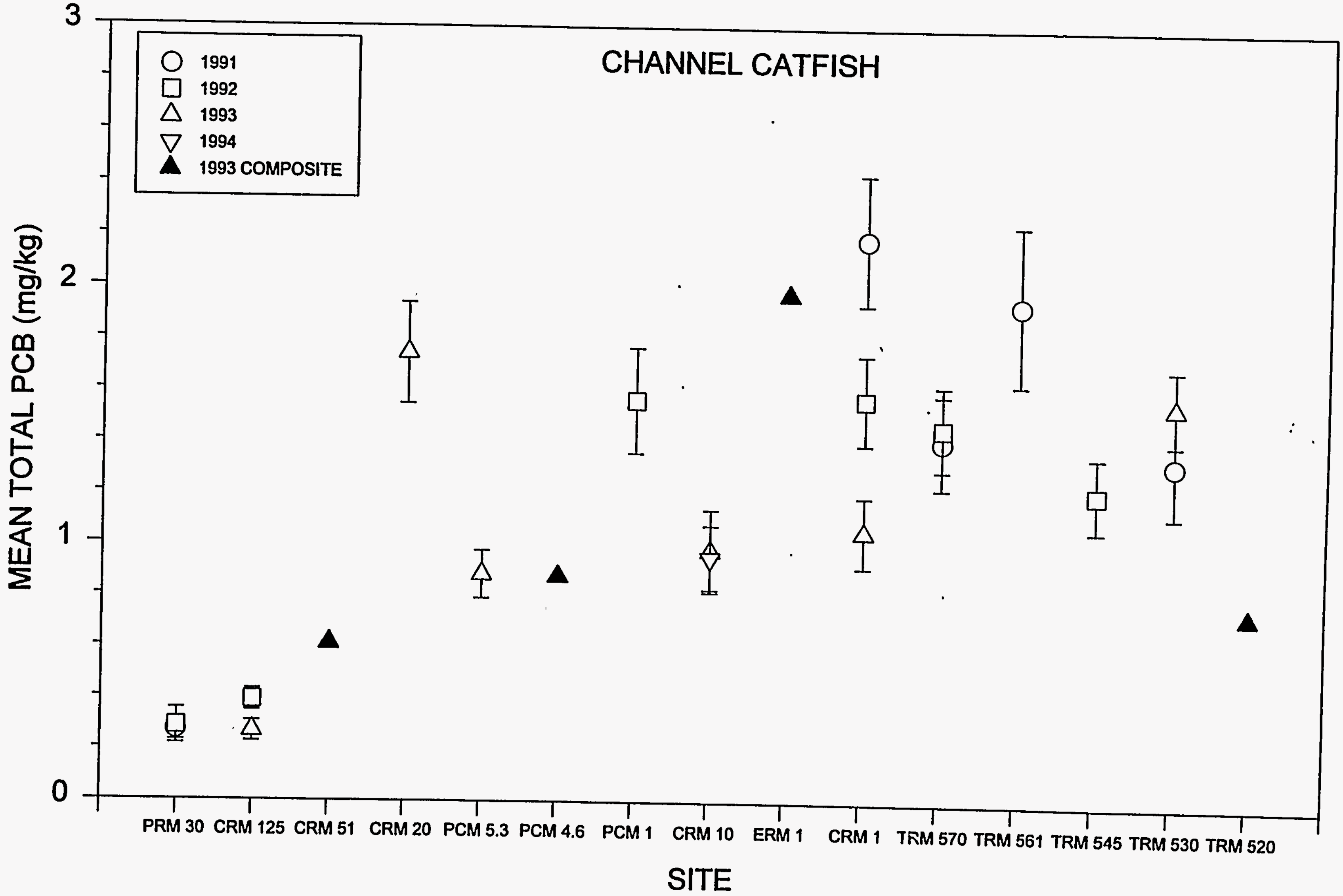


Fig. 3.60. Total PCB concentrations in catfish, Clinch River Remedial Investigation Phase 2 data (mean + SE).

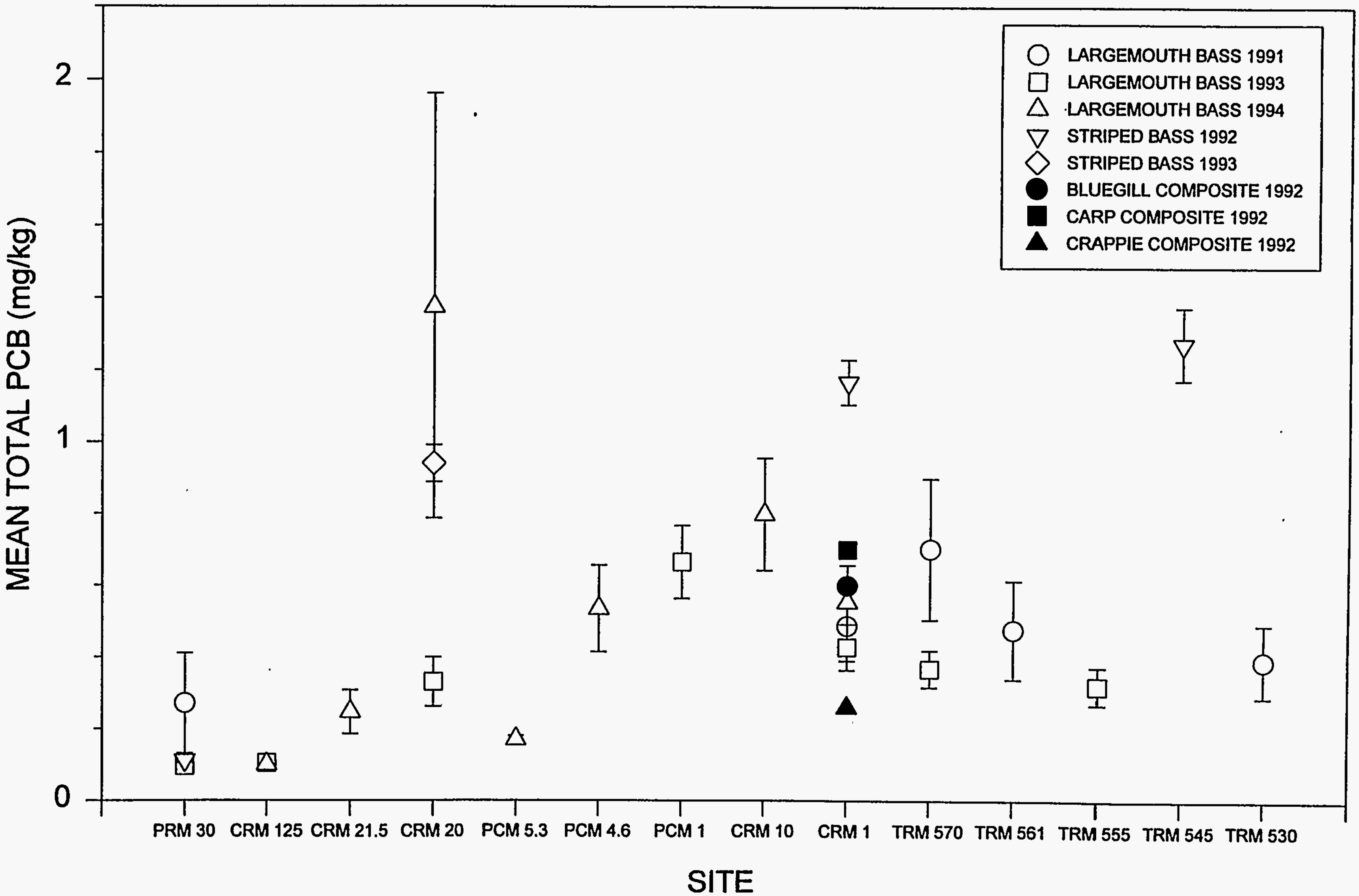


Fig. 3.61. Total PCB concentrations in striped bass, largemouth bass, and other species, Clinch River Remedial Investigation Phase 2 data (mean + SE).

Mercury levels were significantly elevated in mayflies from Poplar Creek in comparison with the other two sites (Table D7). Of the other metals, silver was undetected at any location, and concentrations of the other metals did not differ significantly among sites.

No pesticides were detected in any mayfly samples. Of the PCBs, only Aroclor 1254 was detected (Table D7); mean values at all sites were <0.15 mg/kg and were lowest in the Poplar Creek samples (Fig. 3.62).

Shad (gizzard, *Dorosoma cepedianum*, and threadfin, *D. petenense*) are a primary food for striped and largemouth bass in WBR. Shad were collected from 12 sites and analyzed whole for PCBs and pesticides. Total chlordane concentrations in whole shad did not differ significantly among sites. As with the other fish species (Sect. 3.5.1), total PCB concentrations in shad exhibit a great deal of among-year and among-site variability (Fig. 3.62). On average, total PCB concentrations in shad were roughly 25% that in catfish and striped bass fillets and 70% that in largemouth bass fillets.

This supporting study demonstrated that both mayflies and shad have accumulated body burdens of certain contaminants and therefore represent routes of contaminant exposure for higher trophic organisms. Mayflies and other aquatic insects likely provide one of many mechanisms for the transport of contaminants into fish from the sediment, whereas shad are likely the most important pathway for PCB contamination of piscivorous fish.

3.5.3.2 Contaminant accumulation and effects in great blue heron

Suter (1990) identified that piscivorous wildlife along the Clinch River are potentially at risk from toxicant releases from the ORR. The current study tried to determine if the great blue heron (*Ardea herodias*) populations near the ORR suffer contaminant-induced effects, particularly with respect to mercury and PCB exposure. It was especially important to determine if contaminant exposure has resulted in impaired reproductive capacities, which in turn could threaten the viability of the population. The great blue heron was chosen as an indicator species because it (1) is predominantly piscivorous, foraging along the major waterways on and downstream of the ORR; (2) is at the top of the aquatic food chain; (3) has been suggested to be a good indicator of aquatic health; (4) is well represented in the scientific literature, including the toxicological literature; and (5) satisfies necessary logistical sampling considerations. Important logistical considerations were the presence of study and reference colonies on or in proximity to the ORR and population densities adequate for sampling requirements. In addition, great blue heron are highly visible, facilitating direct observation and site location.

Heron chicks and eggs were collected from four colonies from March 1992 through June 1994. Chicks and eggs were processed and analyzed for several metals and PCBs (Aroclors and individual congeners), and a number of physiological parameters were also measured. Egg and chick collection and processing techniques are described in "Heron Monitoring and Collection Procedures" (Energy Systems 1994) and "Heron Necropsy, Egg Examination, and Tissue Analysis" (Energy Systems 1995).

Two of the colonies are located within 1.8 miles of the ORR (at the K-25 Site and at Melton Hill Reservoir). Herons at these locations are potentially exposed to contaminants occurring on the reservation; these colonies were therefore considered to be on-site. The remaining colonies (at Long Island and Looney Island) are located in the Tennessee River (reach 18) >6 miles from the ORR. Heron in these off-site colonies were assumed not to be exposed to contaminants from the ORR. It should be noted, however, that many of the analyses revealed the Melton Hill colony to be significantly less contaminated than the K-25 colony; therefore, most of the on-site to off-site comparisons that follow are comparisons of the K-25 colony and the two off-site colonies.

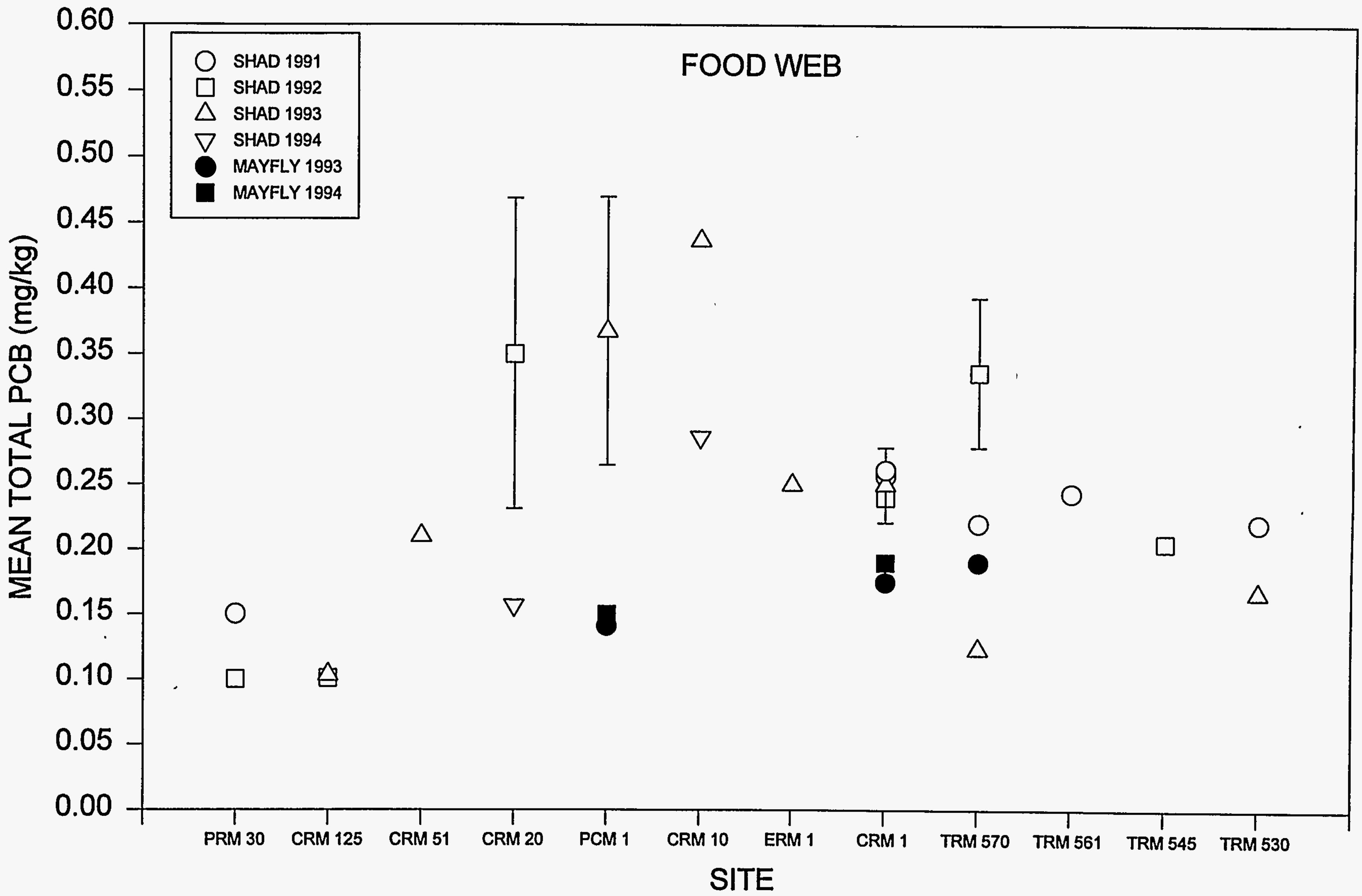


Fig. 3.62. Mean total PCB concentrations (mg/kg) in shad and mayflies, Clinch River Remedial Investigation Phase 2 sampling, 1991–1994.

In addition to chicks and eggs from each colony, fish regurgitated from chicks or found beneath nests were collected for PCB and mercury analyses. Mercury and chromium concentrations were significantly greater in fish collected from colonies located on the ORR than in colonies located off the ORR (Table N1). No significant differences in PCB concentrations existed between on-site and off-site colonies.

The mean mercury concentration in eggs collected on the ORR was significantly greater than the mean concentration in eggs collected off the ORR (Table N2). Overall, the mean concentration was greatest in eggs collected from the K-25 colony and least in the Melton Hill colony. The mean chromium concentration in eggs collected on the ORR was significantly greater than the mean concentration in eggs collected off the ORR (Table N2). Mean chromium concentrations were greatest in eggs collected from the Melton Hill colony (0.22 mg/kg) followed in decreasing order by concentrations in the K-25 (0.15 mg/kg), Long Island (0.11 mg/kg), and Looney Island (0.11 mg/kg) colonies. Arsenic was quantified in only one egg, and the concentration was below the contract required detection limit.

Mean mercury concentrations were significantly greater ($p < 0.05$) in feathers and liver tissue of chicks collected on the ORR in comparison with those collected off the ORR (Table N3). No significant differences existed in mean muscle concentrations between colonies located on or off the ORR. No significant differences in mean liver, muscle, or feather concentrations of arsenic or chromium were detected between chicks collected on and off the ORR (Table N3).

Mean concentrations of Aroclor 1260 and 10 congeners were significantly greater in eggs collected on the ORR in comparison with concentrations in eggs collected off the ORR (Table N4). Overall, concentrations were greatest at the K-25 site. Aroclor 1260 concentration was significantly greater ($p < 0.05$) in fat, liver, and muscle tissue from chicks collected on the ORR in comparison with concentrations in chicks collected off the ORR (Tables N5–N7). Although concentrations of all congeners were greater in fat tissue from chicks collected on the ORR, this difference was significant in only 30% of the congeners quantified.

Chick weight/length ratios, liver somatic indexes, and hematocrit measurements did not differ between on-site and off-site colonies. Paradoxically, mean values for two indicators of contaminant exposure, liver EROD (7-ethoxyresorufin O-deethylase) activity and DNA F values (fraction of double stranded DNA), were significantly greater ($p < 0.05$) in chicks collected from colonies off the ORR compared to those collected on the ORR (Table N8).

No significant differences between on-site and off-site colonies were observed in the number of eggs or chicks per nest (Table N8). The mean weight of eggs collected from on-site colonies was significantly heavier than that of eggs from off-site colonies; however, there was no difference in shell thickness.

Herons occupying colonies on the ORR have significantly elevated body burdens of chromium, mercury (K-25 Site only), and PCBs (K-25 Site only) in comparison with herons occupying colonies off the ORR. However, neither the number of eggs laid per nest nor the survival rate of chicks to fledging differed between on-site and off-site colonies. These results are further evaluated in the ecological risk assessment (Chap. 6), along with other pertinent data, to determine whether heron populations at the ORR are at risk.

3.5.3.3 Reproductive performance of mink

Mink (*Mustela vison*), a mammalian piscivore, was also evaluated for determining possible contaminant-related effects on local populations. Mink were selected for study because they are sensitive to exposure to PCBs (Aulerich and Ringer 1977) and mercury (Calabrese et al. in press) and are known to inhabit the ORR. However, mink are secretive and population densities tend to be low, making assessment of effects in natural populations difficult. Therefore, the potential for effects at the ORR was assessed through the use of a controlled feeding study. The study sought to evaluate the correlation between contaminant bioaccumulation and reproductive effects in mink that were fed fish collected from three sources: (1) the ORR (Poplar Creek below EFPC), (2) the Clinch River above the ORR (above Melton Hill Dam), and (3) the ocean (mackerel from a commercial supplier). The species composition of fish collected from the ORR and the Clinch River above Melton Hill Dam were similar and consisted mostly of benthic species (Table N9).

Fish from each location were homogenized and ten aliquots of homogenate collected for contaminant (mercury and PCBs) analyses. Mean mercury concentrations in fish were significantly different among locations; the concentrations were greatest in fish from Poplar Creek and least in ocean fish (Table N10). Mean (lipid-adjusted) Aroclor 1260 concentrations exhibited an identical pattern (Table N11), as did most of the individual PCB congeners (Table N11).

Five diets, composed of 75% fish and 25% normal mink diet, were prepared. The fish portion of two reference diets consisted entirely of ocean fish (diet A) and of Clinch River fish (diet B). The fish portion of the remaining 3 diets (diets C, D, and E) contained 33, 67, and 100% fish collected from Poplar Creek and 67, 33, and 0% ocean fish, respectively.

Ten aliquots of each diet were collected for contaminant (mercury and PCB) analyses. The mean mercury concentration in each diet differed significantly from all other diets (Table N10), increasing progressively from diet A through diet E. Aroclor 1260 concentrations in diets B and D did not differ significantly; otherwise, mean concentrations in all diets differed (Table N12). Mean concentrations were greatest in diet E, less in diets B and D, still less in diet C, and least in diet A. Many individual PCB congeners exhibited the same pattern.

Fifty adult natural dark mink from the Michigan State University Experimental Fur Farm were randomly divided into five groups with two males and eight females in each group. Each mink group was fed one of the prepared diets from December 1, 1993, through approximately June 30, 1994. Mating began March 1, 1994, and was confined within the respective groups. Kits were whelped by mid-May.

At the conclusion of the study, mercury concentrations in liver, kidney, and hair of adult female mink were found to increase progressively in mink fed diets A through E (Table N13), corresponding to mercury concentrations in fish and in diets. Mercury concentrations in kit kidney tissue and homogenized carcass were not significantly different in offspring of mink fed diets A, B, or C, but were significantly greater ($p < 0.05$) in offspring of mink fed diet E (Table N13).

The mean Aroclor 1260 concentration was significantly greater in liver tissue of female mink fed diet E (Table N14). Mean Aroclor 1260 concentration in fat tissue from female mink fed diet E were significantly greater than mean concentrations from female mink fed diets A or B (Table N15). Concentrations of many PCB congeners exhibited similar patterns. Mean Aroclor 1260 concentrations in liver tissue from 6-week old kits did not differ significantly among diet groups (Table N16), nor did they differ in whole kit carcasses (Table N17).

Mean whole body weights of female mink were not significantly different among diet groups at the beginning of the experiment; however, at its conclusion females in diet group E weighed significantly less ($p = 0.03$) than females in diet group A (Table N18). The mean relative organ weights (organ weights/body weight) of females did not differ significantly among diet groups. At 6 weeks of age, mean whole body weights were significantly lower ($p = 0.004$) in male kits from diet group E than in those from diet group A. A similar trend was observed in 6-week-old female kits, although differences were not statistically significant. Mean relative kidney weights were significantly lower ($p = 0.003$) in kits from diet group B compared with those from diet group E. Kit mean relative liver and spleen weights were not significantly different among diet groups. No histological lesions were attributed to diets.

Mean litter size was significantly reduced ($p = 0.01$) in diet group E in comparison with diet groups A, B, and C but not in diet group D (Table N18). Liver EROD activity, a sensitive biomarker of exposure to PCBs, was significantly increased in adult female mink from diet groups D and E in comparison with those from diet group A (Table N18).

Several effects are noted, particularly for mink whose diet contained the greatest proportion of ORR fish (diet E). Mean concentrations of mercury and PCBs were greater in fish collected from streams located on the ORR, these contaminants were higher in diets that had an increased percentage of ORR fish, and body burdens were correspondingly higher in adult mink that were fed the more contaminated diets. Liver EROD activity was increased in the mink that were fed diets D and E, which contained the greatest proportions of ORR fish. Female mink that were fed diet E weighed less at the end of the study, produced the fewest kits, and their kits weighed the least at 6 weeks of age. The extent to which these results indicate a threat to populations of mink or other piscivorous mammals at the ORR will be further evaluated in the ecological risk assessment (Chap. 6).

3.5.3.4 Bioindicator analysis

Bioindicator analysis was conducted on largemouth bass and bluegill (1) to determine if the health of these species is being adversely affected by contaminants in the Clinch River–Poplar Creek system, (2) to establish baseline conditions for evaluating the effectiveness of any future remedial actions, and (3) to provide information to be used in the ecological risk assessment. The health status of these two species was evaluated by both a functional response group approach (individual response parameters) and an analysis of integrated site responses.

Functional response groups. Individual bioindicator measurements (Tables N19–N22) include five categories: (1) detoxification enzymes, (2) organ dysfunction, (3) histopathology, (4) overall fish health (condition indices), and (5) nutrition and feeding status. In addition, the Health Assessment Index (HAI) was applied to individual fish to provide an overall health profile of the population at each site. These functional groups reflect gradients of both ecological relevance and time-course of responses to stressors such as contaminants (Adams 1990). The variables in categories (1) and (2) are primarily short-term response indicators and have relatively low ecological relevance, whereas the variables in groups (3) through (5) are longer term response indicators and are characterized by lower toxicological but higher ecological relevance.

Detoxification enzymes. The activity of liver detoxification enzymes is used to assess exposure to contaminants; EROD activity is one of the best indicators in field populations (Adams et al. 1992a,b).

Mean EROD activity in largemouth bass was increased at the two lower Poplar Creek sites and in the Clinch River immediately downstream of the mouth of Poplar Creek (CRM 10) in relation to bass from the two reference sites (Fig. 3.63). Bluegill showed a different pattern—only those from CRM 20 near the mouth of WOC demonstrated a significant increase in enzyme activity. Microsomal protein levels showed patterns similar to, but less pronounced than, that of EROD activity in each species. The difference in response between the two species is considered a function of diet—bass consume a diet (principally shad) that is relatively more contaminated than that of bluegill (benthic invertebrates).

Organ dysfunction. Elevated creatinine levels typically indicate kidney malfunction (Tietz 1986). For largemouth bass, creatinine levels at all Clinch River and Poplar Creek sites appear elevated above the reference values but were statistically significant only at PCM 1.0 (Fig. 3.64). Creatinine levels in bluegill were generally similar to reference values; only those sunfish from PCM 1.0 had values that appeared higher.

The enzyme alanine aminotransferase (ALT) is generally used as an indicator of liver damage; elevated levels in humans are a sign of liver cirrhosis, hepatitis, or disease that affects cell integrity. Bluegill from PCM 1.0 had ALT levels that were significantly higher than either reference site (Table N20). Levels of this enzyme in largemouth bass did not suggest liver damage.

Total serum protein concentration, another general indicator of protein metabolism, was significantly lower in largemouth bass from CRM 20 and PCM 1.0 than in reference fish. Bluegill collected from the Clinch River sites and PCM 4.6 all had lower values in comparison with the reference. Abnormally low values of blood protein observed in these fish can reflect anomalies in protein metabolism or nutritional stress (Lockhart and Metner 1984).

Histopathology. Although several indicators of liver pathologies (number of macrophage aggregates, amount of necrotic parenchyma, and abundance of parasites) appeared elevated in bluegill from Poplar Creek and immediately downstream in the Clinch River (CRM 10), most increases were not statistically significant. Similarly, few significant effects were observed for largemouth bass.

The major lesions observed in the liver and spleen tissue of bluegill (Table N23) and largemouth bass (Table N24) were recorded by severity. The average rank of all lesions was used (qualitatively) to indicate the severity of histopathological damage to the major organs of fish at each of the sample sites.

Bluegill from upper Poplar Creek (PCM 5.3 and 4.6) were in the poorest histopathological condition (Table N23). Lower WBR (TRM 530) fish were in the best condition; the condition of fish from the Clinch River was intermediate. It was somewhat surprising that bluegill from CRM 20 (near the mouth of WOC) were in better condition than bluegill 1.5 miles upstream of the WOC mouth (Fig. 3.65).

Largemouth bass from PCM 4.6 were in the poorest histopathological condition, but fish from the other sites in Poplar Creek generally scored better than fish from the Clinch River (Table N24). In contrast to bluegill, bass from CRM 20 had the second poorest overall score. The histopathological condition of largemouth bass improved incrementally with distance downstream from CRM 20 downstream to CRM 1.0 (Fig. 3.65).

Condition indices. The condition factor K 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 stress as a result of contaminants (Adams and Ryon 1994). The K factor was significantly higher for largemouth bass from CRM 1.0 and PCM 1.0 than from the Norris reference; the generally less "plump" fish in Norris Reservoir are primarily explained by its oligotrophic character. Only one site had a significantly reduced K factor; bass from CRM 20 had significantly lower values than bass from either reference site (Fig. 3.66). The K factor was similar for bluegill across all sites.

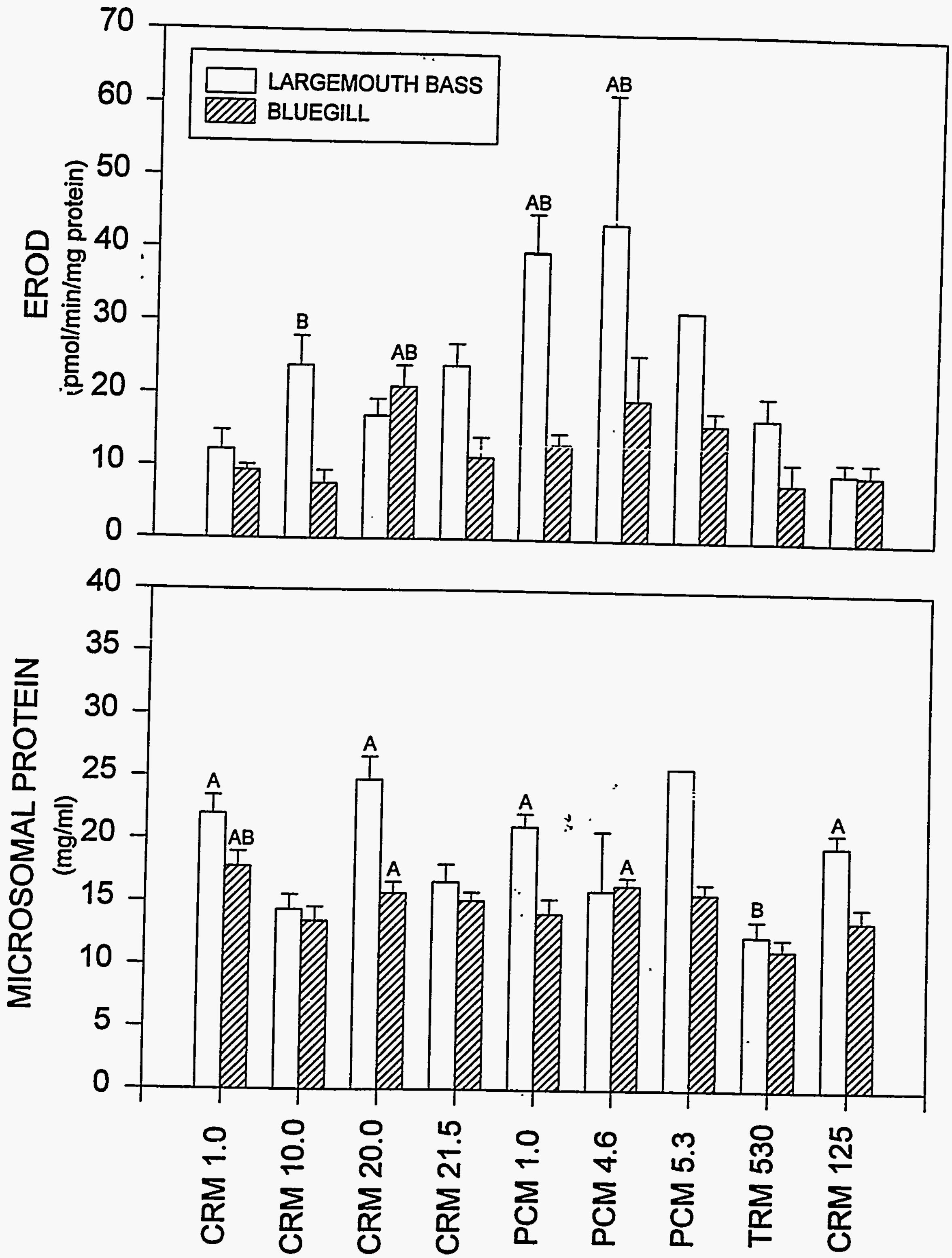


Fig. 3.63. Means + SE for detoxification enzyme indicators [EROD (7-ethoxyresorufin O-deethylase) and microsomal protein] for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site, 1993–1994. A = significantly different from Tennessee River mile 530; B = significantly different from Clinch River mile 125.

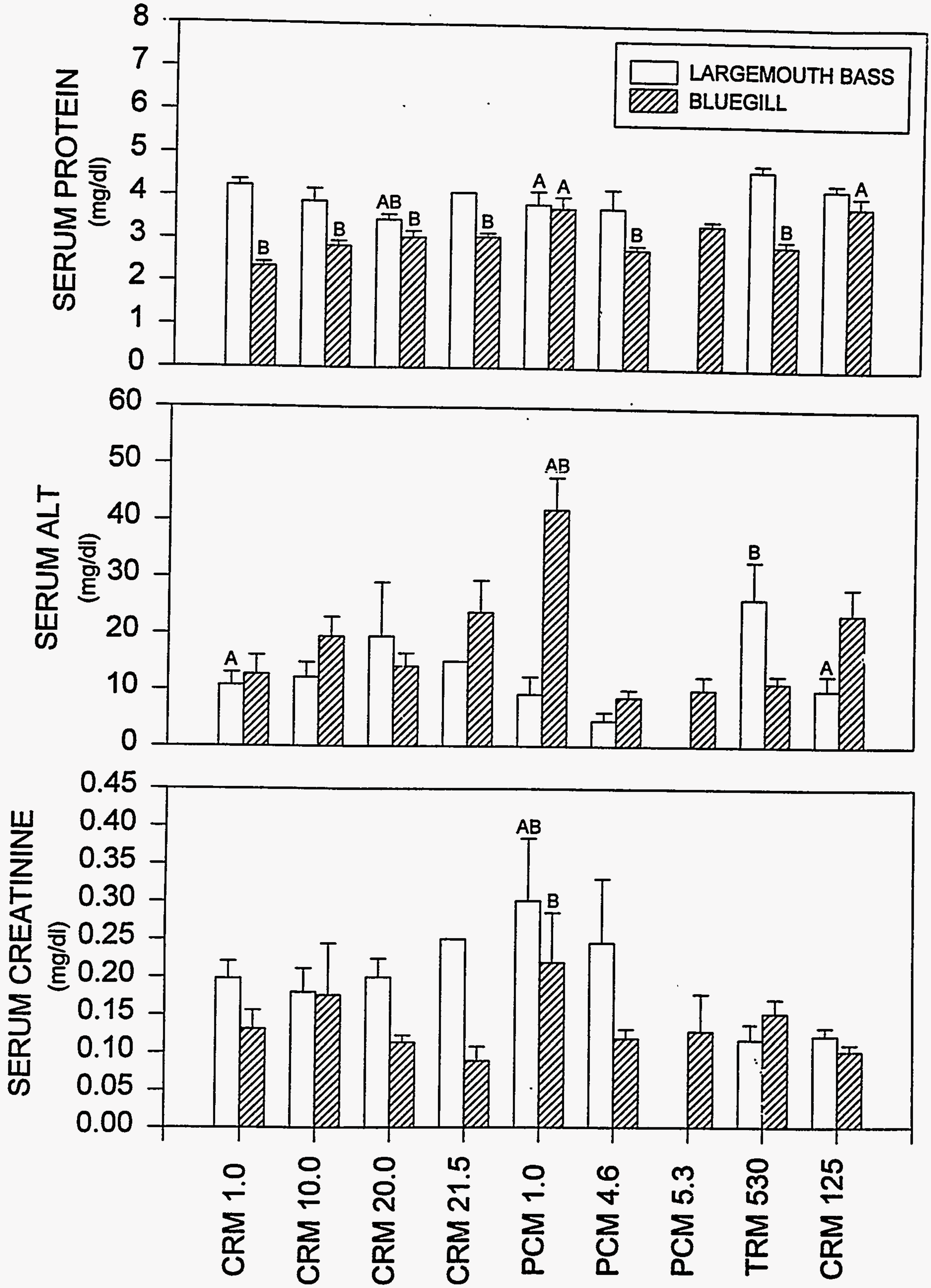


Fig. 3.64. Means + SE for organ dysfunction indicators [serum creatinine, ALT (alanine aminotransferase), and protein] for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site, 1993–1994. A = significantly different from Tennessee River mile 530; B = significantly different from Clinch River mile 125.

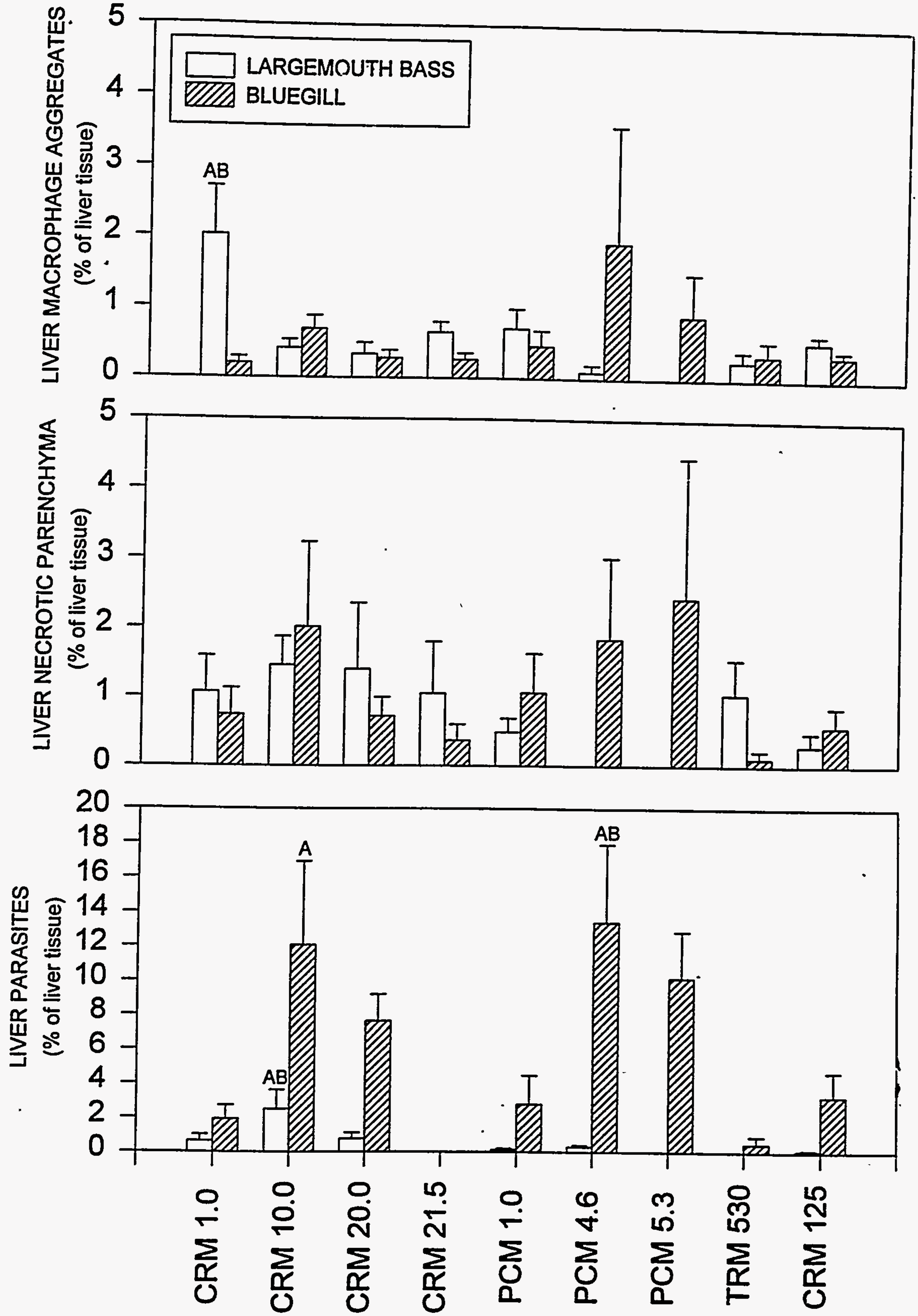


Fig. 3.65. Means + SE for histopathological indicators for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site, 1993–1994. A = significantly different from Tennessee River mile 530; B = significantly different from Clinch River mile 125.

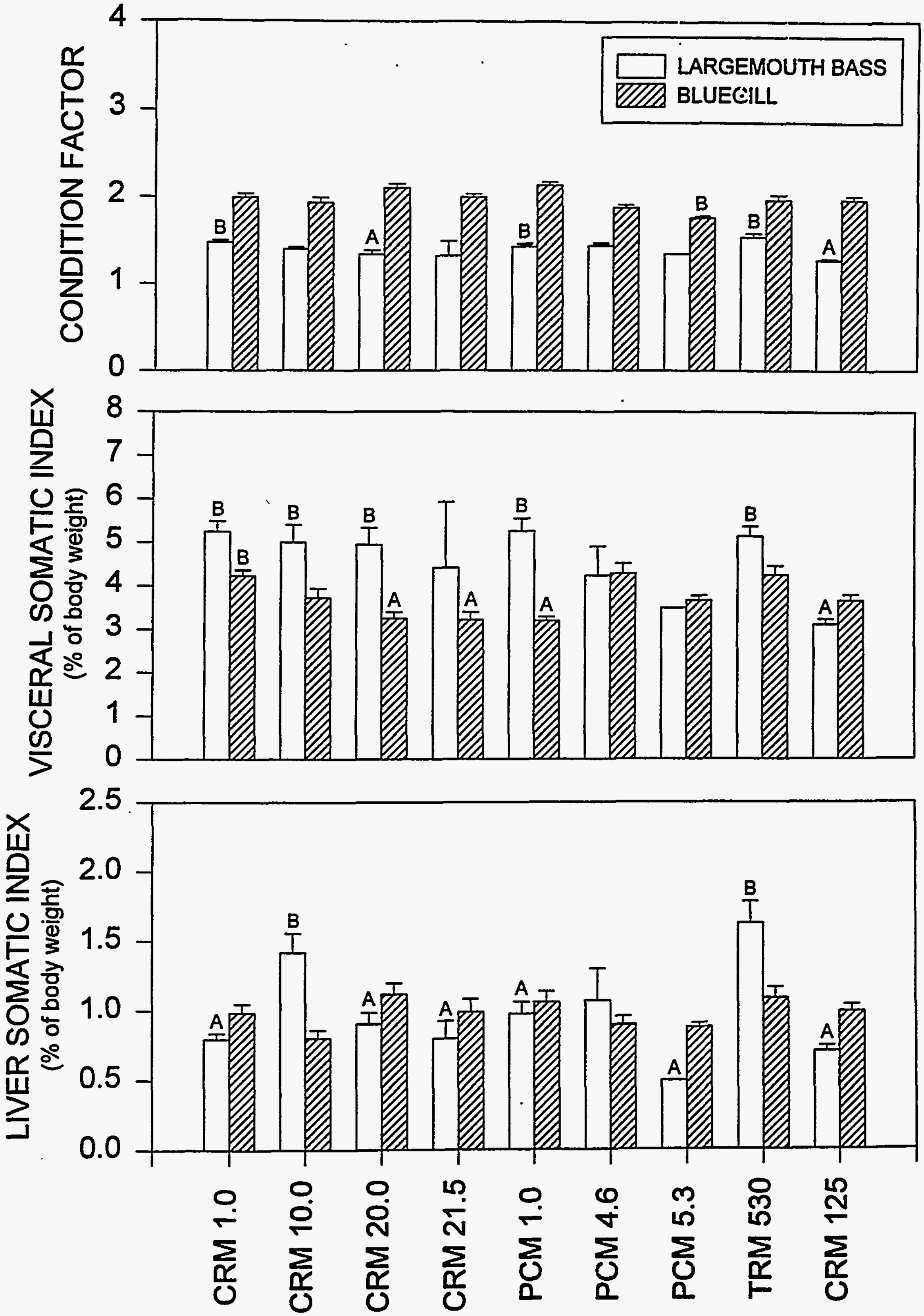


Fig. 3.66. Means + SE for three condition indices for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site, 1993–1994. A = significantly different from Tennessee River mile 530; B = significantly different from Clinch River mile 125.

The visceral-somatic index (VSI) reflects energy stored as lipids in the mesenteries of the viscera and is used to indicate the overall fat storage and bioenergetic condition of fish (Adams and Ryon 1994). Largemouth bass from Poplar Creek and the Clinch River (except CRM 21.5 and PCM 4.6) had significantly higher VSI values than those from the Norris reference, but the values did not differ from the lower Watts Bar reference (Fig. 3.66). In contrast, the VSI values for bluegill from CRM 20 and CRM 21.5 and from lower Poplar Creek were significantly lower than that for bluegill from TRM 530; these lower values indicate levels of bioenergetic dysfunction at these sites.

The liver-somatic index (LSI) uses measures of liver enlargement as an indicator of metabolic energy demands (Adams and McLean 1985). Liver enlargement caused by hyperplasia and hypertrophy has been reported in fish exposed to toxic compounds (Fletcher et al. 1982; Addison 1988; Heath 1987). The LSI values for largemouth bass from almost all Clinch River and Poplar Creek sites were comparable to the Norris reference value and were significantly lower (better) than the values for the Watts Bar reference bass (Fig. 3.66). For bluegill, the LSI values were similar at all sites.

Nutritional and feeding status. The nutritional status of the organism is important in interpreting the effects of contaminant stress on fish. For example, poor nutrition can weaken fish and render them more susceptible to other stresses (Shul'man 1974; Adams et al. 1990). As an indicator of short-term feeding intensity, the percentage of food in the intestine was evaluated among sites (Fig. 3.67). This percentage was significantly lower in bluegill from CRM 1.0, CRM 21.5, and the lower two Poplar Creek sites in comparison with bluegill from at least one reference site. Serum triglyceride levels reflect a more long-term feeding status. Serum levels in largemouth bass from almost all locations were significantly higher than those in bass from Norris and generally did not differ from that in lower WBR bass (Fig. 3.67). Serum levels in bluegill were lower at three sites (CRM 1.0, PCM 1.0, and PCM 5.3) than the WBR reference level but did not differ from the Norris reference level.

Health assessment index. The HAI provides an overall health profile of a fish population (Adams et al. 1993). The HAI for bluegill was similar across all sites (Fig. 3.68), except that bluegill from PCM 4.6 (downstream of Mitchell Branch) were in significantly poorer health than Norris fish. Values for three of the index variables (kidney, parasites, and levels of serum protein) were primarily responsible for the elevated HAI values at this site.

Largemouth bass from three sites (PCM 1.0, PCM 5.3, and CRM 20) had significantly higher HAI scores than the largemouth bass from the Norris reference site, primarily because of anomalies in the gills, spleen, and liver and in blood protein levels. No other Poplar Creek or Clinch River sites had an HAI value different from either reference site.

Integrated site analysis. In addition to the analysis of individual functional groups above, the integrated response of fish to the environmental conditions at each study site was evaluated by incorporating bioindicators representing all five functional response groups and the HAI into a multivariate (canonical discriminant) analysis. This approach provides an integrated assessment of fish health on the basis of multiple, rather than individual, bioindicators. Because of relatively low sample sizes, all data from PCM 5.3 and CRM 21.5 were excluded from this analysis, as were largemouth bass data from PCM 4.6.

The discriminant analysis for largemouth bass indicated that Norris bass were least similar to fish from lower Poplar Creek (PCM 1.0) and most similar to fish from the lower Watts Bar reference site (Fig. 3.69), despite the major differences in the trophic status of these two reference sites. Bass sampled from PCM 1.0 were most similar to fish from CRM 10 and 20.

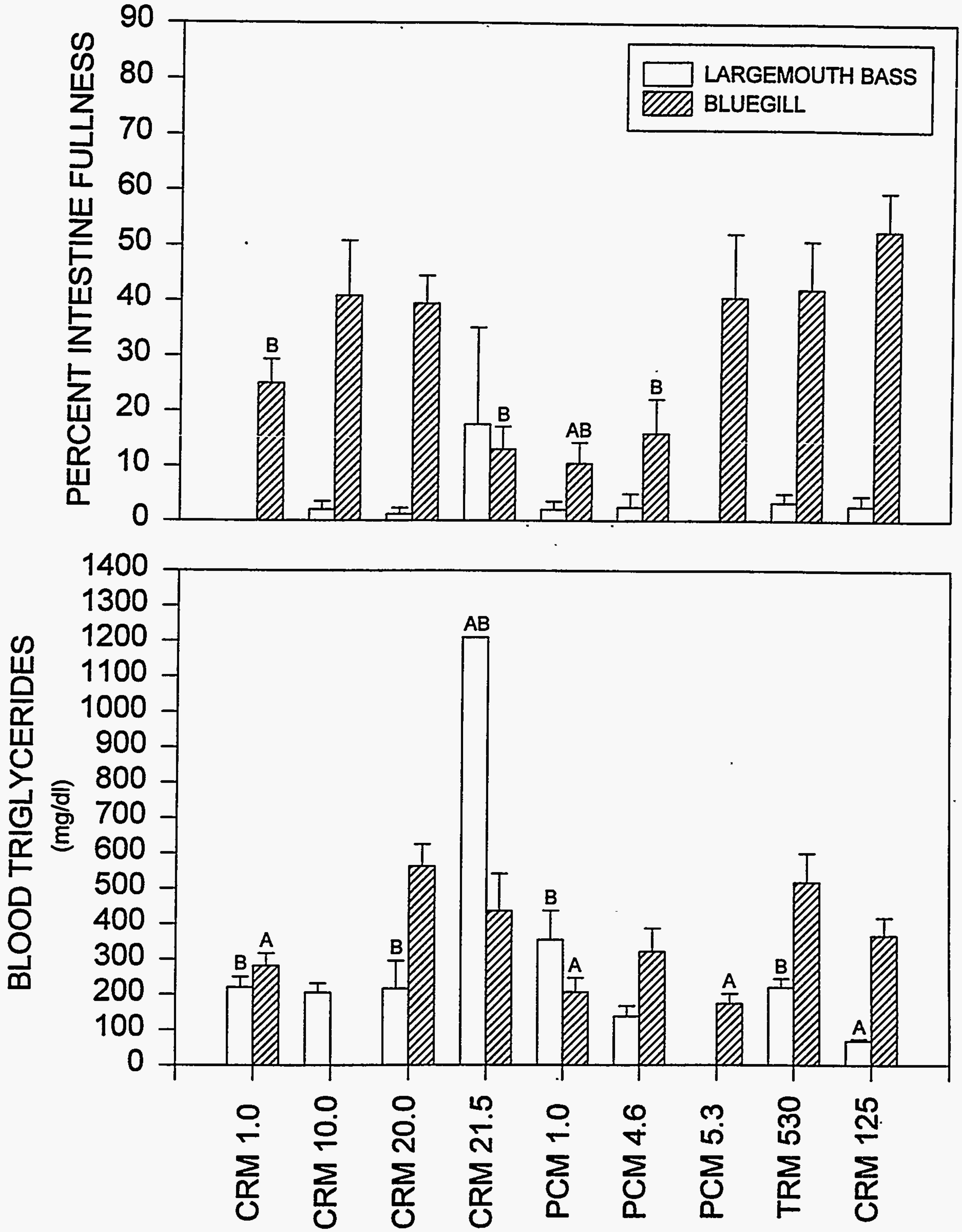


Fig. 3.67. Means + SE for feeding and nutritional indicators for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site, 1993–1994. A = significantly different from Tennessee River mile 530; B = significantly different from Clinch River mile 125.

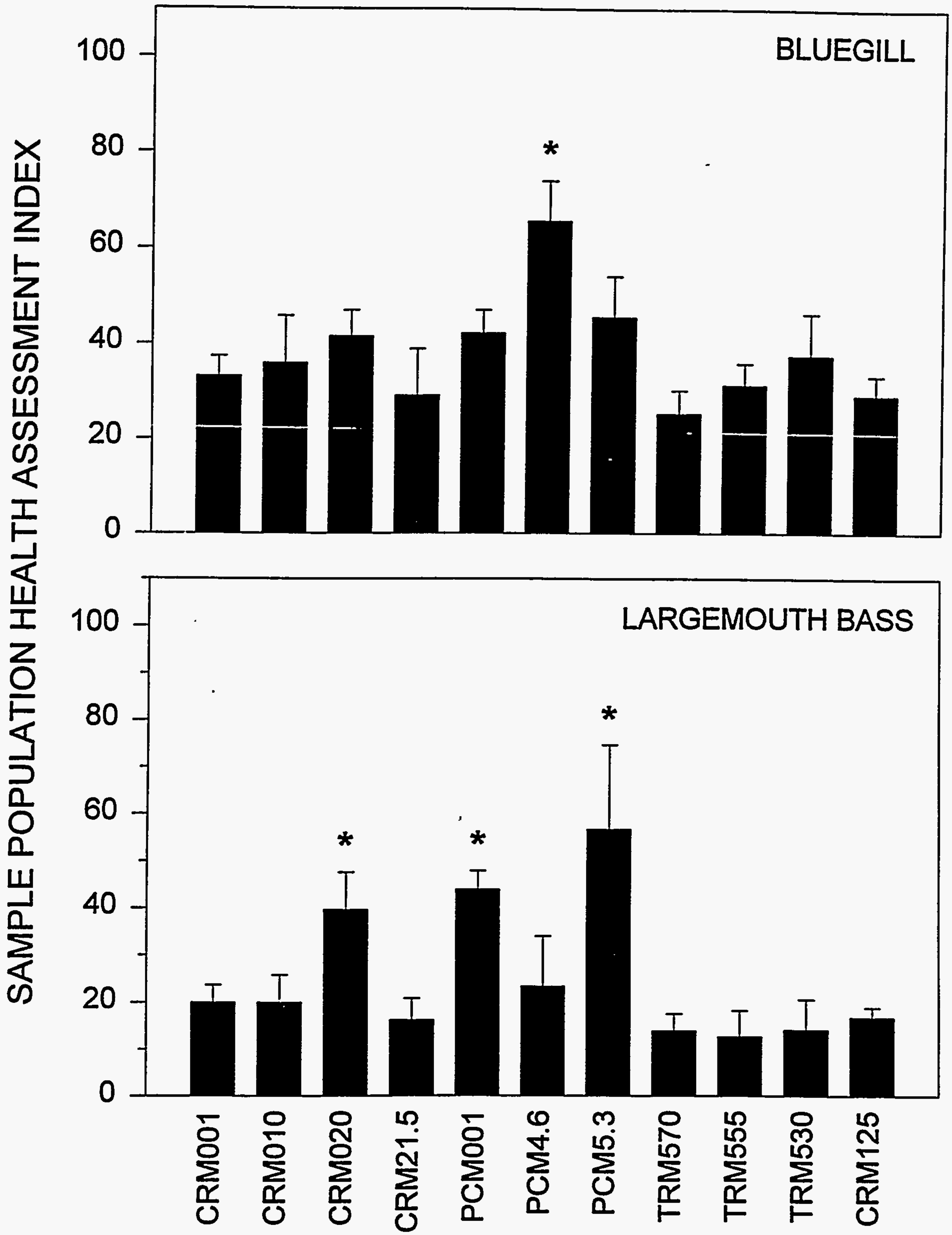


Fig. 3.68. Means + SE for the health assessment index for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site 1993–1994. * An asterisk indicates that the value is significantly different from Clinch River mile 125.

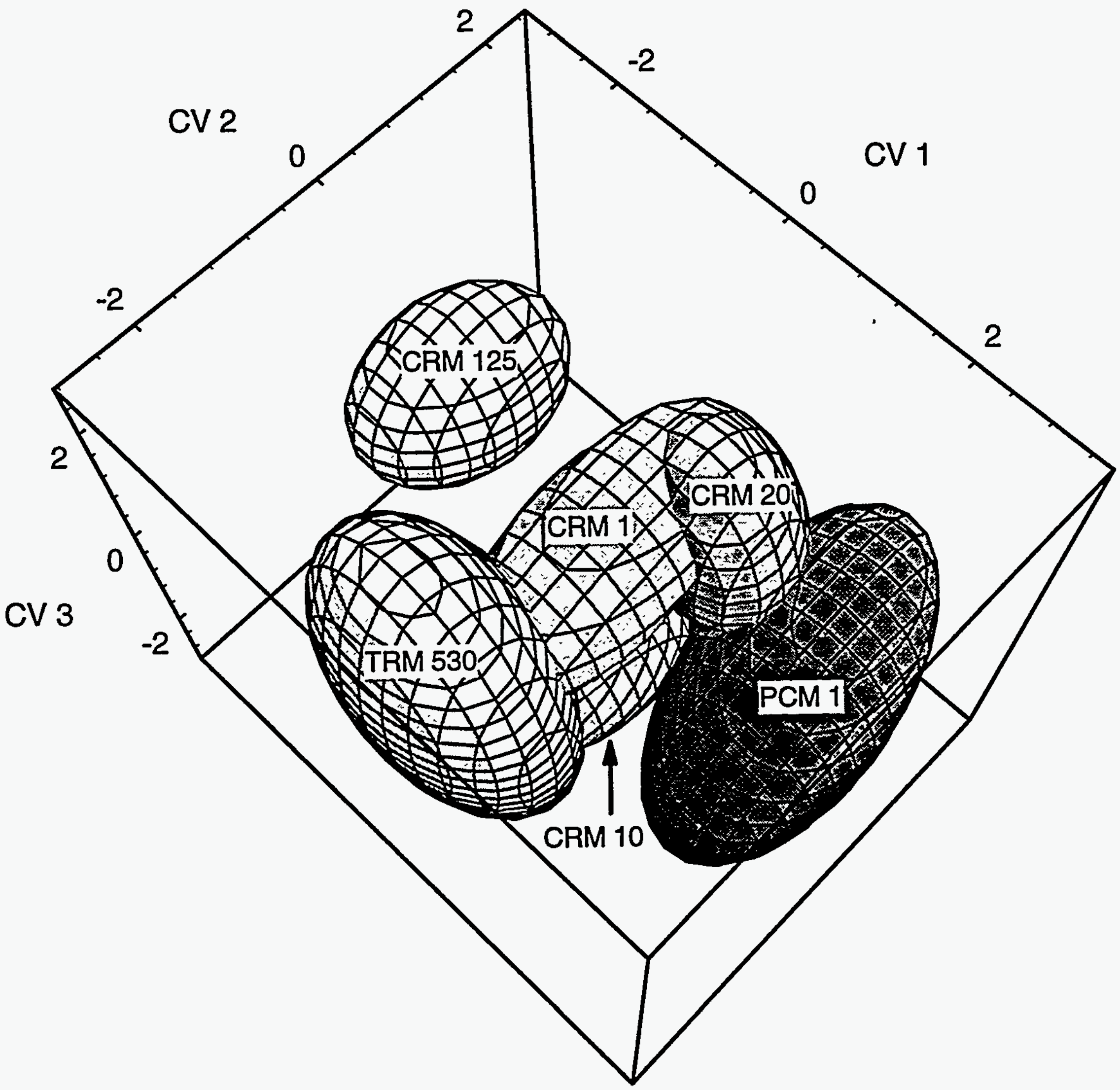


Fig. 3.69. Segregation of integrated health responses for male largemouth bass collected from two reference sites (Clinch River mile 125 and Tennessee River mile 530), three Clinch River sites (Clinch River miles 1.0, 10, 20) and the lower Poplar Creek site (Poplar Creek mile 1.0). Ellipsoids represent the mean integrated responses of bass at a site for 1993–1994.

The individual bioindicator variables which have the most power for distinguishing between sites were, in order of importance, the VSI, EROD, microsomal protein, and the condition factor. The EROD and microsomal protein variables are indicators of contaminant exposure, whereas the VSI and condition factor reflect the bioenergetic and overall condition of fish. These factors indicate that fish in the Clinch River and Poplar Creek have been exposed to contaminants, possibly impairing bioenergetic function.

The discriminant analysis for bluegill indicated that the two reference sites were more similar to each other than they were to any of the Clinch River (Fig. 3.70) or Poplar Creek (Fig. 3.71) sites. Interestingly, the two Poplar Creek sites were more similar to CRM 20 than to the other two Clinch River areas, even though CRM 10 is immediately downstream of the mouth of Poplar Creek. The most important variables for discriminating between sites were EROD, VSI, and microsomal protein. As with largemouth bass, the indication is that bluegill in the Clinch River and Poplar Creek have also been exposed to contaminants, and possibly their bioenergetic function has been impaired as a result.

Summary of bioindicators task. The bioindicator data indicate an apparent gradient in biological effects; fish from Poplar Creek demonstrate the poorest health, and those individuals collected from lower Watts Bar, the best. Some of the major health effects responses observed in fish, such as detoxification enzyme induction and poor histopathological condition, are consistent with levles of contaminant exposure.

3.5.3.5 Reproductive effects assessment

The goal of this task was to evaluate the reproductive health of fish in the CR/WBR downstream of the ORR. Selected reproductive criteria were measured in largemouth bass and bluegill sunfish collected from the Clinch River and Poplar Creek, and they were compared with values from reference sites. Reproductive criteria include the gonadal-somatic index (GSI), levels of testosterone and estradiol in blood plasma, and the frequency of oocyte atresia (death). Each of these measures has been shown to be sensitive to exposure to a number of common environmental contaminants.

Results. For male largemouth bass, mean GSIs (Table N25) were highest in the Norris Reservoir reference sites (CRM 125 and PRM 30) and lowest (poorest) at sites adjacent to the ORR (PCM 1.0 and CRM 20). Plasma concentrations of testosterone were low in male largemouth bass from PCM 1.0 and CRM 20, averaging approximately half the levels measured in fish from the reference sites (Table N25). Together these data indicate a decrease in male reproductive effort at the PCM 1.0 and CRM 20 sites. The estrogenic steroid hormone, 17β-estradiol is generally not detectable in male fish. However, its production may be stimulated in males upon exposure to compounds that act either to mimic or block natural hormones. The mean concentrations of estradiol in plasma from the male bass were very low at all study sites. However, the frequency of detection differed between sites. Measurable levels were found in 88% of the males collected from CRM 20, 69% from CRM 0.5, and 41% from TRM 557. Detectable levels at other sites were 36% for PRM 30, 11% for CRM 125, and 0% for TRM 570 and PCM 1.0. Therefore, sites with the highest frequencies of estradiol "detects" in male largemouth bass were the three sites in the Clinch River and Watts Bar Lake that lie downstream of the ORR.

The mean GSIs of female largemouth bass were lowest at PCM 1.0, CRM 20, and at the two reference sites in Norris Reservoir (Table N26). Sites with relatively higher mean GSIs included TRM 557, TRM 570, and CRM 0.5. Two separate methods were used to derive estimates of fecundity for female bass (Table N26). Neither method revealed statistically different results in mean fecundity between any sites, although by one method it was determined that the mean fecundities of female largemouth bass populations at PCM 1.0 and CRM 20 were ~20% lower than at any of the other study sites. In female fish, 17β-estradiol regulates the production of yolk proteins by the liver for incorporation

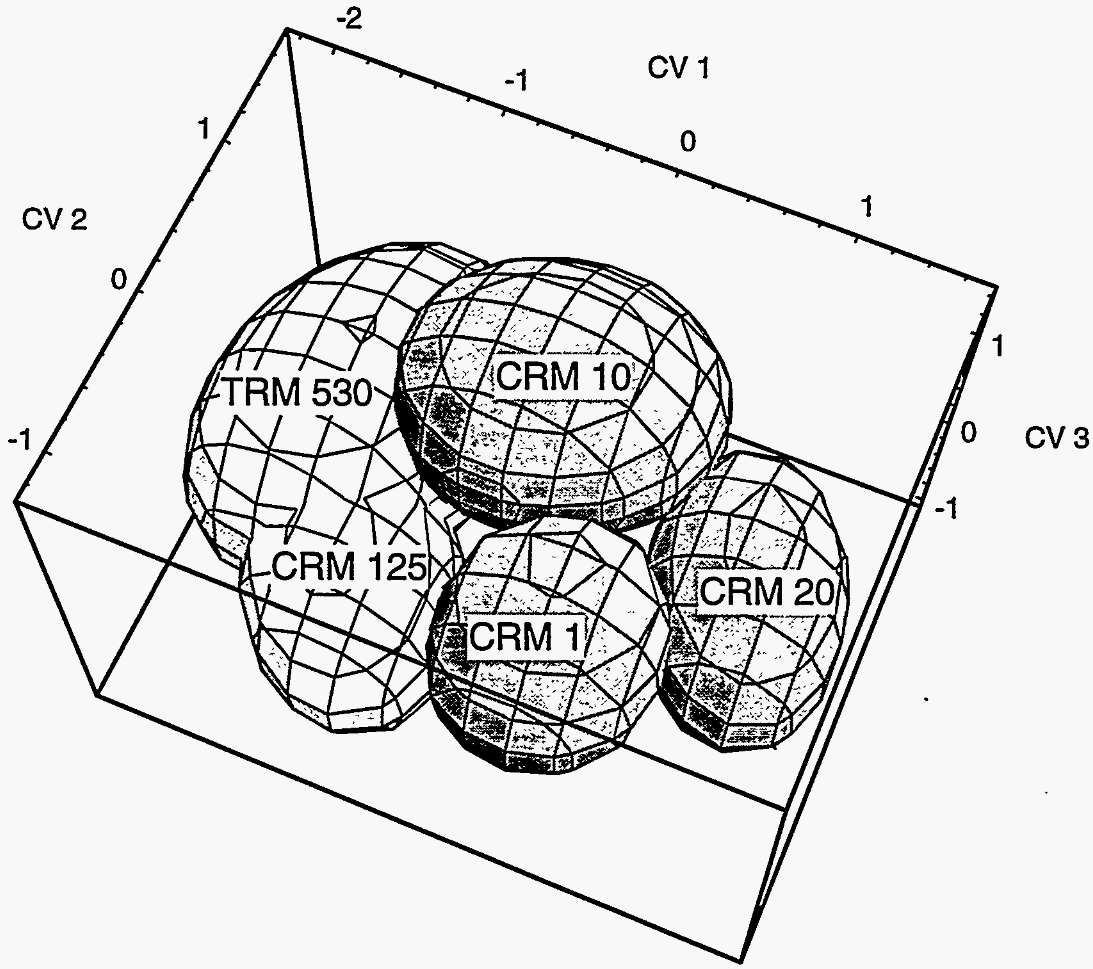


Fig. 3.70. Segregation of integrated health responses for male bluegill sunfish sampled from two reference sites (Clinch River mile 125 and Tennessee River mile 530) and three Clinch River sites (Clinch River miles 20, 10, 1.0). Ellipsoids represent the mean integrated responses of bluegill at a site for 1993–1994.

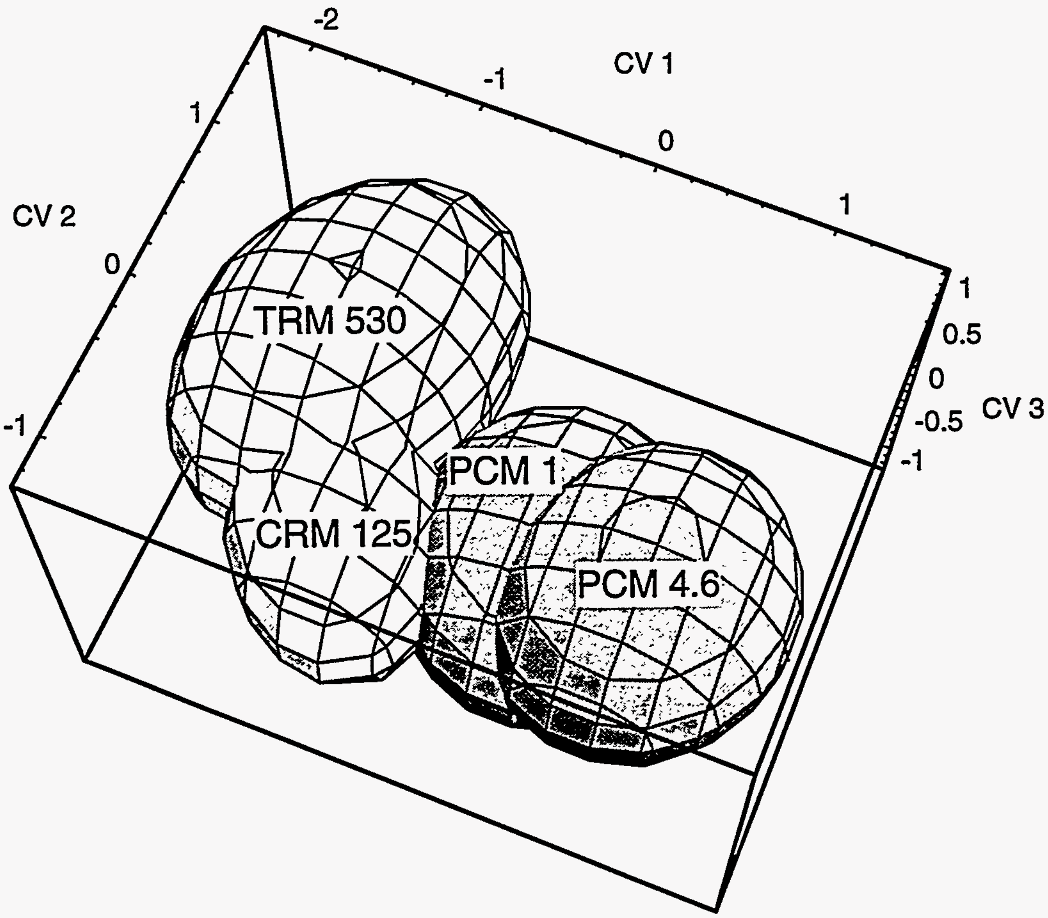
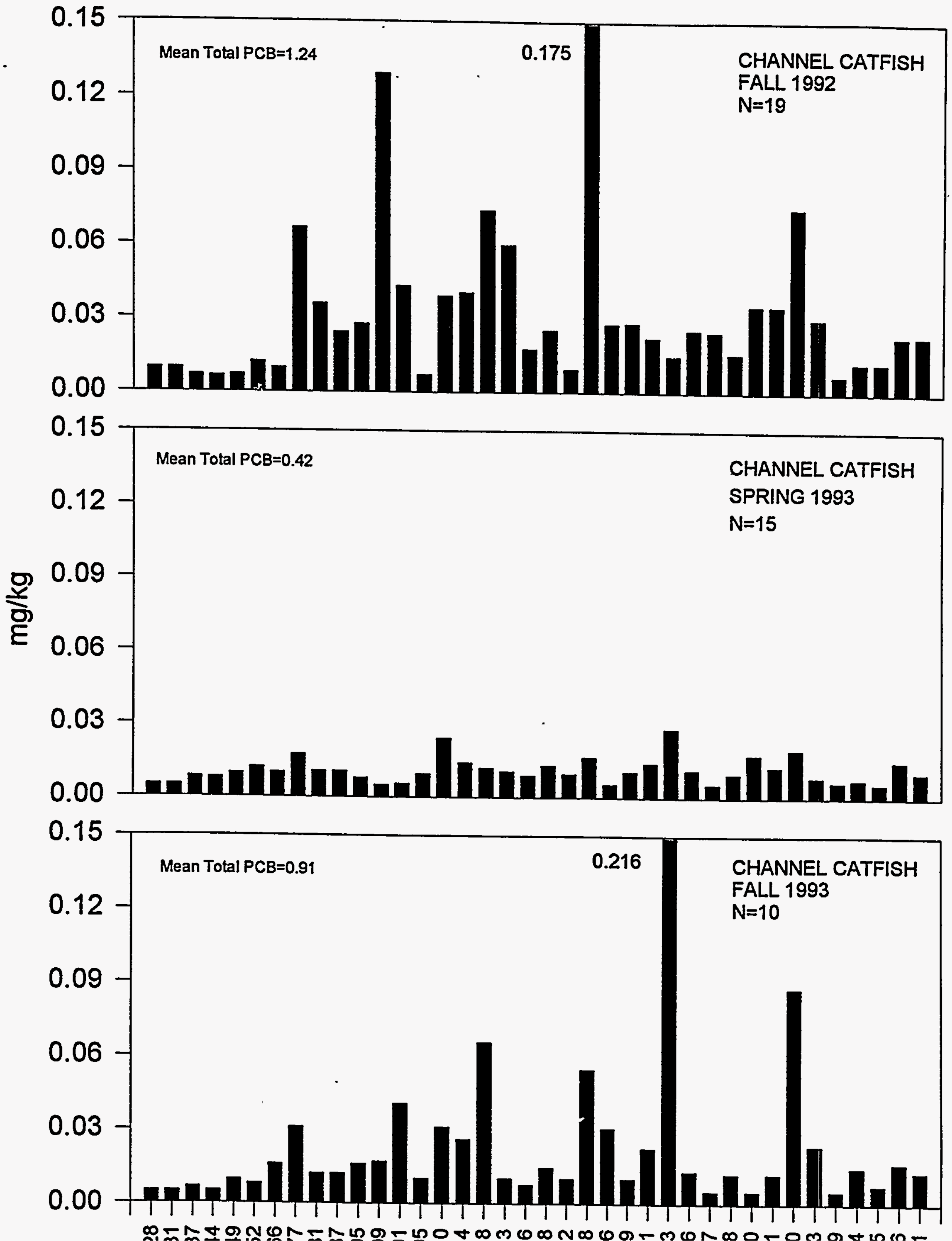


Fig. 3.71. Segregation of integrated health responses for male bluegill sunfish sampled from two reference sites (Clinch River mile 125 and Tennessee River mile 530) and two Poplar Creek sites (Poplar Creek miles 4.6 and 1.0). Ellipsoids represent the mean integrated responses of bluegill at a site for 1993–1994.

into vitellogenic oocytes. Plasma concentrations of estradiol were lowest in female bass from CRM 20



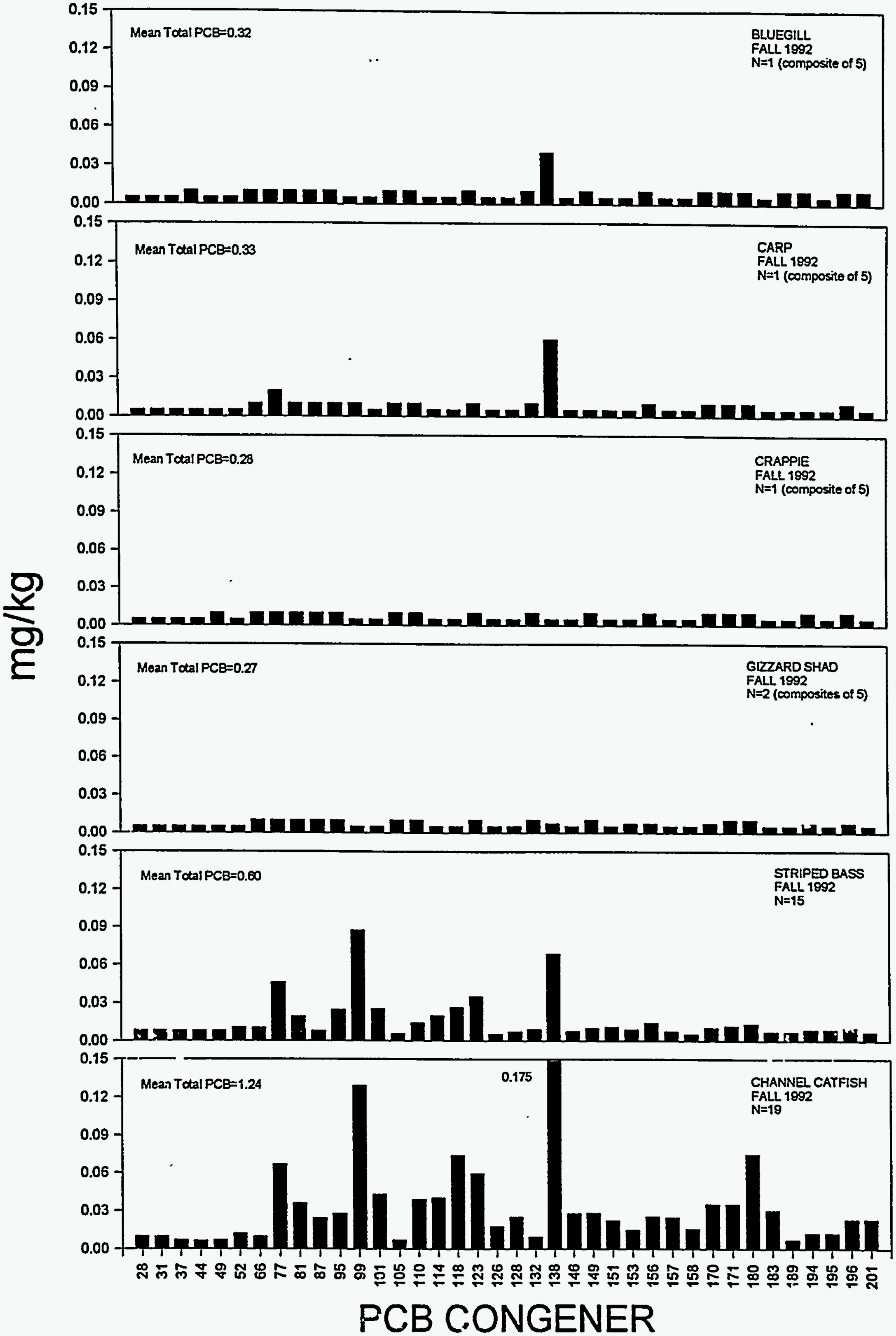


Fig. 3.73. Congener concentrations in sportfish.

of subreach 7.02 (lower McCoy Branch) were significantly lower than those in subreach 7.01 and were comparable to concentrations in Melton Hill Reservoir and upstream. Concentrations of arsenic in sediment were also highest in upper McCoy Branch, and were 6 to 10 times greater than in most other study reaches. Levels in lower McCoy Branch sediment were only slightly elevated in comparison with the other references. No fish data are available for either subreach of McCoy Branch. Despite the elevated levels of arsenic in water and sediment, no clear toxicity was evident in surface water or sediment from either subreach of McCoy Branch.

The increased concentrations of arsenic in surface water and sediment are consistent with the previous study of Ford et al. (1995) and the known release history for McCoy Branch.

3.6.3 Upper Clinch River (Reach 2)

Mean gross alpha and gross beta levels and mean activities of ^{90}Sr and ^{3}H in surface water increased by an order of magnitude immediately below the mouth of WOC (subreach 2.02). Concentrations varied widely, probably reflecting the wide variation in flow in this subreach, and the observed increases were not statistically significant. Nevertheless, the increased activities are consistent with the site model and with the known sources in the WOC watershed. Radionuclide concentrations do not exceed AWQC in subreach 2.02 or in reach 2 as a whole.

Because of flow characteristics, reach 2 accumulates sediment only to a limited extent. Therefore sediment sampling was limited in reach 2 and comparisons with upstream areas are also limited. However, levels of ^{137}Cs are clearly elevated in reach 2 sediment. Levels of selenium and manganese are locally high in subreach 2.04 below the mouth of Grassy Creek.

Mean concentrations of ^{137}Cs were roughly 100 times greater in bluegill sunfish (2 pCi/g) and largemouth bass (5 pCi/g) from reach 2 than in the upstream reference reaches. Mean ^{137}Cs levels in catfish were much lower (0.67 pCi/g) but were still an order of magnitude greater than reference values. Mean total PCB concentrations are significantly higher in largemouth bass and catfish in comparison with the Norris reference, and mean concentrations in catfish are also higher than in Melton Hill Reservoir (no largemouth bass were collected from Melton Hill). Elevated levels of PCBs in sediment and catfish of the WOC embayment indicate that WOC has been a source of this contaminant to reach 2.

No clear evidence of increased toxicity was found in either surface water or sediment of reach 2. However, a suite of biological parameters indicated that largemouth bass from this reach (at CRM 20) were in poorer overall health in comparison with most other sites.

3.6.4 Poplar Creek (Reach 3)

Mercury levels in Poplar Creek were substantially elevated in all media downstream of EFPC. Mean total mercury levels in surface water increase by approximately an order of magnitude and very likely exceed the AWQC of 0.000012 mg/L for protection of aquatic life (CCC) throughout reach 3. Mean concentrations in sediment are increased over reference values by as much as one to two orders of magnitude, depending on the subreach. Mean concentrations in bluegill sunfish and largemouth bass were significantly higher than in fish in the reference reaches but do not exceed the FDA action level of 1.0 mg/kg. Mayflies collected from lower Poplar Creek had significantly higher mercury levels than those collected from the lower Clinch River (reach 4) or the Tennessee River (reach 18). Great blue heron chicks and eggs collected from a rookery on Poplar Creek had higher mercury concentrations than those from rookeries in Melton Hill Reservoir (reach 1) or the Tennessee River (reach 18). Mink that were fed a diet containing Poplar Creek fish developed increased body burdens of mercury; mink whose diets contained the greatest proportion of Poplar Creek fish suffered impaired reproductive capabilities.

The sediment of reach 3 also contained elevated concentrations of a number of analytes, including the metals Ag, As, B, Cd, Cr, Cu, Ni, and V, and the radionuclides ^{234}U, ^{235}U, ^{238}U, ^{99}Tc, ^{137}Cs, and ^{60}Co. Although the frequency of detection is low, Aroclor 1254 and several PAHs were detected in reach 3 as well.

The various species of fish collected from reach 3 also show spatial patterns for contaminants other than mercury, but, unlike the presence of mercury, the pattern is often not consistent across species. Hence, arsenic concentrations in bluegill are greatest in Poplar Creek, but concentrations in largemouth bass are higher in reach 2. Total PCBs in largemouth bass and catfish are elevated in Poplar Creek compared with Norris or Melton Hill reservoirs or upper Poplar Creek (catfish data only). However, peak PCB concentrations in largemouth bass occur in Poplar Creek, whereas PCB concentrations in catfish from Poplar Creek are less than in the Clinch River below WOC (reach 2) and do not differ from levels in either the lower Clinch River (reach 4) or the Tennessee River (reaches 18 and 5). Mean total PCB concentrations do not exceed the FDA action level (2.0 mg/kg) in any reach.

Most of the aqueous toxicity tests conducted in Poplar Creek failed to identify any toxicity of surface water. Sediment toxicity tests revealed intermittent toxicity, but no clear response pattern was identified. However, the Japanese medaka embryo-larval series of aqueous tests did demonstrate a toxic response in lower Poplar Creek. Mean survival of medaka embryos and fry in Poplar Creek was less than in the reference reach, although the incidence of developmental abnormalities in embryos was increased in the former compared with the latter. The pattern of toxic response was also observed in limited testing with a native species (redbreast sunfish).

The suite of physical, physiological, and reproductive indices applied to largemouth bass and bluegill revealed that the overall health of both species is poorer in Poplar Creek than at most of the other study sites and at all reference sites. The overall health effect appears more pronounced in largemouth bass than in bluegill. Many of the individual indicator responses were consistent with contaminant exposure. Multivariate analysis of the bioindicator data indicated that, for each species, fish from widely separate and trophically disparate reference areas (Norris and lower Watts Bar reservoirs) are more similar to each other than fish from either location are to Poplar Creek fish .

3.6.5 Lower Clinch River (Reach 4)

Contaminant concentrations in the Clinch River downstream of Poplar Creek were generally comparable with those in the upstream reference reaches. Mean concentrations of contaminants that were elevated in upstream reaches of the Clinch River and Poplar Creek have returned to levels comparable with those in the reference reaches. Mean mercury levels, even in subreach 4.01, decline by an order of magnitude from those in lower Poplar Creek. However, because the recreation-based AWQC is also approximately an order of magnitude less in the Clinch River (because one of its designated uses is for domestic water supply), concentrations in reach 4 still likely exceed this criterion (as was also the case in Melton Hill Reservoir).

Significant sediment accumulation occurs in the lacustrine environment of reach 4, and many of the particle-associated contaminants released from the ORR have accumulated in these sediments. Previous studies have documented increased levels of radionuclides (particularly ^{137}Cs) and mercury in these sediments in relation to reference values. Because of its relatively long half-life and its affinity for sediment particles, ^{137}Cs is the principal radionuclide of concern today. Surface sediment concentrations of mercury and ^{137}Cs are substantially less than in deeper sediment layers. Because peak releases of these two contaminants occurred almost 40 years ago, the highest concentrations of each are now buried beneath layers of cleaner sediment. Surface concentrations of ^{137}Cs remain relatively constant in the Clinch River downstream of WOC, varying primarily as a result of variations in sediment deposition. Surface concentrations of mercury, however, decline by a factor of 10 in subreach 4.01 from concentrations in Poplar Creek. Overall, the levels of other contaminants in sediment do not differ substantially from reference values.

Sediment modeling studies indicate that, in the absence of major (100-year) storms and at current release rates, ^{137}Cs levels in the Clinch River will decline, primarily as a result of radioactive decay. In the event of a major storm localized in the WOC watershed, a temporary increase in ^{137}Cs levels would occur, but the sediment released from WOC would soon be remobilized and transported downstream. In the event of a major regional storm, ^{137}Cs levels would actually decline in the Clinch River as a result of scouring and subsequent redeposition downstream. A previous RI (DOE 1995) demonstrated that much of this sediment would be redeposited in Watts Bar Reservoir but at greatly diluted concentrations.

Catfish and largemouth bass from reach 4 contain levels of PCBs that are elevated over reference locations in the Clinch River (Norris and Melton Hill reservoirs) but are generally lower than levels in reach 2 and are comparable to levels in the Tennessee River arm of Watts Bar Reservoir (reaches 5 and 18). Mercury levels in bass and bluegill have declined from levels in Poplar Creek but remain slightly above reference levels. However, mean values do not exceed the FDA action level (1.0 mg/kg) in either species. The mean ^{137}Cs activity in bluegill sunfish and largemouth bass has declined by a factor of ten from peak levels in reach 2 but remain roughly an order of magnitude greater than levels in the reference reaches.

Neither surface water nor sediment in reach 4 was toxic to any of the various organisms tested. The overall health of fish in reach 4, as evaluated by the bioindicator data, was generally intermediate to that of fish in the reference reaches and in reach 2 and Poplar Creek.

3.6.6 Conclusions

Environmental media in several subreaches of the Clinch River and Poplar Creek contain increased concentrations of one or more contaminants released from the ORR. The most striking include (1) arsenic in surface water and sediment of upper McCoy Branch; (2) radionuclides in surface water, biota, and sediment of the Clinch River below WOC; (3) mercury in surface water, sediment, and biota of Poplar Creek; and (4) radionuclides and mercury in sediment of the lower Clinch River. The potential human health and ecological impacts of these contaminants, as well as each of the other contaminants discussed in this chapter, will be closely evaluated in Chaps. 5 and 6.

4. APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS

Section 121(d) of CERCLA specifies that remedial actions for cleanup of hazardous substances must comply with requirements or standards under federal or more stringent state environmental laws that are applicable or relevant and appropriate to the hazardous substances or particular circumstances at a site. Inherent in the interpretation of applicable or relevant and appropriate requirements (ARARs) is the assumption that protection of human health and the environment is ensured. Appendix I of this report provides a preliminary list of available federal and state chemical-, location-, and action-specific ARARs for the remediation of the CR/PC OU.

"Chemical-specific requirements set health- or risk-based concentration limits or discharge limitations in various environmental media for specific hazardous substances, pollutants, or contaminants" (53 *FR* 51437). These requirements generally set protective cleanup levels for the chemicals of concern in the designated media or indicate a safe level of discharge that may be incorporated when a specific remedial activity is being considered.

Location-specific requirements "set restrictions upon the concentration of hazardous substances or the conduct of activities solely because they are in special locations" (53 *FR* 51437). The areas along the Clinch River, Poplar Creek, and McCoy Branch contain several sensitive resources that are protected by either federal or state regulations. These resources include wetlands, floodplains, cemeteries, and state-designated natural areas. Preconstruction activities performed in conjunction with an action-based alternative, such as the building of access roads or the removal of bank vegetation in addition to actual containment or removal and treatment activities, may trigger certain location-specific ARARs. The location-specific ARARs summarized in Appendix I will not apply to either the no action alternative or the institutional controls alternative because no such activities will be performed.

The TVA, U.S. Army Corps of Engineers (USACE) and the TDEC regulate activities under §26(a) of the TVA Act, the USACE Nationwide Permit Program, and the TDEC Aquatic Resource Alteration Permit Program, respectively. Although permits are not required for CERCLA on-site actions, activities that would normally require permits under these programs must meet the substantive requirements of the appropriate regulations. Appendix I summarizes these requirements.

Performance, design, or other action-specific requirements set controls or restrictions on particular kinds of activities related to the management of hazardous waste (53 *FR* 51437). The selection of a particular remedial action at a site will invoke appropriate action-specific ARARs, which may specify particular performance standards or technologies, as well as specific environmental levels for discharged or residual chemicals. Action-specific ARARs are presented in Appendix I, and compliance with action-specific ARARs is summarized in Chap. 10.

5. HUMAN HEALTH RISK ASSESSMENT

This baseline human health risk assessment quantifies the carcinogenic risk and noncarcinogenic hazard associated with human exposure to contaminants detected in fish, water fowl, sediment, and surface water collected in the Clinch River and Poplar Creek. Specifically, this baseline risk assessment evaluates the potential for exposure to an individual who might regularly use the resources of either the Clinch River or Poplar Creek under the conditions that would exist if the current institutional controls (i.e., fishing advisories, dredging controls) were removed and no remediation occurs. The results of the baseline human health risk assessment are used in Sect. 5.6 to develop remediation goals and are used in Chap. 8 to develop remedial alternatives for the Clinch River and Poplar Creek.

EPA (Region IV), TDEC, and DOE's Oak Ridge Operation Office agreed in the data quality objectives working group meetings that were held before the development of the *Phase 2 Sampling and Analysis Plan, Quality Assurance Project Plan, and Health and Safety Plan for the Clinch River Remedial Investigation: An Addendum to the Clinch River RCRA Facility Investigation Plan* (Cook et al. 1993) that this study will contain a risk assessment focused on the contaminants of potential concern (COPCs) identified in previous human health risk assessment reports (Table 5.1).

Before the Phase 2 data were collected from both the Clinch River and Poplar Creek, several efforts were undertaken to evaluate the potential risk/hazard posed by exposure to contaminants in the Clinch River. A screening assessment conducted by Hoffman et al. (1991) identified PCBs in fish and external exposure to radiation from ^{137}Cs in dredged sediments as the priority contaminants and exposure pathways that are of potential concern in the Clinch River downstream of the ORR. Hoffman et al. (1991) also identified several inorganics that posed a risk/hazard through the evaluated dredging pathways. On the basis of the CRRI Phase 1 data, Cook et al. (1992) confirmed that these contaminants and exposure pathways are of potential concern and identified chlordane in fish as a COPC. Most recently, the baseline human health risk assessment for the lower Watts Bar Reservoir OU (DOE 1995) identified PCBs, pesticides, and chlordane in fish as COPCs and identified external exposure as a potential pathway for ^{137}Cs and ^{60}Co. Therefore, the Phase 2 sampling and analysis of the Clinch River and Poplar Creek was intended to further evaluate the potential risk/hazard for those contaminants known to be present in the Clinch River and Poplar Creek systems.

5.1 IDENTIFICATION OF CHEMICALS OF POTENTIAL CONCERN

5.1.1 Data Compilation Considerations

As indicated in Sect. 3.2 of this document, multiple studies were conducted on the Clinch River and Poplar Creek environmental media before the Clinch River investigation. Although the sampling and analytical protocols used during the studies were appropriate for their intended purpose, these protocols, in some cases, may not have been appropriate for use in a baseline risk assessment. As a result, before the compilation of the risk assessment database, existing data were compared with specified criteria (as discussed in the following text) to determine their useability for risk assessment purposes.

Table 5.1. Contaminants of potential concern in the Clinch River as identified in previous screening-level risk assessments

Analyte	Exposure pathway for the contaminants of concern identified by Hoffman et al. (1991)[a]	Exposure pathway for the contaminants of concern identified by Cook et al. (1992)[b]	Exposure pathway for the contaminants of concern identified by the U.S. Department of Energy (1995)
Arsenic	Dredging—Agricultural exposure	Fish consumption Dredging—Direct exposure Dredging—Agricultural exposure	Fish consumption[c]
Beryllium	Dredging—Agricultural exposure	Fish consumption Dredging—Direct exposure Dredging—Agricultural exposure	Shoreline use—sediment ingestion[d] Fish consumption[d] Dredging—Sediment ingestion[d] Dredging—Meat and vegetable ingestion[d]
Cadmium		Dredging—Agricultural exposure	Dredging—Milk and vegetable ingestion[c]
Chromium	Dredging—Agricultural exposure	Dredging—Direct exposure	Shoreline use—Sediment inhalation[d] Dredging—Sediment inhalation[d] Dredging—Milk and meat ingestion[c]
Copper		Fish consumption Dredging—Agricultural exposure	
Manganese			Drinking water ingestion[c]
Mercury	Dredging—Agricultural exposure	Fish consumption Dredging—Agricultural exposure	Fish consumption[c] Dredging—Milk, meat, and vegetable ingestion[c]
Nickel		Dredging—Agricultural exposure	
Selenium	Irrigation Dredging—Agricultural exposure		
Silver		Dredging—Direct exposure Dredging—Agricultural exposure	
Zinc	Dredging—Agricultural exposure	Dredging—Agricultural exposure	Dredging—Milk and meat ingestion[d]
Aroclor 1254[e]	Fish consumption	Fish consumption	Fish consumption[c]

Table 5.1. (continued)

Analyte	Exposure pathway for the contaminants of concern identified by Hoffman et al. (1991)[a]	Exposure pathway for the contaminants of concern identified by Cook et al. (1992)[b]	Exposure pathway for the contaminants of concern identified by the U.S. Department of Energy (1995)
Aroclor 1260[e]	Fish consumption	Fish consumption	Fish consumption[c]
Aldrin		Fish consumption	Fish consumption[c]
Chlordane		Fish consumption	Fish consumption[c]
4,4'DDE			Fish consumption[d]
4,4'DDT			Fish consumption[d]
Dieldrin			Fish consumption[d]
Lindane			Fish consumption[d]
^{137}Cs	Dredging—Direct exposure Dredging—Agricultural exposure	Near-shore direct exposure Dredging—Direct exposure	Drinking water ingestion[d] Shoreline use—External exposure[d] Fish consumption[d] Irrigation—External exposure[d] Irrigation—Milk and meat ingestion[d] Dredging—External exposure[d] Dredging—Milk, meat, and vegetable ingestion[d]
^{60}Co	Dredging—Direct exposure	Dredging—Direct exposure	Dredging—External exposure[d] Irrigation—External exposure[d]
^{90}Sr		Dredging—Agricultural exposure	
Uranium, total			Drinking water ingestion[d]

Table 5.1. (continued)

[a]Hoffman et al. (1991) used an excess cancer risk of 10^{-3} and hazard quotient of 1.0 in a nonconservative and conservative screening to indicate "definitely or potentially high priority" and "potentially high priority" contaminants, respectively.
[b]Cook et al. (1992) used an excess cancer risk of 10^{-6} to 10^{-4} and hazard quotients of 0.1 to 1.0 in a conservative and nonconservative screening, respectively, as an indicator of contaminants of potential concern in Phase 1 of the Clinch River Remedial Investigation.
[c]Calculated risk/hazard for these pathways were either $>10^{-4}$ for carcinogens or >1.0 for noncarcinogens (DOE 1995).
[d]Calculated carcinogenic risk for these pathways was between 10^{-5} and 10^{-6} (DOE 1995).
[e]Identified through nonconservative screening in each study.

Sources:

R. B. Cook, S. M. Adams, J. J. Beauchamp, M. S. Bevelhimer, B. G. Blaylock, C. C. Brandt, C. J. Ford, M. L. Frank, M. J. Gentry, S. K. Holladay, L. A. Hook, D. A. Levine, R. C. Longman, C. W. McGinn, J. L. Skiles, G. W. Suter, and L. F. Williams. 1992. *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk and Ecological Risk Screening Assessment.* ORNL/ER-155. Oak Ridge National Laboratory, Oak Ridge, Tenn.

DOE (U.S. Department of Energy). 1995. *Remedial Investigation/Feasibility Study Report for Lower Watts Bar Reservoir Operable Unit.* DOE/OR/01-1282&D4.

F. O. Hoffman, B. G. Blaylock, M. L. Frank, L. A. Hook, E. L. Etnier, and S. S. Talmage. 1991. *Preliminary Screening of Contaminants in the Off-Site Surface Water Environment Downstream of the U.S. Department of Energy Oak Ridge Reservation.* ORNL/ER-9. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Data sets available for use in this risk assessment consisted of data collected as part of the CRRI Phases 1 and 2 and all available historical data. The quality of the CRRI Phase 1 data is well documented (Holladay et al. 1993) as is that of the CRRI Phase 2 data (see Appendix G); however, quality control (QC) records are not available for some historical data. Therefore, two quality assurance (QA) criteria were implemented for the use of quantitative historical data in this risk assessment. First, the quality of the data must have been assessed by the researchers/investigators performing the sampling and analysis. The individuals who planned the historical studies must have, in some form, established criteria against which they then compared the quality of the data. Second, these quality-assured data must have been subsequently published in literature that was available to the public. Because ORNL and the other DOE facilities have established criteria for the publication of reports that include a specified amount of peer review before the release of information to the public, this second criterion ensured that all data used had undergone extensive peer review.

A brief summary of the QA procedures employed in each study from which data were used is presented in Table 5.2. The following subsections describe in detail the data sources that were used, the number and types of samples collected, and the aggregation of data sources, if applicable.

5.1.2 Data Compilation

The data sets used in the baseline human health risk assessment and their sources are shown in Tables 5.3–5.6. Typically, data collected during a remedial investigation comprises the majority, if not all, of the data used in the baseline risk assessment, and all available data are compiled into media-specific databases. Because multiple data sources and collection methods were used, a determination was made as to the appropriateness of combining all data for a particular media into one database. As a result of this evaluation, a single media-specific database was created for use in the baseline risk assessment.

For each of the data sets evaluated in this baseline risk assessment, tables of summary statistics have been prepared and are presented in Appendices B, C, and D. The tables include a reach identifier; the type of chemical (inorganic, organic, or radionuclide); the frequency of detection; the minimum, maximum, and mean concentrations; the representative concentration; units of measure; and a use flag. The frequency of detection column as well as the use flags will be defined in Sect. 5.1.3. The following list provides a brief description of each of the table headings that refer to concentrations:

- Minimum concentration is defined as either the minimum detected value or the minimum detection limit contained within the data set for a given analyte, whichever is smaller.

- Maximum concentration is defined as either (1) the maximum detected value, (2) the maximum detection limit for those analytes within the data set that were all nondetects (i.e., the frequency of detection was zero), or (3) the maximum detection limits from previous studies, in the case where Phase 2 sample detection limits were less than the historical detection limits.

- Mean concentration [listed as a product limit estimate (PLE) mean] is defined as the mean of the sample data set calculated with the product limit estimation technique described in Sect. 5.2.3.1 and is calculated as an average of all the sample values including detection limits as proxy concentrations for those samples that were nondetects when the frequency of detection is >0. Please note, in the case where the frequency of detection is equal to zero, it is not possible to calculate a sample mean.

Table 5.2. Quality assurance review of data sources used quantitatively in the baseline risk assessment

Data sources	Published reference	Description of quality assurance protocols
Oak Ridge Reservation Environmental Monitoring Program	Energy Systems 1993	• Conducted extensive internal quality control programs, participated in several external auality control programs (including the U.S. Environmental Protection Agency's Contract Laboratory Program), and statistically monitored performance on a routine basis. • Certified by State of Tennessee to conduct drinking water analyses • Used standard reference materials for instrument calibration, to standardize methods, and as spike additions for recovery tests. • Used single, blind, and control samples to evaluate laboratory performance
Clinch River Remedial Investigation Phase 1 and Phase 2	Phase 1: Cook et al. 1992 Holladay et al. 1993 Phase 2: Appendix G of this document	• Used field duplicates to assess the overall data precision • Used laboratory control samples, matrix spikes, surrogate spikes, and performance evaluation samples to assess data accuracy • Used standard analytical methods, proper preservation techniques, proper containers; adhered to samples holding times; and used field and laboratory blanks and equipment rinseate samples to ensure that the representativeness of the data was not compromised. • Data were validated by an external laboratory

Sources:

R. B. Cook, S. M. Adams, J. J. Beauchamp, M. S. Bevelhimer, B. G. Blaylock, C. C. Brandt, C. J. Ford, M. L. Frank, M. J. Gentry, S. K. Holladay, L. A. Hook, D. A. Levine, R. C. Longman, C. W. McGinn, J. L. Skiles, G. W. Suter, and L. F. Williams. 1992. *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk and Ecological Risk Screening Assessment.* ORNL/ER-155. Oak Ridge National Laboratory, Oak Ridge, Tenn.

S. K. Holladay, M. S. Bevelhimer, C. C. Brandt, R. B. Cook, W. D. Crumby, C. J. Ford, M. J. Gentry, L. A. Hook, D. A. Levine, R. C. Longman, S. E. Madix, R. L. Moody, C. D. Rash, and L. F. Williams. 1993. *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation.* ORNL/ER-152. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Martin Maritetta Energy Systems, Inc. 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992*, EH/ESH–31/V1. Oak Ridge National Laboratory, Oak Ridge, Tenn.

- Representative concentration is defined as either the upper 95% confidence bound on the mean or the maximum concentration (in the case where either the maximum concentration is less than the upper 95% confidence bound or the frequency of detection is ≤1. A more in-depth discussion of the derivation and use of the representative concentration is included in Sect. 5.2.3.

5.1.2.1 Water data

As indicated in Table 5.3, data from the CRRI Phase 1 and Phase 2 sampling activities were used in this risk assessment. In addition, monitoring data from the Clinch River and Poplar Creek, which were collected by ORREMP, were used to assess human health risk. Because the sampling and analytical methods are comparable for both the CRRI and the ORREMP activities, the various surface water data sets were combined into one media-specific database. The following paragraphs briefly describe each sampling activity. Summary statistics for the water data are listed in Table B3.

The CRRI Phase 1 (Cook et al. 1992) sampling occurred between December 1989 and February 1990. The analytical results were validated and a QA report was generated (Holladay et al. 1993). Analytes of concern for this sampling activity were inorganics, organics, and radionuclides. Sampling locations and analytical results for this activity are described in Chap. 3.

The CRRI Phase 2 (Cook et al. 1993) sampling occurred between November 1993 and September 1994. The original sampling and analysis plan was modified by an addendum at the request of DOE (DOE 1994b). As a result, fewer surface water samples were collected, and the water-biased event study was added.

The CRRI Phase 2 surface water sampling activities included an investigation of current contaminant sources (labeled W1.1), a water-biased event study (labeled W1.1B), contaminant data for toxicity tests (labeled W1.1T), a contaminant remobilization from sediments study (labeled W1.1R), and an in-stream distribution of contaminants from point sources study (labeled W1.4).

The ORREMP surface water data consisted of samples collected between March 1993 and September 1994. Eight surface water sampling locations were included in the data set: CRM 40.9, CRM 36, CRM 49.7, CRM 52.2, CRM 19.9, CRM 14.3, CRM 9.9, and PCM 1.4. Samples were analyzed for inorganics, radionuclides (gross alpha, gross beta, ^{90}Sr, ^{3}H, ^{99}Tc, total U, ^{137}Cs, ^{60}Co), volatile and semivolatile compounds, and pesticides/PCBs (DOE 1992). Because volatile contaminants were analyzed for but not found to be chemicals of concern in previous studies and because they would dissipate rapidly from the system, the volatile contaminants were not included in the risk assessment. ORREMP ran analyses for specific alpha emitters only if the gross alpha counts exceeded 3 pCi/L. Therefore, isotopic analyses for uranium isotopes and other alpha emitters are available for a subset of the total number of samples.

5.1.2.2 Sediment data

The sediment data available for use in this risk assessment were collected as part of the CRRI Phases 1 and 2 sampling and the near-shore surface sediment characterization activities. The data sources and the number of samples per reach for each are listed in Table 5.4.

Table 5.3. Sources of surface water data evaluated quantitatively in the baseline human health risk assessment

Source	Analyses	Number of samples				
		Reach 1	Reach 2	Reach 3	Reach 4	Reach 7
Clinch River Remedial Investigation Phase I (Cook et al. 1992; Holladay et al. 1993)	Inorganics	3	2	4	3	0
	Organics	3	2	4	3	0
	Radionuclides	3	2	4	2	0
Clinch River Remedial Investigation Phase 2—Task W1.1 (Chap. 3)	Inorganics	2	1	41	1	0
	Organics	2	1	41	1	0
	Radionuclides	2	1	41	1	0
Clinch River Remedial Investigation Phase 2—Task W1.1T (Chap. 3)	Inorganics	0	7	12	2	4
	Organics	0	7	8	2	4
	Radionuclides	0	7	12	2	4
Clinch River Remedial Investigation Phase 2—Task W1.1B (Chap. 3)	Inorganics	12	15	3	15	12
	Organics	12	15	3	15	12
	Radionuclides	12	15	3	15	12
Clinch River Remedial Investigation Phase 2—Task W1.1R (Chap. 3)	Inorganics	6	0	0	12	6
	Organics	6	0	0	12	6
	Radionuclides	6	0	0	12	6
Clinch River Remedial Investigation Phase 2—Task W1.4 (Chap. 3)	Inorganics	0	0	26	0	0
	Organics	0	0	0	0	0
	Radionuclides	0	0	30	0	0
Oak Ridge Reservation Environmental Monitoring Program (DOE 1992)	Inorganics	28	18	9	9	0
	Organics	28	18	9	9	0
	Radionuclides	28	18	9	9	0

Sources:

R. B. Cook, S. M. Adams, J. J. Beauchamp, M. S. Bevelhimer, B. G. Blaylock, C. C. Brandt, C. J. Ford, M. L. Frank, M. J. Gentry, S. K. Holladay, L. A. Hook, D. A. Levine, R. C. Longman, C. W. McGinn, J. L. Skiles, G. W. Suter, and L. F. Williams. 1992. *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk and Ecological Risk Screening Assessment.* ORNL/ER-155. Oak Ridge National Laboratory, Oak Ridge, Tenn.

S. K. Holladay, M. S. Bevelhimer, C. C. Brandt, R. B. Cook, W. D. Crumby, C. J. Ford, M. J. Gentry, L. A. Hook, D. A. Levine, R. C. Longman, S. E. Madix, R. L. Moody, C. D. Rash, and L. F. Williams. 1993. *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation.* ORNL/ER-152. Oak Ridge National Laboratory, Oak Ridge, Tenn.

DOE (U.S. Department of Energy). 1992. *Environmental Monitoring Plan for the Oak Ridge Reservation, USDOE-OR.* DOE/OR-1066.

Table 5.4. Sources of sediment data evaluated quantitatively in the baseline human health risk assessment

Source	Analyses	Number of samples				
		Reach 1	Reach 2	Reach 3	Reach 4	Reach 7
	Deep-water data					
Clinch River Remedial Investigation Phase I (Cook et al. 1992; Holladay et al. 1993)	Inorganics	2	2	2	2	0
	Organics	2	2	2	2	0
	Radionuclides	2	2	2	3	0
Clinch River Remedial Investigation Phase 2—Task S1.B (Chap. 3)	Inorganics	0	8	25	33	2
	Organics	0	8	25	33	2
	Radionuclides	0	8	24	33	2
	Near-shore data					
Near-Shore Sediment Characterization (Levine et al. 1994)	Inorganics	0	0	0	0	0
	Organics	0	0	0	0	0
	Radionuclides	21	159	0	150	0
Clinch River Remedial Investigation Phase I (Cook et al. 1992; Holladay et al. 1993)	Inorganics	2	2	2	2	0
	Organics	2	2	2	2	0
	Radionuclides	4	3	4	4	0
Clinch River Remedial Investigation Phase 2–Task S1.B (Chap. 3)	Inorganics	0	9	26	33	2
	Organics	0	9	25	33	2
	Radionuclides	0	9	24	33	2

Sources:

R. B. Cook, S. M. Adams, J. J. Beauchamp, M. S. Bevelhimer, B. G. Blaylock, C. C. Brandt, C. J. Ford, M. L. Frank, M. J. Gentry, S. K. Holladay, L. A. Hook, D. A. Levine, R. C. Longman, C. W. McGinn, J. L. Skiles, G. W. Suter, and L. F. Williams. 1992. *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk and Ecological Risk Screening Assessment.* ORNL/ER-155. Oak Ridge National Laboratory, Oak Ridge, Tenn.

S. K. Holladay, M. S. Bevelhimer, C. C. Brandt, R. B. Cook, W. D. Crumby, C. J. Ford, M. J. Gentry, L. A. Hook, D. A. Levine, R. C. Longman, S. E. Madix, R. L. Moody, C. D. Rash, and L. F. Williams. 1993. *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation.* ORNL/ER-152. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Levine, D. A., W. W. Hargrove, K. R. Campbell, M. A. Wood, and C. D. Rash. 1994. *Data Summary for the Near-Shore Sediment Characterization Task of the Clinch River Environmental Restoration Program.* ORNL/ER-264. Oak Ridge National Laboratory, Oak Ridge, Tenn.

introduced into WBK, were analyzed for PCBs and organic pesticides and herbicides because of their

During the Phase 1 sampling activities (December 1989–June 1990), both surface sediment grab and sediment core samples were collected from most sampling locations. These samples were analyzed for inorganics, organics, and radionuclides. The Phase 1 results are summarized in the *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk And Ecological Risk Screening Assessment* (Cook et al. 1992). A written summary of the data quality can be found in *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation* (Holladay et al. 1993).

After completion of the Phase 1 sampling, an extensive project was initiated in July 1990 to collect surface sediment grab samples from near-shore areas of both the Clinch River and WBR. The

5.1.2.4 Waterfowl data

The waterfowl data that were available for use in the Clinch River and Poplar Creek baseline risk assessment included data on Canada geese and wood ducks collected as part of the ORR BMAP. Evaluation of the available data suggested that Canada geese are the waterfowl species that would be most representative of site conditions and are the most prolific and more heavily hunted on or near the ORR. Furthermore, the Canada geese that were sampled and analyzed were collected from actual waste disposal ponds and on-site surface water bodies. Therefore, these geese potentially represent the maximally contaminated members within the goose population. Summary statistics are listed in Table D9.

5.1.3 Data Evaluation

5.1.3.1 Blanks

The CRRI QA personnel evaluated the analytical data and compared measured concentrations in samples with detected concentrations of a particular contaminant in the associated method and field blanks. By using the protocol outlined in *Risk Assessment Guidance for Superfund: Volume I, Human Health Evaluation Manual (Part A)* (EPA 1989c), common laboratory contaminants were considered as positively detected values only if the measured concentration in the sample was ten times greater than the measured concentration in the blank. For all other analytes, the analyte was considered a positively detected value only if the measured concentration in the sample was five times greater than the measured concentration in the blank. Sample results were then flagged appropriately in the CRRI database. A summary of the blank comparison and the results for the CRRI Phase 2 samples are contained in Appendix G of this document. The CRRI Phase 1 data were validated previously, and the results of that validation are contained in the *Quality Assurance/Quality Control Summary Report for the Phase 1 of the Clinch River Remedial Investigation* (Holladay et al. 1993). No blank comparisons were performed for the historical data incorporated in the data sets because blank information was not available. Thus, no historical contaminants were eliminated on the basis of this comparison.

5.1.3.2 Original samples versus duplicate samples

For those media and sampling locations where both original samples and duplicate or replicate analyses were conducted, the measured concentration in the original sample was averaged with the concentration in the duplicate or replicate to obtain a concentration value for that location.

5.1.3.3 Obtaining a measure for total chlordane

The reporting method used for chlordane provided concentrations for the various chlordane isomers but did not give a measure for total chlordane. However, on the basis of the analytical method and the

Table 5.4. Sources of sediment data evaluated quantitatively in the baseline human health risk assessment

Source	Analyses	Number of samples				
		Reach 1	Reach 2	Reach 3	Reach 4	Reach 7
Deep-water data						
Clinch River Remedial Investigation Phase I (Cook et al. 1992; Holladay et al. 1993)	Inorganics	2	2	2	2	0
	Organics	2	2	2	2	0
	Radionuclides	2	2	2	3	0
Clinch River Remedial Investigation Phase 2—Task S1.B (Chap. 3)	Inorganics	0	8	25	33	2
	Organics	0	8	25	33	2
	Radionuclides	0	8	24	33	2
Near-shore data						
Near-Shore Sediment Characterization (Levine et al. 1994)	Inorganics	0	0	0	0	0
	Organics	0	0	0	0	0
	Radionuclides	21	159	0	150	0
Clinch River Remedial Investigation Phase I (Cook et al. 1992; Holladay et al. 1993)	Inorganics	2	2	2	2	0
	Organics	2	2	2	2	0
	Radionuclides	4	3	4	4	0
Clinch River Remedial Investigation Phase 2–Task S1.B (Chap. 3)	Inorganics	0	9	26	33	2
	Organics	0	9	25	33	2
	Radionuclides	0	9	24	33	2

Sources:

R. B. Cook, S. M. Adams, J. J. Beauchamp, M. S. Bevelhimer, B. G. Blaylock, C. C. Brandt, C. J. Ford, M. L. Frank, M. J. Gentry, S. K. Holladay, L. A. Hook, D. A. Levine, R. C. Longman, C. W. McGinn, J. L. Skiles, G. W. Suter, and L. F. Williams. 1992. *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk and Ecological Risk Screening Assessment.* ORNL/ER-155. Oak Ridge National Laboratory, Oak Ridge, Tenn.

S. K. Holladay, M. S. Bevelhimer, C. C. Brandt, R. B. Cook, W. D. Crumby, C. J. Ford, M. J. Gentry, L. A. Hook, D. A. Levine, R. C. Longman, S. E. Madix, R. L. Moody, C. D. Rash, and L. F. Williams. 1993. *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation.* ORNL/ER-152. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Levine, D. A., W. W. Hargrove, K. R. Campbell, M. A. Wood, and C. D. Rash. 1994. *Data Summary for the Near-Shore Sediment Characterization Task of the Clinch River Environmental Restoration Program.* ORNL/ER-264. Oak Ridge National Laboratory, Oak Ridge, Tenn.

During the Phase 1 sampling activities (December 1989–June 1990), both surface sediment grab and sediment core samples were collected from most sampling locations. These samples were analyzed for inorganics, organics, and radionuclides. The Phase 1 results are summarized in the *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk And Ecological Risk Screening Assessment* (Cook et al. 1992). A written summary of the data quality can be found in *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation* (Holladay et al. 1993).

After completion of the Phase 1 sampling, an extensive project was initiated in July 1990 to collect surface sediment grab samples from near-shore areas of both the Clinch River and WBR. The purpose of this near-shore sampling was to determine the extent to which near-shore surface sediments are contaminated by releases from the ORR and provide data to the WBR Interagency Permitting Group for evaluation of human health risks from exposure to sediments during and following any proposed near-shore dredging operations. The near-shore sediment samples were analyzed by gamma spectrometry for ^{137}Cs and ^{60}Co. The results of the near-shore sediment characterization are contained in the *Data Summary for the Near-Shore Sediment Characterization task of the Clinch River Environmental Restoration Program* (Levine et al. 1994) and are also described in Chap. 3 of this report.

The CRRI Phase 2 sediment data from the following sampling activities were included in the human health risk assessment: (1) sediment contaminant characterization (labeled S1.B) and (2) a continuation of the near-shore surface sediment grab sample collections in areas where a dredging permit had been requested and was undergoing review by the WBR Interagency Permitting Group (labeled S2). Chapter 3 presents the results of the sediment characterization activities. A QA evaluation of the sediment data is contained in Appendix G.

In compiling the sediment data for use in the baseline risk assessment for the Clinch River and Poplar Creek, the data were split into two subsets (near-shore and deep-water). For purposes of the exposure pathways to be evaluated in the risk assessment, the near-shore data set, which comprises all samples collected at an elevation ≥733 ft will be used to assess the shoreline use exposure pathways. The remainder of the cores will be used to assess the deep-water environment through the dredging exposure pathways.

On the basis of the aforementioned criteria and the average water depth maintained above the Melton Hill Dam, no samples collected in either reach 1 or 7 can be defined as "near-shore." Therefore, no shoreline-use exposure pathways will be evaluated for reaches 1 or 7. Summary statistics for the "near-shore" data set are listed in Table C6 and for the "deep-water" data sets are listed in Table C4.

5.1.2.3 Fish data

Sources of data on levels of contaminants in fish include the CRRI Phase 1 and 2 (Table 5.5). Sampling locations are described in Chapter 3. The CRRI Phases 1 and 2 fish samples were analyzed by TVA. All data were combined into one media-specific database. Data were available for the following species of fish: bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), striped bass (*Morone saxatilis*), hybrid bass (*Morone chrysops/Morone saxatilis*), channel catfish (*Ictalurus punctatus*), gizzard shad (*Dorosoma cepedianum*), threadfin shad (*Dorosoma pentense*), carp (*Cyprinus carpio*), and white crappie (*Pomoxis annularis*). For the purposes of this risk assessment, only common food fish (bluegill, bass, and catfish) were used in the analyses. White crappie, a common food fish, was not analyzed because of an insufficient number of samples (one sampling activity at one location).

Not all species of fish accumulate contaminants to the same extent and not all species of fish were analyzed for the same contaminants; therefore, species were evaluated separately. The analyses were designed not only to identify the chemicals of concern (COCs) but also to focus on the species with the highest concentrations. Bluegill were analyzed for inorganics and radionuclides; catfish and largemouth bass were analyzed for inorganics, organics, and radionuclides; and striped and hybrid bass were analyzed for organics.

Table 5.5. Sources of fish data evaluated quantitatively in the baseline human health risk assessment

Source	Analyses	Number of samples[a]				
		Reach 1	Reach 2	Reach 3	Reach 4	Reach 7
Clinch River Remedial Investigation Phase I (Cook et al. 1992; Holladay et al. 1993)	Inorganics	13	29	35	27	0
	Organics	8	22	36	18	0
	Radionuclides	13	51	54	40	0
Clinch River Remedial Investigation Phase 2 (Chap. 3)	Inorganics	0	66	68	105	0
	Organics	3	61	72	142	0
	Radionuclides	0	0	0	0	0

[a]Number of samples per reach indicates a total number of analyses per reach, independent of species type.

Sources:

R. B. Cook, S. M. Adams, J. J. Beauchamp, M. S. Bevelhimer, B. G. Blaylock, C. C. Brandt, C. J. Ford, M. L. Frank, M. J. Gentry, S. K. Holladay, L. A. Hook, D. A. Levine, R. C. Longman, C. W. McGinn, J. L. Skiles, G. W. Suter, and L. F. Williams. 1992. *Phase 1 Data Summary Report for the Clinch River Remedial Investigation: Health Risk and Ecological Risk Screening Assessment*. ORNL/ER-155. Oak Ridge National Laboratory, Oak Ridge, Tenn.

S. K. Holladay, M. S. Bevelhimer, C. C. Brandt, R. B. Cook, W. D. Crumby, C. J. Ford, M. J. Gentry, L. A. Hook, D. A. Levine, R. C. Longman, S. E. Madix, R. L. Moody, C. D. Rash, and L. F. Williams. 1993. *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation*. ORNL/ER-152. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Fish were analyzed for Aroclor 1260 and Aroclor 1254 to determine the concentration of PCBs in fish tissue because these two chemical mixtures are the major contributors to the total PCB concentration. Emphasis was placed on analyzing catfish for PCBs because of their relatively high lipid content, which is directly related to their propensity for accumulating PCBs. Striped bass, a species introduced into WBR, were analyzed for PCBs and organic pesticides and herbicides because of their high lipid content, large size, and food use.

With the exception of striped and hybrid bass, which were lumped into one data set because of limited sample size and like characteristics, separate risk calculations were made for each species. Summary statistics were then calculated for each species of fish (Tables D3–D5 and Table D8). Although some individuals consume fish patties containing fish parts such as bone and roe, this risk assessment evaluates a reasonable maximum exposure only (i.e., the ingestion of fish fillets).

5.1.2.4 Waterfowl data

The waterfowl data that were available for use in the Clinch River and Poplar Creek baseline risk assessment included data on Canada geese and wood ducks collected as part of the ORR BMAP. Evaluation of the available data suggested that Canada geese are the waterfowl species that would be most representative of site conditions and are the most prolific and more heavily hunted on or near the ORR. Furthermore, the Canada geese that were sampled and analyzed were collected from actual waste disposal ponds and on-site surface water bodies. Therefore, these geese potentially represent the maximally contaminated members within the goose population. Summary statistics are listed in Table D9.

5.1.3 Data Evaluation

5.1.3.1 Blanks

The CRRI QA personnel evaluated the analytical data and compared measured concentrations in samples with detected concentrations of a particular contaminant in the associated method and field blanks. By using the protocol outlined in *Risk Assessment Guidance for Superfund: Volume I, Human Health Evaluation Manual (Part A)* (EPA 1989c), common laboratory contaminants were considered as positively detected values only if the measured concentration in the sample was ten times greater than the measured concentration in the blank. For all other analytes, the analyte was considered a positively detected value only if the measured concentration in the sample was five times greater than the measured concentration in the blank. Sample results were then flagged appropriately in the CRRI database. A summary of the blank comparison and the results for the CRRI Phase 2 samples are contained in Appendix G of this document. The CRRI Phase 1 data were validated previously, and the results of that validation are contained in the *Quality Assurance/Quality Control Summary Report for the Phase 1 of the Clinch River Remedial Investigation* (Holladay et al. 1993). No blank comparisons were performed for the historical data incorporated in the data sets because blank information was not available. Thus, no historical contaminants were eliminated on the basis of this comparison.

5.1.3.2 Original samples versus duplicate samples

For those media and sampling locations where both original samples and duplicate or replicate analyses were conducted, the measured concentration in the original sample was averaged with the concentration in the duplicate or replicate to obtain a concentration value for that location.

5.1.3.3 Obtaining a measure for total chlordane

The reporting method used for chlordane provided concentrations for the various chlordane isomers but did not give a measure for total chlordane. However, on the basis of the analytical method and the known physicochemical action of the various isomers, a measure of total chlordane was obtained by summing the measured concentrations of the isomers. As a result, all concentrations of the chlordane isomers (alpha-chlordane, gamma-chlordane, alpha-chlordene, gamma chlordene, chlordene, chlordane, and oxychlordane) were summed and a "chlordane, total" entry was added to the data set.

5.1.3.4 Radionuclide considerations

The ORREMP surface water monitoring data contained entries for uranium-radioactive, whereas the other available surface water data listed individual isotopic measurements for the various uranium isotopes (^{234}U, ^{235}U, ^{236}U, and ^{238}U). As a result, the ORREMP results for uranium-radioactive were conservatively included in the derivation of a representative concentration for ^{238}U.

In addition, the ORREMP surface water monitoring data contained entries for strontium-radioactive. Because the radioactive half-lives of the various strontium isotopes (^{89}Sr, ^{91}Sr, and ^{92}Sr) are so short-lived, the only strontium typically detected in a sample is ^{90}Sr. As a result, the ORREMP results for strontium-radioactive were included in the derivation of a representative concentration for ^{90}Sr.

To ensure that all available data are considered to be representative of current conditions, all radionuclide concentrations were decay-corrected to January 1, 1995.

5.1.3.5 Nondetected contaminants

For each media-specific data set, a determination was made as to whether a contaminant was not detected in any sample. If a contaminant's frequency of detection was zero at all sampling locations within a given exposure unit, that contaminant was eliminated from consideration in the risk assessment.

5.1.3.6 Essential nutrients

Essential nutrients (including Ca, Cu, Fe, Mg, P, K, and Na) (Dunne 1990) are considered to have little or no adverse effects on human health. Furthermore, these nutrients are not expected to have been released in the Clinch River or Poplar Creek. These analytes have been included in the summary statistics for the purpose of reporting all the available information but are not considered to be COPCs.

5.1.3.7 Frequency of detection

As is noted in the *Risk Assessment Guidance for Superfund: Volume I, Human Health Evaluation Manual* (EPA 1989c), "chemicals that are infrequently detected may be artifacts in the data due to sampling, analytical, or other problems, and therefore may not be related to site operations or disposal practices." For the purposes of this baseline risk assessment, the frequency of detection of all analytes was evaluated before the calculation of risk/hazard. Those chemicals that had a frequency of detection <5% (i.e., 1 detect in 20 samples) were eliminated from further consideration in the risk assessment because of the uncertainty associated with the accuracy/precision of the available analytical measurements.

5.1.3.8 Summary statistics

The result of the data manipulations described in the preceding sections is a final data set for each media evaluated in the baseline human health risk assessment. Summary statistics are provided for all contaminants in Appendices B, C, and D. These tables provide a listing of contaminants by exposure unit (reach) along with their frequency of detection; minimum, maximum, mean, and 95% upward bound (see Sect. 5.2.3 for determination of representative concentration). All contaminants remaining after the elimination of the contaminants that were 100% nondetects or that were detected in <5% of the samples are considered COPCs.

5.2 EXPOSURE ASSESSMENT

Exposure, in the context of human health risk, is defined as the contact of a person with a chemical or physical agent. For exposure to occur, a source of contamination or contaminated media must exist (1) that serves as a point of exposure or (2) that transports contaminants away from the source to a point where exposure could occur. In addition, a receptor must come into either direct contact (i.e., ingestion, inhalation, dermal contact, external exposure) or indirect contact (such as ingestion of foodstuffs that have bioaccumulated contaminants within their systems) with the contaminant. This concept is referred to as an exposure pathway. The elements of an exposure pathway are source, environmental transport/transfer media, exposure point, exposure route, and receptor.

An exposure assessment is the determination or estimation (qualitative or quantitative) of the magnitude, frequency, duration, route of exposure, and receptor population for each pathway evaluated in a baseline risk assessment. During the exposure assessment process, the risk assessor

- characterizes the exposure setting in an effort to identify the potentially exposed populations (receptors), their activity patterns, and any other characteristics that might increase or decrease their likelihood of exposure;

- identifies exposure pathwayson the basis of the characterization of the exposure setting [each exposure pathway identifies a unique mechanism by which a population may be exposed to the contaminants (swimming, gardening, private well use, etc.)];

- quantifies the exposure to a contaminant by estimating concentrations to which a receptor may be exposed; and

- calculates a chemical-specific intake or dose typically measured in milligrams per kilogram of body weight per day (mg/kg/d) and a radionuclide-specific dose typically measured in (risk per picocurie of contaminant) for each exposure pathway.

On the basis of the activity patterns of a population, any given individual may be exposed to more than one exposure pathway. For example, an individual who dredged sediment from the Clinch River would be exposed not only through dermal contact with the sediment but also through inhalation of any resuspended particulates, through inadvertent ingestion, or if the dredge soils were used as garden soil, through ingestion of homegrown produce. Therefore, the exposure assessment must include an evaluation of the activity patterns of the potential receptors and determine what combination, if any, of exposure pathways could affect an individual. This evaluation results in the generation of exposure scenarios. Exposure scenarios represent the combination (if applicable) of exposure pathways that an individual could be exposed to on the basis of his/her activity patterns.

Once the appropriate exposure pathways and scenarios have been identified, the risk assessor must select the appropriate equations and associated parameter values to calculate the amount of contaminant that is in contact with the body (skin, lungs, gut) per unit body weight per unit time (intake or dose) for each exposure pathway. The output of this activity is used in conjunction with the output from the toxicity assessment (Sect. 5.3) to quantify risks/hazards to receptors during risk characterization (Sect. 5.4). The exposure assessment process as applied to the Clinch River is detailed in the following subsections of this report.

5.2.1 Characterization of Exposure Setting

A detailed description of the geography, demography, climate, and topography of the area surrounding the Clinch River and Poplar Creek is contained in Chap. 2. With respect to the baseline human health risk assessment, the area under evaluation is the Clinch River from CRM 50.5 to CRM 1.0 and Poplar Creek from PCM 5.5 to 0.0 (See Chap. 3 for a detailed map). Because this area is quite large, a decision was made before the initiation of Phase 2 sampling to divide the OU into smaller sections or reaches. The boundaries of each reach were delineated on the basis of the location of a specific physical/topographic feature (e.g., dams, streams or river confluences). Each of the reaches is described in detail in Chap. 3. Furthermore, each reach was divided into subreaches on the basis of information about zones of sediment deposition.

Before the preparation of this baseline risk assessment, the risk assessors had to determine whether surface water, sediment, and fish data would be evaluated on a reach or subreach basis. The sampling locations for waterfowl negate the need for such an evaluation (Sect. 5.1.2.4). The evaluation process involved examination of the existing data to determine (1) if concentrations varied significantly enough between subreaches to indicate that "hot spots" of contamination were present and (2) if the quantity of existing data for each of the subreaches was both adequate and appropriate for drawing conclusions regarding risk/hazard. The results of this evaluation were as follows:

1. for fish and surface water data collected within the Clinch River, risks/hazards would be calculated and presented on a reach basis for reaches 1, 2, 4, and 7;

2. for deep-water sediment collected in the Clinch River, risks/hazards would be calculated for reaches 1 and 7 and subreaches 2.04, 4.01, 4.02, 4.03, and 4.04;

3. for "near-shore" sediment collected in the Clinch River below the Melton Hill Dam risks/hazards would be calculated for subreaches 2.04, 4.01, 4.02, 4.03, and 4.04; and

4. for surface water and sediment (both near-shore and deep-water) collected in Poplar Creek risks/hazards would be evaluated for subreaches 3.01, 3.02, 3.03, and 3.04.

As discussed in Chaps. 2 and 3, land surrounding the Clinch River is used for residential, farming, industrial, and recreational purposes. The Clinch River and Poplar Creek both flow through agricultural and industrial areas, where contaminants from both point and nonpoint sources can enter the water. The fate of a contaminant depends not only on the flow rate of the water but also on the physical and chemical properties of the contaminant. Dissolved substances are usually flushed through the system in a matter of weeks, whereas particle-associated contaminants may accumulate in the sediment and remain indefinitely. This is evidenced by the fact that peak concentrations of contaminants known to have been released in the past are buried at depths that can be correlated with release histories. These buried contaminants are indicative of areas of sediment accumulation that have remained stable for 40 years.

The Clinch River and to a lesser degree, Poplar Creek provide a variety of recreational opportunities, such as sport fishing, hunting, boating, swimming, water skiing, and camping. The greatest participation in these activities is in the summer, but because eastern Tennessee has a temperate climate, some activities are pursued year round. The area also draws a substantial amount of out-of-town visitors. This risk assessment focuses on local inhabitants who regularly engage in activities where they can come in contact with or ingest fish, sediment, and/or surface water from the Clinch River and Poplar Creek.

Currently, a fish advisory exists which limits the consumption of fish from the Clinch River and Poplar Creek because of concern over PCB contamination. PCBs have been released from the ORR (Loar et al. 1992); however, PCBs also occur in many streams and lakes in eastern Tennessee.

5.2.2 Identification of Exposure Scenarios

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

As is indicated in the site conceptual model (Fig. 5.1), several exposure scenarios were selected for evaluation in this baseline risk assessment: ingestion of untreated surface water, consumption of fish, ingestion of waterfowl, exposure to near-shore sediment through shoreline use, exposureto surface water through swimming, exposure through irrigation, and through the use of deep-water dredged sediments for agricultural purposes. Most of the scenarios included in this risk assessment were previously evaluated in the screening assessments of Hoffman et al. (1991) and Cook et al. (1992). Because both the Clinch River and Poplar Creek are readily accessible to the public and are not protected property of DOE, all exposure pathways are considered residential. In addition, because not all individuals use the resources of the Clinch River or Poplar Creek in the same way, the risk assessment process includes a variety of exposure scenarios to account for the different ways an individual might be exposed to potentially harmful contaminants.

Exposure scenarios were designated as either current or future in this risk assessment. Current exposure scenarios evaluate those exposure pathways that are considered to occur or have the potential to occur; they use measured contaminant concentrations for the media of interest. These scenarios include consumption of fish, ingestion of waterfowl, exposure to near-shore sediments through shoreline use, exposure to surface water through swimming, and ingestion of untreated surface water.

Future exposure scenarios evaluate exposure pathways that are not currently occurring but have the potential to occur at a later date. For this assessment, the deep-water dredging and irrigation scenarios are considered future exposure scenarios; they use modeled estimates of contaminant concentrations for the media of interest. For example, contaminant concentrations for soil irrigated with surface water were not available; nevertheless, concentrations were predicted with mathematical models. Other models were used to predict the uptake of the contaminants in the soil by pasture forage and the subsequent transfer into the milk and meat from cattle that consume the forage. Consumption of milk and meat from cattle that have grazed on pastures irrigated with Clinch River or Poplar Creek surface water are examples of future exposure pathways.

5.2.3 Quantification of Exposure

As stated previously, exposure, in the context of risk to human health, is defined as the contact of a person with a chemical or physical agent. To quantify exposure, a representative concentration must be determined for each analyte and then chemical intakes must be calculated for the various exposure pathways identified for the site.

5.2.3.1 Derivation of an exposure concentration

Before the derivation of the chemical-specific intake values for each pathway, an exposure concentration must be developed. The exposure concentration, the amount of each chemical in the media of interest, is the first variable to be defined in the exposure assessment process. EPA (1989c) recommends the use of the arithmetic average (mean) of the chemical concentration in a medium as the exposure concentration. Because of the variability inherent in sampling natural systems where the degree and extent of contamination is unknown, accurately estimating the true mean becomes difficult. Because it is not possible to know the true mean concentration of a contaminant, the UCL_{95} of the mean is used

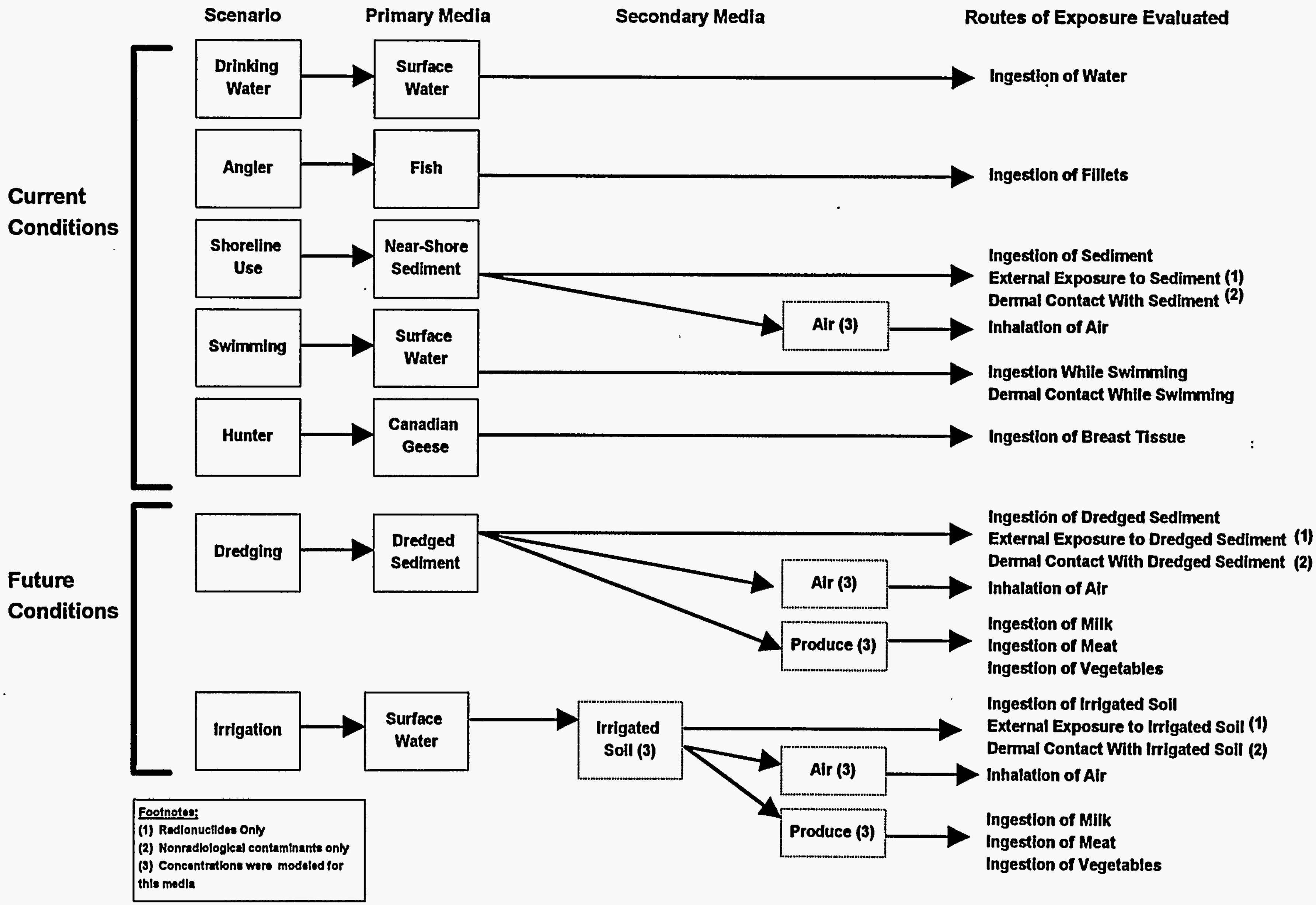


Fig. 5.1. Site model for human health risk assessment.

as a conservative estimate of the true mean. Measured contaminant concentrations were not available for all exposure media so mathematical models were used to estimate the concentrations. For media with limited sample data or extreme variability in the measured or modeled concentrations, the UCL_{95} can be greater than the highest measured or modeled concentration. In these cases, the maximum measured or modeled value is used as the exposure concentration instead of the UCL_{95}. It should be noted, however, that the true (but unknown) mean could be higher than the maximum value, especially if the most contaminated portion of the site has not been sampled. This issue is discussed in the uncertainty analysis (Sect. 5.5).

Before the derivation of summary statistics, estimates and confidence bounds for mean concentrations were computed by three methods: (1) by substituting detection limits for nondetect results and computing the usual arithmetic mean and standard error, (2) by computing the mean and standard error using the lognormal model and the method of maximum likelihood, and (3) by computing a mean and standard error from the PLE. The purpose of this evaluation was to determine the most appropriate statistical method to use to derive summary statistics for the Clinch River data sets because the data sets were comprised not only of CRRI Phase 1 and 2 data, but also existing historical data.

The PLE is a nonparametric distribution function estimate that statistically accounts for nondetects. It was originally defined for right-censored (failure-time) data (Kaplan and Meier 1958), but applies to left-censored (nondetect) data, as well. The left-censored version is given in Schmoyer et al. (1995).When there are no nondetects, the PLE reduces to the usual empirical distribution function. If there are nondetects, however, and if the smallest observation (concentration or detection limit) is a nondetect, then, the PLE is not uniquely defined for concentration values below that smallest observation. In such cases, the mean of the PLE is likewise undefined.

To accommodate these cases, various ad hoc definitions of the PLE have been proposed. Perhaps the most conservative is to take the PLE to be zero for concentrations smaller than the smallest detected value. This conservative definition is the one used, for example, in the SAS Lifetest procedure (for right-censored data). Simulation results of Schmoyer et al. suggest that even this conservative PLE mean tends to be slightly anticonservative for nonnormal data distributions. We therefore adopted the conservative definition for the PLE and PLE mean estimate. From the same simulation results, the authors further concluded that the PLE mean is preferable to either approaches (1) or (2), which tend to be too conservative (when no physical basis exists for the lognormal model).Therefore, we selected PLE means and standard errors (and confidence bounds) as the primary statistics for this RI. Means and standard errors computed by the other two methods were carried along as a check but are not reported.

5.2.3.2 Derivation of the chronic daily intake

Once the exposure concentration is calculated, exposure to contaminants is evaluated quantitatively by deriving the chronic daily intake of a chemical or radionuclide. For this baseline human health risk assessment, intake is defined as the amount of a contaminant that an individual takes into his/her body per day through ingestion, inhalation, or dermal contact; dose is defined as the total absorbed energy from exposure to a particular radionuclide.

The general equation for calculating chemical-specific intakes for receptors and the exposure pathways to be evaluated in the risk assessment has been defined by EPA (1989c) as follows:

$$Intake = \frac{C \times IR \times EF \times ED}{BW \times AT}\,, \tag{5.1}$$

where

C	=	concentration of a chemical in the environmental media of concern;
IR	=	ingestion rate of the media per day or per hour, as appropriate;
EF	=	exposure frequency;
ED	=	exposure duration;
BW	=	body weight; and
AT	=	averaging time.

Ingestion rates for the various environmental media as well as recommendations for exposure frequencies and durations for various exposure pathways have been provided by the EPA. EPA's standard default exposure factors (EPA 1989c, 1991a) are used in calculating the pathway-specific intakes where appropriate. Knowledge of site-specific conditions and receptor activity is used when guidance is not available or when professional judgment deems necessary. The default values are typically either the average value, the 50th percentile value, or the 90th percentile value.

Equation 5.1 must be modified on the basis of pathway-specific considerations. Depending on the pathway under evaluation, the exposure frequency might differ or an exposure time might be included in the equation to more reasonably estimate the intake of COCs. The pathway-specific equations that were used in this baseline human health risk assessment are listed in Tables E1–E5, E7–E18, E20–E24 and E27. These tables contain a detailed listing of the exposure parameters used in the pathway-specific equations and a source for each of the parameters.

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

5.2.3.3 Surface water ingestion scenario

The surface water ingestion scenario assumes that an adult weighing 70 kg and a child weighing 15 kg drinks untreated water from the reservoir at a rate of 2 L/d and 1 L/d, respectively. The duration of exposure for adults is 30 years, whereas the duration of exposure for a child is 6 years. As far as is known, no one routinely drinks untreated reservoir water, but because the possibility exists, this exposure scenario is included in the risk assessment. The exposure equation for the ingestion of drinking water is listed in Table E1.

5.2.3.4 Shoreline use scenario

The shoreline use pathways, as evaluated for subreaches 2.04–4.04, consider persons who spend time along the shoreline of the reservoir where they may come in contact with near-shore sediment through activities such as walking, searching for artifacts, and wading. Some activities such as walking and searching for artifacts are often more popular in the fall and winter when the water level in the reservoir is at its lowest. In many locations, the winter water level provides a broader beach and easier access. The contaminant exposure concentrations for the shoreline use pathways are listed in the representative concentration column in Table C6.

Four exposure pathways are included in the shoreline use pathways: inadvertent ingestion of near-shore sediment, inhalation of the sediment in the form of dust, external exposure to gamma-emitting radionuclides in the sediment, and dermal contact with near-shore sediment (Fig. 5.1). It is reasonable to assume that an individual exposed through inhalation would also be exposed by external radiation. Therefore, these two pathways are summed for evaluating an individual exposed by inhalation. The parameter values and equations used for the human health risk assessment are given in Appendix E for exposure to the following pathways: (1) inadvertent ingestion of near-shore sediment (Table E2), (2) dermal contact with near-shore sediment (Table E3), (3) inhalation of resuspended sediment (Table E4), and (4) external exposure to radiation from gamma-emitting radionuclides in near-shore sediments (Table E5). Each pathway is discussed briefly in the following text.

Shoreline use: inadvertent ingestion of near-shore sediment. An individual who spends time along the shore of the Clinch River or Poplar Creek is assumed to inadvertently ingest a small quantity of sediment. Table E2 lists the intake equations and exposure parameters used to assess the risk of adverse health effects from the inadvertent ingestion of near-shore sediment. Parameter values used in the exposure equation have been derived from standard intake rates, exposure frequencies, exposure durations, and averaging times. These values are based on EPA standard default values for residential settings.

Shoreline use: dermal contact with near-shore sediment. An individual who spends time at a beach area is assumed to come into physical contact with the sediment and to absorb some fraction of the contaminant(s) in the sediment through the skin. Radionuclides were not evaluated for dermal exposure because it was assumed that the dose from external exposure to the radionuclides would provide a much greater health risk than the dose from dermal contact with those same radionuclides. In addition, it was assumed conservatively that an individual would only visit the beach area for a maximum of 6 months/year or half of the standard exposure frequency of 350 d/year. Pathway-specific intake parameters are listed in Table E3.

Shoreline use: inhalation of resuspended particles from near-shore sediment. The evaluation of the exposure of an individual to contaminants in near-shore sediment that are resuspended in the air is dependent on the quantity of fugitive dust (resuspended sediments) generated by the wind. Although no air monitoring data were available for the Clinch River or Poplar Creek area, the quantity of fugitive dust generated by the wind was empirically derived by the method described in Eckerman and Young (1980). This method was also used in the previous screening risk assessments conducted by Hoffman et al. (1991) and Cook et al. (1992). The risk assessment equations and parameter values are given in Table E4.

Shoreline use: external exposure to near-shore sediment. The external exposure pathway is concerned only with external exposure to gamma radiation from radionuclides while an individual spends time along the shoreline. Radionuclides are carcinogens, and as noted earlier, only adults are considered for exposure to carcinogens. Exposure equations and related parameter values are listed in Table E5. These equations and parameter values were obtained from the National Council on Radiation Protection (NCRP) (1984) and EPA (1989c).

5.2.3.5 Swimming scenario

An individual can be exposed to water in the Clinch River or Poplar Creek through recreational activities such as swimming, water skiing, and wading. The swimming pathways are included in the risk assessment because swimming offers the greatest potential for dermal exposure. The exposure frequency for the swimming pathways is limited to 45 days because swimming is a local summer activity and EPA (Region IV) has established 45 days as a reasonable maximum exposure frequency for the southeast (EPA 1991c). Water data were collected during the Phase 1 and Phase 2 sampling activities. In addition, surface water data collected as part of the ORREMP were used to supplement the surface water data set. The exposure concentrations detected in the surface water samples are listed in the representative concentration column in Table B3.

Swimming: inadvertent ingestion of surface water. Many swimmers will ingest some of the surrounding water while swimming. Standard EPA default intake equations and parameter values were used to calculate exposure estimates from this exposure pathway. The equations and parameter values are listed in Table E7.

Swimming: dermal contact with surface water. Chemicals that are dissolved in the water can be absorbed through the skin while swimming. The rate of absorption is chemical specific and is represented by a permeability constant. (These constants are discussed in Sect. 5.3 and are presented in Table E6). As in the case of dermal contact with near-shore sediments, radionuclides were not included in the dermal absorption pathway because the risk of health effects from external exposure to radiation would be much greater than the chemical effects. The exposure equations and associated parameter values for dermal contact with surface water are listed in Table E8.

5.2.3.6 Fish consumption scenario

Many individuals who live in the vicinity of the Clinch River and Poplar Creek are avid anglers. The fish consumption scenario represents an avid angler who regularly catches fish in the Clinch River or Poplar Creek and uses the fish to feed the family. Because many anglers are selective in the species of fish they eat and not all species of fish accumulate contaminants in their tissues to the same extent, several species of fish are included in this assessment; these species are bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), striped bass (*Morone saxatilis*), hybrid bass (*Morone chrysops/Morone saxatilis*), and channel catfish (*Ictalurus punctatus*). With the exception of striped and hybrid bass, which were lumped into one data set because of limited sample size and like characteristics (see Sect. 5.1.2.3), separate risk calculations were made for each species.

The equations and parameter values used for calculating the baseline human health risk assessment for the fish consumption exposure pathway are given in Table E9. The contaminant exposure concentrations for this scenario are listed in the representative concentration columns in Tables D3–D5 and Table D8.

Separate risks were calculated for the ingestion of different species of fish to provide risk factors for individuals who eat only one species of fish. The EPA-recommended ingestion rate of 54 g of fish per day was used to calculate the risk for each species; therefore, the risk for each species should not be added to obtain a risk to an individual who eats all species. Instead, the risk for the species that has the maximum value should be used as the upper-bound risk for the ingestion of fish.

5.2.3.7 Waterfowl consumption scenario

As described in Sect. 5.1.2.4, the waterfowl data available for use in the baseline human health risk assessment were limited to data collected as part of the CRRI Phase 1 and Phase 2 sampling activities. Of the available data, only the tissue data for Canada geese were used to evaluate risk to human health. Standard EPA default exposure parameters are not available for intake and exposure duration for the ingestion of waterfowl. Therefore, on the basis of personal communication with the Tennessee Wildlife Resources Agency (TWRA), posted bag limits for the various species of waterfowl, and the duration of the hunting season, best professional judgement was used to establish an ingestion rate and exposure duration for the consumption of Canada geese.

The ingestion rate that was established is based on the assumption that the average hunter would harvest two geese per year. Although the hunting season is typically 45 days and the bag limit is two geese per day, the TWRA provides estimates of the number of hunting permits issued and the estimated kills based on their tag and check-out system. For 1992–93, the number of permits issued for the Watts Bar hunt unit was 639 and the estimated kill for those years was 173 (TWRA 1994). Therefore, the assumption that the average hunter would harvest two geese per year is considered by the assessors to be reasonable. The exposure frequency was considered to be 14 d/year, or 2 weeks.

An average Canada goose weighs ~12 pounds. Individuals typically eat only the breast tissue, which is ~1/3 of the weight of the goose, or 4 lb. In addition, the typical hunter would share half his kill with his family or friends. Therefore, for use in this baseline risk assessment each hunter was assumed to consume ~4 lb of goose per year (equals 12 lb per goose × 2 geese per year × 1/3 × 0.5). The exposure equation and parameter values for the ingestion of Canada geese are listed in Table E10.

5.2.3.8 Dredging scenario

Historical and recently collected data indicate that the highest concentrations of ^{137}Cs and mercury in the sediment of the Clinch River are buried under 40 to 80 cm (1 to 2 ft) of less-contaminated sediment in depositional zones along the old river channel (Energy Systems 1992; Olsen et al. 1992). This situation occurs because releases of contaminants from the ORR have decreased significantly since the 1950s and 1960s (Chap. 3), and the more recently deposited sediments are much less contaminated. As long as the sediments with the highest levels of contamination remain buried, there is no realistic exposure pathway for humans and no risk to human health. Therefore, the most viable pathway for human exposure would result from dredging the deep-water sediment to maintain a navigational channel or for other reasons. If the dredged spoils are placed on the shore, they provide a potential pathway for human exposure. Because dredging is an action that could occur in the future, a dredging scenario was used to evaluate the health risks associated with the use of deep sediments for agricultural purposes.

Dredging activities in the Clinch River and Poplar Creek are regulated by TVA and USACE so that the uncontrolled disposal of dredged spoils is unlikely. The dredging scenario assumes that deep-water sediment is removed from the reservoir and deposited upon agricultural land, where it is used to grow crops and raise livestock. The sediment is assumed to be spread deep enough so that plowing would not dilute the material by mixing it with the underlying soil. Therefore, this exposure scenario assumes worst-case conditions.

Occasionally, a lakefront property owner will request permission from TVA to remove sediment from an area near his/her property to provide navigable water to a boat dock. In such instances, the dredged spoils may be placed onshore where humans could come in contact with them or where the spoils could be used for agricultural purposes. However, because the shoreline use pathways include direct exposure pathways for near-shore surface sediment and the quantity of sediment that would be dredged is typically only several cubic yards or less, dredged near-shore surface sediment is not considered in the dredging scenario.

The five exposure pathways evaluated under the dredging scenario are also evaluated for the irrigation scenario (Fig. 5.1). Direct exposure occurs through ingestion of dredged sediment, dermal contact with the sediment, inhalation of air containing dust derived from the sediment, and external exposure to gamma-emitting radionuclides. Indirect exposures occur through the ingestion of milk, meat, and vegetables produced on the dredged sediment.

Data for the dredging scenario came from cores taken during the CRRI Phase 1 (Cook et al. 1992) and Phase 2. Individual core concentrations were calculated from mass-weighted averages over the entire length of the core. Data from all cores were combined into one data set and summary statistics were calculated from this data set (Table C4). Contaminant concentrations for milk and meat, and vegetables produced on dredged sediment were calculated in the same way as those produced on irrigated soil (Tables E11–E14).

Dredging scenario: inadvertent ingestion of dredged sediment. Table E15 lists the exposure parameters for the inadvertent ingestion of dredged sediments. The parameter values used in the exposure equation are standard EPA default values for residential exposure.

Dredging scenario: dermal contact with dredged sediment. The sediment concentrations for dermal contact with the dredged sediments are the same as those for ingestion of dredged sediment (Table C4). Radionuclides were not evaluated in this pathway because the dose from external exposure to the radionuclides was assumed to be much greater than the dose received from dermal absorption of the same radionuclides. Exposure equations and pathway-specific intake parameters are listed in Table E16.

Dredging scenario: inhalation of resuspended particles from dredged sediment. No air monitoring data were available for the Clinch River or Poplar Creek area; therefore, transport equations were used for estimating air concentrations associated with the dredged sediments. Equations and parameter values used to estimate exposure to a receptor are given in Table E17.

Dredging scenario: external exposure to dredged sediment. The equation and parameter values used to estimate the dose from external exposure to radionuclides are listed in Table E18 . The pathway-specific exposure equations and parameter values listed in Table E18 were taken from NCRP (1984) and EPA (1989c).

Dredging scenario: ingestion of vegetables, meat, and milk produced on dredged sediment. As indicated in the site conceptual model (Fig. 5.1), the consumption of contaminated produce is a potential exposure pathway if dredged sediment is used for agricultural purposes. The contaminant concentrations used for the ingestion of dredged sediment were used to model the expected concentrations in vegetables, meat, and milk. Models for estimating plant and animal uptake are the same as those used in the irrigation scenario (Tables E11–E14). Modeled contaminant concentrations in milk, meat, and vegetables are given in Table E19. The equation and parameter values used to estimate the intake from ingestion of vegetables, meat, and milk produced on dredged sediment are listed in Table E20.

5.2.3.9 Irrigation scenario

The irrigation scenario assumes that an individual uses surface water from the Clinch River or Poplar Creek in an overhead irrigation system to grow vegetables for his/her family and forage for cows. Surface water is not widely used for irrigation purposes because of the amount of rainfall occurring in eastern Tennessee and the types of crops grown in the area. However, a limited amount of irrigation is used for lawns, pastures, and vegetable gardens because of ease of access and occasional dry periods. In addition, in some instances, irrigation is used to protect crops from damage by frost and freezing temperatures. For these reasons, irrigation cannot be ruled out as a potential exposure pathway. However, irrigation is considered an assessment of future conditions because such activity is limited currently but could increase in the future. Contaminant concentrations are modeled values that represent a 30-year buildup period. The exposure pathways associated with the irrigation scenario are shown in Fig. 5.1.

The EPA recommends that 30 years be used as a default lifetime exposure period in human health risk calculations (EPA 1991a). If an individual uses lake water to irrigate crops for 30 years, certain contaminants present in the water may build up in soil. For this reason, the values that are used as concentrations for contaminants in irrigated soil are based on a 30-year accumulation minus losses attributable to leaching, harvesting, and radioactive decay of radionuclides. Therefore, irrigated soil is defined as soil that has been irrigated with water from the Clinch River or Poplar Creek for a period of 30 years. The methodology for calculating the concentrations of contaminants in the irrigated soil and produce is provided in Appendix E. The exposure equations are given in Appendix E for the following exposure pathways: (1) inadvertent ingestion of irrigated soil (Table E21); (2) dermal contact with irrigated soil (Table E22); (3) inhalation of resuspended irrigated soil (Table E23); (4) external exposure to radionuclides in irrigated soil (Table E24) ; and (5) ingestion of vegetables, milk, and meat produced on irrigated soil (Table E27).

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

The equations for calculating contaminant concentrations in the irrigated soil after a 30-year buildup period are given in Tables E11–E14. The water data are the same as those used in the water ingestion and swimming scenarios. The calculated contaminant concentrations for irrigated soil are listed in Table E25 and E26.

Irrigation scenario: inadvertent ingestion of irrigated soil. Table E21 lists the exposure parameters associated with the inadvertent ingestion of irrigated soil. This ingestion occurs as a result of working with the soil as well as from the ingestion of produce contaminated with soil. The parameters used in this exposure equation have been derived from standard intake rates, exposure frequencies, exposure durations, body weights, and averaging times.

Irrigation scenario: dermal contact with irrigated soil. The soil exposure concentrations for dermal contact with irrigated soil are the same as those for the ingestion of irrigated soil. Radionuclides were not evaluated in this scenario because the health risk from external exposure to the radionuclides in the soil was assumed to be much greater than the risk from dermal absorption of the radionuclides. Pathway-specific intake parameters are listed in Table E22.

Irrigation scenario: inhalation of resuspended particles from irrigated soil. Table E23 lists the exposure parameters associated with the inhalation of resuspended particles resulting from wind-generated dust. The concentration term used in this equation is a modeled value. The parameters used in this exposure equation have been derived from standard intake rates, exposure frequencies, exposure duration, body weights, and averaging times.

Irrigation scenario: external exposure to irrigated soil. The external exposure equation and pathway-specific intake parameters are listed in Table E24. The external exposure pathway was limited to an adult exposure because gamma radiation is considered carcinogenic. As explained in Sect. 5.2.2, the calculated risk of adverse carcinogenic health effects for a child would be less than for an adult because of the shorter exposure period.

Irrigation scenario: ingestion of vegetables, meat, and milk grown on irrigated soil. As shown in the site-conceptual model (Fig. 5.1), the consumption of food produced on soil irrigated with river water is a potential exposure pathway. The soil concentrations used in the soil ingestion pathway were used to model the expected concentrations in vegetables, meat, and milk. Equations used for estimating plant and animal uptake were taken from the International Atomic Energy Agency (IAEA) (1982) and DOE (1987) and are listed in Tables E11–E14. For inorganics and radionuclides, contaminant-specific uptake coefficients were used to estimate the contaminant concentrations in produce on the basis of the modeled contaminant concentrations in the irrigated soil. Each transfer coefficient represents a ratio of the contaminant concentration in the plant to the concentration in the soil (Table E28). Therefore, the product of the contaminant concentration in irrigated soil and the transfer coefficient is the contaminant concentration in the plant. The soil-to-plant elemental transfer coefficients for the reproductive portions of plants as given by Baes et al. (1984) were used to estimate the contaminant concentrations for the ingestion of produce.

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

$$\log B_v = 1.588 - 0.578 \log K_{ow} , \tag{5.2}$$

where

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

Modelled concentrations of contaminants in milk, meat, and vegetables grown on irrigated soil are listed in Tables E26 and 27. Note that in deriving the concentration of contaminants in milk and meat, incidental ingestion of soil and ingestion of surface water by cattle are included in the modeling equations.

5.3 TOXICITY ASSESSMENT

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

Data used to discuss the chemical-specific potential health effects in both this section and in Appendix E are from human and laboratory animal research and from occupational studies to characterize likely health effects resulting from exposure to the COPCs. Refer to the *BEIAS Toxicity Profiles* report (1994) for further information regarding specific chemicals. Tables 5.6–5.8 summarize the available toxicity information for the Clinch River and Poplar Creek COPCs. Chemical-specific information aimed at providing the reader with general qualitative information about carcinogenic and noncarcinogenic health effects related to those COCs identified in this baseline human health risk assessment (Sect. 5.4) will be presented in Appendix E.

5.3.1 Toxicity Information for Noncarcinogenic Effects

Noncarcinogenic effects are evaluated by comparing an exposure experienced over a specified time period (e.g., 30 years) with a reference dose (RfD) [or reference concentration (RfC)] derived for a similar exposure period. The RfDs available for the COPCs present in the Clinch River are given in Table 5.6. To evaluate the noncarcinogenic effects from exposure to the COPCs in the Clinch River and Poplar Creek, the hazard quotient (HQ) [the ratio of the exposure dose (i.e., intake) to the RfD] is calculated for each COPC. An HQ is used to evaluate exposure to noncarcinogens only and assumes that, below a given level of exposure (i.e., the RfD), even sensitive populations are unlikely to experience adverse health effects. If the exposure level (intake) exceeds this threshold (i.e., if intake/RfD > 1.0), there may be concern for potential systemic health effects. The level of concern does not necessarily increase linearly as the HQ approaches or exceeds unity; the HQ is not a percentage or probability.

Chronic RfDs are developed for protection from long-term exposure to a chemical (7 years to a lifetime); subchronic RfDs are used to evaluate short-term exposure (2 weeks to 7 years) (EPA 1989a). No short-term or acute exposure scenarios were evaluated as part of this baseline risk assessment. Although exposure to a child was evaluated and the child exposure duration is <7 years, subchronic RfDs were not used to evaluate exposure to children because the subchronic RfDs are derived from an average cohort and do not take into account age-specific physiochemical or physiokinetic processes.

5.3.2 Toxicity Information for Carcinogenic Effects

For carcinogens, risks are estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to the carcinogen. Cancer risk from exposure to contamination is expressed as excess cancer risk or, stated differently, cancer incurred in addition to normally expected rates of cancer development in the general population. An excess cancer risk of 1.0E-06 indicates that an individual has a one in a million chance of developing cancer from exposure to this contamination level over a 70-year lifetime.

To evaluate the carcinogenic risk from exposure to Clinch River and Poplar Creek COPCs, the risk is calculated for each COPC [the multiplication of the exposure dose (i.e., CDI and/or dose) by the slope factor (SF), which is a chemical-specific value based on carcinogenic dose-response data]. Because the SFs are the UCL_{95} on the probability of a carcinogenic response, the carcinogenic risk estimate represents an upper confidence bound estimate. Therefore, a 5% probability exists that the actual risk will be higher than the estimate presented, and the actual risk may well be less than the estimate. Slope

Table 5.6. Toxicity information for the Clinch River and Poplar Creek noncarcinogenic contaminants of potential concern

Chemical	Chronic oral RfD[a,b] (mg/kg/d)	Confidence level[a]	GI absorption[c] (%)	Chronic oral RfD absorbed[d,e] (mg/kg/d)	Chronic inhalation RfD[a,b,f] (mg/kg/d)	RfD basis (vehicle)[a]	Critical effect[a]	Uncertainty factor[a,g]/Modifying factor
				Inorganics				
Antimony	4.0E-04	Low	2.0	8.0E-06	na	oral	gastrointestinal disorders	UF = 1000 MF = 1
Arsenic, inorganic	3.0E-04	Medium	41	1.2E-04	na	oral	CNS, cardiovascular system, skin	UF = 3 MF = 1
Beryllium	5.0E-03	Low	1.0	5.0E-05	na	oral	reduced weight, rickets, lung, skin	UF = 100 MF = 1
Boron and Borates only	9.0E-02	Medium	90	8.1E-02	5.7E-03	na	testicular atrophy, spermatogenic arrest (dog)	UF = 100 MF = 1
Cadmium (diet)	1.00E-03	High	1.0	1.0E-05	na	oral	renal toxicity, osteomalacia, osteoporosis	UF = 10 MF = 1
Cadmium (water)	5.0E-04	High	1.0	5.0E-06	na	oral	renal toxicity, osteomalacia, osteoporosis	UF = 10 MF = 1
Chromium VI	5.0E-03	Low	2.0	1.0E-04	na	oral water	hepatotoxicity, nephrotoxicity, dermatitis	UF = 500 MF = na
Manganese (diet)	1.4E-01	Medium	4.0	5.6E-03	1.4E-05	oral	lethargy, tremors mental disturbance, muscle tonus, CNS	UF = 1 MF = 1

Table 5.6. (continued)

Chemical	Chronic oral RfD[a,b] (mg/kg/d)	Confidence level[a]	GI absorption[c] (%)	Chronic oral RfD absorbed[d,e] (mg/kg/d)	Chronic inhalation RfD[a,b,f] (mg/kg/d)	RfD basis (vehicle)[g]	Critical effect[g]	Uncertainty factor[a,g]/Modifying factor
Manganese (water)	5.0E-03	Medium	4	2.0E-04	1.4E-05	oral	lethargy, tremors mental disturbance, muscle tonus, CNS	UF = 1 MF = 1
Mercury, inorganic	3.0E-04	na	7	2.10E-05	8.6E-05	oral	CNS, kidney, gastrointestinal tract toxicity	UF = 1000 MF = na
Nickel, soluble salts	2.0E-02	Medium	27	5.4E-03	na	oral	nephrotoxicity, reduced weight	UF = 100[g] MF = 3[g]
Selenium	5.0E-03	High	44	2.2E-03	na	oral	methemoglo-binemia, dermatitis	UF = 3 MF = na
Silver	5.0E-03	Low	18	9.00E-04	na	oral	Argyria	UF = 3 MF = 1
Vanadium, metallic	7.0E-03	na	1	7.0E-05	na	oral, oral-water	reduced pup weight (rat)	UF = 100
Zinc, metallic	3.0E-01	Medium	20	6.0E-02	na	oral	gastrointestinal system, kidney, renal function	UF = 10 MF = na
				Organics				
Aldrin	3.0E-05	Medium	50	1.5E-05	na	oral	liver toxicity (rat)	UF = 1000 MF = 1
Aroclor 1254	2.0E-05	na	90	1.8E-05	na	na	na	na
Aroclor 1260	na	na	90	na	na	na	na	na
Benzo(a)anthracene	na	na	31	na	na	na	na	na

Table 5.6. (continued)

Chemical	Chronic oral RfD[a,b] (mg/kg/d)	Confidence level[a]	GI absorption[c] (%)	Chronic oral RfD absorbed[d,e] (mg/kg/d)	Chronic inhalation RfD[a,b,f] (mg/kg/d)	RfD basis (vehicle)[a]	Critical effect[g]	Uncertainty factor[a,g]/Modifying factor
Benzo(a)pyrene	na	na	31	na	na	na	na	na
Benzo(b)fluoranthene	na	na	31	na	na	na	na	na
Bis(2-ethyl-hexyl)phthalate	2.0E-02	Medium	19	3.8E-03	na	oral	increased liver and kidney weight in animals	UF = 1000 MF = 1
Chlordane	6.00E-05	Low	50	3.0E-05	na	oral	regional liver hypertrophy (rat)	UF = 1000 MF = 1
Cresol, p-	5.0E-03	Low	65	3.3E-03	na	oral	maternal death, hypactivity, cyanosis, eye discharge, and respiratory system distress (rabbit)	UF = 1000
DDD	na	na	70	na	na	na	na	na
DDE	na	na	70	na	na	na	na	na
DDT	5.0E-04	na	70	3.5E-04	na	na	na	na
Dibenzo(a,h)anthracene	na	na	31	na	na	na	na	na
Indeno(1,2,3-cd)pyrene	na	na	31	na	na	na	na	na

na = no data available or data inconclusive; GI = gastrointestinal; RfD = reference dose; CNS = central nervous system.

[a]Based on Integrated Risk Information System (EPA 1994).

[b]Based on Health Effects Assessment Summary Tables (EPA 1993).

[c]Biomedical and Environmental Information Analysis Section 1994.

[d]The absorbed RfD = (RfD × GI absorption percent); the absorbed RfD is used for dermal pathway calculations. RfD absorbed = RfD (i.e, GI = 100%) when the GI absorption percent value is unknown (na), and when the GI is >80%.

[e]The chronic oral absorbed RfD is used for the dermal contact pathway calculations (RfD oral absorbed = RfD oral × GI percent).

[f]Reference concentrations (RfC) were converted to units of mg/kg/d (i.e., RfD units) with the inhalation rate and body weight of an adult [i.e., RfC × 20 m^3/d × (1/70 kg) = RfD] (EPA 1989).

[g]Biomedical and Environmental Information Analysis Section Toxicity Profiles (BEIAS 1994).

Table 5.6. (continued)

Sources:

Biomedical and Environmental Information Analysis Section. 1994. *Toxicity Profiles for Use in Hazardous Waste Risk Assessment and Remediation*. Vol. I. ES/ER/TM-77. Martin Marietta Energy Systems, Health Sciences Research Division. Oak Ridge National Laboratory, Oak Ridge, Tenn.

EPA (U.S. Environmental Protection Agency). 1989. *Exposure Factors Handbook*. EPA/600/8-89/043. Office of Health and Environmental Assessment, Washington, D.C.

EPA (United States Environmental Protection Agency). 1993. *Health Effects Assessment Summary Tables*. Office of Research and Development and Office of Emergency and Remedial Response, Washington, D.C.

EPA (United States Environmental Protection Agency). 1994. *Integrated Risk Information System. IRIS Database*. Office of Research and Development. Washington, D.C.

factors used in the evaluation of risk from exposure to chemicals and radionuclides in the Clinch River and Poplar Creek are listed in Tables 5.7 and 5.8, respectively.

5.3.3 Estimation of Toxicity Values for Dermal Exposure

Oral RfDs and SFs are often adjusted for evaluation of the dermal exposure pathway (EPA 1989a); it is conservative to evaluate both carcinogenic risk and noncarcinogenic hazard to adjust the toxicity values in the manner described in the following discussion. Most RfDs/SFs are expressed as the amount of substance administered per unit time and body weight (administered dose); however, dermal exposure to chemicals in soil and water should be expressed as absorbed doses.

For the dermal assessments in this baseline human health risk assessment, the oral RfD and/or SF for each chemical was adjusted by the percent gastrointestinal absorption efficiency (%GI) for that chemical (EPA 1992). The %GI is known for only a limited number of chemicals; for those chemicals where a %GI is currently not available in the literature, 100% was assumed. For many chemicals, estimates of %GI were based on qualitative information on the rate and extent of GI absorption; rapid or extensive absorption was assumed to be essentially complete (i.e., %GI = 100%). Wide ranges of %GI values can be found for some chemicals and in the absence of chemical-specific absorption data, estimates can be made based on data for related chemical structures. Most organic compounds are readily absorbed (i.e., %GI =100) from the GI tract; for this baseline human health risk assessment, no adjustments were made to chemicals with %GI ≥ 80%.

Minor adjustments to the oral RfDs and SFs, used in the dermal assessments only, which favor conservatism, were made for this baseline human health risk assessment. The oral RfD was multiplied by the %GI/100, and the SF was divided by the %GI/100 to give the absorbed dose RfD and absorbed dose SF, respectively. It should be noted that this approach may result in overly conservative risk estimations. If unacceptable risks are identified for this exposure route, the associated chemicals should be examined in detail to ascertain the credibility of the dermal toxicity value before making decisions on the basis of the dermal results.

5.3.4 Chemicals for Which No EPA Toxicity Values Are Available

Slope factors and RfDs are not currently available for all chemicals analyzed because (1) their carcinogenic and/or noncarcinogenic effects have not yet been determined, (2) epidemiological studies have proven them not to be carcinogenic and/or toxic, or (3) existing or previously determined toxicity values have since been withdrawn by the EPA. Therefore, several of the COPCs for the Clinch River and Poplar Creek cannot be quantitatively evaluated at the present time. However, literature research has been conducted for most of these chemicals and a qualitative summary of the available information can be found in *Toxicity Profiles for Use in Hazardous Waste Risk Assessment and Remediation* (Energy Systems 1994).

5.3.5 Uncertainties Related to Toxicity Information

The methodology used in developing a noncarcinogenic toxicity value (RfD or RfC) involves identifying a threshold level below which adverse health effects will not occur. The RfD or RfC values are generally based on studies of the most sensitive animal species tested and the most sensitive end point measured (unless adequate human health data are available). From these studies, the experimental exposure that represents the highest dose level tested at which no adverse effects were demonstrated [the no-observed-adverse-effect level (NOAEL)] was derived; in some cases, only a lowest-observed-adverse-effect level (LOAEL) is available. The RfD or RfC is derived from the

Table 5.7. Toxicity information for the Clinch River and Poplar Creek carcinogenic contaminants of potential concern

Chemical	Oral slope factor[a] (kg/d/mg)	GI absorption[b] (%)	Oral slope factor absorbed[c] (kg/d/mg)	TEF[d]	Inhalation slope factor[a] (kg/d/mg)	EPA class[e]	Type of cancer
			Inorganics				
Antimony	na	2.0	na	na	na	na	na
Arsenic, inorganic	na	41	na	na	5.0E+01	A	skin, liver, bladder, respiratory, gastrointestinal
Beryllium	4.3E+00	1.0	4.3E+02	na	8.4E+00	B2	breast, bone, uterus, lung
Boron and Borates only	na	90	na	na	na	na	na
Cadmium (diet)	na	1.0	na	na	6.1E+00	B1	respiratory tract, lung
Cadmium (water)	na	1.0	na	na	6.1E+00	B1	respiratory tract, lung
Chromium (VI)	na	2.0	na	na	4.1E+01	A	tumors, lung
Manganese (diet)	na	4.0	na	na	na	D	na
Manganese (water)	na	4.0	na	na	na	D	na
Mercury, inorganic	na	7.0	na	na	na	D	na
Nickel, soluble salts	na	27	na	na	na	D	na
Selenium	na	44	na	na	na	D	na
Silver	na	18	na	na	na	D	na
Vanadium, metallic	na	1	na	na	na	D	na
Zinc, metallic	na	20	na	na	na	D	na

Table 5.7. (continued)

Chemical	Oral slope factor[a] (kg/d/mg)	GI absorption[b] (%)	Oral slope factor absorbed[c] (kg/d/mg)	TEF[d]	Inhalation slope factor[a] (kg/d/mg)	EPA class[e]	Type of cancer
			Organics				
Aldrin	1.7E+01	50	3.4E+01	na	1.7E+01	B2	liver carcinoma
Aroclor 1254	na	90	na	na	na	B2	liver, biliary tract, gall bladder
Aroclor 1260[f]	7.7E+00	90	8.6E+00	na	na	B2	liver, biliary tract, gall bladder
Benzo(a)anthracene	7.3E-01	31	2.43E+00	0.1	na	B2	tumors
Benzo(a)pyrene	7.3E+00	31	2.4E+01	1.0	na	B2	stomach tumors
Benzo(b)fluoranthene	7.3E-01	31	2.4E+00	0.1	na	B2	tumors
Bis(2-ethylhexyl)-phthalate	1.4E-02	19	7.4E-02	na	na	B2	liver neoplastic nodule and hepatocellular carcinoma
Chlordane	1.3E+00	50	2.6E+00	na	1.3E+00	B2	(O): liver carcinoma (mouse); (I): CNS depression
Cresol,p-	na	65	na	na	na	C	na
DDD	2.4E-01	70	3.4E-01	na	na	B2	liver tumors (mouse)

Table 5.7. (continued)

Chemical	Oral slope factor[a] (kg/d/mg)	GI absorption[b] (%)	Oral slope factor absorbed[c] (kg/d/mg)	TEF[d]	Inhalation slope factor[a] (kg/d/mg)	EPA class[e]	Type of cancer
DDE	3.4E-01	70	4.9E-01	na	na	B2	hepatocellular carcinoma and hepatoma (mouse)
DDT	3.4E-01	70	4.9E-01	na	3.4E-01	B2	(O): liver tumors (mouse, rat)
Dibenzo(a,h)anthracene	7.3E+00	31	2.4E+01	1.0	na	B2	tumors
Indeno(1,2,3-cd)pyrene	7.3E-01	31	2.4E+00	0.1	na	B2	tumors

na = No data available or data inconclusive;TEF = toxicity equivalency factor; EPA = U.S. Environmental Protection Agency.
[a]Based on Integrated Risk Information System (EPA 1994) or Health Effects Assessment Summary Tables (EPA 1993a).
[b]Biomedical and Environmental Information Analysis Section. 1994.
[c]The absorbed oral slope factor (SF) is used for the dermal contact pathway calculations; the absorbed oral SF = (slope factor/percent gastrointestinal absorption).
[d]The oral SFs for these polycyclic aromatic hydrocarbons are derived by multiplying the benzo(a)pyrene oral SF (7.3E+00) by the chemical specific toxicity equivalency factor (EPA 1993b).
[e]U.S. Enviornmental Protection Agency Weight of Evidence Classification System for Carcinogenicity was used to characterize the extent to which available data indicate that an agent is a human carcinogen: A = human carcinogen; B1 or B2 = probable human carcinogen (B1 indicates that limited data on humans are available and B2 indicates sufficient evidence in animals and inadequate or no evidence in humans); C = possible human carcinogen; D = not classifiable as to human carcinogenicity; E = evidence of noncarcinogenicity for humans.
[f]For these Aroclors, the oral SF (7.7E+00) for polychlorinated biphenyls was used.

Sources:

Biomedical and Environmental Information Analysis Section. 1994. *Toxicity Profiles for Use in Hazardous Waste Risk Assessment and Remediation.* Vol. I. ES/ER/TM-77. Martin Marietta Energy Systems, Health Sciences Research Division. Oak Ridge National Laboratory, Oak Ridge, Tenn.
EPA (United States Environmental Protection Agency). 1993a. *Health Effects Assessment Summary Tables.* Office of Research and Development and Office of Emergency and Remedial Response, Washington, D.C.
EPA (United States Environmental Protection Agency). 1993b. *Provisional Guidance for Quantitative Risk Assessment of Polycyclic Aromatic Hydrocarbons.* EPA600/R/93/089. Office of Research Development, Washington, D.C.
EPA (United States Environmental Protection Agency). 1994. *Integrated Risk Information System. IRIS Database.* Office of Research and Development. Washington, D.C.

Table 5.8. **Toxicity information for exposure to the Clinch River and Poplar Creek radionuclide contaminants of potential concern**

Chemical	External exposure slope factor[a,b] (g/pCi/y)	Oral slope factor[a,b] $[(pCi)^{-1}]$	Inhalation slope factor[a,b] $[(pCi)^{-1}]$	ICRP lung class[a,c]	EPA class[d]	Type of cancer
^{137}Cs+D	2.0E-06	2.8E-11	1.9E-11	D	A	various
^{60}Co	8.6E-06	1.5E-11	1.5E-10	Y	A	various
^{90}Sr+D	0	3.6E-11	6.2E-11	D	A	various

EPA = U.S. Environmental Protection Agency; ICRP = International Commission on Radiological Protection.
[a]Based on Health Effects Assessment Summary Tables (EPA 1993).
[b]The radionuclide slope factors include contributions from daughter products.
[c]Lung clearance classification recommended by the International Commission on Radiological Protection: Y = year;D = day.
[d]EPA Weight of Evidence Classification System for Carcinogenicity was used to characterize the extent to which available data indicate that an agent is a human carcinogen: A = human carcinogen; B1 or B2 = probable human carcinogen (B1 indicates that limited data on humans are available and B2 indicates sufficient evidence in animals and inadequate or no evidence in humans); C = possible human carcinogen; D = not classifiable as to human carcinogenicity; E = evidence of noncarcinogenicity for humans.

Source: EPA (United States Environmental Protection Agency). 1993. *Health Effects Assessment Summary Tables*. Office of Research and Development and Office of Emergency and Remedial Response, Washington, D.C.

NOAEL (or LOAEL) for the critical toxic effect by dividing the NOAEL (or LOAEL) by uncertainty factors. These factors usually are in multiples of ten, with each factor representing a specific area of uncertainty in the extrapolation of the data. An uncertainty factor of 100 is typically used when extrapolating animal studies to humans; additional uncertainty factors are sometimes necessary when other experimental data limitations are found. Because of the large uncertainties (10–10,000) on some RfD/RfC toxicity values, exact/precise safe levels of exposure for humans are not possible.

A two-part evaluation exists for assessing the carcinogenic potential of a chemical: (1) evaluating the likelihood that a chemical is a carcinogen (i.e., a weight-of-evidence assessment) and (2) determining the quantitative dose-response relationship (i.e., potency factor or SF); uncertainties occur with each evaluation. On the basis of weight-of-evidence studies that used human and laboratory animal research, chemicals fall into one of five groups (EPA 1989a, 1993a): (1) Group A, human carcinogen; (2) Group B1, probable human carcinogen based on limited human data; (3) Group B2, probable human carcinogen based on sufficient evidence in animals and inadequate or no evidence in humans; (4) Group C, possible human carcinogen; (5) Group D, not classified as to human carcinogenicity; and (6) Group E, evidence of noncarcinogenicity for humans. The SF for a chemical is a plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime; it is derived by applying a mathematical model to extrapolate from a relatively high administered dose (to animals) to the lower exposure levels expected for humans. The SF represents the UCL_{95} on the linear component of the slope of the tumorigenic dose-response curve in the low-dose region. A number of low-dose extrapolation models have been developed, and EPA generally uses the linearized multistage model in the absence of adequate information to support other models.

5.3.6 Summary of Toxicity Information

The potential toxicological effects of the COPCs included in the evaluation of the Clinch River and Poplar Creek have been discussed. The majority of the information pertaining to specific contaminants was taken from *Toxicity Profiles for Use in Hazardous Waste Risk Assessment and Remediation* (BEIAS 1994) prepared by the Biomedical and Environmental Information Analysis Section (BEIAS) of the Health Sciences Research Division. However, if no toxicity profile was available from BEIAS for a specific contaminant, then EPA's Integrated Risk Information System (IRIS) database, the Health Effects Assessment Summary Tables (HEAST), or library data were used, if available.

Toxicity profiles for COPCs for which no toxicity information is currently available is presented in Appendix E. In addition, Appendix E contains toxicity profiles for contaminants for which toxicity information was available that have been determined by the risk characterization (Sect. 5.4) to be COCs for the Clinch River and/or Poplar Creek.

5.4 RISK CHARACTERIZATION

The purpose of the risk characterization is to integrate and summarize the information presented in the exposure and toxicity assessment; it is the final step in the human health risk assessment process. Potential carcinogenic effects are characterized by estimating the probability that an individual will develop cancer over a lifetime by multiplying projected intakes that result from exposure to carcinogenic chemicals by chemical-specific dose-response data (i.e., slope factors). This probability is presented as an excess cancer risk (ECR). Potential noncarcinogenic (systemic/toxic) effects are characterized by dividing projected intakes of contaminants by noncarinogenic toxicity values (i.e., reference doses) resulting in an HQ. The numerical ECR and hazard quotients produced during this phase of the risk assessment process must be interpreted in the context of the uncertainties and assumptions associated with the risk assessment process and with the data upon which the risk estimates are based.

5.4.1 Risk Characterization Approach for Carcinogens

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

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

The ECR for carcinogens is calculated by multiplying the calculated intake/dose for each contaminant by the appropriate EPA-approved slope factors. This estimate of ECR represents the potential of an individual developing excess cancer over a lifetime, above and beyond the normal (unavoidable) incidence of developing cancer.

Consideration is given to exposure to multiple chemicals as well as multiple exposure pathways when calculating the risk of an individual developing cancer. This is accomplished through summing ECRs for each chemical both within a given pathway and across pathways within a scenario. The EPA has established a target risk range of 10^{-4} to 10^{-6} (55 *FR* 46). Where the baseline risk assessment indicates that a cumulative site risk to an individual (calculated by using RME assumptions) for either current or future land use exceeds the 10^{-4} lifetime ECR end of the target risk range, action under CERCLA is generally warranted at the site. For sites where the cumulative site risk to an individual based on RME for both current and future land use is $<10^{-4}$, action generally is not warranted but may be warranted if a chemical-specific standard that defines acceptable risk is violated or there are noncarcinogenic effects or an adverse environmental impact that warrants action.

In this risk assessment, pathways of concern relative to carcinogenic risk are those that have a pathway total $>10^{-4}$. Within the pathways that have a total risk $>10^{-4}$, the carcinogenic COCs are defined as those contaminants whose ECR is $>10^{-6}$.

5.4.2 Risk Characterization Approach for Noncarcinogens

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

To characterize potential noncarcinogenic effects from contaminants, the ratio of projected intakes of substances to EPA-approved RfDs is calculated. In this assessment, only long–term, chronic exposures to contaminants are evaluated. A chronic RfD is an estimate of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effect during a lifetime (EPA 1989c). Noncarcinogenic effects are not expressed as the probability of an individual suffering an adverse effect as in the case of carcinogens but as a comparison of a daily exposure level averaged over a specified period of time with an RfD. The ratio of the average daily exposure level of a single toxicant to the RfD for that toxicant is defined as an HQ. The sum of more than one HQ for multiple toxicants and/or multiple exposure pathways is called a hazard index (HI). An HQ or an HI > 1 is considered unacceptable. Both a child and an adult were included in the assessment of noncarcinogens because in some instances the parameter values used for a child are more conservative than those used for an adult. Exposure to noncarcinogens can occur through ingestion, inhalation, and dermal contact. For this risk assessment, pathways of concern for noncarcinogenic effects are defined as those with an HI >1.0. Within the pathways that have an HI >1.0, noncarcinogenic chemicals of concern are defined as those with HIs >0.1.

5.4.3 Risk Characterization Results for Current Conditions

Results of the human health risk assessment for the direct exposure pathways to primary media for current exposure scenarios (Fig. 5.1) are presented in the following order: drinking water ingestion, shoreline use, swimming, fish ingestion, and water fowl ingestion. Brief descriptions of the types of analytes and number of samples in the databases used in each scenario are given in the following text. Before the preparation of the risk characterization tables, the various pathways and contaminants were screened to limit the information presented to only those contaminants that had either an ECR >10^{-6} or an HQ >0.1. This screen was applied to reduce the volume of material generated and to highlight those contaminants and pathways of potential concern. In the case where a chemical had an ECR >10^{-6} or a HQ >0.1 for one pathway in a given scenario, the ECR and hazard quotients for that chemical are presented for all pathways within the scenario regardless of the ECR or HQ.

5.4.3.1 Drinking water ingestion scenario

Human health risks for the water ingestion scenario were calculated separately for the Clinch River reaches and the Poplar Creek subreaches. Within the Clinch River, reach 7 was further broken down into two subreaches: 7.01 (upper McCoy Branch) and 7.02 (lower McCoy Branch). The methods, equations, and parameter values in the human health risk assessment for radiological and nonradiological contaminants in drinking water are listed in Table E1.

Carcinogenic and noncarcinogenic chemicals of concern for the drinking water ingestion pathway for reaches 1, 2, 4, and 7 of the Clinch River. Within the Clinch River system, risks were calculated separately for each of the defined reaches (reaches 1, 2, 4, and 7) and are presented in Table E29. For the drinking water ingestion pathway, none of the reaches evaluated exhibited a pathway ECR

$>10^{-4}$. Thus, no carcinogenic chemicals of concern were identified for the drinking water ingestion pathway for reaches 1, 2, 4, or 7.

Noncarcinogenic HIs were calculated for both an adult and a child for the drinking water ingestion pathway. As shown in Table E30, no noncarcinogenic HIs were >1.0 for the adult water ingestion pathway for the Clinch River reaches. However, the HIs for the child water ingestion pathway for the Clinch River reaches was >1.0 for each of the Clinch River reaches (Table E31). Several inorganic contaminants (As, Sb, Mn, and nitrate) were found to have an individual HQ >0.1 for this pathway. Individually, none of these contaminants have a HQ >1.0 and the HIs for each reach range from 1.0 to 1.5. The chemical that contributes >50% of the total HI for each reach is manganese. There are two RfDs currently available for manganese, one for water and one for diet. In evaluating the drinking water ingestion pathway, the RfD for water ingestion of manganese was used. Because unfiltered water samples were used in the evaluation of hazard/risk for the Clinch River and Poplar Creek, the actual hazard from the ingestion of manganese may have been overestimated as a result of the differential bioavailability of manganese within the system. In addition, the distribution of total and dissolved manganese in the surface water of the Clinch River and Poplar Creek does not vary significantly from reach to reach or between the reference reaches and the study area (see Chap. 3).

Carcinogenic and noncarcinogenic chemicals of concern for the drinking water ingestion pathway for Poplar Creek. Within Poplar Creek, carcinogenic risks were calculated separately for each of the defined subreaches. For the drinking water ingestion pathway, none of the subreaches exhibited a pathway ECR $>10^{-4}$ (Table E32). Therefore, no carcinogenic COCs were identified.

Noncarcinogenic contaminants whose HIs were >1.0 are listed in Table E33 for adults and Table E34 for children for each of the Poplar Creek subreaches. With the exception of subreach 3.04 of Poplar Creek, the HIs for the adult drinking water ingestion pathway were <1.0 and no COCs were identified. In subreach 3.04, the HI = 1.0, and manganese contributed ~50% of the total hazard. For the child drinking water ingestion pathway, several inorganic contaminants (As, Sb, and Mn) and one organic contaminant (Aroclor 1254) were found to have an individual HQs >0.1 for this pathway. In addition, within subreach 3.03, the child's HI for nitrate was 0.2. The two contaminants that were the primary drivers of the noncarcinogenic hazard for subreach 3.01 and 3.04 for this pathway are manganese (52% and 48%, respectively) and Aroclor 1254 (40% and 29%, respectively). For subreach 3.02, Mn contributed 53%, As 31%, and Sb 15% of the HI. For subreach 3.03, Mn contributed 65%, As 18%, and nitrate 12% of the HI. Each of these chemicals (As, Sb, Mn, nitrate, and Aroclor 1254) are considered COCs for Poplar Creek surface water.

5.4.3.2 Shoreline use pathways

As discussed previously, near-shore sediment data were used to evaluate the shoreline use pathways. As previously defined, the "near-shore" data set consists of samples collected at elevations >733 ft. No such data are available for reaches 1 or 7, therefore, the shoreline use pathways were evaluated for subreaches 2.04–4.04 only. The risk characterization results for these subreaches are presented in Tables E35–E37.

For the subreaches evaluated, the only subreach with a pathway ECR $>10^{-4}$ was subreach 3.02 (Table E35). The ECR for the combined inhalation and external exposure pathways for subreach 3.02 was 1.8×10^{-4}; the following chemicals have an ECR $>10^{-6}$: Cr, As, ^{234}U, ^{238}U, and ^{235}U.

The only chemical identified as having an HQ >1.0 for the adult shoreline use sediment pathways was manganese through the inhalation of resuspended sediment. Asshown in Table E36, the HQ for the inhalation of resuspended manganese is >1.0 for all of the subreaches evaluated. As was discussed in Chap. 3, the distribution of manganese in the Clinch River is significantly higher in subreach 2.04, this is also evidenced in a higher HQ for this subreach. This peak concentration of manganese suggests that the K-700 area at the K-25 Site is a potential source of manganese to the system.

For the child shoreline use sediment pathways, the only pathway with an HI >1.0 was the inhalation of resuspended sediment. Once again, manganese was identified in all subreaches as the primary driver of the HIs. For all subreaches evaluated, manganese contributed >97% of the HI.

Barium was also found to have a HQ >0.1 for the child inhalation pathway (Table E37) for all subreaches except 3.01, 4.01, and 4.02, but its relative contribution to the HI was <2%.

5.4.3.3 Swimming pathways

The swimming scenario considers the potential risks of exposure to contaminants in water while swimming in the Clinch River and Poplar Creek (Fig. 5.1). The potential routes of exposure while swimming are dermal contact and inadvertent ingestion of water. Equations and parameter values used in the human health risk assessment for exposure to carcinogens and noncarcinogens in the pathways of the swimming scenario are given in Tables E7 and E8.

Carcinogenic and noncarcinogenic chemicals of concern for the Clinch River swimming pathways. No carcinogenic or noncarcinogenic COCs were determined for the swimming pathways for the Clinch River surface water.

Carcinogenic and noncarcinogenic chemicals of concern for the Poplar Creek swimming pathways. No carcinogenic COCs were determined for the swimming pathways for the Poplar Creek surface water. A phthalate ester, di-n-octylphthalate was detected in subreach 3.02 at levels that would result in an HQ of 1.6 for an adult and an HQ of 2.9 for a child through dermal contact (Table E38 and E39). However, the frequency of detection for this phthalate ester was 1/12. In general, the phthalate esters are common laboratory contaminants and are extensively used as solvents and as plasticizers of synthetic polymers such as polyvinyl chloride and cellulose acetate (EPA 1989c). As a result, the phthalate esters are commonly found as environmental contaminants and may not have been introduced to the environment solely as a result of activities on the ORR. However, for this baseline risk assessment this phthalate ester is considered a COC. In subreach 3.01, Aroclor 1254 was determined to have an HQ of 1.1. This PCB will also be considered a COC.

5.4.3.4 Fish consumption scenario

Nine contaminants detected in fish fillets produced cancer risks $>10^{-6}$ (Table E40). All species of fish for which Aroclor 1260, a PCB, was analyzed had calculated cancer risks $>10^{-4}$. The excess lifetime cancer risks for this PCB in catfish and striped bass were $>10^{-3}$. The pesticides (aldrin, chlordane, and 4,4'-DDT) are carcinogenic COCs with cancer risks $>10^{-6}$. In addition, the ECR for 4,4'-DDE, a chemical compound found only in the environment as a degradation product of 4,4'-DDT was also $>10^{-6}$. At one time, these pesticides were commonly used in residential, farming, and industrial areas; therefore, they are not unique to the ORR but are ubiquitous contaminants in eastern Tennessee streams and reservoirs. The remaining carcinogenic contaminants of concern for fish include two inorganics (As and Be) and two radionuclides (^{137}Cs and ^{90}Sr). The highest concentrations of the inorganics are found in largemouth bass and catfish. The radionuclides are associated with the ingestion of catfish. With the exception of the pesticides, aldrin, 4,4'-DDT, and 4,4'-DDE, all of the aforementioned contaminants will be retained as carcinogenic contaminants of concern.

Noncarcinogenic COCs (Tables E41 and E42) that contributed to a pathway HI >1.0 included several inorganic contaminants (As, Hg, and Se), two pesticides (chlordane and 4,4-'DDT), and Aroclor 1254. The highest HQs for the fish pathway were for the ingestion of Aroclor 1254 and chlordane in catfish and striped/hybrid bass by a child.

In interpreting the risk assessment results for this scenario, one important data limitation must be taken into account. Several bluegill, a species of fish previously not considered to be an effective bioindicator of PCB contamination on the basis of their feeding habits and relatively low lipid content, was collected and analyzed for PCBs in reach 4 only. The ECR calculated for Aroclor 1260 in these bluegill was 5.4×10^{-4}, a number well over the EPA risk range of 10^{-6} to 10^{-4}. Currently, existing fishing advisories for the lower WBR and Melton Hill do not apply to bluegill. The lack of sufficient data for this species of fish from other reaches warrants further investigation and should be incorporated into any subsequent monitoring of the Clinch River.

Because the fish consumption scenario is a significant pathway relative to risks/hazards in the Clinch River and Poplar Creek, a quantitative uncertainty analysis was performed for the COCs for this pathway. The results of the uncertainty analysis are contained in Sect. 5.5.

5.4.3.5 Waterfowl consumption

As discussed previously, the consumption of Canada Geese was evaluated as a current exposure pathway. The analytes evaluated in the Canada Geese were radionuclides, inorganics, and organics. Calculated risks/hazards for the ingestion of Canada Geese (Tables E43–E45) indicate that no carcinogenic or noncarcinogenic COCs for this pathway.

5.4.4 Risk Characterization Results for Future Conditions

5.4.4.1 Irrigation scenario

As discussed in Sect. 5.2, six exposure pathways were evaluated in the irrigation scenario. For all pathways evaluated (ingestion, inhalation, and external exposure to irrigated soil and the ingestion of milk, meat, and vegetables produced on irrigated soils) the ECRs calculated for each pathway for each reach within the Clinch River (Table E46) and all subreaches within Poplar Creek (Table E49) were all $<10^{-4}$. As a result, no carcinogenic COCs were determined for these pathways. The only noncarcinogenic contaminant of concern identified for the adult irrigation pathways for the Clinch River reaches was nitrate (Table E47) through the ingestion of milk from cattle pastured on irrigated soils. Noncarcinogenic chemicals identified as having an HI>1.0 for the child irrigation pathways for the Clinch River (Table E48) are (1) nitrate through ingestion of milk, meat, and vegetables; (2) As and Sb through ingestion of leafy vegetables; (3) Ni through ingestion of milk; and (4) acetone through ingestion of vegetables.

The only noncarcinogenic contaminants of concern identified for the adult irrigation pathway for the Poplar Creek subreaches are di-n-octylphthalate (subreach 3.02) and nitrate (subreach 3.03). As discussed in Sect. 5.4.3.3, the frequency of detection was 1/12 and the phthalate esters are commonly found as environmental contaminants and are not considered contaminants of concern in this risk assesment. Noncarcinogenic chemicals identified as having an HI >1.0 for the child irrigation pathways for the Poplar Creek subreaches are shown in Table E51. The HI for the ingestion of milk by a child for subreaches 3.01 and 3.04 of Poplar Creek indicated that the current concentrations of Aroclor 1254 when modeled to a concentration in milk produced by cattle grazing on irrigated soil and two inorganics, nitrate and nickel would result in an HQ >1.0. The HI for the ingestion of vegetables grown in irrigated soil indicated that two inorganics (As and nitrate) and one organic (acetone) would result in an HI = 1.0.

Lastly, the HI as calculated for the ingestion of beef cattle pastured on irrigated soils indicates that for subreach 3.03, nitrate concentrations result in an HQ >1.0.

5.4.4.2 Dredging scenario

As discussed in Sect. 5.2, seven exposure pathways were evaluated in the dredging scenario. The deep-water sediment data were aggregated into two data sets for risk assessment purposes. The first data set evaluates the Clinch River reaches 1 and 7; the second data set evaluates subreaches 2.04, 3.01, 3.02, 3.03, 3.04, 4.01, 4.02, 4.03, and 4.04.

The ECRs calculated for direct dredging pathways (ingestion, inhalation, external, and dermal) for Clinch River reaches 1 and 7 were $<10^{-4}$ for all pathways except external exposure. In reach 1, ^{60}Co and ^{137}Cs had ECRs (as a result of external exposure) of 1.3E-04 and 1.2E-05, respectively (Table E52). Previous discoveries of ^{60}Co in reach 1 have been traced to the former American Nuclear Corporation (ANC), which was located on Braden Branch Creek. Braden Branch Creek discharges to the Melton Hill Reservoir. ANC was under the jurisdiction of the U.S. Atomic Energy Commission until 1965 at which time the State of Tennessee assumed jurisdiction. ANC used ^{60}Co as a radiation source for medical instruments between 1962 and 1970. The ANC site is currently listed on the State of Tennessee's Superfund list.

The ECRs calculated for the meat and vegetable ingestion pathways were $<10^{-4}$ for both Clinch River reach 1 and 7. The ingestion of milk was $>10^{-4}$ for Clinch River reach 7 only. In reach 7, four PAHs, and two radionuclides (^{90}Sr and ^{137}Cs) were found to have an ECR $>10^{-6}$ (Table E52).

The noncarcinogenic hazards calculated for the Clinch River reaches 1 and 7 (Table E53) indicate that for the adult dredging pathways, exposure to manganese through the sediment inhalation and vegetable ingestion for reach 7 has an HQ >1.0. For the child dredging pathways, the only direct pathway of concern is the inhalation of manganese from resuspended sediments for reach 7. As discussed in Chap. 3, the highest average surface sediment concentrations of manganese are found in reaches 7 (McCoy Branch) and 8 (Walker Branch); this suggests an upstream source. Reach 8 is a reference reach and drains a "clean" watershed. This suggests that there is a natural geologic source of manganese in this area. Therefore, in making remedial decisions regarding manganese as a COC, the risk managers should determine if the concentrations of manganese are to be considered as indicative of naturally occurring background levels. The noncarcinogenic hazard calculated for the child indirect pathways (Table E54) indicate that exposure to children through the ingestion of meat, milk, and vegetables would pose a potential threat of noncarcinogenic health effects. COCs for these pathways are 4,4'-DDT (milk), As (milk, meat, and vegetables), Hg (vegetables), Cr (meat and vegetables), Ni (milk and vegetables), Se (milk and meat), Zn (milk, meat, and vegetables), V (milk), and Mn (milk and vegetables).

The ECRs calculated (Table E55) for the direct exposure pathways (ingestion, dermal, inhalation, and external exposure) for the Clinch River and Poplar Creek subreaches indicated that external exposure to ^{137}Cs and ^{60}Co (subreaches 2.04, 3.04, 4.01, 4.03, and 4.04) U^{238} (subreach 3.04), and ^{90}Sr (subreach 4.04) would result in a pathway ECR $>10^{-4}$. In addition, inhalation of resuspended sediments for subreach 3.04 indicate that concentrations of As, Cr, Cd, ^{238}U, and ^{234}U could pose an ECR $>10^{-6}$.

The indirect pathways (ingestion of milk, meat, and vegetables) evaluated for all of the Clinch River and Poplar Creek subreaches indicated that dredging sediments from the deep-water depositional zones and using them as fill for agricultural purposes would pose unacceptable risks/hazards to the public. The majority of carcinogenic contaminants that contribute to unacceptable ECRs (Table E55) are PAHs. PAHs [benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, dibenz(a,h)anthracene, and ideno(1,2,3-cd)pyrene] are natural products produced by forest fires, microbial synthesis, and anthropogenic activities. The levels of PAHs detected in the deep-water sediments of the Clinch River are well below the concentrations of PAHs detected during the Background Soils Characterization Project. Thus, the PAH levels are considered to be indicative of background levels and are not considered COCs. Therefore, the only carcinogenic COCs identified for the indirect pathways (Table E55) in the dredging scenario are Be (vegetables), ^{137}Cs (milk), ^{90}Sr (milk, meat, and vegetables), ^{99}Te (milk and vegetables), ^{234}U (milk), ^{238}U (milk), As (milk, meat, and vegetables), and Aroclor 1260 (milk, meat, and vegetables).

The noncarcinogenic COCs for the adult and child dredging pathways for the Clinch River and Poplar Creek subreaches are listed in Tables E56 and E57. Twelve inorganics and two organics were found to have HQs >0.1 within those pathways having an HI >1.0.

5.4.5 Special Case—Lead

Measurable concentrations of lead were found in sediment from both the Clinch River and Poplar Creek (Table C4); however, EPA toxicity values (RfDs) for lead are not presently available for risk assessment purposes. Although the concentrations of lead in the sediment samples approximated concentrations in background soils (DOE 1993a), lead was included in the risk assessment because of the public's concern with lead toxicity. An EPA uptake/biokinetic model (EPA 1991b) was used to evaluate the health risk from exposure to lead. This model uses site-specific, multimedia input values to estimate levels of lead in blood. Young children are more sensitive to lead toxicity than adults; therefore, the model was used to estimate the blood lead levels for a population of children 0–6 years of age. The critical blood lead level for children is 0.010 mg/dL of blood. EPA Region IV considers the population to be at risk if more than 5% of the children have blood lead levels that exceed the critical value.

The model allows for input of lead concentrations for soil, dust, food, and water. Lead concentrations were available for all environmental media in the Clinch River and Poplar Creek with the exception of fish. In addition, modeled concentrations of lead in milk, meat, and vegetables were included. The model allows the programmer to input concentrations of contaminants in water but has no field for milk input. Therefore, the assumption was made that children drink equal amounts of water and milk and a simple average of the two values (water and milk) was generated and used as a concentration of lead in water. Alternate dietary sources included data for homegrown vegetables and meat. The model input parameters, however, do not include homegrown meat as an alternate dietary source. Therefore, the concentration of lead in meat (beef) was modeled as game meat by using 44% as the average proportion (EPA 1990) and a reasonable maximum proportion of 75% (EPA 1993a) of beef in the diet as homegrown. The concentration of lead in homegrown vegetables has a specific model input as an alternate dietary source; the average proportion of homegrown vegetables in the diet is 25% and the reasonable maximum proportion is 75% (EPA 1993a). The lead concentrations in sediment were modeled as soil; concentrations of lead in household dust were not provided. As a result, multiple source analysis was used to estimate the concentration of lead in household dust with air and soil being contributing factors. Values for dust were based on a 70% conversion of soil to dust and an air to dust conversion of 100 μg Pb in dust/μg Pb/m^3 of air. Default values were used for the concentration of lead in air (0.1 μg Pb/m^3).

The model was run with both the average and reasonable maximum consumption of homegrown vegetables and beef. The results (Table 5.9) indicate that with the exception of reaches 1 and 2, that >95% of children 0–6 years of age would have blood lead levels below the 0.010 mg/dL critical value. Thus, no adverse health effects would be expected as a result of exposure to lead for reaches 3, 4, or 7.

For reach 1, the probability of the blood lead level exceeding 10µg/dL for both the average and maximum exposure to available lead concentrations was greater than the cutoff level of 5%. The frequency of detection for lead in surface water and sediment within reach 1 was 2/33 and 2/2, respectively. The frequency of detection for lead in surface water within reach 1 was 6%, just greater than the cutoff of 5%. The small number of samples that were analyzed for lead in reach 1 sediments should be considered a limiting factor when evaluating these results. However, lead will be listed as a COC for reach 1.

For reach 2, the probability of the blood lead level exceeding 10µg/dL for the maximum exposure to available lead concentrations at 8.79% was slightly greater than the 5% cutoff. The frequency of detection of lead in sediment was 10/10 while the frequency of detection in surface water was 3/27. Thus, lead will also be considered a COC for reach 2.

5.4.6 Risk Characterization Summary

An overall summary of the scenarios and pathways of concern by reach is presented in Table 5.10. The two scenarios that posed the greatest risk to the public are the fish consumption and the dredging scenarios. The fish ingestion pathway was the most significant exposure pathway in terms of adverse health effects. Fish contained the only contaminants with cancer risks $>10^{-4}$: Aroclor 1260 had excess lifetime cancer risks $>10^{-3}$ for the ingestion of channel catfish and striped/hybrid bass and $>10^{-4}$ for largemouth bass. The number of fish samples and their spatial distribution appear to be sufficient to be representative of PCB concentrations in these three species of Clinch River and Poplar Creek fish.

Other carcinogenic compounds found in fish tissue with ECRs within the range of concern included several pesticides (aldrin; chlordane; 4,4'-DDE; and 4,4'DDT). These pesticides were widely used until the 1970s when their use and production was banned. The widespread use of these pesticides before their being banned has resulted in residual levels of these contaminants in the environment.

Relatively large releases of radionuclides from the ORR have occurred in the past. However, the only scenarios for which radionuclides pose an unacceptable ECR are the fish consumption, dredging, and shoreline use scenarios. The ECRs for two radionuclides (^{137}Cs, and ^{90}Sr) detected in fish were $>10^{-6}$.

For the dredging scenario, several radionuclides posed a risk $>10^{-5}$ for both the direct and indirect pathways evaluated. Cobalt-60 and ^{137}Cs posed an ECR $>10^{-6}$ for the external exposure to dredged sediment for reach 1. Three isotopes of Uranium (234, 235, and 238) posed an ECR $>10^{-6}$ for the inhalation of resuspended dredged sediment. In addition to the aforementioned radionuclides, ^{90}Sr and ^{99}Te were also found to contribute to the ECR for the indirect exposure pathways evaluated as part of the dredging scenario (milk, meat, and vegetable ingestion). One inorganic (beryllium) and one organic (pentachlorophenol) were also detected in samples from the Clinch River and Poplar Creek.

Beryllium poses an ECR $>10^{-5}$ for the ingestion of meat and an ECR of $>10^{-6}$ for the ingestion of vegetables Pentachlorophenol was infrequently detected but poses a $>10^{-6}$ ECR for the ingestion of milk and vegetables produced on dredge spoils.

Mercury, which has been released in large quantities from facilities at the Y-12 Plant to the Clinch River via East Fork Poplar Creek and Poplar Creek, produced HQs >1.0 for the ingestion of bluegill, catfish, and largemouth bass by a child. The HQs for several inorganics (As, Hg, and Se) detected in fish were also determined to pose a noncarcinogenic hazard, mainly through the ingestion of bluegill and largemouth bass.

Table 5.9. Lead concentrations for the Clinch River and Poplar Creek baseline human health risk assessment

Location	Sediment (μg/g)	Water (μg/L)	Milk (μg/L)	Water/milk[a] (μg/L)	Vegetable[b] (μg/g)	Meat[c] (μg/g)	Fish (μg/L)	PbB concentration (μg/dL)[d]		Probability of exceedance (%)[e]	
								Average	Maximum	Average	Maximum
Reach 1	39	1.8	34	17.9	0.16	0.40	nd	6.1	7.9	13.64	28.52
Reach 2	36	1.7	19	10.4	0.092	0.23	nd	4.3	5.4	3.45	8.79
Reach 3	26	2.1	14	8.1	0.066	0.16	nd	3.5	4.3	1.16	3.45
Reach 4	28	1.5	15	8.3	0.071	0.18	nd	3.7	4.6	1.56	4.42
Reach 7	24	1.7	13	7.4	0.062	0.15	nd	3.3	4.1	0.92	2.70

[a]Average water and milk consumption combined to estimate the average consumption of liquids.

[b]Average consumption of home-grown vegetables si 25%; the reasonable maximum (worst case) consumption is 40% (EPA 1991).

[c]Average consumption of home-grown beef is 44% (EPA 1989); the reasonable maximum (worst case) consumption is 75% (EPA 1991).

[d]Geometric mean of blood lead level assuming average or maximum consumption of homegrown vegetables and beef.

[e]Probability of the blood lead level exceeding 10 μg/dL assuming average or maximum consumption of home-grown vegetables and beef.

Sources:

EPA (U.S. Environmental Protection Agency). 1989. *Exposure Factors Handbook.* EPA/600/8-89/043. Office of Health and Environmental Assessment, Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1991. *Human Health Evaluation Manual, Supplemental Guidance; Standard Default Exposure Factors*. OSWER Directive 9285.6-03. U.S. Environmental Protection Agency.

Table 5.10. Summary of scenarios and pathways of concern by reach and subreach

Scenario	Exposure pathway	Reach	Contaminant types of concern
Drinking water	Ingestion	1	Inorganic
		2	Inorganic
		4	Inorganic
		3.01	Organic; inorganic
		3.02	Inorganic
		3.03	Inorganic
		3.04	Organic; inorganic
		7.01	Inorganic
		7.02	Inorganic
Near-shore	Inhalation	2.04	Inorganic
		3.01	Inorganic
		3.02	Inorganic
		3.03	Inorganic
		3.04	Inorganic
		4.01	Inorganic
		4.02	Inorganic
		4.03	Inorganic
Swimming	Dermal contact	3.01	Organic
		3.02	Organic
Fish	Ingestion	1	Inorganic (bluegill); organic (catfish); radionuclide (catfish)
			Organic (largemouth bass); inorganic (largemouth bass); radionuclide (largemouth bass)
			Organic (striped/hybrid bass)
		3	Inorganic (bluegill)
			Organic (catfish); radionuclide (catfish)
			Organic (largemouth bass); inorganic (largemouth bass); radionuclide (largemouth bass)
		4	Organic (bluegill); inorganic (bluegill); radionuclide (bluegill)

Table 5.10. (continued)

Scenario	Exposure pathway	Reach	Contaminant types of concern
			Organic (catfish); inorganic (catfish); radionuclide (catfish)
			Organic (largemouth bass); inorganic (largemouth bass); radionuclide (largemouth bass)
			Organic (striped/hybrid bass)
Irrigation	Ingestion of milk	1	Inorganic
		2	Inorganic
		3.01	Organic; inorganic
		3.02	Organic; inorganic
		3.03	Inorganic
		3.04	Organic; inorganic
		4	Inorganic
	Ingestion of meat	1	Inorganic
		2	Inorganic
		3.02	Organic
		3.03	Inorganic
		4	Inorganic
	Ingestion of leafy vegetables	1	Inorganic
		2	Organic; inorganic
		3.03	Organic; inorganic
		4	Inorganic
Dredging	External exposure	1	Radionuclide
		2.04	Radionuclide
		3.04	Radionuclide
		4.01	Radionuclide
		4.03	Radionuclide
		4.04	Radionuclide
	Ingestion of soil	3.04	Inorganic
	Ingestion of milk	1	Organic; inorganic
		2.04	Organic; inorganic; radionuclide

Table 5.10. (continued)

Scenario	Exposure pathway	Reach	Contaminant types of concern
		3.01	Inorganic
		3.02	Organic; inorganic; radionuclide
		3.03	Organic; inorganic; radionuclide
		3.04	Organic; inorganic
		4.01	Organic; inorganic; radionuclide
		4.02	Inorganic
		4.03	Organic; inorganic; radionuclide
		4.04	Organic; inorganic; radionuclide
		7	Organic; inorganic; radionuclide
	Ingestion of meat	1	Organic; inorganic
		2.04	Inorganic
		3.02	Organic; inorganic; radionuclide
		3.03	Inorganic
		3.04	Organic; inorganic
		4.01	Organic; inorganic; radionuclide
		4.03	Organic; inorganic; radionuclide
		4.04	Organic; inorganic; radionuclide
		7	Inorganic
	Ingestion of vegetables	2.04	Inorganic
		3.01	Inorganic
		3.02	Organic; inorganic
		3.03	Inorganic
		3.04	Organic; inorganic
		4.01	Organic; inorganic; radionuclide
		4.02	Inorganic
		4.03	Inorganic
		4.04	Organic; inorganic
		7	Inorganic
	Inhalation	2.04	Inorganic

Table 5.10. (continued)

Scenario	Exposure pathway	Reach	Contaminant types of concern
		3.01	Inorganic
		3.02	Inorganic
		3.03	Inorganic
		3.04	Inorganic
		4.01	Inorganic
		4.02	Inorganic
		4.03	Inorganic
		4.04	Inorganic
		3.04	Inorganic; radionuclide
		7	Inorganic

The milk, meat, and vegetable pathways in the dredging scenario were the most important in terms of health effects for noncarcinogens and also contributed to unacceptable cancer risks; however, the concentrations that were used to calculate the cancer risks and HQs for these foods were modeled concentrations, which have a high degree of uncertainty and are usually conservatively biased.

The results of the drinking water ingestion scenario indicate that multiple noncarcinogenic chemicals (As, Sb, Mn, nitrate, and Aroclor 1254) pose an unacceptable hazard through direct ingestion.

With the exception of the inhalation of resuspended near-shore sediment, the shoreline use scenario does not pose any unacceptable risk to the public. No COCs were identified for the swimming pathways nor for dermal contact or external exposure to near-shore sediments. The noncarcinogenic COCs for the shoreline use scenario through the inhalation of resuspended sediments are barium and manganese. The carcinogenic COCs are As, Cr, ^{234}U, ^{235}U, and ^{238}U.

No carcinogenic COCs were determined for the irrigation scenario. Noncarcinogenic chemicals of concern for this scenario are nitrate, As, Sb, Ni, and acetone.

5.5 UNCERTAINTY ANALYSIS

The results of any risk assessment are uncertain in that the sample data, exposure parameters, and chemical toxicity values used to characterize risk each contain a range of possibly correct values, each of which could be used in the risk assessment. EPA risk assessment guidance favors the selection of a conservative value for a given parameter so as to "err on the side of safety." The risk assessor must communicate the level of uncertainty contained in the risk estimate to those persons responsible for implementing remedial actions (i.e., the risk managers). The risk managers may conclude that the risk assessment contains too much uncertainty to warrant a costly and possible unnecessary remedial action at a particular site.

Two basic means are used to describe uncertainty in the risk assessment: (1) a qualitative uncertainty analysis, in which the sources of uncertainty with respect to particular findings are identified and discussed qualitatively, and (2) a quantitative uncertainty analysis, in which the amount of uncertainty in the final risk characterization is mathematically quantified. Section 5.5.1 contains a qualitative assessment of uncertainty with those pathways identified in this risk assessment as posing a potentially unacceptable risk. Section 5.5.2 contains a quantitative uncertainty analysis of the fish ingestion and dredging scenarios.

5.5.1 Qualitative Uncertainty Analysis

The following pathways were identified as potentially posing an unacceptable risk to persons exposed to contaminants at one or more locations within the OU: the water ingestion pathway under the drinking water scenario, the dermal absorption pathway under the swimming scenario, the fish ingestion pathway and scenario, the inhalation pathway in the shoreline use scenario, and several pathways in the dredging scenario. The qualitative uncertainty associated with the risk characterization for each pathway is discussed below.

5.5.1.1 Drinking water scenario

The drinking water scenario conservatively assumes that an individual drinks 2 L of surface water daily. (Unfiltered water samples were used in the evaluation of hazard/risk for the Clinch River and Poplar Creek.) Manganese, which has a relatively large distribution coefficient (k_d value), will be mainly attached to the particulates in the water and would be easily filtered during the water treatment process. Evidence does show that manganese occurs in surface waters both in suspension in the quadrivalent state and in the trivalent state in a relatively stable, soluble complex (APHA 1989). Comparatively little is known about the toxicity of manganese (Kimball n.d.). However, in concentrations not causing unpleasant tastes, manganese is regarded by most investigators to be of no toxicological significance in drinking water (Negus 1938; Muehlberger 1951). In addition, manganese has relatively high background values in the environment. Manganese makes up about 0.10% of the earth's crust and is the 12th most abundant element.

Arsenic is found in all living organisms, including those in aquatic systems. McCoy Branch, including the embayment, is not designated for use as a domestic water supply by the state of Tennessee nor is it currently used as a residential supply. There are currently no plans to use any position of the creek for residential supply in the near future. Also, a considerable amount of uncertainty is associated with the RfD value for arsenic. Other sources of arsenic to the Clinch River would include the fly ash pond located at Bull Run Steam Plant.

5.5.1.2 Swimming scenario

The swimming scenario assumes that an individual swims in the Clinch River and Poplar Creek for 45 days out of the year. As noted previously, a phthalate ester, di-n-octylphthalate was detected in subreach 3.02 at levels that would result in an HQ of 1.6 for an adult and an HQ of 2.9 for a child through dermal contact (Table E38 and E39). However, the frequency of detection for this phthalate ester was 1/12. These low frequencies of detection make it difficult to rule out upstream sources or laboratory contamination. In general, the phthalate esters are common laboratory contaminants and are extensively used as solvents and as plasticizers of synthetic polymers such as polyvinal chloride and cellulose acetate (Patnaik 1992). Poplar Creek is not developed for recreational access and has no beach areas or parks. There is uncertainty as to any such development in the foreseeable future.

5.5.1.3 Shoreline use scenario

The shoreline use scenario assumes four potential routes of exposure: (1) ingestion of near-shore sediment, (2) dermal contact with near-shore sediment, (3) inhalation of resuspended near-shore sediment, and (4) external exposure to near-shore sediment. Although the inhalation of resuspended chromium at PCM 3.1 leads to a risk level of concern, the risk estimate conservatively assumes that all chromium is in the chromium IV valence state, which is the most toxic. Previous studies have indicated that most of the chromium in the Clinch River is not chromium IV and should not be a problem.

The evaluation of the exposure of an individual to contaminants in near-shore sediment that are resuspended in the air is dependent on the quantity of fugitive dust generated by the wind. No air monitoring data were available for the Clinch River or Poplar Creek area. Therefore, the quantity of fugitive dust generated by the wind was empirically derived by the method described in Eckerman and Young (1980). In addition, there are uncertainties associated with the input parameters for this method (listed in Table E4).

The parameter values used in the exposure equation have been derived from standard intake rates, exposure frequencies, exposure durations, and averaging times. These values are based on EPA standard default values for residential settings. However, the reach of Poplar Creek identified as being of concern is not currently developed for residential use. There are no plans to do so in the near future.

5.5.1.4 Irrigation scenario

The risk evaluations for the irrigation scenario have a significant additional layer of uncertainty as a result of the use of modeled exposure concentrations. The concentrations that were used to calculate the cancer risks and HQs for this scenario were modeled concentrations, which have a high degree of uncertainty and are usually conservatively biased. Modeling uncertainties arise from having to simulate an infinitely complex system with a finite number of variables. In addition, irrigation is considered an assessment of future conditions because this activity is limited currently but could increase in the future.

5.5.1.5 Fish consumption scenario

Because the fish consumption scenario is a significant pathway relative to risks/hazards in the Clinch River and Poplar Creek, a quantitative uncertainty analysis pertaining to this pathway of exposure is found in Sect. 5.5.2. PCBs and chlordane are classified as "B2" carcinogens, which implies that all information is derived from animal data and that the actual risk to humans may be zero.

5.5.1.6 Dredging scenario

The risk evaluations for the dredging, like the irrigation scenario, have a significant additional layer of uncertainty due to the use of modeled exposure concentrations. As noted previously, the milk, meat, and vegetable pathways in the dredging scenario were the most important pathways in terms of health effects for noncarcinogens and also contributed to unacceptable cancer risks. However, the concentrations that were used to calculate the cancer risks and HQs for these foods were modeled concentrations which have a high degree of uncertainty and are usually conservatively biased. Modeling uncertainties arise from having to simulate an infinitely complex system using a finite number of variables. Like the fish pathway, the dredging scenario is a significant pathway and is included in Sect. 5.5.2.

5.5.2 Quantitative Uncertainty Analysis

This section presents the results of a quantitative uncertainty analysis in which Monte Carlo simulation was used to propagate the subjective probability distributions for the parameters involved in the risk assessment into a subjective probability distribution for both the excess lifetime cancer risk for carcinogens and the HI for noncarcinogens. The sampling method used for the Monte Carlo simulation was Latin Hypercube Sampling (LHS).

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

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

5.5.2.1 Analysis for the fisherman scenario

For the fish ingestion pathway, selected radionuclides and chemicals were identified for use in the quantitative uncertainty analysis. The contaminants given priority for the uncertainty analysis were determined by considering several factors: (1) adequacy of data, (2) contributing sources of contamination, and (3) the risk estimated in the baseline risk assessment. The contaminant was included in the uncertainty analysis if the following conditions were met: if the data were considered adequate, if the contaminant was released by DOE facilities, and if the 10^{-4} risk level was exceeded or if the HQ was >1 for the child or the adult in the baseline risk assessment. The risk estimates from the baseline risk assessment that meet the above criteria are listed for each contaminant in Table 5.11 for noncarcinogens and Table 5.12 for carcinogens. The contaminants analyzed in the uncertainty analysis for the fish ingestion pathway were ^{137}Cs, Aroclor 1260, Aroclor 1254, chlordane, and methyl mercury.

The uncertainties involved with the slope factors for Aroclor 1260, ^{137}Cs, and chlordane and with the reference doses for methyl mercury were propagated through the uncertainty analysis.

Subjective confidence intervals obtained for the excess lifetime cancer risk from select radionuclides and carcinogenic chemicals. The uncertainties associated with the parameters in the risk assessment equations for the fish ingestion pathway are discussed in Appendix E. The uncertainties for each parameter were propagated through the use of 500 iterations of LHS to obtain subjective probability distributions for the excess lifetime cancer risk associated with human ingestion of fish from the Clinch River and Poplar Creek which are contaminated with ^{137}Cs, Aroclor 1260, and chlordane. The excess lifetime cancer risks from ^{137}Cs, Aroclor 1260, and chlordane are analyzed for the adult only. From these distributions, the medians, the lower 5% subjective confidence limits, and the UCL_{95} were obtained.

The subjective confidence intervals for the excess lifetime cancer risk resulting from the contamination of ^{137}Cs are summarized in Table 5.13. The median risk for ^{137}Cs in largemouth bass is 9.3E-05, and there is 90% confidence that the true but unknown risk lies between 1.5E-05 and 6.2E-04.

The subjective confidence interval given in Table 5.13 provides valuable information that can be used to guide decision making. For example, if a 5% lower confidence limit is above a regulatory standard of concern, then remediation is most likely needed. If the 95% confidence limit is below the standard, remediation is most likely not required. If the 95% upper confidence limit is above the standard, but the 50th percentile is below the standard, further study of those parameters which dominate the overall uncertainty is recommended to increase confidence in the decision. However, if the 50th percentile is above the standard, further study may still be recommended. The decision as to whether or not to proceed with remediation will be driven by a cost vs benefits analysis. Because the 95% upper confidence limit is above the regulatory standard of concern but the median value is below the standard, further study should be recommended for those parameters which dominate the overall uncertainty.

The subjective confidence intervals for the excess lifetime cancer risk resulting from Aroclor 1260 and chlordane are summarized in Table 5.14. The median risk for Aroclor 1260 in the catfish of reach 1, for instance, is 2.4E-04; there is 90% confidence that the true but unknown risk lies between 5.6E-05 and 8.4E-04 (or there is 95% confidence that the true but unknown value does not exceed 8.4E-04).

The highest potential excess lifetime cancer risk results from eating fish contaminated with Aroclor 1260. The subjective confidence intervals given in Table 5.14 provide valuable information that can be used to guide decision making. For example, because the 5% lower confidence limit is above a 10^{-4} lifetime cancer risk for the catfish in reaches 2, 3, and 4 and for the striped/hybrid bass in reach 4, it would be recommended that warning signs be posted near these reaches. For all species and reaches studied, the median values for Aroclor 1260 exceed the 10^{-4} lifetime cancer risk. Again, these results indicate the need for action (provided that 10^{-4} is to be used as an action limit and that the evidence for carcinogenicity of PCBs in animals is applicable to human populations).

Results of the sensitivity analysis for carcinogens. A sensitivity analysis was performed for the "Total Excess Lifetime Cancer Risk" for fish ingestion. The method used for this sensitivity analysis was to square the Spearman rank correlation coefficients of each parameter and normalize them to 100% (Decisioneering, Inc., 1994). From this method, an approximate percentage of the parameter's contribution to the overall uncertainty is obtained. These percentages are referred to as the sensitivity index (SI) of the parameter. The higher the SI, the more important the parameter is to the overall uncertainty.

The top sensitivity indices for the total risk resulting from ^{137}Cs contaminated largemouth bass are presented in Table 5.15.

Table 5.11. Noncarcinogenic chemicals of concern for the fish ingestion pathway

			Risk estimate	
Pathway	**Contaminant**	**Reach**	**Adult**	**Child**
Bluegill	Mercury	3	1.1	2.8
		4	na	1.2
Largemouth bass	Mercury	2	na	1.8
		3	1.8	4.7
		4	na	2.2
	Chlordane	2	na	2.3
		3	na	2.5
		4	na	2.3
	Aroclor 1254	2	6.8	18
		3	14	35
		4	8.1	21
Catfish	Mercury	4	na	1.9
	Chlordane	1	4.2	11
		2	2.6	6.7
		3	1.2	3
		4	1.6	4.2
	Aroclor 1254	1	4.2	11
		2	24	61
		3	14	35
		4	20	52
Striped/hybrid bass	Chlordane	2	1.8	4.7
		4	1.3	3.5
	Aroclor 1254	2	23	59
		4	18	47

na = not applicable (these are not contaminants of concern for adults).

Table 5.12. Carcinogenic chemicals of concern for the fish ingestion pathway

Contaminant	Pathway	Reach	Risk estimate
Aroclor 1260	Catfish	1	9.0E-04
		2	2.8E-03
		3	1.9E-03
		4	2.1E-03
	Largemouth bass	2	4.7E-04
		3	9.2E-04
		4	5.9E-04
	Striped/hybrid bass	2	1.1E-03
		4	1.9E-03
	Bluegill	4	9.8E-04
Chlordane	Catfish	1	1.4E-04
^{137}Cs	Largemouth bass	2	1.5E-04

Table 5.13. Results obtained for the excess lifetime cancer risk from ^{137}Cs in the Clinch River and Poplar Creek

Species	Radionuclide	5% subjective confidence limit	Median	95% subjective confidence limit
Largemouth bass	^{137}Cs	1.5E-05	9.3E-05	6.2E-04

Table 5.14. Results obtained for the excess lifetime cancer risk from select carcinogenic chemical contaminants in the Clinch River and Poplar Creek

Reach	Species	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
1	Catfish	Aroclor 1260	5.6E-05	2.4E-04	8.4E-04
	Catfish	Chlordane	1.4E-05	8.6E-05	4.2E-04
	Total risk[a]		8.8E-05	3.5E-04	1.2E-03
2	Largemouth bass	Aroclor 1260	3.1E-05	1.5E-04	5.0E-04
	Catfish	Aroclor 1260	2.1E-04	8.2E-04	2.9E-03
	Striped/hybrid bass	Aroclor 1260	7.8E-05	3.2E-04	1.1E-03

The subjective confidence intervals obtained for the health risk to adults from the fish are

Table 5.14 (continued)

Reach	Species	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
3	Largemouth bass	Aroclor 1260	6.3E-05	2.9E-04	1.0E-03
	Catfish	Aroclor 1260	1.4E-04	5.6E-04	2.1E-03
4	Largemouth bass	Aroclor 1260	3.9E-05	1.7E-04	5.6E-04
	Catfish	Aroclor 1260	1.6E-04	6.9E-04	2.3E-03
	Striped/hybrid bass	Aroclor 1260	1.3E-04	5.8E-04	1.9E-03

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

Table 5.15. Results obtained for the sensitivity analysis performed for excess lifetime risk from ^{137}Cs contamination associated with the

Table 5.17. Results obtained for the hazard quotients for adults from select noncarcinogenic chemical contaminants in the Clinch River and Poplar Creek

Reach	Species	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
1	Catfish	Chlordane	0.1	1.9	22.5
	Catfish	Aroclor 1254	0.1	0.7	4.8
	Total hazard index[a]		0.3	3.0	23.7
2	Largemouth bass	Mercury	0.1	0.8	8.0
	Largemouth bass	Chlordane	0.1	1.6	18.6
	Largemouth bass	Aroclor 1254	0.1	1.0	6.9
	Total hazard index[a]		0.7	4.5	25.8
	Catfish	Chlordane	0.1	1.1	13.0
	Catfish	Aroclor 1254	0.5	4.2	28.8
	Total hazard index[a]		1.0	6.1	38.8
	Striped/hybrid bass	Chlordane	0.1	0.8	9.0
	Striped/hybrid bass	Aroclor 1254	0.4	3.2	24.6
	Total hazard index[a]		0.8	4.7	31.3
3	Bluegill	Mercury	0.1	1.2	12.2
	Largemouth bass	Mercury	0.2	1.7	19.5
	Largemouth bass	Chlordane	0.0	0.4	4.5
	Largemouth bass	Aroclor 1254	0.3	1.9	15.5
	Total hazard index[a]		0.8	5.2	31.8
	Catfish	Chlordane	0.0	0.5	5.4
	Catfish	Aroclor 1254	0.2	1.9	14.0
	Total hazard index[a]		0.5	2.8	18.5
4	Bluegill	Mercury	0.0	0.4	4.7

Table 5.12. Carcinogenic chemicals of concern for the fish ingestion pathway

Contaminant	Pathway	Reach	Risk estimate
Aroclor 1260	Catfish	1	9.0E-04
		2	2.8E-03
		3	1.9E-03
		4	2.1E-03
	Largemouth bass	2	4.7E-04
		3	9.2E-04
		4	5.9E-04
	Striped/hybrid bass	2	1.1E-03
		4	1.9E-03
	Bluegill	4	9.8E-04
Chlordane	Catfish	1	1.4E-04
^{137}Cs	Largemouth bass	2	1.5E-04

Table 5.13. Results obtained for the excess lifetime cancer risk from ^{137}Cs in the Clinch River and Poplar Creek

Species	Radionuclide	5% subjective confidence limit	Median	95% subjective confidence limit
Largemouth bass	^{137}Cs	1.5E-05	9.3E-05	6.2E-04

Table 5.14. Results obtained for the excess lifetime cancer risk from select carcinogenic chemical contaminants in the Clinch River and Poplar Creek

Reach	Species	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
1	Catfish	Aroclor 1260	5.6E-05	2.4E-04	8.4E-04
	Catfish	Chlordane	1.4E-05	8.6E-05	4.2E-04
	Total risk[a]		8.8E-05	3.5E-04	1.2E-03
2	Largemouth bass	Aroclor 1260	3.1E-05	1.5E-04	5.0E-04
	Catfish	Aroclor 1260	2.1E-04	8.2E-04	2.9E-03
	Striped/hybrid bass	Aroclor 1260	7.8E-05	3.2E-04	1.1E-03

uncertainty associated with its RfD. The subjective confidence intervals given in Tables 5.17 and 5.18

Table 5.14 (continued)

Reach	Species	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit

Table 5.17. Results obtained for the hazard quotients for adults from select noncarcinogenic chemical contaminants in the Clinch River and Poplar Creek

Reach	Species	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
1	Catfish	Chlordane	0.1	1.9	22.5
	Catfish	Aroclor 1254	0.1	0.7	4.8
	Total hazard index[a]		0.3	3.0	23.7
2	Largemouth bass	Mercury	0.1	0.8	8.0
	Largemouth bass	Chlordane	0.1	1.6	18.6
	Largemouth bass	Aroclor 1254	0.1	1.0	6.9
	Total hazard index[a]		0.7	4.5	25.8
	Catfish	Chlordane	0.1	1.1	13.0
	Catfish	Aroclor 1254	0.5	4.2	28.8
	Total hazard index[a]		1.0	6.1	38.8
	Striped/hybrid bass	Chlordane	0.1	0.8	9.0
	Striped/hybrid bass	Aroclor 1254	0.4	3.2	24.6
	Total hazard index[a]		0.8	4.7	31.3
3	Bluegill	Mercury	0.1	1.2	12.2
	Largemouth bass	Mercury	0.2	1.7	19.5
	Largemouth bass	Chlordane	0.0	0.4	4.5
	Largemouth bass	Aroclor 1254	0.3	1.9	15.5
	Total hazard index[a]		0.8	5.2	31.8
	Catfish	Chlordane	0.0	0.5	5.4
	Catfish	Aroclor 1254	0.2	1.9	14.0
	Total hazard index[a]		0.5	2.8	18.5
4	Bluegill	Mercury	0.0	0.4	4.7
	Largemouth bass	Mercury	0.1	0.8	8.9
	Largemouth bass	Chlordane	0.0	0.4	4.3
	Largemouth bass	Aroclor 1254	0.2	1.2	8.2
	Total hazard index[a]		0.5	3.1	17.0
	Catfish	Mercury	0.1	1.5	17.5
	Catfish	Chlordane	0.1	0.7	7.6
	Catfish	Aroclor 1254	0.5	3.6	29.2
	Total hazard index[a]		1.1	6.8	48.1
	Striped/hybrid bass	Chlordane	0.0	0.6	6.4
	Striped/hybrid bass	Aroclor 1254	0.3	2.7	19.4

Table 5.18. Results obtained for the hazard quotients for children from select noncarcinogenic chemical contaminants in the Clinch River and Poplar Creek

Reach	Species	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
1	Catfish	Chlordane	0.1	1.6	40.7
	Catfish	Aroclor 1254	0.1	0.6	5.0
	Total hazard index[a]		0.3	3.0	44.1
2	Largemouth bass	Mercury	0.1	0.9	8.8
	Largemouth bass	Chlordane	0.1	1.3	32.2
	Largemouth bass	Aroclor 1254	0.1	1.0	8.5
	Total hazard index[a]		0.7	5.2	38.8
	Catfish	Chlordane	0.1	1.0	22.8
	Catfish	Aroclor 1254	0.6	4.3	31.8
	Total hazard index[a]		1.0	6.7	44.1
	Striped/hybrid bass	Chlordane	0.0	0.7	14.9
	Striped/hybrid bass	Aroclor 1254	0.5	3.3	26.5
	Total hazard index[a]		0.8	5.0	35.8
3	Bluegill	Mercury	0.1	1.2	13.2
	Largemouth bass	Mercury	0.2	1.9	20.0
	Largemouth bass	Chlordane	0.0	0.4	7.6
	Largemouth bass	Aroclor 1254	0.3	2.1	15.5
	Total hazard index[a]		1.0	5.7	35.4
	Catfish	Chlordane	0.0	0.4	8.7
	Catfish	Aroclor 1254		2.0	15.3
	Total hazard index[a]		0.5	3.0	20.9
4	Bluegill	Mercury	0.1	0.4	5.1
	Largemouth bass	Mercury	0.1	0.9	9.9
	Largemouth bass	Chlordane	0.0	0.3	7.4
	Largemouth bass	Aroclor 1254	0.2	1.2	9.3
	Total hazard index[a]		0.6	3.5	20.6
	Catfish	Mercury	0.1	1.3	33.4
	Catfish	Chlordane	0.0	0.6	12.9
	Catfish	Aroclor 1254	0.5	3.8	26.4
	Total hazard index[a]		1.1	7.5	66.6
	Striped/hybrid bass	Chlordane	0.0	0.5	11.0
	Striped/hybrid bass	Aroclor 1254	0.4	2.7	20.6
	Total hazard index[a]		0.7	4.0	29.2

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

provide valuable information that can be used to guide decision making. For example, if a 5% lower confidence limit is above a regulatory standard of concern, then remediation is most likely needed. If the UCL_{95} is below the standard, remediation is most likely not required. If the UCL_{95} is above the standard, but the 50th percentile is below the standard, further study may be necessary for those parameters which dominate the overall uncertainty. However, if the 50th percentile is above the standard, further study may still be worthwhile, although under some circumstances it may be prudent to proceed with remediation. In all cases, the UCL_{95} exceeds an HQ of 1.0 for both adults and children.

Results of the sensitivity analysis for select noncarcinogens. A sensitivity analysis was performed for each "Total Hazard Index" for each species of fish. The top sensitivity indices for the total HI in largemouth bass in reach 3 for adults are presented in Table 5.19.

The two most sensitive parameters for the total HI from largemouth bass are the number of fish-meals per day and the RfD for Aroclor 1254 and methyl mercury. These results indicate that, if necessary, the uncertainty involved with the total HI from the human ingestion of fish from the Clinch River and Poplar Creek would be most effectively lowered by taking local surveys of consumption patterns and by further analysis of the RfD for PCBs and methyl mercury rather than by further sampling of fish. The uncertainty in the overall HI was almost dominated by the number of fish-meals per day.

A parameter that does not show up as a sensitive parameter is the contaminant concentration in fish. However, the statistical analyses that were performed on the data show a relatively small uncertainty for the majority of the contaminants and reaches. In most cases, the uncertainty seems too small (1.09 uncertainty factor) and may be suspect.

Table 5.19. Results obtained for the sensitivity analysis performed for the total hazard index from select noncarcinogen contamination associated with the adult ingestion of fish from the Clinch River and Poplar Creek

Species	Sensitive parameter	Sensitivity index (%)	Rank
Largemouth bass	I_f: meals per day	62.3	1
(reach 3)	Mercury RfD: mg/kg/d	14.2	2
	Aroclor 1254 RfD: mg/kg/d	10.1	2
	I_m: kilograms per meal	6.3	3

I_f = mass of fish consumed per meal; RfD = reference dose; I_m = frequency of meals consisting of fish caught in the Clinch River–Poplar Creek system.

Conclusions for the fish ingestion pathway. The following conclusions can be drawn from the quantitative uncertainty analysis on the risk of adverse health effects resulting from the ingestion of select radionuclide- and chemical-contaminated fish from Clinch River and Poplar Creek.

- The total hazard quotients for all contaminants and fish exceed the recommended level of 1.0 at the 95% subjective confidence limit as well as at the median value in many cases for children and adults. The uncertainty cannot be improved without additional information regarding the toxic effects of chlordane, Aroclor 1254, and mercury. The large uncertainty in the hazard quotients for chlordane result in part from the uncertainty in the reference dose.

- The highest potential excess lifetime cancer risk from carcinogenic chemicals results from Aroclor 1260. The median values exceed 10^{-4} risk for each of the species and reaches studied for both chlordane and Aroclor 1260.

- Results of the sensitivity analysis indicate that, if necessary, the overall estimate of uncertainty in the total risk estimates would be most efficiently reduced by taking local surveys to obtain better estimates for the numbers of fish-meals eaten per week and the amount of fish eaten per meal for the avid angler.

5.5.2.2 Analysis for the dredging scenario resulting from contaminated deep sediment

The contaminants given priority for the uncertainty analysis were determined by considering the following factors: (1) adequacy of data, (2) contributing sources of contamination, and (3) the risk estimated in the baseline risk assessment. The contaminant was included in the uncertainty analysis if the following conditions were met: if the data was considered adequate, if the contaminant was released by DOE facilities, and if the 10^{-4} risk level was exceeded or if the HQ was >1 for adults in the baseline risk assessment. The noncarcinogenic contaminant and radionuclide identified on the basis of these criteria for the uncertainty analysis for the dredging pathway was inorganic mercury and ^{137}Cs, respectively. The reaches of concern are shown in Table 5.20.

Table 5.20. Contaminants of concern for the dredging pathway

Pathway	Contaminant	Subreach	Risk estimate	
			Adult	Child
Milk	Mercury	3.04	1.3	13
Meat	Mercury	3.04	2.6	12
Vegetables	Mercury	3.04	7.1	33
External	^{137}Cs	2.04	2.7E-04	na
	^{137}Cs	3.04	1.0E-04	na
	^{137}Cs	4.03	2.3E-04	na

na = not applicable (cancer risks are not evaluated for children because they are based on a 30-year exposure).

Although concern from Aroclor 1254 is shown in the dredging scenario of the baseline risk assessment calculations, there was insufficient data information to bound the uncertainty in the sediment concentration for this contaminant. This was due in large part to the aggregation of the data into subreaches. Therefore, Aroclor 1254 was not considered in the uncertainty analysis.

Subjective confidence intervals obtained from the uncertainty analysis for inorganic mercury. The uncertainties associated with the parameters in the risk assessment equations for the dredging scenario are discussed in Appendix E. The uncertainties for each parameter were propagated through the use of 500 iterations of LHS to obtain subjective probability distributions for the hazard quotient associated with dredging deep sediments from the Clinch River and Poplar Creek that are contaminated with inorganic mercury. From these distributions, the medians, the lower 5% subjective confidence limits, and the UCL_{95} were obtained.

The subjective confidence intervals for the adult are summarized in Table 5.21, and the subjective intervals for children are presented in Table 5.22. The median risk for the ingestion of milk from mercury, for instance, is 0.09; there is 90% confidence that the true but unknown risk lies between 0.01 and 2.25 (or there is 95% confidence that the true but unknown value does not exceed 2.25).

The subjective confidence intervals given in Table 5.21 and Table 5.22 provide valuable information that can be used to guide decision making. For example, if a 5% lower confidence limit is above a regulatory standard of concern, then remediation is most likely needed. If the 95% upper confidence limit is below the standard, remediation is most likely not required. If the 95% upper confidence limit is above the standard, but the 50th percentile is below the standard, further study of those parameters which dominate the overall uncertainty (milk pathways) would be needed to reduce overall uncertainty. However, if the 50th percentile is above the standard, further study may still be worthwhile. The decision as to whether or not to proceed with remediation will be driven by a cost vs benefits analysis.

Table 5.21. Results obtained for the excess health risk to adults from select noncarcinogenic contamination in deep sediments of the Clinch River and Poplar Creek

Subreach	Pathway	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
3.04	Milk	Mercury	0.01	0.09	2.25
	Meat	Mercury	0.59	5.89	287.27
	Leafy vegetables	Mercury	0.24	2.35	68.10
	Nonleafy vegetables	Mercury	0.13	1.96	43.95

Table 5.22. Results obtained for the excess health risk to children from select noncarcinogenic contamination in deep sediments of the Clinch River and Poplar Creek

Subreach	Pathway	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
3.04	Milk	Mercury	0.05	0.48	18.48
	Meat	Mercury	1.54	15.28	338.69
	Leafy vegetables	Mercury	0.61	5.99	126.11
	Nonleafy vegetables	Mercury	0.24	3.57	74.18

Results of the sensitivity analysis for the dredging scenario for inorganic mercury. A sensitivity analysis was performed for the different pathways. The method used for this sensitivity analysis was to square the Spearman rank correlation coefficients of each parameter and normalize them to 100% (Decisioneering, Inc., 1994). From this method, an approximate percentage of the parameter's contribution to the overall uncertainty is obtained. These percentages are referred to as the SI of the parameter. The higher the SI, the more important the parameter is to the overall uncertainty.

The top sensitivity indices for the risk from dredging deep sediments in the Clinch River and Poplar Creek which are contaminated with inorganic mercury are presented in Table 5.23.

The top four sensitive parameters for the uncertainty are the uncertainty in the soil-to-plant uptake factor for pasture, the RfD for mercury, the milk and meat uptake factors, and the sediment concentration data. Uncertainty in the estimate of risk for inorganic mercury in deep sediments (if the deep sediment is dredged) could be dramatically reduced with more information pertaining to these parameters. A very large source of uncertainty for the transfer factors is the lack of knowledge about the solubility of mercury and lack of knowledge about its chemical form.

Subjective confidence intervals obtained from the uncertainty analysis for ^{137}Cs. The uncertainties associated with the parameters in the risk assessment equations for the dredging scenario are discussed in Appendix E. The uncertainties for each parameter were propagated through the use of 500 iterations of LHS to obtain subjective probability distributions for the excess lifetime cancer risk associated with dredging deep sediments from the Clinch River and Poplar Creek that are contaminated with inorganic mercury. From these distributions, the medians, the lower 5% subjective confidence limits, and the UCL_{95} were obtained.

The subjective confidence intervals for the adult are summarized in Table 5.24. The median risk for the external exposure from ^{137}Cs, for instance, is 1.1E-04; there is 90% confidence that the true but unknown risk lies between 1.0E-05 and 7.2E-04 (or there is 95% confidence that the true but unknown value does not exceed 7.2E-04).

Table 5.23. Results obtained for the sensitivity analysis performed for total excess health risk from mercury contamination associated with the dredging scenario from the Clinch River and Poplar Creek

Pathway	Sensitive parameter	Sensitivity index (%)	Rank
Milk	$B_{v,past}$	30.1	1
	F_m	25.0	2
	RfD (mercury)	19.9	3
	C_{sed}	14.1	4
Meat	$B_{v,past}$	36.9	1
	RfD (mercury)	23.8	2
	C_{sed}	16.5	3
	F_f	12.8	4
Vegetables	RfD (mercury)	38.5	1
	B_v	34.2	1
	C_{sed}	18.5	2

$B_{v,past}$ = soil-to-plant uptake factor for pasture; F_m = milk transfer coefficent (d/L); RfD = reference dose; C_{sed} = sediment concentration; F_f= meat transfer coefficient (d/kg).

Table 5.24. Results obtained for the excess health risk to adults from ^{137}Cs contamination in deep sediment of the Clinch River and Poplar Creek

Subreach	Pathway	Contaminant	5% subjective confidence limit	Median	95% subjective confidence limit
2.04	External	^{137}Cs	1.0E-05	1.1E-04	7.2E-04
3.04	External	^{137}Cs	6.1E-06	8.1E-05	4.7E-04
4.03	External	^{137}Cs	9.6E-06	5.2E-05	2.4E-04

The subjective confidence intervals given in Table 5.24 provides valuable information that can be used to guide decision making. For example, if a 5% lower confidence limit is above a regulatory standard of concern, then remediation is most likely needed. If the 95% upper confidence limit is below the standard, remediation is most likely not required. If the 95% upper confidence limit is above the standard, but the 50th percentile is below the standard, further study may be necessary for those parameters which dominate the overall uncertainty (subreaches 3.04 and 4.03). However, if the 50th percentile is above the standard, further study may still be worthwhile (subreach 2.04). The decision as to whether or not to proceed with remediation will be driven by a cost vs benefits analysis.

Results of the sensitivity analysis for the dredging scenario for ^{137}Cs. A sensitivity analysis was performed for the different pathways. The method used for this sensitivity analysis was to square the Spearman rank correlation coefficients of each parameter and normalize them to 100% (Decisioneering, Inc., 1994). From this method, an approximate percentage of the parameter's contribution to the overall uncertainty is obtained. These percentages are referred to as the SI of the parameter. The higher the SI, the more important the parameter is to the overall uncertainty.

The top sensitivity indices for the total risk from dredging deep sediments in the Clinch River and Poplar Creek which are contaminated with ^{137}Cs are presented in Table 5.25.

The top three sensitive parameters for the uncertainty are the uncertainties in the: (1) ^{137}Cs concentration data, (2) the risk conversion factor (RCF), and (3) the external exposure frequency. If necessary, the uncertainty in the estimate of the excess lifetime cancer risk would be most effectively reduced by taking additional samples in order to reduce the uncertainty in the sediment concentration.

Conclusions for the dredging scenario. The uncertainty analysis conducted for the dredging scenario verifies the results of the baseline risk assessment (i.e., that ^{137}Cs and mercury in the deep sediment are COPCs). If necessary, additional sediment samples would improve the uncertainty associated with the risk estimates of both contaminants. Finally, caution should be used when interpreting the estimates of the HQ for inorganic mercury because of the lack of knowledge about the solubility of mercury in deep sediment.

5.6 DEVELOPMENT OF REMEDIAL GOAL OPTIONS

The EPA (Region IV) has requested that each baseline risk assessment include a section which outlines the remedial goal options (RGOs) for the chemicals and media of potential concern that have been identified. Typically, this sections includes both ARARs and health-based cleanup goals, but for the purpose of this report the ARARs have been presented in Chap. 4.

Remedial goal options are chemical-specific, medium-specific numerical concentration limits that are identified for all contaminants and all pathways found to be of concern during the baseline human health risk assessment process. Remedial goal options are not the first or the final set of cleanup levels in the CERCLA process, but they can be viewed as modified preliminary remediation goals based on site characterization and the baseline risk assessment findings.

Remedial goal options for the Clinch River and Poplar Creek were established on the basis of the exposure parameters and equations used to perform the risk assessment. No modifications were made to the parameter values. Risk/hazard-based concentrations were back-calculated for each chemical and pathway identified as a concern and are presented in Tables 5.26–5.33. For the carcinogenic chemicals of concern, the RGOs were calculated based on an ECR of 10^{-4}, 10^{-5}, and 10^{-6}. For the noncarcinogenic chemicals of concern, the RGOs were calculated based on an individual hazard quotient of 0.1, 1.0, and 10. The purpose of the RGOs is to provide the risk managers with the maximum risk-related media level options on which to develop remedial alternatives.

Table 5.25. Results obtained for the sensitivity analysis performed for total excess health risk from ^{137}Cs contamination associated with the dredging scenario from the Clinch River and Poplar Creek

Pathway	Sensitive parameter	Sensitivity index (%)	Rank
External (subreach 2.04)	C_{sed} (sediment concentration)	38.4	1
	RCF (risk conversion factor)	21.2	2
	D_l (dilution factor)	9.6	3
External (subreach 3.04)	C_{sed}	38.5	1
	RCF	22.2	2
	λ_{eff} [loss rate from radioactive decay and from harvesting/ leaching (year)$^{-1}$]	10.7	3
External (subreach 4.03)	RCF	31.7	1
	External exposure frequency	15.3	2
	λ_{eff}	14.4	2

Table 5.26. Remedial goal options for the carcinogenic chemicals and pathways of concern for soil for the dredging scenario

Pathway	Chemical	RGOs based on an ECR = 10^{-4}	RGOs based on an ECR = 10^{-5}	RGOs based on an ECR = 10^{-6}	Units of measure
External exposure	^{60}Co	4.60E+00	4.60E-01	4.60E-02	pCi/g
	^{137}Cs	7.84E+00	7.84E-01	7.84E-02	pCi/g
	^{238}U	2.29E+02	2.29E+01	2.29E+00	pCi/g
Inhalation of resuspended sediments	Arsenic	6.34E+01	6.34E+00	6.34E-01	mg/kg
	Cadmium	5.19E+02	5.19E+01	5.19E+00	mg/kg
	Chromium	7.73E+01	7.73E+00	7.73E-01	mg/kg
	^{234}U	1.27E+02	1.27E+01	1.27E+00	pCi/g
	^{238}U	1.43E+02	1.43E+01	1.43E+00	pCi/g
Ingestion of milk produced by cattle pastured on dredged sediments	^{137}Cs	4.18E+01	4.18E+00	4.18E-01	pCi/g
	^{90}Sr	2.48E+01	2.48E+00	2.48E-01	pCi/g
	^{99}Te	1.58E+03	1.58E+02	1.58E+01	pCi/g
	^{234}U	8.49E+02	8.49E+01	8.49E+00	pCi/g
	^{238}U	5.90E+02	5.90E+01	5.90E+00	pCi/g
	Arsenic	3.01E+02	3.01E+01	3.01E+00	mg/kg
	Pentachlorophenol	3.45E+01	3.45E+00	3.45E-01	mg/kg
	Aroclor 1260	3.10E-02	3.10E-03	3.10E-04	mg/kg

Table 5.26 (continued)

Pathway	Chemical	RGOs based on an ECR = 10^{-4}	RGOs based on an ECR = 10^{-5}	RGOs based on an ECR = 10^{-6}	Units of measure
Ingestion of beef grazed on pasture forage produced on dredged sediments	^{137}Cs	7.64E+01	7.64E+00	7.64E-01	pCi/g
	^{90}Sr	8.67E+01	8.67E+00	8.67E-01	pCi/g
	Arsenic	1.01E+02	1.01E+01	1.01E+00	mg/kg
	Aroclor 1260	1.10E-01	1.10E-02	1.10E-03	mg/kg
Ingestion of vegetables produced on dredged sediments	^{137}Cs	3.50E+02	3.50E+01	3.50E+00	pCi/g
	^{99}Te	6.80E+01	6.80E+00	6.80E-01	pCi/g
	^{90}Sr	2.80E+01	2.80E+00	2.80E-01	pCi/g
	Arsenic	1.24E+02	1.24E+01	1.24E+00	mg/kg
	Pentachlorophenol	3.75E+02	3.75E+01	3.75E+00	mg/kg
	Aroclor 1260	1.87E+01	1.87E+00	1.87E-01	mg/kg

RGOs = remedial goal options; ECR = excess cancer risk.

Table 5.27. Remedial goal options for the carcinogenic chemicals and pathways of concern for near-shore sediment for the shoreline use scenario

Pathway	Chemical	RGOs based on an ECR = 10^{-4}	RGOs based on an ECR = 10^{-5}	RGOs based on an ECR = 10^{-6}	Units of measure
Shoreline use: inhalation	Arsenic	1.27E+02	1.27E+01	1.27E+00	mg/kg
	Chromium	1.55E+02	1.55E+01	1.55E+00	mg/kg
	^{234}U	2.53E+02	2.53E+01	2.53E+00	pCi/g
	^{235}U	2.73E+02	2.73E+01	2.73E+00	pCi/g
	^{238}U	2.86E+02	2.86E+01	2.86E+00	pCi/g

RGOs = remedial goal options; ECR = excess cancer risk.

Table 5.28. Remedial goal options for the carcinogenic chemicals and pathways of concern for fish tissue for the fish ingestion scenario

Pathway	Chemical	RGOs based on an ECR = 10^{-4}	RGOs based on an ECR = 10^{-5}	RGOs based on an ECR = 10^{-6}	Units of measure
Fish ingestion	Arsenic	2.10E-01	2.10E-02	2.10E-03	mg/kg
	Beryllium	7.34E-02	7.34E-03	7.34E-04	mg/kg
	Aroclor 1260	4.10E-02	4.10E-03	4.10E-04	mg/kg
	Chlordane	2.43E-01	2.43E-02	2.43E-03	mg/kg
	^{137}Cs	5.58E+00	5.58E-01	5.58E-02	pCi/g
	^{90}Sr	3.16E+00	3.16E-01	3.16E-02	pCi/g
	4,4'-DDE	9.28E-01	9.28E-02	9.28E-03	mg/kg
	4,4'-DDT	9.28E-01	9.28E-02	9.28E-03	mg/kg
	Aldrin	1.86E-02	1.86E-03	1.86E-04	mg/kg

RGOs = remedial goal options; ECR = excess cancer risk.

Table 5.29. Remedial goal options for soil for the noncarcinogenic chemicals and pathways of concern for the dredging scenario

Pathway	Chemical	Adult			Child		
		RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)	RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)
Inhalation of resuspended sediment	Barium	1.94E+04	1.94E+03	1.94E+02	4.36E+03	4.36E+02	4.36E+01
	Manganese	1.94E+03	1.94E+02	1.94E+01	4.36E+02	4.36E+01	4.36E+00
Ingestion of milk produced by cattle pastured on dredged sediment	Arsenic	5.58E+03	5.58E+02	5.58E+01	6.22E+02	6.22E+01	6.22E+00
	Barium	1.57E+05	1.57E+04	1.57E+03	1.68E+04	1.68E+03	1.68E+02
	Boron	1.81E+04	1.81E+03	1.81E+02	1.94E+03	1.94E+02	1.94E+01
	Cadmium	8.63E+02	8.63E+01	8.63E+00	9.24E+01	9.24E+00	9.24E-01
	Chromium	5.37E+05	5.37E+04	5.37E+03	5.75E+04	5.75E+03	5.75E+02
	Mercury	1.92E+02	1.92E+01	1.92E+00	2.05E+01	2.05E+00	2.05E-01
	Molybdenum	1.75E+03	1.75E+02	1.75E+01	1.88E+02	1.88E+01	1.88E+00
	Nickel	9.90E+02	9.90E+01	9.90E+00	1.06E+02	1.06E+01	1.06E+00
	Selenium	2.98E+02	2.98E+01	2.98E+00	3.20E+01	3.20E+00	3.20E-01
	Silver	3.99E+04	3.99E+03	3.99E+02	4.28E+03	4.28E+02	4.28E+01
	Vanadium	8.14E+03	8.14E+02	8.14E+01	8.72E+02	8.72E+01	8.72E+00
	Zinc	7.22E+03	7.22E+02	7.22E+01	7.73E+02	7.73E+01	7.73E+00
	4,4'-DDT	3.00E+01	3.00E+00	3.00E-01	3.21E+00	3.21E-01	3.21E-02
	Aroclor 1254	2.46E+00	2.46E-01	2.46E-02	2.64E-01	2.64E-02	2.64E-03

Table 5.29 (continued)

Pathway	Chemical	Adult			Child		
		RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)	RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)
Ingestion of beef grazed on pasture forage produced on dredged sediments	Arsenic	1.95E+03	1.95E+02	1.95E+01	4.17E+02	4.17E+01	4.17E+00
	Chromium	6.67E+03	6.67E+02	6.67E+01	1.43E+03	1.43E+02	1.43E+01
	Manganese	7.53E+05	7.53E+04	7.53E+03	1.61E+05	1.61E+04	1.61E+03
	Mercury	1.01E+02	1.01E+01	1.01E+00	2.16E+01	2.16E+00	2.16E-01
	Selenium	3.34E+02	3.34E+01	3.34E+00	7.15E+01	7.15E+00	7.15E-01
	Vanadium	3.96E+04	3.96E+03	3.96E+02	8.48E+03	8.48E+02	8.48E+01
	Zinc	8.07E+03	8.07E+02	8.07E+01	1.73E+03	1.73E+02	1.73E+01
	Aroclor 1254	8.71E+00	8.71E-01	8.71E-02	1.87E+00	1.87E-01	1.87E-02
Ingestion of vegetables produced on dredged sediments	Arsenic	2.40E+03	2.40E+02	2.40E+01	5.14E+02	5.14E+01	5.14E+00
	Boron	2.67E+04	2.67E+03	2.67E+02	5.72E+03	5.72E+02	5.72E+01
	Cadmium	9.00E+02	9.00E+01	9.00E+00	1.93E+02	1.93E+01	1.93E+00
	Chromium	1.40E+04	1.40E+03	1.40E+02	2.99E+03	2.99E+02	2.99E+01
	Manganese	1.27E+04	1.27E+03	1.27E+02	2.72E+03	2.72E+02	2.72E+01
	Mercury	3.61E+01	3.61E+00	3.61E-01	7.74E+00	7.74E-01	7.74E-02
	Molybdenum	1.77E+03	1.77E+02	1.77E+01	3.79E+02	3.79E+01	3.79E+00
	Nickel	1.38E+04	1.38E+03	1.38E+02	2.95E+03	2.95E+02	2.95E+01

Table 5.29 (continued)

Pathway	Chemical	Adult			Child		
		RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)	RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)
	Silver	8.99E+02	8.99E+01	8.99E+00	1.93E+02	1.93E+01	1.93E+00
	Zinc	2.72E+04	2.72E+03	2.72E+02	5.82E+03	5.82E+02	5.82E+01
	Aroclor 1254	4.64E+01	4.64E+00	4.64E-01	9.94E+00	9.94E-01	9.94E-02

RGOs = remedial goal options; HQ = hazard quotient.

Table 5.30. Remedial goal options for the noncarcinogenic chemicals and pathways of concern for near-shore sediment for the shoreline use scenario

Pathway	Chemical	Adult			Child		
		RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)	RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)
Inhalation of resuspended sediment	Barium	3.88E+04	3.88E+03	3.88E+02	8.73E+03	8.73E+02	8.73E+01
	Manganese	3.88E+03	3.88E+02	3.88E+01	8.73E+02	8.73E+01	8.73E+00

RGOs = remedial goal options; HQ = hazard quotient.

Table 5.31. Remedial goal options for the noncarcinogenic chemicals and pathways of concern for water

Pathway	Chemical	Adult			Child		
		RGOs based on an HQ of 10 (mg/L)	RGOs based on an HQ of 1 (mg/L)	RGOs based on an HQ of 0.1 (mg/L)	RGOs based on an HQ of 10 (mg/L)	RGOs based on an HQ of 1 (mg/L)	RGOs based on an HQ of 0.1 (mg/L)
Ingestion of milk produced by cattle pastured on soil irrigated with surface water	Nitrate	2.54E+01	2.54E+00	2.54E-01	2.72E+00	2.72E-01	2.72E-02
	Nickel	3.91E+00	3.91E-01	3.91E-02	4.19E-01	4.19E-02	4.19E-03
	Aroclor 1254	7.61E-03	7.61E-04	7.61E-05	8.16E-04	8.16E-05	8.16E-06
Ingestion of vegetables irrigated with surface water	Arsenic	3.14E-01	3.14E-02	3.14E-03	6.73E-02	6.73E-03	6.73E-04
	Nitrate	1.93E+02	1.93E+01	1.93E+00	2.72E+00	2.72E-01	2.72E-02
Ingestion of beef drinking surface water	Nitrate	9.47E+01	9.47E+00	9.47E-01	2.03E+01	2.03E+00	2.03E-01
Direct ingestion of untreated surface water	Antimony	1.46E-01	1.46E-02	1.46E-03	6.26E-02	6.26E-03	6.26E-04
	Arsenic	1.10E-01	1.10E-02	1.10E-03	4.69E-02	4.69E-03	4.69E-04
	Manganese	5.11E+01	5.11E+00	5.11E-01	2.19E+01	2.19E+00	2.19E-01
	Nitrate	5.84E+02	5.84E+01	5.84E+00	2.50E+02	2.50E+01	2.50E+00
	Aroclor 1254	7.30E-03	7.30E-04	7.30E-05	3.13E-03	3.13E-04	3.13E-05

RGOs = remedial goal options; HQ = hazard quotient.

Table 5.32. Remedial goal options for the noncarcinogenic chemicals and pathways of concern for fish tissue

Pathway	Chemical	Adult			Child		
		RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)	RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)
Ingestion fish tissue	Arsenic	4.06E+00	4.06E-01	4.06E-02	1.56E+00	1.56E-01	1.56E-02
	Mercury	4.06E+00	4.06E-01	4.06E-02	1.56E+00	1.56E-01	1.56E-02
	Selenium	6.76E+01	6.76E+00	6.76E-01	2.61E+01	2.61E+00	2.61E-01
	Chlordane	8.11E-01	8.11E-02	8.11E-03	3.13E-01	3.13E-02	3.13E-03
	4,4'-DDT	6.76E+00	6.76E-01	6.76E-02	2.61E+00	2.61E-01	2.61E-02
	Aroclor 1254	2.7E-01	2.7E-02	2.7E-03	1.04E-01	1.04E-02	1.04E-03

RGOs = remedial goal options; HQ = hazard quotient.

Table 5.33. Remedial goal options for the noncarcinogenic chemicals and pathways of concern for swimming scenario

Pathway	Chemical	Adult			Child		
		RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)	RGOs based on an HQ of 10 (mg/kg)	RGOs based on an HQ of 1 (mg/kg)	RGOs based on an HQ of 0.1 (mg/kg)
Dermal contact	Di-N-octylpthalate	6.25E-02	6.25E-03	6.25E-04	3.45E-02	3.45E-03	3.45E-04

RGOs = remedial goal options; HQ = hazard quotient.

6. BASELINE ECOLOGICAL RISK ASSESSMENT REPORT

The purpose of this ecological risk assessment is to provide an estimate of ecological risks due to contaminants in the Clinch River/Poplar Creek Operable Unit (CR/PC OU). The CR/PC OU includes Poplar Creek embayment, McCoy Branch embayment, and the Clinch River adjacent to and below the DOE's ORR (Fig. 3.1). The ecological risk assessment is structured in terms of the standard paradigm for risk assessment (NRC 1983, Risk Assessment Forum 1992). It begins with a problem formulation phase that defines the contaminant sources, the receiving environment, and the assessment end points. Then, for each end point, there is an analytical phase consisting of exposure assessment and effects assessment and a risk characterization that combines the components of the analysis phase.

6.1 ECOLOGICAL PROBLEM FORMULATION

The problem formulation consists of the description of the relevant features of the environment, description of the sources of contamination, identification of ecological end points, and summarization of that information in terms of a conceptual model of the hazard posed by the contaminants to the end point biota.

6.1.1 Environmental Description

The environment considered in this assessment is the Clinch River adjacent to and downstream of the ORR including the embayments of Poplar Creek and McCoy Branch. A general description of this system is provided above (Chap. 2). It consists of the Clinch River arm of Watts Bar Reservoir, the lower portion of Melton Hill Reservoir, Poplar Creek embayment up to the confluence of the East Fork, and the McCoy Branch embayment up to . The reservoirs are narrow and relatively fast flowing (Chap. 3). Because there is no floodplain, there are currently no known contaminated terrestrial systems. A contaminated terrestrial ecosystem would be created under the baseline dredging case, but the simple assessment performed for that hypothetical case does not include site-specific ecological characteristics. The federal and state-listed threatened or endangered (T&E) species and other state-listed rare or declining species are listed in Appendix J.

6.1.2 Sources

The proximate sources considered in this assessment are the contaminated media in the CR/PC OU: water, sediment, and fish tissues. The ultimate sources of the contaminants are the DOE's operations of the ORR including waste disposal, spills, and use of chemicals such as pesticides in the environment and other upstream sources. The intermediate sources are the streams that drain the ORR, all of which are tributaries of the Clinch River. McCoy Branch has been contaminated by coal ash and potentially by other currently undefined materials deposited in Rogers Quarry. Poplar Creek has been contaminated by releases from Y-12 to EFPC (particularly mercury), by various contaminants released by the K-25 plant directly to the embayment, and by various contaminants released by the cities of Oliver Springs and Oak Ridge, and by other sources in the Poplar Creek watershed. Although the DOE's operations did not contribute all of the contaminants in the Clinch River or Poplar Creek, the assessment must characterize the overall risks to the biota, not simply the risks that would have occurred if the DOE emissions had been released to a pristine environment.

Contaminants of potential ecological concern (COPECs) for ecological risks have been identified in previous screening assessments (Suter 1991, Cook et al. 1992, Blaylock et al. 1993a). However, because the number of COPECs was large for all media, the revisions that have occurred in the ecological screening benchmarks, and the new contaminant concentration data that have been added, all contaminants are rescreened against benchmarks for this assessment.

6.1.3 Ecological End Points

The problem formulation must identify both the assessment end points, which are explicit statements of the characteristics of the environment that are to be protected, and the measurement end points, which are quantitative summaries of a measurement or series of measurements that are related to effects on an assessment end point.

6.1.3.1 Assessment end points

The following assessment end points for aquatic and terrestrial risks have been selected for this assessment.

- Reduction in species richness or abundance or increased frequency of gross pathologies in fish communities resulting from toxicity.
- Reduced species richness or abundance of benthic macroinvertebrate communities resulting from toxicity.
- Reduction in abundance or production of piscivorous wildlife populations resulting from toxicity.
- Reduction in abundance or production of flying insectivorous wildlife populations resulting from toxicity.
- Reduction in production of terrestrial plant communities resulting from toxicity.
- Reduction in abundance or production of terrestrial wildlife populations resulting from toxicity.
- Reduction in viability or fecundity of individuals of any threatened or endangered species resulting from toxicity.

The ecological assessment end points have been selected based on DQO meetings that included representatives of the DOE, EPA Region IV, and TDEC. The selected end points are explained below.

1. The fish community is considered to be an appropriate end point community because it is ecologically and societally important, susceptible, and has a scale appropriate to the site. The fish community is ecologically significant since much of the energy flow in temperate reservoirs passess through fishes and fishes are a major nutrient reservoir in those systems; its societal importance is due to recreational fisheries. In addition, the scale is appropriate because most of the fish species in the CR/PC OU move within an area smaller than the OU.

2. The benthic invertebrate community is considered to be an appropriate end point community because they are highly susceptible, and have a scale appropriate to the site. This community is highly susceptible because of its association with the sediment, which is the repository of most of the COPECs. In addition, the insects and crustaceans that dominate this community are sensitive to a variety of contaminants. Because these organisms are sedentary, their scale is highly appropriate to the scale of the site.

3. Piscivorous wildlife are considered to be an appropriate end point trophic group because they are highly susceptible and have a scale appropriate to the site. Two of the contaminants of potential concern, PCBs and mercury, biomagnify in aquatic food webs leading to high levels of exposure in piscivorous wildlife. Mink have been shown to be highly sensitive to the toxic effects of PCBs and mercury in toxicity tests. Reproduction of piscivorous birds has been shown to be highly sensitive to PCBs. Prior screening assessments have shown that there is a potential for toxic exposures of wildlife to a variety of contaminants in Watts Bar Reservoir (Suter 1991, Cook et al. 1992). Although populations of piscivorous wildlife are not as clearly associated with the OU as the aquatic communities, they forage within an area smaller than the Clinch River arm (at least during the breeding season) so the scale is appropriate. The chosen end point populations are mink and great blue heron.

4. Flying insectivorous wildlife (i.e., bats and birds that catch insects on the wing) are considered to be an appropriate end point trophic group because they are susceptible and have a scale appropriate to the site. Two of the contaminants of potential concern, PCBs and mercury, accumulate in sediments and biomagnify in food webs leading to potentially high levels of exposure in wildlife that feed on benthic invertebrates. Certain species of swallows and bats tend to forage over open water and feed on the emergent stages of aquatic insects. Although populations of insectivorous wildlife are not as clearly associated with the OU as the aquatic communities, they forage within an area smaller than the Clinch River arm (at least during the breeding season) so the scale is appropriate. The chosen end point populations are the little brown bat and the rough-winged swallow.

5. The terrestrial plant community is considered to be an appropriate end point community because it is ecologically important, susceptible, and has a scale appropriate to the site. Since the plant community is responsible for primary production it is ecological significant. This community is susceptible because it would be directly exposed to the contaminants in the dredge spoil. Finally, the scale is appropriate because plants are immobile and because a distinct plant community would develop on dredge spoil deposited on land.

6. EPA Region IV recommended that the ecological risk assessment for animals in the dredge spoil scenario be based on the human health assessment garden model with a herbivorous wildlife species substituted for the human (Lynn Wellman, personal communication to Glenn Suter, November 2, 1993). The chosen representative species is the eastern cottontail.

7. T&E species are, in general, too rare to be ecologically significant, and may not be particularly susceptible or clearly associated with the site. However, they have high societal and policy importance. Because of the particular societal value placed on these organisms, as reflected in the Endangered Species Act, the end point properties for these species are any toxic effect that would reduce survivorship or fecundity of individuals of listed species that currently occur on the OU or may occur in the future. State and federally listed T&E species that occur in the region are listed in Appendix J. Their treatment in this assessment are as follows:

   - Terrestrial plants are not currently exposed to contaminants in this OU. Because there is no basis for estimating risks to particular plant species and because the available plant toxicity data for effects on individual plants, the results of the assessment of dredge spoil phytotoxicity (Sect. 6.7) are applicable to T&E plants.

- Four species of fish-eating birds are listed (bald eagle, double-crested cormorant, black-crowned night heron, and osprey). Of these, the osprey was chosen for specific assessment because it nests in the OU (Sect. 6.4). The others are assumed to be represented by the osprey.

- None of the T&E terrestrial birds have food habits that would significantly expose them to contaminants in the OU.

- Two T&E bat species occur in the area. The gray bat was chosen to represent this group because if forages over water and therefore is likely to be more heavily exposed to contaminants than the others (Sect. 6.5).

- The only T&E mammal other than bats with food habits that could lead to significant exposure is the river otter. Otters do not currently occur on the OU but are expected to reoccupy the area in the future.

- Several T&E species of riverine molluscs (mussels and snails) are listed. These species are assumed to be represented by the assessment of risks to the benthic invertebrate community, particularly by the toxicity tests with the native mussel *Anodonta imbecillis* (Sects. 3.4.5 and 6.3).

- A state listed aquatic turtle (Cumberland slider) and salamander (hellbender) may occur in the OU. Toxic effects on these species can not be assessed with existing data. Therefore it must be assumed that the assessment of risks to fish will serve to protect these species as well.

In addition to defining the assessment end points in terms of environmental entities (e.g., the fish community) and a properties of those entities (e.g., species richness), it is necessary to identify a level of effect on those properties. The level of effect is used to help determine how much sampling is needed during the DQO process and to provide a benchmark for comparison of estimates of effects obtained through different lines of evidence. A 20% or greater reduction in one of the end point properties measured in the field or a 20% reduction in survivorship, growth, or reproduction in a toxicity test is considered to be potentially significant. The figure 20% used in the definition of ecological assessment end points is based on an analysis of U.S. EPA and Tennessee regulatory practice; it was adopted by the FFA parties in a DQO process (Ashwood et al. 1994) and incorporated into the plan for ecological risk assessment for the ORR (Suter et al. 1994).

6.1.3.2 Measurement end points

Three basic types of effects data are potentially available to serve as measurement end points: results of biological surveys, toxicity tests performed on media from the CR/PC OU, and toxicity test end points for chemicals found in the CR/PC OU. Measurement end points are presented below for each assessment end point.

Fish. The following measurement end points were used for the fish assessment.

Biological survey data. (a) Species richness and abundance from TVA reservoir surveys and fish community surveys conducted for this assessment. These measurement end points are assumed to be direct estimates of that assessment end point.

Biological indicators data. (b) Frequencies of gross pathologies are a direct measure of one aspect of the assessment end point. (c) Measures of fish fecundity in largemouth bass and bluegill provide an indication of the potential contribution of reproductive toxicity to community effects. (d) Measures of the levels of physiological and histological condition in largemouth bass and bluegill help to confirm that exposures have occurred and may suggest mechanistic connections between exposure and effects on the fish community.

Media toxicity data. (e) Reductions in growth and survivorship of larval fathead minnows and in fecundity and survivorship of *Ceriodaphnia dubia (C. dubia)* in 7 day tests of ambient water. (f) Reductions in hatching and larval survival and increases in terata in Japanese medaka (*Oryzias latipes*) eggs and larvae exposed to ambient water from shortly after fertilization to 48 hours post-hatch. (g) The *Daphnia magna (D. magna)* test of water overlying ambient sediments (discussed below) may be indicative of toxicity to epibenthic fish or to early life stages of benthic spawning fish. Responses that are statistically significantly different or are inhibited by 20% or greater relative to control or reference waters are assumed to be indicative of waters that are toxic to fish.

Single chemical toxicity data. (h) Chronic toxicity thresholds for freshwater fish expressed as chronic EC20s or Chronic Values (CVs). These test end points correspond to the assessment end point for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of CR/PC OU species, and exceedence of the CVs and EC20s are assumed to correspond to 20% or greater reductions in abundance, with some uncertainty.

Benthic invertebrates. The following measurement end points were used for the benthic invertebrate assessment.

Biological survey data. (a) Species richness and abundance of benthic invertebrates reported from the TVA reservoir surveys and surveys performed by the TVA for this assessment. These measurement end points are assumed to be direct estimates of the assessment end point for this community.

Media toxicity data. (b) Reductions in survival of an amphipod [*Hyalella azteca (H. azteca)*] in 10 day exposures to whole sediment. (c) Reductions in survival of a cladoceran (*C. dubia*) in 48 hour exposures to sediment pore water. (d) Reductions in survival and fecundity of another cladoceran (*D. magna*) in 12 to 14 day exposures to water overlying sediment. Responses that are statistically significantly different or are inhibited by 20% or greater relative to reference sediments are assumed to be indicative of sediments that are toxic to benthic biota.

Single chemical toxicity data. (e) Chronic toxicity thresholds for freshwater invertebrates expressed as chronic EC20s or CVs. These test end points correspond to the assessment end point for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of CR/PC OU species, and exceedence of the CVs and EC20s are assumed to correspond to 20% reductions in population abundance or greater, with some uncertainty. (f) Toxic concentrations in ambient sediments reported by the State of Florida. Two types of values were extracted from that data set. First, thresholds for modification of benthic invertebrate communityproperties based on co-occurrence analyses, which are assumed to correspond to the assessment end point. Second, thresholds for lethality in toxicity tests of contaminated sediments, which are also assumed to correspond to the assessment end point effect, but with greater uncertainty due to the extrapolation to the field.

Piscivorous wildlife. The following measurement end points were used for the piscivorous wildlife assessment.

Biological survey data. (a) Reproductive success of great blue herons at rookeries on Poplar Creek embayment and in Melton Hill Reservoir relative to reference rookeries. Assuming that each colony constitutes a population, this is a direct measure of the assessment end point for avian piscivores.

Media toxicity data. (b) Reproductive success, survivorship, and pathologies in mink fed diets containing various proportions of fish from Poplar Creek embayment. The reproductive success and survivorship test end points are assumed to correspond to effects on individual mink with implications for populations. An extrapolation must be made to populations if effects on individuals are estimated to occur.

Single chemical toxicity data. (c) Chronic toxicity thresholds for contaminants of concern in birds and mammals with greater weight given to data from long-term feeding studies with wildlife species. Preference was given to tests that included reproductive end points. After allometric scaling for the end point species, these test end points are assumed to correspond to effects on individuals that could result in exceedence of the population-level assessment end point. An extrapolation must be made to populations if effects on individuals are estimated to occur.

Flying insectivorous wildlife. The following measurement end point was used for the flying insectivorous wildlife assessment. No biological survey data or media toxicity data end points were used.

Single chemical toxicity data. (a) Chronic toxicity thresholds for contaminants of concern in birds and mammals with greater weight given to data from long-term feeding studies with wildlife species. Preference was given to tests that included reproductive end points. After allometric scaling for the end point species, these test end points are assumed to correspond to effects on individuals that could result in exceedence of the population-level assessment end point. An extrapolation must be made to populations if effects on individuals are estimated to occur.

Terrestrial plants. The following measurement end point was used for the terrestrial plants assessment. No biological survey data or media toxicity data end points were used.

Single chemical toxicity data. (a) EC20s for growth or production of vascular plants or equivalent chronic toxicity thresholds for contaminants of concern in soil. These test end points are assumed to correspond to the assessment end point for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of species that would colonize Clinch River dredge spoil, exceedence of the test end points is assumed to correspond to 20% reductions in abundance or productivity, with some uncertainty, and a distinct plant community is assumed to occur on a spoil disposal area.

Terrestrial wildlife. The following measurement end point was used for terrestrial wildlife assessment. No biological survey data or media toxicity data end points were used.

Single chemical toxicity data. (a) Chronic toxicity thresholds for contaminants of concern in mammals with greater weight given to data from long-term feeding studies with wildlife species. Preference was given to tests that included reproductive end points. After allometric scaling for the end point species, the eastern cottontail, these test end points are assumed to correspond to effects on individuals that could result in exceedence of the population-level assessment end point. Since a distinct population is assumed to occur on a spoil disposal area, exceedence of the benchmarks is assumed to correspond to significant effects.

Threatened and endangered species. The following measurement end points were used for the threatened and endangered species assessment.

Biological survey data. Except for osprey reproduction, no surveys of abundance or production of other T&E species were conducted. (a) The reproductive success of osprey nesting in the OU is assumed to be a direct measure of the assessment end point for that species. (b) The surveys of taxonomically and trophically related species described above are used as evidence to help estimate risks to the T&E species.

Media toxicity data. (c) T&E species of aquatic animals, benthic invertebrates, or piscivorous mustellids are assumed to be represented by the test species used in the water, sediment, and fish tissue toxicity tests as described above.

Single chemical toxicity data. (d) The same chronic toxicity thresholds for contaminants of concern in invertebrates, fish, birds and mammals used as measurement end points for other species are used with the T&E species but are interpreted in the risk characterization so as to provide the higher level of protection specified in the assessment end point.

6.1.4 Conceptual Models

Two baseline cases are assessed in the RI (Chap. 1). The current baseline case is the state of the CR/PC OU as it now exists and would continue to exist if no remedial actions are taken and the system is not significantly altered. That is, contaminants occur in the sediments and enter the OU from the ORR and from upstream, but no dredging is allowed without authorization. The hypothetical future dredging case involves unregulated dredging of sediments and disposal of the spoils on land. The ecological conceptual models presented here elaborate on those cases by defining the ecological receptors and their modes of exposure.

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

A conceptual model of exposure of the terrestrial food web in the dredging case is depicted in Fig. 6.2. It is assumed that ecological succession is allowed to occur on the spoil fill. Plants take up contaminants directly from the spoil. Herbivorous wildlife consume the plants, and incidentally ingest some spoil with their food or during grooming.

6.1.5 Organization of the Ecological Risk Assessment

The risks of chemicals to each of the ecological risk assessment end points in the current baseline case are assessed separately (Sects. 6.2–6.5). Each includes an exposure assessment, effects assessment, risk characterization, including characterization of uncertainty. Risks from radionuclides are discussed separately (Sect. 6.6) as are risks from unregulated dredge spoil disposal (Sect. 6.7). Finally, ecological risks are summarized and compared to human health risks (Sect. 6.8). The components of the assessment are explained below.

Exposure assessment characterizes the distribution in space and time of the concentrations of contaminants to which organisms are exposed. Risk from undetected chemicals are not assessed, as instructed by the FFA parties (September 1993). Exposure calculations are performed for each subreach. (See Sect. 3.1.1 and Fig. 3.1 for descriptions of the reaches and subreaches.) The concentrations used in the exposure assessment are summarized in Chap. 3.

Effects assessment characterizes the evidence concerning effects of contaminant exposure. The principle lines of evidence concerning effects are biological survey data that indicate the actual state of the receiving environment, media toxicity data which indicate whether the contaminated media are toxic under controlled conditions, bioindicator data which are biochemical and histological indications of the potential mechanisms and causes of effects, and single chemical toxicity data which indicate the toxic effects of the concentrations measured in site media.

Risk characterization is the phase of risk assessment in which the information concerning exposure and the information concerning the potential effects of exposure are integrated to estimate risks (the likelihood of effects given the exposure). Risk characterization in ecological risk assessment is performed by a weight-of-evidence analysis. Procedurally, the risk characterization is performed for each assessment end point by (1) screening all measured contaminants against toxicological benchmarks and background concentrations, (2) estimating the effects of the contaminants retained by the screening analysis, (3) estimating the toxicity of the ambient media based on the media toxicity test results, (4) estimating the effects of exposure on the end point biota based on the results of the biological survey data, (5) logically integrating the lines of evidence to characterize risks to the end point, and (6) listing and discussing the uncertainties in the assessment.

6.1.6 Risk Characterization Techniques

Three risk characterization techniques are used to apply literature toxicity information to chemical concentrations measured in CR/PC media. Because all three techniques are common to all end points, they are explained here. The techniques are calculation of hazard quotients (HQs), calculation and plotting of sums of toxic units (ΣTU), and comparison of distributions of exposure concentrations to distributions of effects.

HQs are used in the screening phase of the risk assessment to help identify COPECs. They are quotients of the ratios of exposure concentrations divided by ecotoxicological benchmark concentrations. HQs greater than one indicate that a chemical is a COPEC for a particular medium and end point, unless other information such as concentrations in upstream samples indicate that it should not be of concern. Because the intent of this analysis is to screen out only chemicals that are clearly not of concern, conservative estimates of exposure [e.g., maxima or 95% upper confidence limits (UCL)] and conservative benchmarks (e.g., NOECs) or suites of benchmarks are used.

After the screening, the COPECs are compared to each other and their distributions across the reaches is examined. Since the relative importance of COPECs is a function of their potential toxicity rather than their concentration, toxicity normalized concentrations or toxic units (TUs) are calculated. This is a common technique for dealing with exposures to multiple chemicals by expressing concentration relative to a standard test end point (Finney 1971). TUs are quotients of the concentration of a chemical in a medium divided by the standard test end point concentration for that chemical. They are similar to HQs except that a common test end point is used rather than conservative benchmarks because TUs are used for comparative purposes rather than to draw conclusions. The expression of concentration and the test end point vary among media; for water they are the upper 95% confidence limit concentration and the 48 hour EC50 for *D. magna*. If the TU for a chemical equals one, the

CLINCH RIVER CONCEPTUAL MODEL: AQUATIC FOOD WEB

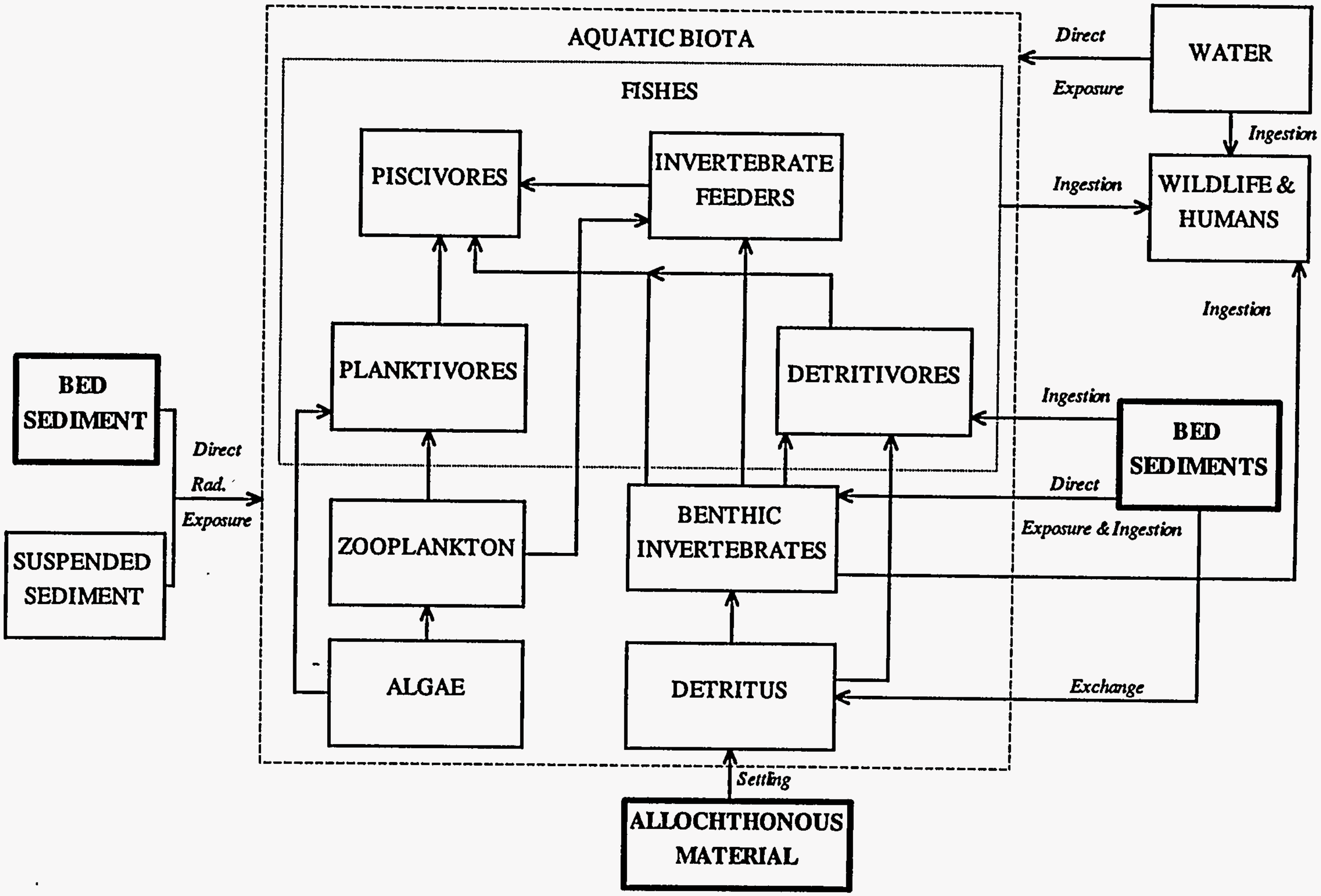


Fig. 6.1. Conceptual model of contaminant sources, transport, and exposure in the aquatic food web. Dark-lined boxes represent sources that may be remediated. Arrows represent routes of transport and exposure, described by italic labels. Unlabeled arrows represent food web transfers by ingestion.

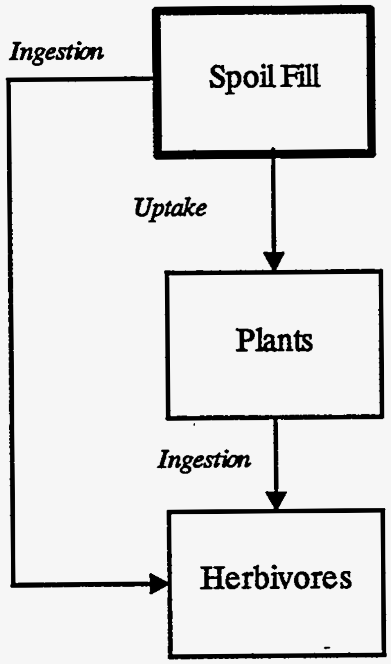


Fig. 6.2. Conceptual model of contaminant sources, transport, and exposure in the terrestrial food web for the baseline dredging case. Dark-lined boxes represent sources that may be remediated. Arrows represent routes of transport and exposure, described by italic labels.

interpretation is that the aquatic community in that reach is exposed to a conservatively estimated average concentration sufficient to kill or immobilize *Daphnia* within 48 hours. The chemicals that constitute a major component of toxicity (i.e., TUs > 0.01) are plotted for each negative reference reach and OU subreach for water, sediment, and wildlife intake (e.g., Fig. 6.3). The height of the plot at each subreach is the ΣTU for that medium and subreach. This value can be conservatively interpreted as the total toxicity-normalized concentration and therefore as a relative indication of the toxicity of the medium in that subreach. In addition, the ΣTU is commonly assumed to estimate the absolute toxicity of the medium. That is, if the ΣTU equals one, then the end point effect (e.g., *Daphnia* acute lethality) will occur. This will be the case if all of the chemicals have the same mode of action. For heterogeneous chemical mixtures it is a likely to be a conservative assumption because combined toxic effects of chemicals in environmental samples have been found to be additive or less than additive (Alabaster and Lloyd 1982). Because the test end points are chosen for their consistency rather than their relationship to an assessment end point, the plots of TUs are heuristic, providing an indication of the relative toxicity of sites and the relative contributions of chemicals to that toxicity.

Inferences about the risk posed by the COPECs is based on the distribution of concentrations relative to the distribution of effects. Distributions provide a better basis for inference than point estimates because they allow consideration of variance in concentration over space or time and of sensitivity across species, measures of effects, media properties, or chemical forms. In all cases, risk is a function of the overlap between the exposure and effects distributions, but the interpretation depends on the data that are used.

- In the case of exposure of fish to chemicals in water, the exposure distributions are distributions of aqueous concentrations over time and the effects distributions are distributions of sensitivities of species to acutely lethal effects (e.g., LC50s) and chronically lethal or sublethal effects (CVs). Overlap of these two distributions indicates the approximate proportion of the time when aqueous concentrations of the chemical are acutely or chronically toxic to a particular proportion of aquatic species. For example, 10% of the time copper concentrations in subreach 4.01 are at levels chronically toxic to 80% of aquatic animals and acutely toxic to 30% of aquatic invertebrates (Fig. F2.2).
- In the case of exposure of benthic invertebrates to sediment pore water, the exposure distributions are interpreted as distributions over space since sediment composition varies little over the period in which samples were collected, but samples were distributed in space within reaches. The effects distributions are the same as for surface water. Therefore, overlap of the distributions indicates the proportion of locations in the reach where concentrations of the chemical in pore water are acutely or chronically toxic to a particular proportion of species. For example, copper concentrations in sediment pore water from more than 90% of locations in subreach 4.04 are below chronically toxic concentrations for more than 90% of aquatic animal species (Fig. F3.6).
- In the case of exposure of benthic invertebrates to chemicals in whole sediment, the exposure distributions are, as with pore water, distributions in space within reaches. Two effects distributions are presented for each sediment COPEC: a distribution of concentrations reported to be thresholds for reductions in benthic invertebrate community parameters in various locations, and a distribution of concentrations reported to be thresholds for lethal effects in toxicity tests of various sediments. If we assume that the effects data set are drawn from studies of a random sample of sediments so that the CR/PC sediments can be assumed to be a random draw from the same distribution, and if we assume that the two types of reported effects are independent estimates of the threshold for significance for this end point, then the effects distributions can be treated as distributions of the probability that the chemical causes significant toxic effects on the CR/PC benthic community. Overlap of the exposure and effects distributions represents the probability of significant toxic effects on benthic communities in a given proportion of locations in a reach. For example, copper concentrations in whole sediment from <20% of locations in subreach 3.02 are above the concentration at which there is a 50% likelihood of toxic effects on benthic invertebrates (Fig. F3.9b).
- In the case of wildlife exposure to chemicals in food and water, the exposure distributions are distributions of total intake rate of the chemical across individuals in the populations. The distributions are obtained by the Monte Carlo simulation using the distributions of observed concentrations in water and various food items. If we assume that the members of a population occurring in each modelled area independently sample the water and food items, then the proportions of the exposure distributions represent estimates of the proportion of a population receiving a particular intake rate. In keeping with practice in wildlife toxicology, the effects are

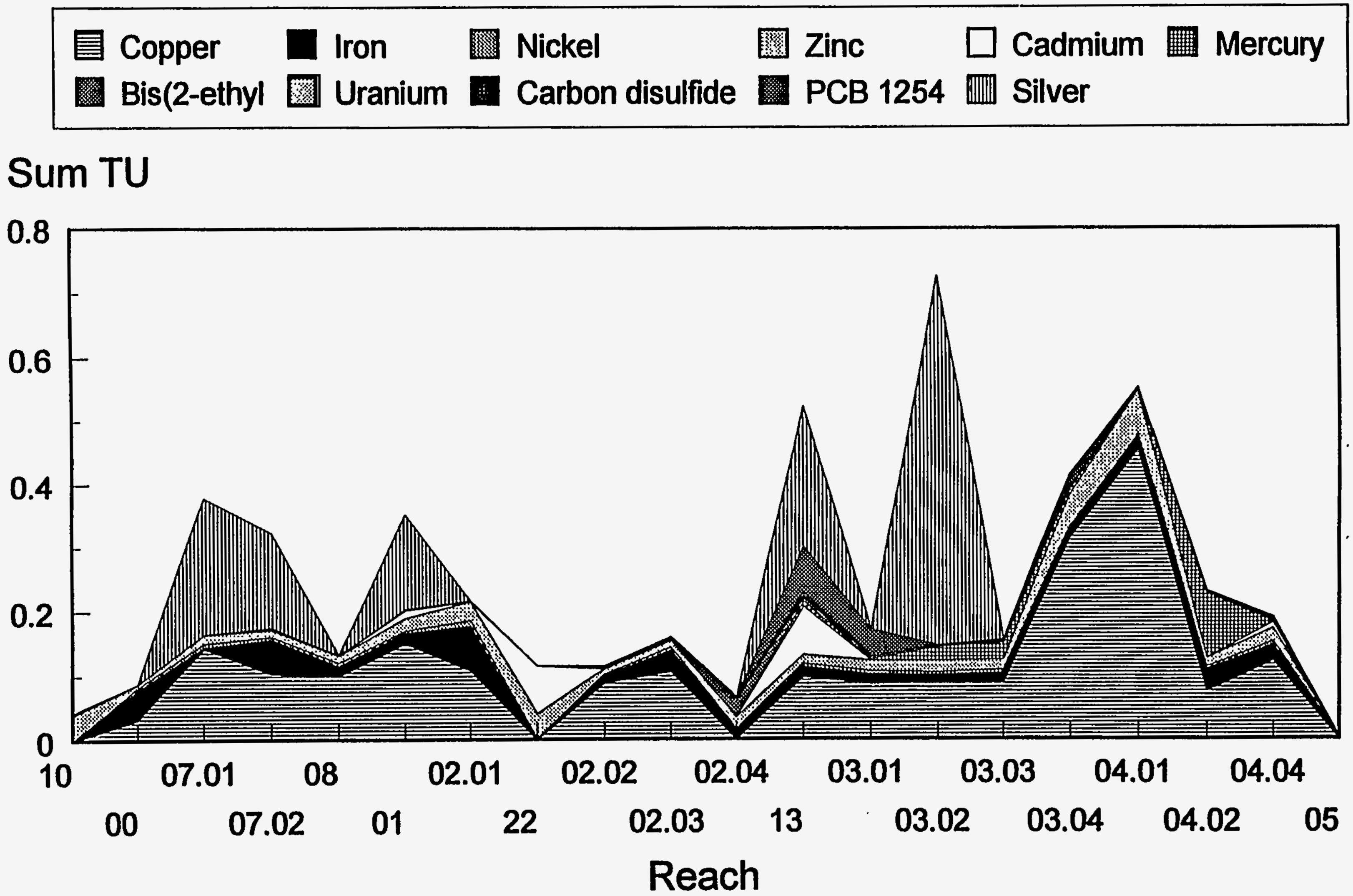


Fig. 6.3. Plot of concentration of contaminants of potential ecological concern normalized to toxic units (TU). The upper contour of the graph is the sum of TU for the reaches.

treated as point estimates (i.e., NOAEL and LOAELs). Therefore, the intersection of these points with the exposure distributions represent estimates of the proportion of the exposed population with exposure levels less than the NOAEL and LOAEL. For example, <1% of the members of a rough-winged swallow colony located in subreach 3.02 would receive a dose rate greater than the LOAEL (Sect. 6.5).

- The case of T&E wildlife is a little different from that of other wildlife species. Risks to these species are assessed at the organism level, so the exposure models represent exposure of individuals within their foraging areas. Therefore, the exposure distribution is a distribution of the likelihood of various exposure rates to individuals foraging in the specified areas. Effects are specified by the same point estimates as with other wildlife. Therefore, the intersection of the point estimates of effects with the exposure distributions represent the likelihood that an individual will have an intake rate less than the NOAEL and LOAEL. For example, the likelihood that the osprey foraging exclusively in subreach 3.02 will receive a dose rate greater than the LOEL is >50% (Sect. 6.4).

These risk characterization techniques apply to one line of evidence, the concentrations of chemicals in ambient media. Other interpretation techniques applied to other lines of evidence are more specific to the end point, so they are not summarized here. However, for all end points the risk characterization ultimately depends on weighing of all of the lines of evidence.

6.1.7 Background, Reference, and Operable Unit Reaches

OU reaches are those that are potential candidates for remediation based on the findings of this RI. They are the Clinch River adjacent to and downstream of the ORR (reaches 1, 2, & 4) and the Poplar Creek embayment (reach 3). Interpretation of the chemical concentrations and biological characteristics of an OU are made by comparison to reference reaches. These include positive reference reaches which are identified *a priori* to be sources of contamination. They are the streams draining the ORR (reaches 20-22). Positive reference reaches are used to indicate where contaminants in the OU may have come from and what sorts of biological effects occur in more highly contaminated areas. There are also negative reference reaches that are upstream of the OU on the Clinch River (reach 0) and Poplar Creek (reach 13). These reaches were believed *a priori* to be relatively uncontaminated and are at least not contaminated by DOE releases. They are used to indicate what chemicals in the OU may have come from upstream and what sorts of biological conditions exist in the absence of exposure to DOE releases. It should be noted that the negative reference reaches do not constitute background conditions. Both the CR/PC are influenced by chemical releases and physical disturbances associated with municipalities, agriculture, and other activities in their watersheds. As a result, chemical concentrations in the OU reaches can not be screened against background.

The reaches are described in Sect. 3.1.1. and a map is available in Fig. 3.1.

6.2 RISKS TO FISH

6.2.1 Exposure Assessment for Fish

6.2.1.1 Aqueous exposures

Fish are exposed primarily to contaminants in water. Contaminants in water may come from upstream aqueous sources including the ORR, exchange of materials between the surface water and contaminated sediments, or exchange of contaminants between the biota and the water column. The consensus of the scientific community and of the EPA Office of Water is that aquatic biota should be assumed to be exposed to the dissolved fraction of the chemicals in water, because that is the bioavailable form (Prothro 1993). However, the EPA Region IV, prefers to use total concentrations as conservative estimates of the exposure concentration. Therefore, both dissolved phase and total concentrations of metals are used in the exposure assessment for fish.

Because water in the OU is likely to be more variable in time than space, due to the rapid replacement of water, the mean water concentration within a subreach is an appropriate estimate of the chronic exposure experienced by fishes. The upper 95% confidence bound on the mean is an appropriately conservative estimate of this exposure for use in the contaminant screening. However, the distribution of observed concentrations is used to estimate risks.

Some fish spend most of their lives near the sediment and the eggs and larvae of some fish (particularly sunfish and black bass) develop at the sediment water-interface. These epibenthic species and life stages may be more highly exposed to contaminants than is suggested by analysis of samples from the water column. The water samples collected 1 m above the sediments by the contaminant remobilization task (Sect. 3.3) provide an estimate of this exposure. However, those samples were not found to contain significantly higher chemical concentrations (Sect. 3.3), so they were combined with the other samples. Another exposure that is relevant to this route is the exposure of *D. magna* to water above sediment in one of the sediment toxicity tests (Sect. 3.3).

6.2.1.2 Fish body burdens

Although nearly all toxicity data for fishes is expressed in terms of aqueous concentrations, fish body burdens potentially provide an exposure metric that is more strongly correlated with effects (McCarty and Mackay 1993). This is particularly likely to be the case for chemicals that bioaccumulate in fish and other biota to concentrations greater than in water. For such chemicals dietary exposure may be more important than direct aqueous exposures and concentrations that are not detectable in water may result in high body burdens in fish. Three contaminants that were determined to be COPECs in prior assessments of the CR/PC OU bioaccumulate in that manner: mercury, PCBs, and selenium. Only mercury and PCBs are considered in this assessment, because selenium was not found to be significantly elevated in the data used in the RI (Chap. 3). Since the individual body burden measurements correspond to an exposure level for an individual fish, the maximum value is used for screening purposes and the risk estimate is based on the distribution of individual observations for each measured species. Measurements were performed on muscle (fillet), carcass (residue after filleting), or whole fish. Since whole fish measurements are most commonly used in the literature, whole fish concentrations either measured directly or reconstructed from fillet and carcass data are used (Sect. 6.4).

6.2.2 Effects Assessment for Fish

6.2.2.1 Single chemical aqueous toxicity

The screening benchmarks for aquatic biota are taken from Suter and Mabrey (1994). Because there are no standard screening benchmarks, sets of alternative benchmarks (described in Table 6.1) were calculated for each chemical. The benchmark preferred by the regulatory agencies is the chronic National Ambient Water Quality Criteria (NAWQC), but they are available for relatively few industrial chemicals. Secondary chronic values (SCV), which are conservative estimates of chronic NAWQC, were calculated for chemicals that do not have NAWQC. Other benchmarks are included to provide greater assurance of detecting all COPECs.

NAWQC that are functions of water hardness are corrected for site-specific conditions. For purposes of screening, conditions were chosen that would constitute reasonable maximum toxicity, defined as conditions that would persist for seven days. This was a hardness of ~100 mg/L. The minimum, mean, and maximum hardness in the Clinch River are 105, 122, and 132 and in Poplar Creek are 68, 114, and 162.

Toxicity profiles are presented in Appendix F for COPECs. The toxicity profiles summarize the existing toxicity information for each chemical including concentrations causing acute lethality and chronic lethal and sublethal effects, and physical-chemical conditions that modify toxicity. For chemicals with hardness-dependent criteria, test end points were corrected to 100 mg/L hardness using the EPA's formula unless the test water was between 80 and 120 mg/L hardness.

6.2.2.2 Fish body burdens

There are no standard benchmarks for effects on fish of internal exposures. The body burdens associated with effects in toxicity tests and field studies and body burdens found at other sites are discussed below. To be consistent with EPA practices in calculating CVs, thresholds for toxic effects are expressed as geometric means of body burdens measured at the NOEC and LOEC. Only mercury and PCBs were considered because they are the COPECs known to significantly bioaccumulate in fish through dietary exposure.

Mercury. Estimates of body burdens of mercury that constitute a threshold for toxic effects on fish range from 0.66 mg/kg to 5.6 mg/kg wet weight. The lower figure is the geometric mean of body burdens in fathead minnows showing significant reproductive effects and those of fish showing no significant effects in a chronic test of mercuric chloride that included dietary exposures (Snarski and Olson 1982). The higher figure is the geometric mean of body burdens associated with reproductive effects and no significant effects in exposures of brook trout to methyl mercury (McKim et al. 1976). Mercury concentrations in sunfish from streams in the vicinity of the ORR that were not mercury contaminated ranged from 0.06 to 0.11 mg/kg wet weight (Southworth et al. 1994). Concentrations in reservoirs that do not receive ORR effluents ranged from 0.03 to 0.13 mg/kg wet weight with the highest concentrations in Norris Reservoir (Southworth et al. 1994).

PCBs. Estimates of body burdens of PCBs that constitute thresholds for toxic effects in fish are presented in Table F1.11. They are presented on a whole-body basis, and, when possible, on a lipid normalized basis. There is a very large range of values that have been reported to constitute thresholds for toxic effects in toxicity tests, including one incredibly low value for lake trout. When possible, studies of the species (or closely related species) in which body burdens were measured will be used.

6.2.2.3 Ambient water toxicity

Methods and results of toxicity tests of ambient water are presented in Sect. 3.7. The tests employed include the standard 7-d tests of growth and survival in fathead minnow larvae and fecundity and survival of *C. dubia*, and an early life stage test with Japanese medaka eggs and larvae. Because these subchronic tests were developed for testing effluents and ambient waters rather than pure chemicals, their relationship to levels of contaminant exposure must be related to the patterns of contamination of the OU and reference reaches and related to effects levels for individual contaminants in other tests (Sect. 6.2.2.1).

Table 6.1. Descriptions of the ecotoxicological screening benchmarks for aquatic biota. More details are presented by Suter and Mabrey (1994)[a].

Benchmark	Abbreviation	Description
Acute national ambient water quality criteria	NAWQC_ACU	Current national criteria for protection of aquatic life from lethal effects in episodic exposures.
Chronic national ambient water quality criteria	NAWQC_CHR	Current national criteria for protection of aquatic life from lethal and sublethal effects in extended exposures. Criteria for uses of aquatic life (i.e., fish consumption) are not included.
Secondary acute value	S_ACU_V	Values estimated with 80% confidence to not exceed the unknown acute NAWQC. Used when data are inadequate to calculate the acute criterion.
Secondary chronic value	S_CHR_V	Values estimated with 80% confidence to not exceed the unknown chronic NAWQC. Used when data are inadequate to calculate the chronic criterion.
Lowest chronic value for fish	LCV_FISH	The lowest value, from acceptable fish chronic toxicity tests, of the geometric mean of the lowest observed rffect voncentration (LOEC) and the no observed effect concentration (NOEC).
Lowest chronic value for daphnids	LCV_DAPH	The lowest value, from acceptable daphnid chronic toxicity tests, of the geometric mean of the LOEC and the NOEC.
Lowest chronic value for non-daphnid invertebrates	LCV_ND	The lowest value of the geometric mean of the LOEC and the NOEC from acceptable chronic toxicity tests of nondaphnid invertebrate species.
Lowest plant value	LCV_AQPL	The lowest value from an acceptable daphnid chronic toxicity test of the geometric mean of the LOEC and the NOEC.
Lowest test EC20 for fish	LTV_FISH	The lowest value, from acceptable fish chronic toxicity tests, of the lowest concentration causing at least a 20% reduction in the weight of young per female or the weight of young per egg.
Lowest test EC20 for daphnids	LTV_DAPH	The lowest value from an acceptable daphnid chronic toxicity test of the lowest concentration causing at least a 20% reduction in the product of survivorship, growth, and fecundity.

[a]Suter, G. W., II., and J. B. Mabrey. 1994. *Toxicological Benchmarks for Screening of Potential Contaminants of Concern for Effects on Aquatic Biota: 1994 Revision.* ES/ER/TM 96/R1. Oak Ridge National Laboratory, Oak Ridge, Tenn.

6.2.2.4 Fish community survey

Analysis of fish population and community survey data provides a direct measure of impacts of human activities on aquatic ecosystems. These data can provide an additional line of evidence relating to potential impacts of DOE operations on the Clinch River. Two sources of data were available for this assessment: (1) data from TVA's Reservoir Vital Signs Monitoring Program, and (2) data from the Clinch River System collected for the ER Program using the same sampling techniques. These data are described in Appendix F8.

6.2.2.5 Fish bioindicators

Two of the classes of bioindicators discussed in Sect. 3.5.4 are sufficiently associated with effects on fish to be used as measures of effects and show a sufficiently large and consistent variance relative to exposure of fish to the DOE sources to be of interest for estimating risks. They are histopathology and reproductive indicators. The relationship of these indicator responses to levels of exposure to the contaminants are poorly specified, so they must be related to the patterns of contamination of the OU and reference reaches in the risk characterization.

6.2.3 Risk Characterization for Fish

6.2.3.1 Single chemical aqueous toxicity

Screening aqueous exposures against benchmarks. All chemicals detected in whole or filtered water were screened against benchmarks. This was done by dividing the 95% upper confidence bound (UCB) concentration by each of the aqueous screening benchmarks (Table F2.1). Chemicals that exceeded any benchmark (i.e., the quotient of UCB/ASB was >1) were examined further to determine whether they were credible COPECs (Table 6.2). Chemicals that exceeded benchmarks were eliminated as COPECs if they met any of the following criteria.

- They are metals, do not exceed benchmarks in filtered samples, and are not associated with an identifiable source. Such metals are deemed to be associated with background particulate material which is not bioavailable.

- They exceed a highly conservative benchmark and the exceedence occurs at all or nearly all locations and not in particular association with an identifiable source. The benchmark is deemed to be effectively below background and too conservative for this use.

- They are detected in only one or two samples per reach, those detects constitute 5% or less of the total samples, the detections are only in unfiltered water, and they are not associated with any identifiable source. Such detections are deemed to be anomalies and not relevant to risks to aquatic biota.

A second step in the screening process involved determining the relative contribution of the COPECs to the distribution of toxicity across reaches. This was done by converting the concentrations to equivalent toxicity normalized concentrations and then plotting the values for each reach or subreach (Fig. 6.3, see also Fig. 3.1 for locations of subreaches). The standard toxic value used in the normalization is the *D. magna* EC50, the only test end point that is available for all of the aqueous COPECs. Concentration in TUs was calculated as

$$TU = UCB/EC50$$

If the combined toxic effects of the COPECs are concentration additive, then, if the sum across chemicals of TU is >1, one would expect acute lethality to *Daphnia*. ΣTU values >0.2 are suggestive of chronic toxic effects. The ΣTU graph reveals that toxicity is likely to be high in subreaches 3.04 and 4.01, primarily due to copper. Silver, is detected only in reaches 7, 1, and 13 and in subreach 3.02, but, because the detection limit is high relative to its toxicity, where it is detected it is a large contributor to ΣTU. The relatively high total toxicity in reach 13 in Poplar Creek upstream of the reservation and the diversity of chemicals contributing to that toxicity, is a surprising result.

Exposure effects profiles for aqueous exposures. For each COPEC, the distribution of observed concentrations is compared to the distribution of toxic concentrations for aqueous fish and invertebrates. If sufficient toxicity data exist, the empirical distribution functions are presented graphically. Toxicity information is drawn from the toxicity profiles (Appendix F).

Bis(2-ethylhexyl)phthalate. The highest concentrations and greatest frequencies of detections of bis(2-ethylhexyl)phthalate are in reach 13. However, the distribution of concentrations there and in other reaches is poorly defined because detections are rare and often occur at concentrations equal to or less than the limits of detection in other samples. The maximum concentration in filtered water from reach 13 (0.23 mg/L) exceeds the CV for rainbow trout and the two highest concentrations exceed the species mean CV for *D. magna*. The maximum whole water concentration is lower than filtered water, exceeding only the *D. magna* CV. The only other sample exceeding reported toxic concentrations is the single detect from subreach 2.04 which barely exceeds the *D. magna* CV. Hence the risk to aquatic biota in the OU from bis(2-ethylhexyl)phthalate appears to be negligible.

Carbon disulfide. Carbon disulfide was seldom detected in this study. The highest observed concentration, which is from reach 13, is approximately a factor of 100 lower than the lowest observed toxic concentration. The exceedence of the SCV in the screening assessment is attributable to the large safety factors used with minimal data sets. Hence, there appears to be negligible risk to aquatic biota from carbon disulfide

Cadmium. The distributions of ambient cadmium concentrations and aqueous test end points are shown in Fig. F2.1. The ambient concentrations are dissolved phase concentrations in the subreaches in which Cd was detected (22, 13, 4.04, and 1). Because Cd was detected in only one sample from each subreach, the single concentrations are depicted as 100th percentile points. The highest Cd concentrations exceeded more than half of CVs (CVs for two cladoceran genera) and the chronic NAWQC, but these were the Poplar Creek upstream reach (13) and the White Oak Creek embayment (22) which is a source, not a component of the OU. The two detected values in the OU (reach 1 and subreach 4.04) were below the NAWQC but above one cladoceran CV in 4.04 and both cladoceran CVs in 1. Because Cd was not detected more than once in any reach and the detected values in the OU barely exceed CVs for members of a highly sensitive family of zooplanktors, it appears to pose negligible risk to aquatic biota.

Copper. The distributions of ambient copper concentrations and aqueous test end points are shown in Fig. F2.2. The ambient concentrations are dissolved phase concentrations in the subreaches with potentially hazardous levels of Cu (3.04 and 4.01). The toxic concentrations are those presented in the ecotoxicity profile (Table F1.2). The ambient concentrations fall into two phases. Concentrations below 0.01 mg/L display a fairly smooth increase suggestive of a log-normal distribution. The upper end of this phase of the distribution (above the 75th percentile of 4.01 and the 80th percentile of 3.04) exceed the lowest CV (a bluntnose minnow CV for reproductive effects).

Table 6.2. Results of screening of chemicals that exceed benchmarks in whole or filtered water for contaminants of potential ecological concern (COPECs)

Chemical	COPEC	Reason for inclusion or rejection
Aluminum	No	Exceeds chronic national ambient water quality criteria (NAWQC) in all reaches and acute NAWQC in 18, 21, and 22 in whole water but no benchmarks in dissolved phase. The exceedences are due to filterable particulate Al which is ubiquitous and not bioavailable.
Barium	No	Exceed the SCV, but no other benchmark, in both whole and filtered water in all reaches. The SCV for barium is apparently below background due to the conservative factors used in its derivation.
Boron	No	Exceeds the LTV Daphnids, but no other benchmarks, in whole and filtered water in all reaches. That benchmark is apparently below the local background due to the compounding of effects across life stages in its derivation.
Cadmium	Yes	Dissolved phase cadmium from 13 and 22 exceeds chronic NAWQC and from 1 and 4.04 exceeds CVs.
Cobalt	No	Detected in only whole samples and in only one sample from 1, 2.02, 18, and 22 and on only 2 out of 18 samples in 21. Deleted because it is rarely detected and never in a bioavailable form.
Copper	Yes	Exceeds several benchmarks including the chronic NAWQC in filtered water from 4.01.
Cyanide	No	Detected in only 1/20 filtered samples from 20 and not in whole water there. That detect is deemed to be an anomaly.
Iron	Yes	Exceeds Daphnid LCV in filtered samples from 0, 2, 4, and 7.
Mercury	Yes	Exceeds fish LCV in filtered water from 4.02 and whole water from 3.04.
Nickel	Yes	Exceeds Daphnid and plant LCVs in filtered water from 2.01, 2.03, 3.01, 3.03, 4.04, 7.01, 8, and 13.
Silver	Yes	Exceeds the fish LCV in filtered water from 1, 3.02, 7.01, 7.02, and 13, and the daphnid LCV in 3.02.
Thallium	No	Not detected in filtered water but detected in a single whole water sample out of 48 samples from 1 at a concentration that exceeds several benchmarks. Deleted because anomalous and not bioavailable.
Uranium	Yes	Exceeds the SCV in filtered water from 2.02, 3.02, 3.03, and 3.04.
Zinc	Yes	Exceeds fish and plant LCVs in filtered water from 4.01.
Bis (2-ethylhexyl) phthalate	Yes	Exceeds multiple benchmarks in filtered water from 1, 2, 3, 4, and 13 and in whole water from 7 and 10.
Carbon disulfide	Yes	Exceeds the SCV in whole water from 2.04 and 3.03.
Aroclor 1254	Yes	Exceeds the plant LCV and SCV in whole water from 3.01, 3.04, and 13.

However, the distributions above the 90th percentile are not continuous with the other points. The break in the curve suggests that some episodic phenomenon causes exceptionally high concentrations. The two points in 4.01 and one in 3.04 that lie above this break exceed ~90% of the CVs, ~30% of the acute values, and both the acute and chronic NAWQCs. These results are suggestive of a small risk of chronic toxicity from routine exposures, but a high risk of short-term toxic effects of Cu during episodic exposures in lower Poplar Creek embayment and the Clinch River immediately downstream. Toxic concentrations were found in no other subreaches.

Iron. A single iron concentration in subreach 7.02 and in reach 13 exceed the chronic NAWQC. However, neither concentration exceeds toxic concentrations except for the study from Dave (1984) which simulated dilution of an acidic effluent. Since the NAWQC is also based on acidic effluents, which are not present in releases to this OU, risks to aquatic biota from iron are deemed to be minimal.

Mercury. The distributions of ambient mercury concentrations and aqueous test end points are shown in Fig. F2.3. The ambient concentrations are dissolved phase concentrations in the subreaches with potentially hazardous levels of Hg (3.02, 3.03, 3.04 and 13). The toxic concentrations are those presented in the ecotoxicity profile (Table F1.3). The ambient concentrations fall into two phases. Concentrations below 0.00001 mg/L display a fairly smooth increase within a relatively small range of concentrations. The upper end of this phase of the distribution (above the 85th to 95th percentile) is well below toxic concentrations. However, the observations above this break point are up to a factor of 100 higher and not continuous with the other points. The break in the curve suggests that some episodic phenomenon causes exceptionally high concentrations. The highest observed concentrations in subreaches 3.02 and 3.03 and reach 13 exceed the CV for fathead minnows and approximately equal the CV for *D. magna* (the only two CVs available for inorganic Hg). These results suggest that routine exposures are not toxic, but there is a small risk of short-term toxic effects of Hg during episodic exposures in Poplar Creek. Mercury concentrations in Poplar Creek upstream of the ORR (reach 13) are routinely lower than those on the ORR, but they are approximately equal to reach 3 extremes during the episodes. Toxic concentrations were found in no other subreaches, including EFPC (subreach 20.1) which is the presumed source.

Nickel. The distributions of ambient nickel concentrations and of aqueous toxicity test end points are shown in Fig. F2.4. Unlike other COPECs, episodic high concentrations of Ni were not detected. Rather, in subreaches 3.03, 3.04, 4.04, 7.02, and reach 13, Ni concentrations display a relatively smooth increase over approximately an order of magnitude with the very highest concentrations exceeding the CV for *C. dubia*, but no other toxic values. The highest concentrations and highest frequency of exceedences of the CV were observed in reach 13, upstream of the ORR. These results suggest a small risk to the fish community in the OU due to Ni.

PCBs. The distributions of ambient PCB concentrations and aqueous test end points are shown in Fig. F2.5. The ambient concentrations are whole water concentrations in the subreaches in which PCBs were detected (13, 3.01, and 3.04). Because PCBs were detected in only one sample from each subreach, the single concentrations are depicted as 100th percentile points. The sum of Aroclor 1254 and 1260 in reach 13 exceeds more than half of the CVs, but that reach is upstream of the ORR. Observed PCB values in water from reach 3 were well below toxic concentrations.

Silver. The distributions of ambient silver concentrations and aqueous test end points is shown in Fig. F2.6. The ambient concentrations are dissolved phase concentrations in the subreaches in which Ag was detected (3.02, 7.01, and reach 13). Because Ag was detected in only one or two samples per reach, only the upper end of the concentrations are displayed; the remainder of the distributions is undefined. One observed concentration in subreach 3.02, which was more than 10 times the second highest concentration, exceeded the acute NAWQC as well as a *D. magna* EC50, LC50s for two fish (speckled dace and mottled sculpin), and both of the available CVs for Ag (*D. magna* and rainbow trout), but the second highest concentration in that reach was below all toxic levels. The single detected Ag concentration in reach 13 and the highest of two detects in subreach 7.01 exceeded the rainbow trout CV. These results suggest that Ag poses little risk to the fish communities of the OU. However, the combination of a detection limit that is barely lower than the lowest CV and the small toxicity data set makes interpretation uncertain.

Uranium. None of the observed uranium concentrations fall within a factor of 100 of the lowest reported toxic concentration (a fathead minnow LC50 of 2.8 mg/L), and all but a few observations were more than a factor of 1000 lower. The exceedence of the SCV in the screening assessment is attributable to the large safety factors used with minimal data sets.

Zinc. The distributions of ambient zinc concentrations and aqueous test end points is shown in Fig. F2.7. The ambient concentrations are dissolved phase concentrations in the subreach with potentially hazardous levels of Zn (4.01). The toxic concentrations are those presented in the ecotoxicity profile (Table F1.7). The ambient concentrations fall into two phases. Concentrations below 0.01 mg/L display a fairly smooth increase suggestive of a log-normal distribution. This phase of the distribution (below roughly the 85th percentile) does not exceed any toxic concentrations. However, the distributions above the 90th percentile are not continuous with the other points. The break in the curve suggests that some episodic phenomenon causes exceptionally high concentrations. The two points that lie above this break exceed ~15% of the CVs (flagfish and *D. magna*), the very lowest acute values for both fish (cutthroat trout LC50) and invertebrates (*Ceriodaphnia reticulata* EC50), and chronic NAWQC. The highest concentration exceeds the acute NAWQC. These results are suggestive of a small risk of short-term toxic effects of Zn to sensitive species during episodic exposures in the Clinch River immediately downstream of Poplar Creek. Toxic concentrations were found in no other subreaches.

6.2.3.2 Single chemical internal toxicity

Mercury. The only fish with body burdens of mercury that fell within the potentially toxic range were largemouth bass in reach 3. When fillet/whole body corrections were applied, the concentration in the most contaminated largemouth bass (1.28 mg/kg) exceeded the 0.66 mg/kg concentration that corresponded to a CV for fathead minnows but was well below the value for brook trout (5.6 mg/kg). The median concentration for largemouth bass and all concentrations for bluegill, gizzard shad, and channel catfish were below 0.66 mg/kg, but shad and catfish were not analyzed for mercury in reach 3. Body burdens in the OU can also be compared to those in EFPC, the primary source of mercury to Poplar Creek and the Clinch River. The fish community at kilometer 13.8 of EFPC has species richness and abundance that approximately equals background. At that site, average mercury concentrations in redbreast sunfish have been in the range of 0.6–1 mg/kg since 1984. Since that range is considerably higher than the median concentration on reach 3 for largemouth bass (0.29 mg/kg) or bluegill sunfish (0.25 mg/kg), we may conclude that mercury accumulation is unlikely to significantly reduce fish community richness or abundance in the OU. That conclusion, and the marginal exceedence of the lowest benchmark based on laboratory testing, suggest that the risk to fish from accumulation of mercury appears to be quite small.

PCBs. PCB concentrations are highest in channel catfish and in certain positive reference reaches: White Oak Creek (22), Tennessee River (18), and Emory River (6). The highest average whole-body concentrations (estimated from fillets) in the OU are from reach 2 with median and maxima of 1.9 and 5.7 mg/kg for Aroclor 1254 and 4.1 and 13.1 mg/kg for Aroclor 1260. A test of dietary exposure of channel catfish to Aroclor 1242 found reduced growth and liver hypertrophy at a body burden of 14.3 mg/kg (Hansen et al. 1976). This concentration is approximately half of the median concentration of total PCBs in reach 22 but much higher than the median in other reaches. It is, however, less than the total PCB concentration in the most exposed individual catfish in reaches 0, 2, 3, 4, 5, and 18. The test was run with Aroclor 1242 which should be less toxic than the more highly chlorinated PCBs found in catfish from the OU. However, fingerlings used in the test may be more sensitive than the adult fish analyzed, and the body burdens of fingerlings are unknown.

6.2.3.3 Ambient water toxicity

Two expressions of the results of aqueous toxicity tests are used, the average magnitude of the effects and the frequency of effects that are significant. Test results are deemed significant if the difference between the response of the organisms exposed to water from the OU and organisms in control or reference water is at least 20% (i.e., biologically significant) or if the likelihood that the difference is due to chance is <5% (i.e., statistically significant). Test results are related to two expressions of exposure, the location of the sample relative to sources and the relationship between the responses and the concentrations of aqueous COPECs in the tested water. Three responses in two tests are potentially indicative of toxicity. Mortality of fathead minnow larvae was frequently more than 20% greater than in control water, *Ceriodaphnia* fecundity was occasionally reduced more than 20% relative to control water, and the mortality and frequencies of abnormalities in medaka embryos were frequently more than 20% greater than in control water. The methods and results of these tests are discussed in Sect. 3.3.7. This section analyzes their implications for risks to fish in the OU.

Fathead minnow. Mortality of fathead minnow larvae in Poplar Creek is elevated relative to controls, but mortality in reach 3 is not appreciably elevated relative to reference subreach 13.06 which is 0.9 miles upstream of any DOE sources (Fig. 6.4). High levels of mortality in Poplar Creek occurred primarily in the April 1994 testing period when water levels were high, suspended particulate levels were high, and hardness was low due to storms (Sect. 3.3). The professional judgement of the individuals who conducted the tests, and who have considerable experience with this test on the ORR, is that the mortality was due to causes other than toxicity, possibly associated with high suspended particulate levels. This judgement is supported by the fact that the proportional increase in fathead minnow mortality relative to controls was statistically significantly positively correlated with total suspended solids levels in samples taken in conjunction with the tests ($\alpha = 0.003$, $r^2 = 0.28$), but not with concentrations of any of the COPECs. However, the water analyses were conducted on only one day, so contaminant concentrations or other conditions on other days of the test do not contribute to the correlations. In sum, these test results do not provide good evidence concerning risks of toxic effects to the fish community of Poplar Creek.

No statistically significant differences in fathead minnow mortality were found in McCoy Branch (reach 7) water relative to either control or reference water, but in one of the two tests of subreach 7.02, there was a 21% increase in mortality relative to controls (accounting for the 50% significant tests in Fig. 6.4) which corresponded to a 14% reduction relative to reference. The facts that the results are marginal and that the greatest potential effect occurred in the subreach furthest from the source suggest that sources in the McCoy Branch watershed are not causing significant toxicity to fathead minnow larvae.

Ceriodaphnia. Fecundity of *Ceriodaphnia* is enhanced by ambient water relative to controls at most sites (Fig. 6.5). However, one test each in subreaches 3.02, 13.11, and 7.01 had a >20% reduction in fecundity which was also statistically significant (Sect. 3.3). Since effects were greater in the upstream reference subreach (13.11) than in the OU portion of Poplar Creek (3.02) and effects are rare, there is little evidence that DOE sources are causing toxicity to *Ceriodaphnia*. The reduction in fecundity in upper McCoy Branch embayment (reach 7.01) in August 1994 is large relative to both controls (52%) and reference (59%) and is supported by a small but statistically significant reduction (16%) relative to reference on the other tested date. Hence, although only two tests were run at that site, they both suggest that water from upper McCoy Branch is toxic to *Ceriodaphnia*. Correlation of *Ceriodaphnia* fecundity against concentrations of COPECs does not reveal any contaminants to be potential causes of the variance across reaches in test results.

Medaka. The proportion of medaka embryo tests showing significant increases in mortality or abnormalities is greatest in Poplar Creek embayment (Fig. 6.6 and 6.7). Mean rates of abnormalities were <10% in embryos incubated in water from reach 8 and subreaches 7.01, 7.02, 2.01, and 4.01. However, mean rates were >30% in all subreaches of reach 3, and were >50% in subreach 3.02 (Fig. 6.7). More than 40% of tests conducted in reach 3 water had abnormality rates that were significant (statistically significant or more than 20% greater than reference) (Sect. 3.3). Proportional reductions in medaka survival were quite erratic because of variance in survival in reference water, but the proportion of tests with significant reductions in survival relative to the reference site (subreach 13.11) consistently increased downstream in Poplar Creek (Fig. 6.6). Hence, reach 3 water is clearly toxic to medaka embryos relative to upstream water. However, the toxicity observed at 13.06, although less severe and frequent than in reach 3, does indicate that some component of the water coming from upstream of the DOE sources contributes to the toxicity. Although one out of two tests in upper McCoy Branch embayment (subreach 7.01) had a reduction in survival that was statistically significantly greater than the reference (reach 8), survival was an acceptable 84%, and abnormality rates were <10%.

The proportional reduction in survival of medaka embryos relative to reference was regressed against concentrations of COPECs and other properties of the water collected in conjunction with the toxicity tests. Medaka survival was statistically significantly correlated with concentrations of nickel ($r^2 = 0.64$), but not other variables. The nickel concentrations in the tests with the highest medaka mortality were less than a tenth of the chronic NAWQC, but approximately equal the *C. dubia* CV for ORR water [0.0106 mg/L (Table F1.7)]. Hence, the apparent exposure-response relationship is credible if medaka embryos are at least a sensitive to Ni as *Ceriodaphnia*, if Ni concentrations were higher on other days of the test than on the day when water was analyzed, or if other contaminants present in the test water contributed to the total toxicity. Another limitation of this exposure-response analysis is the fact that, by chance, the three out of seven medaka tests for which the water was analyzed showed the least mortality and abnormalities. Analysis of the more toxic waters would be likely to have shown more clearly the causes of the toxicity to medaka.

Sunfish and Largemouth Bass. To determine the relevance of the medaka embryo test to fish that occur in the OU, a single set of tests was run on embryos of medaka, redbreast sunfish and largemouth bass at six sites (Sect. 3.3.7.3). The bass results were erratic, with considerable variance among pairs. However, the sunfish results support the medaka results in that both showed biologically and statistically significant embryo lethality in subreaches 3.02 and 3.04 (3.01 and 3.03 were not tested) but not in 2.01 or 4.01. However, unlike the medaka, the sunfish showed no significant increase in abnormalities.

Sediment toxicity tests. The toxicity tests performed with sediment pore water and water above sediment showed no toxicity (Sect. 6.3). This result suggests that the epibenthic water inhabited by bottom-dwelling fish and the eggs and larvae of bottom-spawning fish does not pose a particular risk relative to the near surface water that was used in the tests described above.

concentrations of copper, mercury, silver, and zinc described above did not occur during any of the test periods. Of the COPECs, only nickel occurred at its highest concentrations during a test period. Water in reach 3 is regularly toxic to medaka embryos and is toxic to redbreast sunfish embryos in the one test of that species relative to the reference site. Although the medaka test has not been validated against field survey data like the standard fathead minnow and *Ceriodaphnia* tests, the nature and magnitude of the effects suggest a significant risk to sensitive fish populations in reach 3 that could be sufficient to significantly affect community properties.

6.2.3.4 Biological indicators

Histopathology. Histopathologies of bluegill liver and spleen were most common in subreaches 3.01, 3.02, and 4.01 (Sect. 3.5.4). The lowest frequencies were in lower Watts Bar (5), the Clinch immediately below White Oak Creek (2.02), and Norris Reservoir (10). The highest frequency of histopathologies were in largemouth bass from subreach 3.02, followed by 2.02 and 4.01. The lowest frequencies were in 5, 10, and, surprisingly, 3.04. Hence, histopathologies are consistently common in Poplar Creek below Mitchell Branch and in the Clinch immediately below Poplar Creek. Of the reaches in the OU, histological measures were not obtained from 1, 2.03, 2.04, 3.03, 4.02, 4.03, or 7. These histopathologies are characteristic of exposure to PCBs, so had they been absent, that absence would have constituted evidence against PCB toxicity. However, because they are suborganismal effects, their presence does not necessarily imply effects on the fish community. They simply indicate a potential causal link between exposure and higher level effects.

Reproductive indicators. Results of the studies of reproductive indicators are presented in Sect. 3.5.5. These indicators were measured in the OU only in subreaches 2.02, 3.04, and 4.04. At both subreaches 2.02 and 3.04 the gonadosomatic index of male bass was reduced, and in female bass death of oocytes (atresia) increased and fecundity decreased relative to reference fish from Norris Reservoir. Bluegill did not show clear or consistent differences in the indicators that might be related to DOE emissions. Although these indicators can not be interpreted as indicative of effects on the fish community end point, they are suggestive of inhibited reproduction in individuals of one species.

6.2.3.5 Biological surveys

Fish community survey data were analyzed at two scales. Tennessee Valley Authority (TVA) data were used to compare Watts Bar Reservoir to other reservoirs in the system (Sec. F8). Watts Bar was found to contain a typical fish community in that it had roughly average species richness by both sampling methods (Fig. 6.8 & 6.9), and it fell near the intersection of the axes of the multivariate discriminant analyses (Fig. F8.1.1-8). The pattern of the reservoirs in the discriminant analyses suggests that the primary variable controlling fish species composition is the position of the reservoirs in the system and the central position of Watts Bar in the discriminant analyses is consistent with its roughly central position in the system at the confluence of the Clinch and Tennessee Rivers. Hence, at the reservoir scale, there appear to be no effects on the fish community.

Fish surveys conducted by the CRRI were used to examine effects at local scales. Nine sites were sampled by two methods in 1993 and 1994. The results appeared to show low species richness in Poplar Creek embayment, except near the mouth of the creek (reach 3.04) which may be colonized by fish from the Clinch River. However, those results were ambiguous because the comparisons were

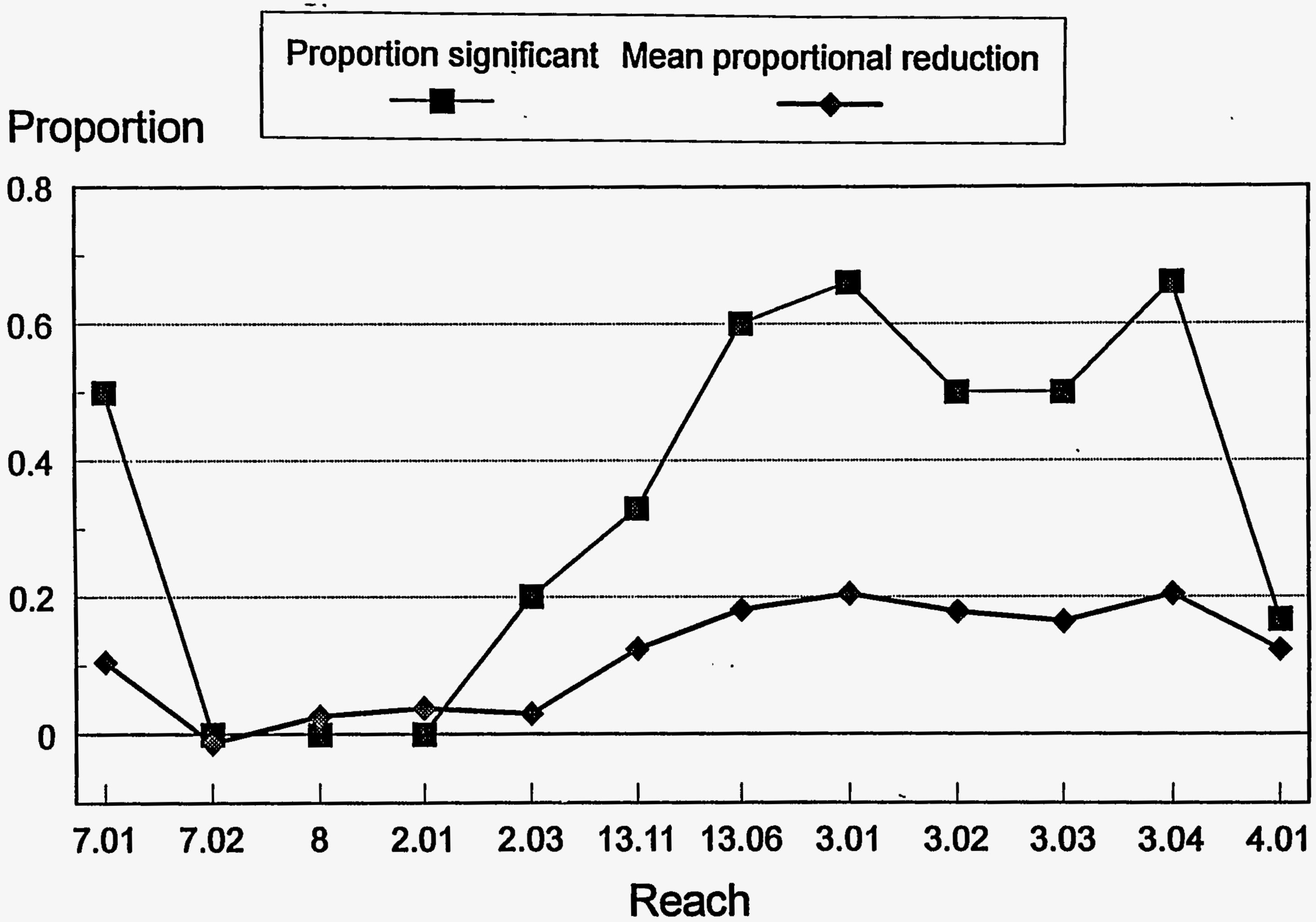


Fig. 6.4. Proportional reduction in fathead minnow larval survival relative to controls and proportion of tests in which survival was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly decreased). Note for comparison the responses in the Clinch River reference subreach (2.01), the far upstream reference subreach in Poplar Creek (13.11), and the near upstream reference (13.06).

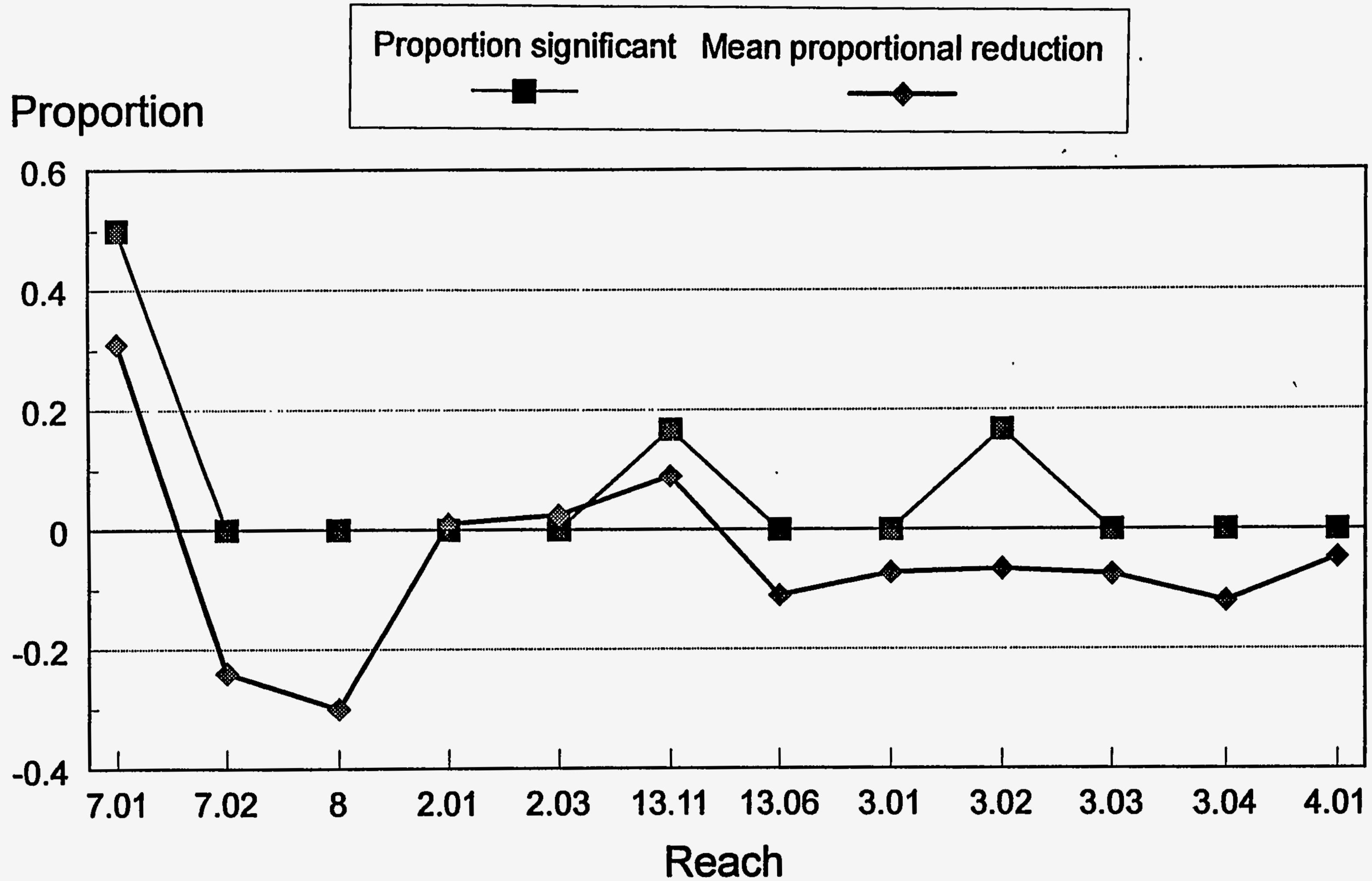


Fig. 6.5. Proportional reduction in *Ceriodaphnia dubia* fecundity relative to controls and proportion of tests in which fecundity was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly decreased). Note for comparison the responses in the Clinch River reference subreach (2.01), the far upstream reference subreach in Poplar Creek (13.11), and the near upstream reference (13.06).

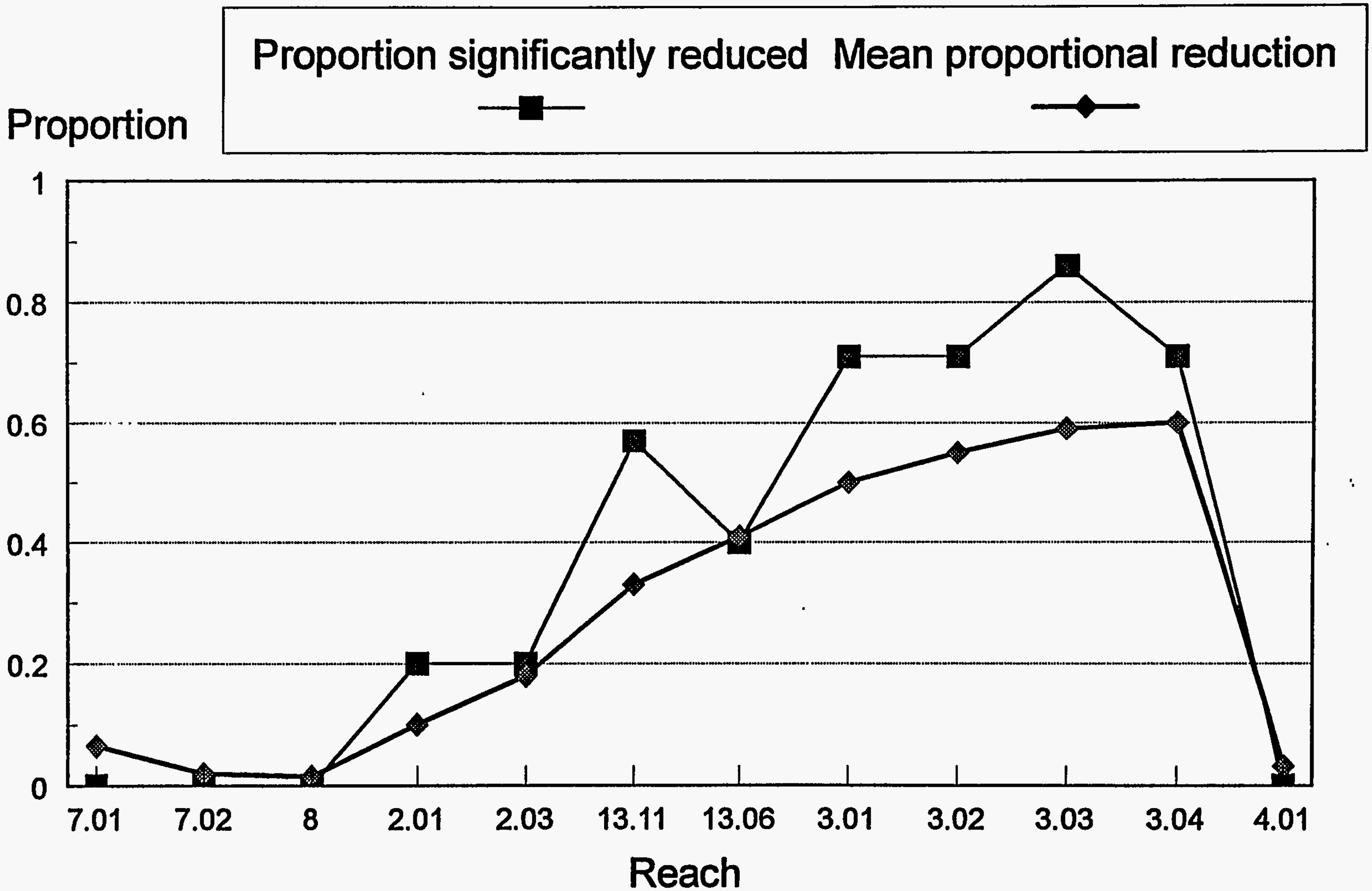


Fig. 6.6. Proportional reduction in survival of medaka eggs relative to controls and proportion of tests in which survival was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly increased). Note for comparison the responses in the Clinch River reference subreach (2.01), the far upstream reference subreach in Poplar Creek (13.11), and the near upstream referenc (13.06).

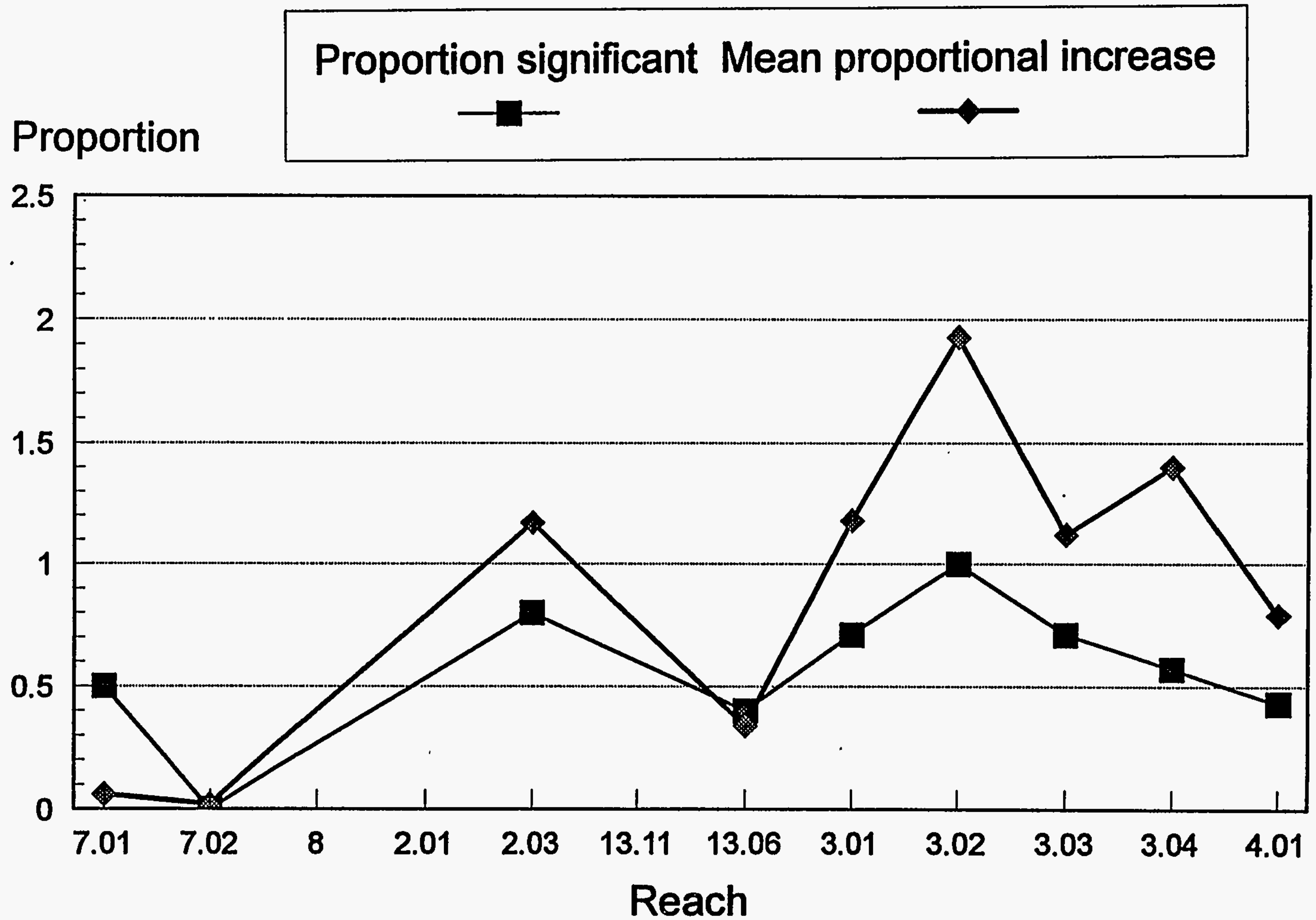


Fig. 6.7. Proportion of abnormal medaka larvae and proportion of tests in which abnormalities were statistically significantly increased relative to reference or were increased by at least 20% relative to reference (i.e., significantly increased). Note for comparison the frequencies of abnormalities in the Clinch River reference subreach (2.01), the far upstream reference subreach in Poplar Creek (13.11), and the near upstream reference (13.06).

to reservoir sites rather than to embayments. In 1995 one site in Poplar Creek embayment (subreach 3.02) was sampled along with a site in the ecologically similar and relatively uncontaminated Bull Run Creek embayment. Bull Run Creek was found to have higher species richness by both sampling methods than any Poplar Creek, Tennessee River, or Clinch River site (Fig. 6.10 & 6.11). In the 1995 samples, total species richness for Poplar Creek subreach 3.02 was 29% lower by gill netting, 65% lower by electrofishing, and 22% lower for both methods combined than Bull Run Creek. Also, for 1993 and 1994 and for both methods, species richness in subreaches 3.01 and 3.02 was > 20% lower than in the 1995 Bull Run Creek samples. The Bull Run Creek samples were not replicated across years and no other reference embayments were sampled to establish the variance among embayments, so the possibility that the results are confounded by habitat differences has not been eliminated. However, these results clearly suggest that the fish community of Poplar Creek embayment has fewer species than expected.

6.2.3.6 Weight of evidence for fish

The weighing of evidence is performed by asking the following questions concerning each reach in the OU:

1. Is the fish community less species-rich or abundant than would be expected?
2. Do individual fish display injuries that are indicative of significant toxic effects?
3. Is the water toxic to aquatic organisms?
4. Does that water contain chemicals in toxic amounts?
5. Do the fish contain chemicals in toxic amounts?
6. What factors account for apparent discrepancies in the results?
7. What is the likelihood that the fish community is at least 20% less species rich or abundant than it would be in the absence of contamination?

Poplar Creek embayment (reach 3). It is more likely than not that the fish community end point is violated in Poplar Creek embayment . This conclusion is based on the results indicating that the fish in reach 3 are intermittently exposed to levels of contaminants in water that have been shown to be toxic, that water collected between those episodes is toxic to fish embryos, and that individual fish are experiencing sublethal injuries. Although the relatively depauperate state of the community consistent with significant toxic effects, it is not clearly attributable to toxicity. However, it is not clear what chemicals from what sources are the cause of the observed effects. The conclusion is mitigated by the fact that the effects of contaminants on the fish community are not entirely attributable to DOE (Poplar Creek upstream is contaminated and somewhat toxic). The lines of evidence concerning risks to fish in Poplar Creek embayment (reach 3) are summarized in Table 6.3, and are discussed below.

If significant toxic effects were occurring the community would have fewer species and individuals than other sites, and it does. The fish surveys found that for both collection methods the number of species and the abundance of fish was lower than in a reference embayment and lower than in reservoir sites other than Norris (a low-productivity reservoir). This could be attributed to contaminants or to relatively poor habitat structure. However, the upper subreach (3.01) has relatively good habitat structure, but when fish were collected there for analysis, the abundance was qualitatively determined to be as low as in other subreaches. Hence, the survey results are consistent with a significant toxic effect, but habitat quality may also contribute.

If significant toxic effects were occurring, then the fish would display suborganismal signs of the toxic effects of the site contaminants, and they do. The elevated histopathology frequencies in 3.01 and 3.02 and reproductive abnormalities are consistent with exposure to PCBs, and possibly other contaminants. However, although they are adverse to individual fish, they do not have clear implications for properties of the fish community because of the compensatory properties that act at the individual, population, and community levels. Therefore, they are indicative of toxicity but not necessarily of significant effects on the community.

If significant toxic effects were occurring, then the water should be toxic, and it is. The water from Poplar Creek was toxic to medaka and sunfish embryos, but apparently not to fathead minnow larvae or *Ceriodaphnia*. However, by chance water was not collected during periods of high contaminant levels, so the lack of clear toxicity in the standard tests does not disprove the occurrence of episodic toxicity to those species. The toxicity to medaka embryos appears to be associated with nickel concentrations.

If significant toxic effects were occurring due to chemicals that bioaccumulate in fish, then concentrations in fish should be as high as those that caused effects in toxicity tests, and for some fish they are. The most highly contaminated channel catfish had whole-body concentrations of PCBs greater than those that were reported to cause reduced growth in that species in the laboratory. However, those results are from tests of young fish so they may not be applicable to the adult fish which were measured.

If significant toxic effects were occurring, then chemicals should be found in water at concentrations that have been shown to be toxic in the laboratory, and on occasion they were. However, because the cause and duration of the episodes is unknown, it is not certain that they actually cause sufficient toxic effects to cause significant decrements in community properties.

Although each of these lines of evidence is consistent with significant toxic effects, each has associated counter arguments. The contribution of contaminants to community properties could be negligible relative to habitat quality. The bioindicators and embryo toxicities may be indicative of effects for which the community may compensate. The toxic effects associated with PCB body burdens may not be relevant to adult fish or may not have community-level consequences. The high contaminant levels may be associated with extremely brief episodes that do not cause significant effects. Hence, no one of the line of evidence is convincing by itself, but the fact that they are all consistent with toxic effects makes it unlikely that the results are all false positives.

Upper Clinch River arm (reach 2). Although there is evidence of sublethal toxic effects on individual fish in the Clinch River above Poplar Creek but below the Melton Hill dam (reach 2), the community does not appear to be significantly affected. The lines of evidence concerning risks to fish are summarized in Table 6.4. This reach resembles reach 3 in that the fish community is relatively depauperate, reproductive indicators differ from reference fishes, and the most contaminated catfish have PCB body burdens that are greater than those causing effects in toxicity tests. However, there is much more fish survey data for this reach and better reference data against which to compare it than in Poplar Creek. Those data allow us to conclude that the fish community properties are approximately what would be expected given the nature of the habitat. In addition, no chemicals were detected at toxic concentrations in reach 2. Finally, rates of histopathologies were low in largemouth bass and intermediate in bluegill below White Oak Creek (2.02).

Lower Clinch River arm (reach 4). Although there are episodically high concentrations of metals and evidence of sublethal toxic effects on individual fish in the Clinch River below Poplar Creek (reach 4), the community does not appear to be significantly affected. The lines of evidence concerning risks to fish in the Clinch River below Poplar Creek (reach 4) are summarized in Table 6.5.

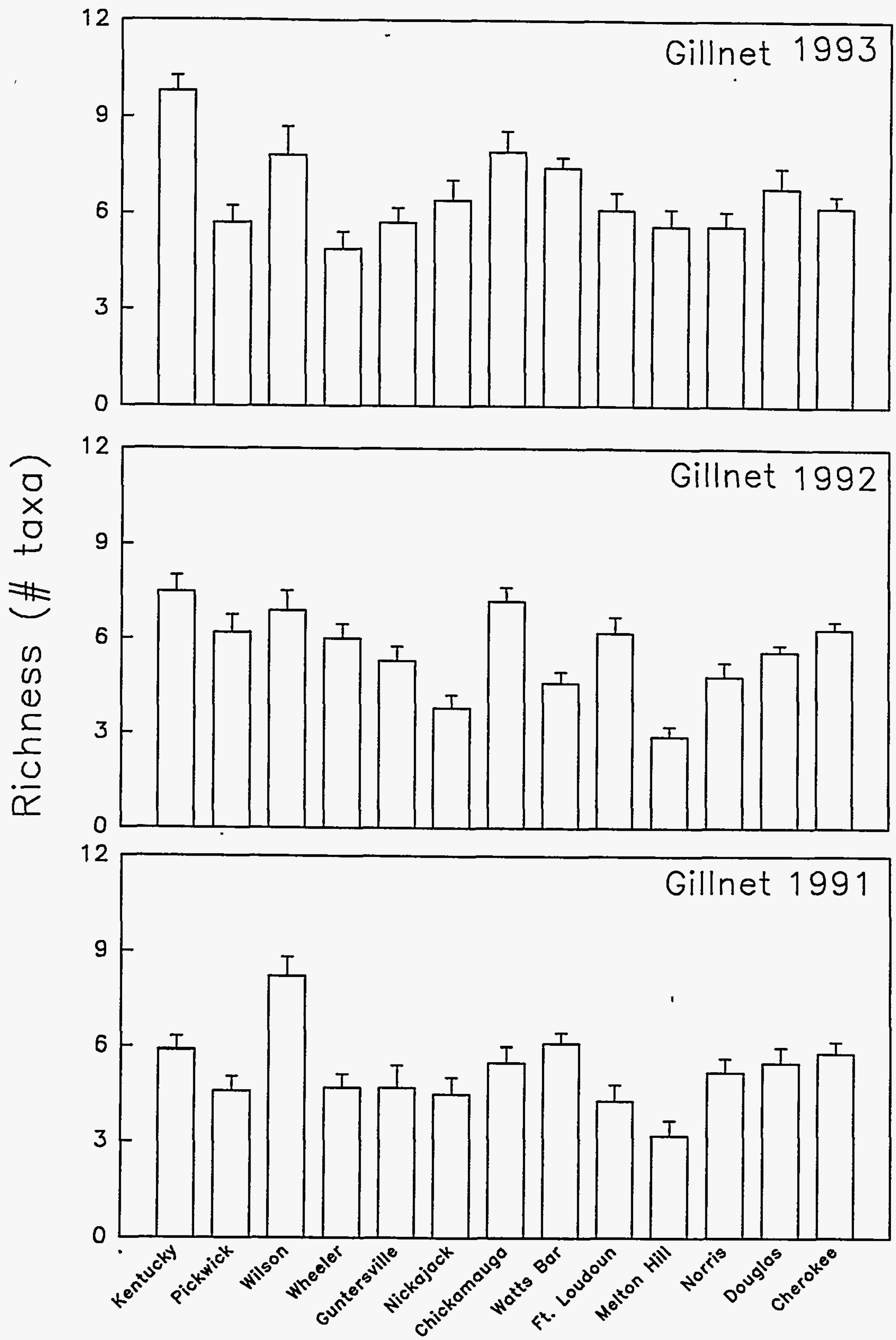


Fig. 6.8. Species richness of fish sampled by gill netting in individual reservoirs in 1991–1993.

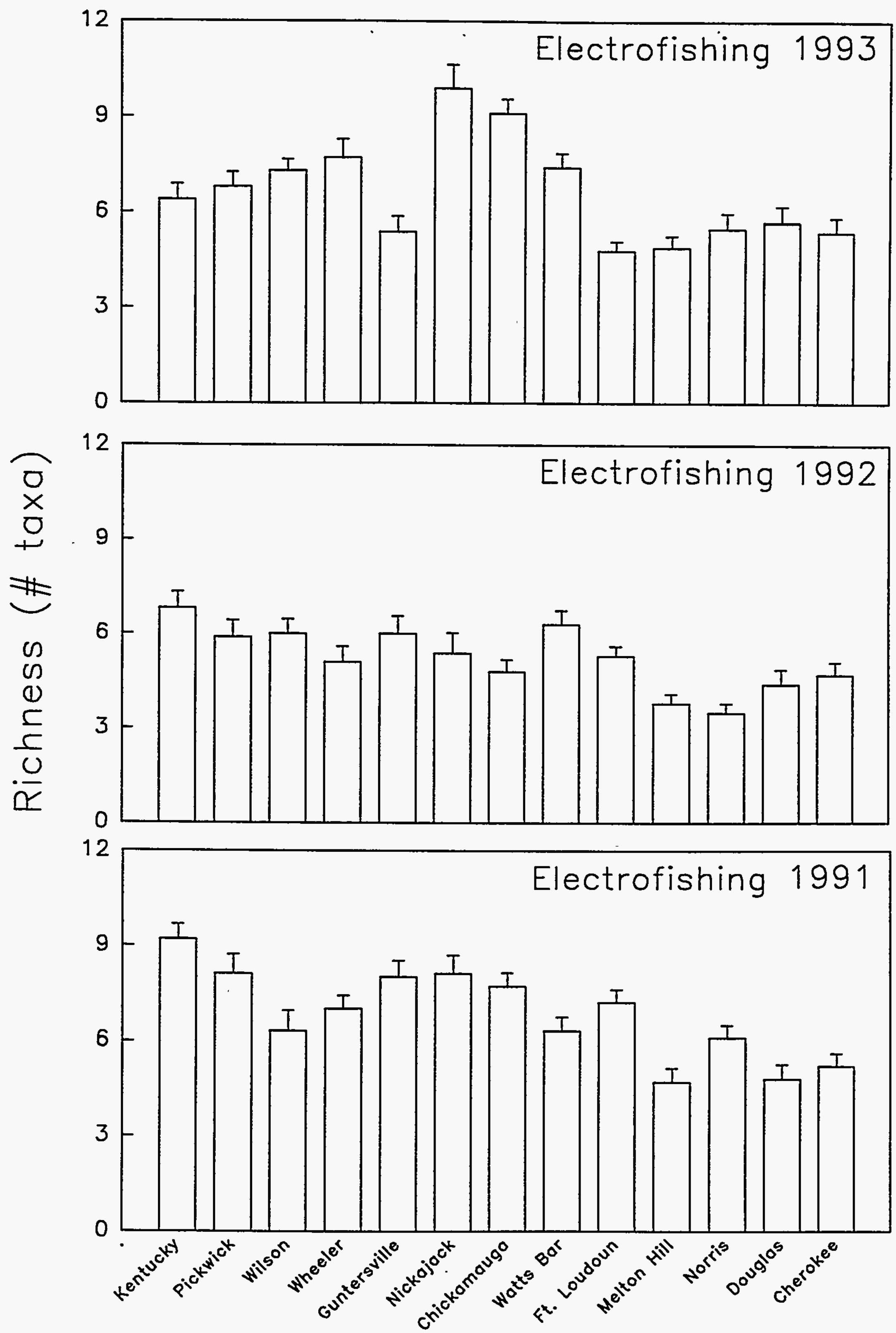


Fig. 6.9. Species richness of fish sampled by electrofishing in individual reservoirs in 1991–1993.

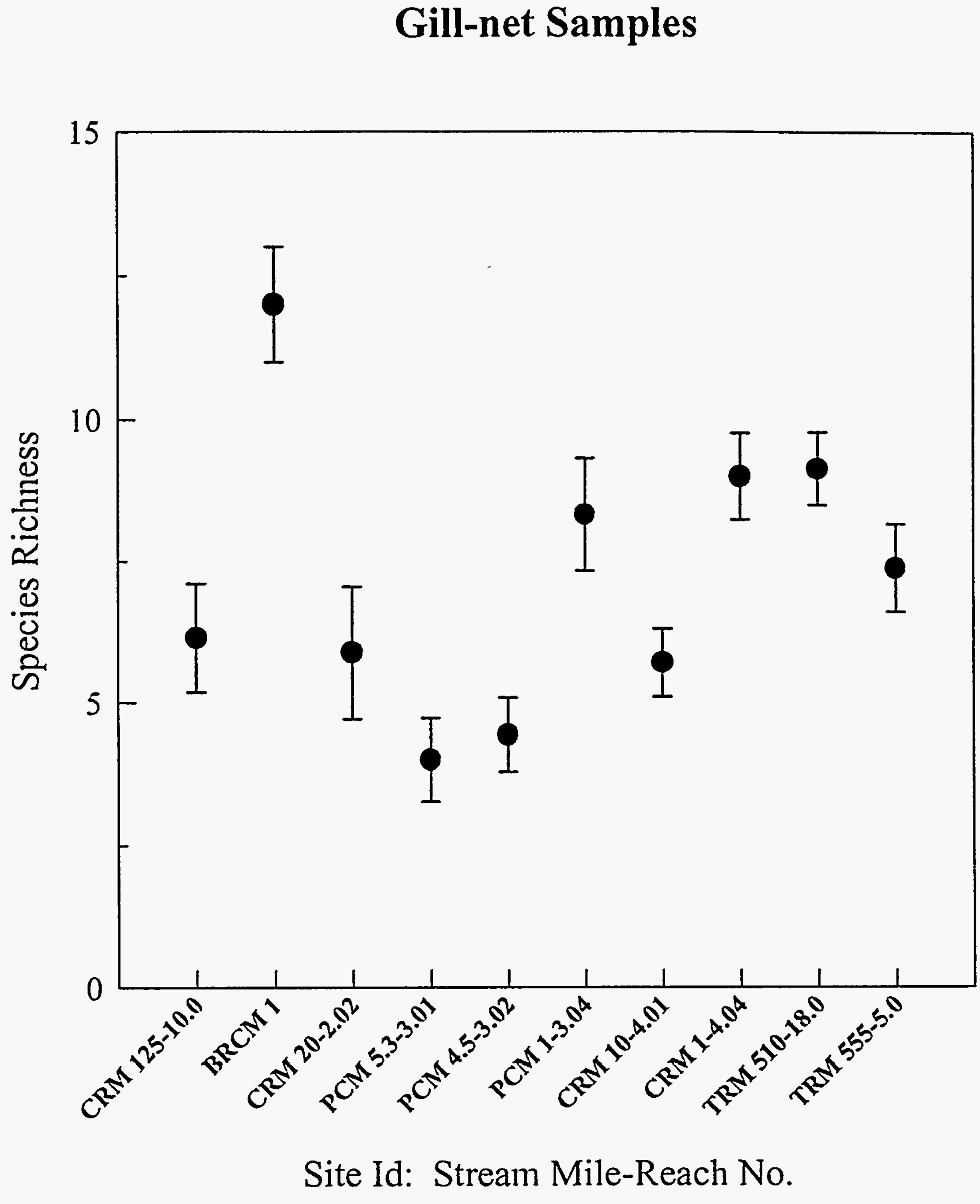


Fig. 6.10. Species richness of fish sampled by gill netting in the Clinch River/Poplar Creek system and reference reaches. Means and standard errors for samples taken in 1993–1995 are shown. Sampling locations are identified by mile number and reach number. CR = Clinch River, BRC = Bull Run Creek, PC = Poplar Creek, and TR = Tennessee River.

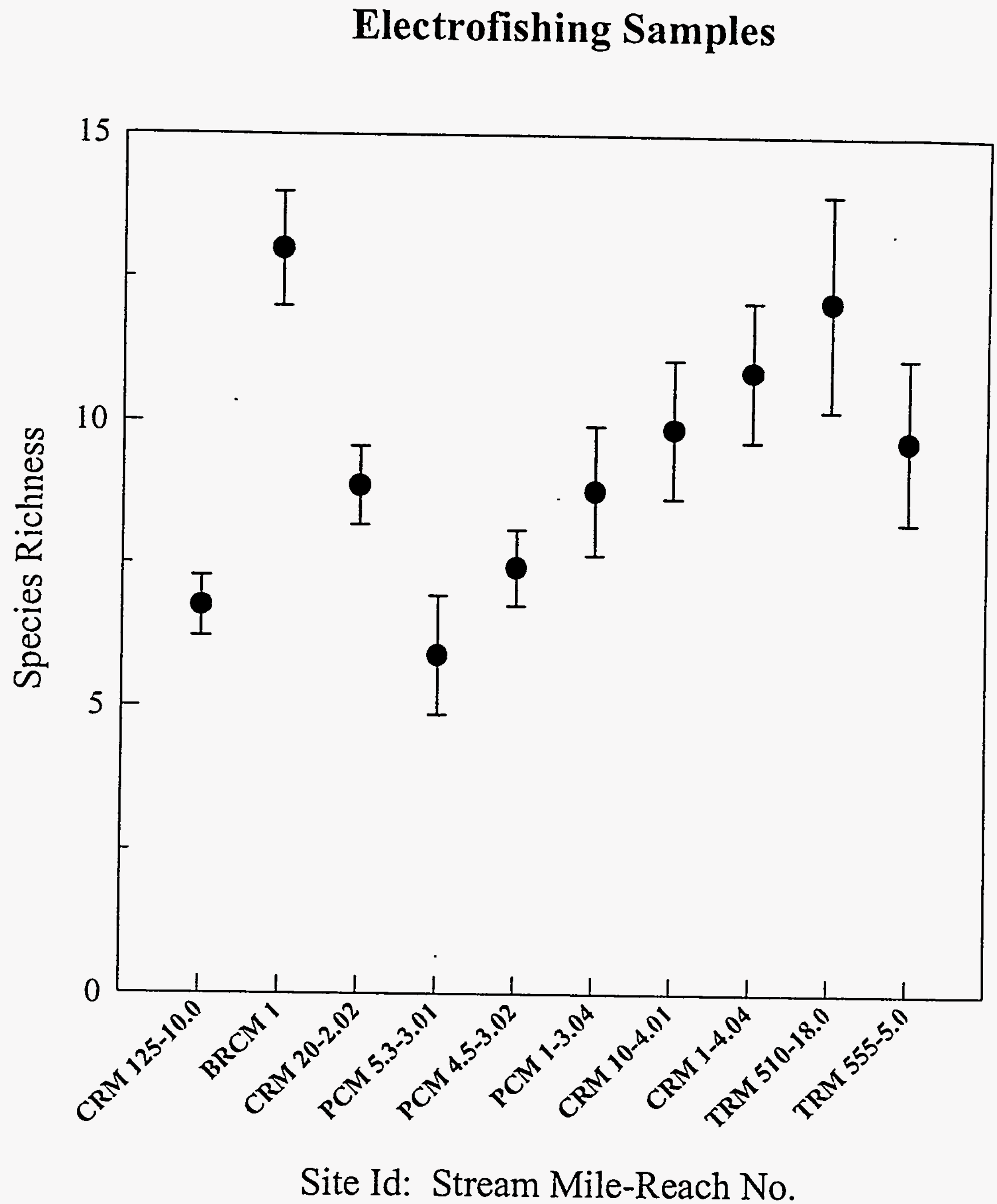


Fig. 6.11. Species richness of fish sampled by electrofishing in the Clinch River/Poplar Creek system and reference reaches. Means and standard errors for samples taken in 1993–1995 are shown. Sampling locations are identified by mile number and reach number. CR = Clinch River, BRC = Bull Run Creek, PC = Poplar Creek, and TR = Tennessee River.

If significant toxic effects were occurring, then the community would have fewer species and individuals than other sites. This does not appear to be the case on balance. The species richness in 4.04 by both methods of measurement and in 4.01 by electrofishing are similar to potential reference sites. However, in both years the species richness in 4.01 by gill netting is lower than upstream in the Clinch River (2.02), downstream (4.04), and lower Poplar Creek (3.04).

If significant toxic effects were occurring, then the fish would display suborganismal signs of the toxic effects of the site contaminants, and they do. Subreach 4.01 has the third highest frequency of histopathology for both bluegill and largemouth bass.

If significant toxic effects were occurring, then the water should be toxic, but it is not. The water from 4.01 did not cause significant toxic effects on any species.

Table 6.3. A summary of risk characterization for the fish community in Poplar Creek (reach 3)

Evidence	Result[a]	Explanation
Biological surveys	+	Fish community abundance and species richness are both low, but habitat quality may be a causal factor.
Bioindicators	±	Frequencies of histopathologies were elevated in largemouth bass and bluegill, and levels of fecundity and other indicators of reproductive condition were reduced in largemouth bass in relation to reference fish.
Toxicity tests	+	Water was toxic to medaka and redbreast sunfish embryos but apparently not to fathead minnows and *Ceriodaphnia.*
Fish analyses	±	Concentrations in maximally contaminated channel catfish exceed the concentration causing reduced growth and liver pathology in that species, but most concentrations are much lower.
Media analyses	+	Cu, Hg, and possibly Ni and Ag were detected episodically at toxic concentrations, but average concentrations are nontoxic.
Weight-of-Evidence	+	Although the evidence is not strong, reach 3 waters appear to pose a significant risk to the fish community from episodes of high metal concentrations and from PCBs, which are mostly undetectable in water.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
± indicates that the evidence is too ambiguous to interpret.

If significant toxic effects were occurring due to chemicals that bioaccumulate in fish, then concentrations in fish should be as high as those that caused effects in toxicity tests, and for some individual fish they are. The most highly contaminated channel catfish had whole-body concentrations of PCBs greater than those that were reported to cause reduced growth in that species in the laboratory. However, those results are from tests of young fish so they may not be applicable to the adult fish which were measured.

If significant toxic effects were occurring, then chemicals should be found in water at concentrations that have been shown to be toxic in the laboratory, and on occasion they were. However, only copper and zinc were found at toxic levels, and, because the cause and duration of the episodes is unknown, it is not certain that they actually cause sufficient toxic effects to cause significant decrements in community properties.

In sum, while it is possible that significant effects on the fish community are occurring, none of the lines of evidence is strong enough to indicate such effects, and the lines of evidence are not sufficiently consistent to indicate such effects based on the weight of evidence.

Table 6.4. A summary of risk characterization for the fish community in the upper Clinch River arm of Watts Bar Reservoir (reach 2)

Evidence	Result[a]	Explanation
Biological surveys	±	Fish community abundance and species richness are both low but not in relation to other Clinch River sites.
Bioindicators	±	Reproductive indicators were modified in largemouth bass below White Oak Creek (subreach 2.02) in relation to reference fish.
Toxicity tests		No toxicity tests were performed with water from the portion of this reach downstream of U.S. Department of Energy sources (subreaches 2.02–2.04). Subreach 2.01 water was apparently nontoxic.
Fish analyses	±	Concentrations in maximally exposed channel catfish exceed the concentration causing reduced growth and liver pathology in that species, but most concentrations are much lower.
Media analyses	-	No chemicals were detected at potentially toxic concentrations.
Weight-of-evidence	-	The community does not appear to be significantly affected by contaminant exposures.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
± indicates that the evidence is too ambiguous to interpret.

McCoy Branch embayment (reach 7). The lines of evidence concerning risks to fish in the McCoy Branch embayment (reach 7) are summarized in Table 6.6. Although water from 7.01 is toxic to *Ceriodaphnia*, only two tests were performed so the evidence is not strong. Single measurements of silver and nickel constituted the only evidence of toxicity from chemical analyses. No fish community surveys, bioindicator analyses, embryo toxicity tests, or analyses of fish were conducted.

Since there is evidence of toxicity but no chemicals in clearly toxic amounts, the evidence is ambiguous. Significant risks can not be said to occur but can not be excluded.

6.2.4 Uncertainties Concerning Risks to Fish

The following issues constitute the major sources of uncertainty in the risk assessment for the fish community.

The nature of the episodic exposures to high chemical concentrations is poorly specified. It is not known whether they are due to episodic releases from the ORR or mobilization of materials in the Poplar Creek system. Also their duration is unknown. Since they represent more than 5% of the samples, it was assumed that their duration is >7 days (2% of a year) which is the duration of the EPA's standard tests for chronic effects of effluents and ambient waters. However, if the observed high concentrations are associated with multiple episodes of very short duration, then they may not cause significant toxic effects.

The toxicity of water during the episodes of high exposure are unknown because tests were not conducted during those periods.

Table 6.5. A summary of risk characterization for the fish community in the lower Clinch River arm of Watts Bar Reservoir (reach 4)

Evidence	Result[a]	Explanation
Biological surveys	-	Fish community abundance and species richness are not atypical.
Bioindicators	±	Frequencies of histopathologies were elevated in subreach 4.01 in relation to reference fish.
Toxicity tests	-	Water from subreach 4.01 is not toxic.
Fish analyses	±	Concentrations in maximally contaminated channel catfish exceed the concentration causing reduced growth and liver pathology in that species, but most concentrations are much lower.
Media analyses	+	Copper and zinc were detected episodically at toxic concentrations in subreach 4.01, but average concentrations are nontoxic.
Weight-of-evidence	-	The community does not appear to be significantly affected by contaminant exposures.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
± indicates that the evidence is too ambiguous to interpret.

Table 6.6. A summary of risk characterization for the fish community in McCoy Branch embayment (reach 7)

Evidence	Result[a]	Explanation
Biological surveys		The fish community was not surveyed.
Bioindicators		Bioindicators were not measured
Toxicity tests	±	Water was toxic to *Ceriodaphnia* and possibly fathead minnows in one test each in the upper embayment, but apparently not to medaka. Only two testing periods were conducted.
Fish analyses		Body burdens were not measured
Media analyses	-	No chemicals were detected at toxic concentrations except for single values of silver and nickel.
Weight-of-evidence	±	Reach 7 has a possibly significant risk to fish, but the evidence is weak.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the fish community.
± indicates that the evidence is too ambiguous to interpret.

The high observed mortality and abnormalities in fish embryós are believed to constitute toxic effects that would result in significant effects on the fish community. However, these tests have not been validated against biological survey data at sites where clear toxic effects are occurring as has been done with the standard 7-day tests.

The relationship of the histopathologies and reproductive indicators to effects on populations and ecosystems is unknown.

The reference embayment, Bull Run Creek, was sampled in only one year, and it was the only reference embayment sampled. Therefore, the variance in reference embayments for comparison to Poplar Creek embayment is unknown.

6.3 RISKS TO BENTHIC INVERTEBRATES

6.3.1 Exposure Assessment for Benthic Invertebrates

Two different expressions of sediment contamination are used, whole sediment concentrations and filtered pore water concentrations. The use of pore water is based on the assumption that chemicals associated with the solid phase are largely unavailable and therefore sediment toxicity can be estimated by measuring or modelling the pore water concentration. This is the approach used by the EPA to calculate sediment quality criteria. Whole sediment concentrations do not account for effects of sediment properties on bioavailability. However, they are required by the EPA Region IV and may provide a better estimate of risk for highly particle-associated chemicals.

For purposes of screening chemicals, the appropriate estimate of exposure is a concentration that protects the most exposed organisms. Because benthic invertebrates are relatively immobile and inhabit a medium that changes little over time, the maximum concentration is used. For risk estimation, the estimate of exposure of the community is the percentage of samples exceeding particular effects levels.

The sediment toxicity tests include direct exposure to whole sediment (*H. azteca*), exposure to sediment pore water (*C. dubia*), and exposure to epibenthic water (*D. magna*). It is assumed that the form and bioavailability of chemicals is not altered by sediment collection and handling, so these tests represent exposures in the field.

6.3.2 Effects Assessment for Benthic Invertebrates

6.3.2.1 Single chemical sediment toxicity

Because there are no standard screening benchmarks and sediment quality criteria for only a few chemicals, sets of alternative sediment benchmarks were derived for each chemical (Table 6.7). Whole sediment concentrations are compared to these alternative benchmarks. Pore water concentrations are compared to the ecotoxicological benchmarks for aquatic biota (Table 6.1). The use of multiple benchmarks provides greater assurance of detecting all COPECs. Sediment quality criteria are corrected for site-specific conditions. Sediment benchmarks derived using the equilibrium partitioning method are calculated using location-specific percent organic carbon. As in the aqueous chemical screening (Sect. 6.2.2.1), hardness-dependent criteria are conservatively corrected to 100 mg/L hardness based on the pore water mean across reaches of 134 mg/L and range of 82-160 mg/L.

Toxicity profiles are presented in Appendix F1 for those chemicals that exceed benchmarks. The toxicity profiles summarize the existing toxicity information for each chemical including concentrations causing acute lethality and chronic lethal and sublethal effects, and physical-chemical conditions that modify toxicity. Pore water COPEC concentrations are evaluated using aqueous toxicity data. Most of the available sediment toxicity data are for marine and estuarine systems.The sediment toxicity profiles address primarily freshwater sediment data, which are not included in the ER-Ls and ER-Ms (Long et al. 1995). An extensive marine data set was presented in MacDonald et al. (1994). Lethal and community level effects from these sources are presented graphically for most COPECs in Appendix F3.

6.3.2.2 Ambient sediment toxicity

Because there is no dilution series for the sediment toxicity tests, the appropriate exposure-response relationship is the association between the frequency of significant toxic responses and either the concentrations of contaminants in the sediment or the location relative to known sources of contaminants. Responses that are statistically significantly different, or are inhibited by 20% or more relative to reference sediments or control water, are assumed to be indicative of sediments that are toxic to benthic biota. Test responses used in this assessment include reductions in survival of an amphipod (*H. azteca*) in 10-day exposures to whole sediment, reductions in survival of a cladoceran (*C. dubia*) in 48-hour exposures to sediment pore water, and reductions in survival and fecundity of another cladoceran (*D. magna*) in 12- to 14-day exposures to water overlying sediment.

Table 6.7. Descriptions of the ecotoxicological screening benchmarks for benthic biota exposed to contaminated sediments. More details are presented by Hull and Suter (1994), Long et al. (1995), and EPA Region IV (1994). The last three benchmarks are used only when none of the first five are available.

Benchmark	Abbreviation	Description
Effects range—low	ER_L	The tenth percentile of estuarine sediment concentrations reported to be associated with some level of toxic effects.
Effects range—median	ER_M	The fiftieth percentile of estuarine sediment concentrations reported to be associated with some level of toxic effects.
Region IV benchmark	REG_IV	The higher of two values, the U.S. Environmental Protection Agency Contract Laboratory Program Practical Quantification Limit (CLP PQL) and the Effects Value, which is the lower of the ER-L and the Florida no-observed-effects level.
National sediment quality criteria	EPA_SQC01	Sediment quality criteria based on toxicity in water expressed as chronic water quality criteria (recalculated after adding some benthic species) and partitioning of the contaminant between organic matter (1% of sediment by weight) and pore water. Sediment quality criteria were adjusted to the site-specific percent total organic matter content.
Equilibrium partitioning benchmark	EQ_PART	Benchmarks derived in the same manner as sediment quality criteria except that the expression of aqueous toxicity is the chronic National ambient water quality criteria or the Secondary Chronic Value, and location-specific percent organic matter is used.
Ontario Ministry of the Environment Lowest Effect Level	LEL_MOE	Concentrations determined by the Ontario MOE to constitute thresholds for toxic effects in Ontario sediments.
Region V benchmark	REG_V	A concentration determined to constitute background for sediments in Illinois.
Apparent effect threshold	AET	A concentration above which toxic effects occurred at all sites in Puget Sound.

Sources:

EPA (United States Environmental Protection Agency). 1994 *Draft Region IV Waste Management Division Sediment Screening Values for Hazardous Waste Sites.* 2/16/94 version. Region IV, Atlanta.

Hull, R. N., and G. W. Suter, II. 1994. *Toxicological Benchmarks for Screening of Potential Contaminants of Concern for Effects on Sediment-Associated Biota: 1994 Revision.* ES/ER/TM-95/R1. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Long, E. R., D. D. MacDonald, S. L. Smith, and F. D. Calder. 1995. "Incidence of Adverse Biological Effects within Ranges of Chemical Concentrations in Marine and Estuarine Sediments." *Environ. Manage.*

6.3.2.3 Invertebrate community survey

Benthic macroinvertebrate communities were evaluated on two spatial scales. Reservoirs in the study area were compared to other mainstem TVA reservoirs and reaches within the study area were compared to each other. Two sets of benthic macroinvertebrate data were evaluated: TVA Vital Signs Monitoring Program data from thirteen mainstem reservoirs for the years 1992–1994 and ER Program data collected by TVA in 1994 from eight sites in the Clinch River and eight sites in Poplar Creek. These data should be comparable because the same sampling and statistical analysis methods were used for both sets of data. Patterns in community structure across reservoirs were evaluated using canonical discriminant analysis and richness and abundance were analyzed using ANOVA and multiple-comparison tests. Reduction of these end points by 20 %, relative to reference sites, would indicate that actual impacts to the benthos are occurring. However, it would be necessary to infer the cause from the distribution of known or suspected sources of contaminants or toxicity. Towards this end, a multiple linear regression analysis was performed to investigate the influence of physical characteristics and contaminant concentrations on benthic community structure. Methods of data collection and analysis are detailed in Appendix F8.

6.3.3 Risk Characterization for Benthic Invertebrates

6.3.3.1 Single chemical sediment toxicity

Screening against benchmarks. Chemicals detected in whole sediments or filtered pore water were screened against benchmarks and evaluated as COPECs. A HQ was calculated for each chemical by dividing the maximum concentration by the sediment (Table F3.1) or aquatic benchmarks (Table F3.2). Chemicals that exceeded any benchmark (e.g., HQ>1) were examined further to determine whether they were credible COPECs (Table 6.8). The criteria for eliminating COPECs were those used for evaluating aqueous COPECs (Sect. 6.2.3.1.1), where filtered and unfiltered water samples correspond to pore water and whole sediment samples, respectively. None of the candidate COPECs could be eliminated without a more detailed evaluation of their distribution among reaches and the relevant effects data.

The ΣTUs for each subreach was calculated to show the relative toxicity of sites and the relative contribution of COPECs to that toxicity. This holistic overview provides a context for the detailed discussions of exposure-effects that follow. The generic derivation and interpretation of TUs in this assessment is presented in Sect. 6.1.6. The TUs for pore water COPECs were calculated using the same procedure used for the aqueous COPECs (Sect. 6.2.3.1.1). That is, Daphnid EC_{50}s were used as the denominator and they were adjusted to a hardness of 100 mg/L for hardness dependant chemicals (e.g., copper). The ER-L from Long et. al. (1995) was the denominator for most whole sediment COPECs. The AET or REG_V value was used only if an ER-L was not available. TUs were calculated for each COPEC using the maximum concentration in each subreach. These TUs were summed by subreach for the pore water COPECs and whole sediment COPECs. Maximum concentrations may over estimate exposure. The concentrations represented by the maximum may actually be present in only one location in a subreach and the maxima for all COPECs may not occur at the same location. Therefore, ΣTUs were also calculated using median concentrations in pore water and whole sediment to better estimate typical exposures for each subreach. Pore water maximum and median ΣTUs are presented in Figures 6.12 and 6.13, respectively. Both maximum and median ΣTUs for whole sediment exceeded 1.0 at all sites. Therefore, only the more representative median ΣTUs are presented (Fig. 6.14a-d).

Table 6.8. Results of screening of chemicals that exceed benchmarks in whole sediments or filtered pore water for contaminants of potential ecological concern (COPECs)

Chemical	COPEC	Reason for inclusion or rejection
Aluminum	Yes	Exceeds chronic National ambient water quality criterai (NAWQC) in pore water in reaches 0 and 18 and in subreaches 3.04, 4.01, 4.02, 4.04, and 7.02. Exceeds Daphnid LTV and the LCV of all tested aquatic organisms (plants) at subreach 4.01. A sediment benchmark is not available.
Arsenic	Yes	Exceeds the ER_M in whole sediment in subreach 7.01 and the ER-L, LEL_MOE, and/or REG_IV in reaches 1,7, 8, 10, subreach 2.04, the three downstream subreaches of 3, and reach 4. Did not exceed NAWQC or any other aqueous benchmark in any reach.
Barium	Yes	Exceeds EPA_REG V background value in all reaches. Exceeds the SCV in pore water in all reaches and the SAV in reaches 0 and 13, in subreaches 2.01 and 2.04, and the three downstream subreaches of 3 and 4, but the Daphnid LCV was not exceeded in any reach.
Boron	Yes	Exceeds the Daphnid LTV in pore water in all reaches. That benchmark is apparently below the local background due to the compounding of effects across all life stages in its derivation. Exceeds the SCV in subreach 3.04, but does not exceed this or any other benchmark in any other reach. A sediment benchmark is not available.
Cadmium	Yes	Exceeds the ER-L, LEL_MOE, and/or REG_IV in reaches 1, 3, 5, and 22 and in subreaches 4.04 and 7.02. Reported at the detection limit in 1 of 11 pore water samples from 4.04, where it exceeds the LCV and LTV for Daphnids but no other benchmarks.
Chromium	Yes	Exceeds the REG_IV in subreaches 3.02, 3.03, 3.04, 4.03, and 7.02, and in reach 22 and the ER_L in 3.02, 3.03, and 22, but is less than the ER_M in all reaches. Did not exceed any benchmarks in pore water in any reach.
Cobalt	Yes	Exceeds the SCV and the Daphnid LTV and LCV in pore water in subreaches 3.01, 3.03, 4.03, and 4.04, and in reach 13; the SCV in reach 0 and subreach 3.02; no other benchmarks in any reach. A sediment benchmark is not available.
Copper	Yes	Exceeds the REG_IV and ER-L in reach 1, the three downstream subreaches of 3, 4.03, 4.04, 5, 7.02, 10, and 22, but is less than the ER_M in all reaches. Exceeds the chronic NAWQC in pore water in subreach 3.04; the Daphnid LTV and LCV and Aquatic Plant LCV in all reaches; the Fish LCV and/or LTV in 3, 4.01, 4.03, 4.04, 7.01, and 18; and the Non-Daphnid Invertebrate LCV in 3, 4.03, and 4.04.
Iron	Yes	Marginally exceeds the LEL_MOE in subreaches 3.03 and 4.04, and in reach 7. Exceeds the chronic NAWQC and the Fish LCV in pore water in subreaches 3.02, and 4.04; the Daphnid LTV in all reaches; and the Daphnid LCV in 0, 2, the three downstream subreaches of 3, 4.01, 4.04, and 18.

Table 6.8 (continued)

Chemical	COPEC	Reason for inclusion or rejection
Lead	Yes	Exceeds the REG_IV and/or ER-L in reaches 0 and 1, in subreach 2.04, the three downstream subreaches of 3, 4, 5, 7.02, 8, 10, 18, and 22, but is less than the ER_M in all reaches. Detected in 3 of 7 pore water samples from 4.03, but did not exceed any of the seven benchmarks.
Manganese	Yes	Exceeds the LEL_MOE in all reaches except 2.01. Exceeds all benchmarks for pore water in all reaches except 2.03.
Mercury	Yes	Exceeds the REG_IV and/or ER-L in all reaches except 0, 2.03, and 10, and the ER_M in reaches 3, 4, 5, 8, 13, and 22. Exceeds the LCV for all aquatic organisms (LCV FISH) in subreaches 3.01, 3.02, and 4.04, but no other benchmarks in any reach.
Nickel	Yes	Exceeds the REG_IV and ER-L in 2.04, 3, 4, 5, 7, 8, 10, 13, and 22, and the ER_M in subreaches 3.02, 3.03, and 3.04. Exceeds the Aquatic Plants and Daphnids LCVs in all reaches except 2.01, and 18; the Fish LCV in 3.01 and 3.02; and the Daphnid and Fish LTVs in 3.01, but no other benchmarks, including NAWQC, in any reach.
Silver	Yes	Exceeds the ER-L and/or REG_IV in the three downstream subreaches of 3, 4.04, and 22, the ER_M in subreaches 3.02 and 3.03, and in reach 22. Not detected in pore water in any reach.
Uranium	Yes	Detected in pore water in reaches 3, 4, and 13. Exceeds SCV in 3.02, 3.03, 3.04, and 4.04, the SAV in 3.02, but not the Fish LCV or LTV in any reach. A sediment benchmark is not available.
Zinc	Yes	Exceeds the REG_IV in all reaches except 2.01, 2.02, 2.03, and 4.01, the ER_L in 3.02, 4.04, and 5, but not the ER_M in any reach. Exceeds the Aquatic Plant LCV in 3.03 and 4.03; the Fish LCV and LTV and the Daphnid LCV in 4.03; no other benchmarks, including NAWQC, in any reach.
2-Methylnaphthalene	Yes	Exceeds the ER_L in reaches 0, 4, and 13, and in subreaches 2.04, 3.02, 3.03, 3.04, and 7.02; the REG_IV (CLP PQL) in 0, 2.04, 3.02, 3.03, and 13; and the ER_M in 0, and 13. Not detected in pore water in any reach.
4,4'-DDD	Yes	Detected in 1 of 2 sediment samples from reach 1 and exceeds the ER_L , ER_M, and the EQ_PART. Not detected in any reach in pore water but reach 1 was not sampled.
4,4'-DDT	Yes	Detected in 1 of 2 sediment samples from reach 1 and exceeds the ER_L, ER_M, and REG_IV, but not the EQ_PART. Not detected in any reach in pore water but reach 1 was not sampled.

Table 6.8 (continued)

Chemical	COPEC	Reason for inclusion or rejection
Acenaphthene	Yes	Exceeds the ER_L in reaches 0 and 13 and in subreaches 3.02, 3.03, 3.04, and 4.01; does not exceed the REG_IV (CLP PQL), the ER_M, or the EPA SQC01 in any reach. Not detected in pore water in any reach.
Anthracene	Yes	Exceeds the EQP_PART in subreaches 3.02, 3.03, 3.04, 4.01, and 4.02 and in reach 13; the ER_L in 3.02, 3.03, 3.04, and 4.01; but not the REG_IV (CLP PQL) or ER-M in any reach. Not detected in pore water in any reach.
Benzo(a)anthracene	Yes	Exceeds the EQ_PART in 3.01, 3.03, 3.04, 4.01, and 4.02; the ER_L and/or the REG_IV (CLP PQL) in 3.02, 3.03, and 18; but not the ER_M in any reach. Not detected in pore water in any reach.
Benzo(a)pyrene	Yes	Exceeds the EQ_PART in 2.04, 3.03, 3.04, 4.02, 4.03, 4.04, 7 and 8; the ER_L and/or the REG_IV (CLP PQL) in 2.04, 3.02, 3.03, 4.03, 4.04, 7, 8, and 13; but not the ER_M in any reach. Not detected in pore water in any reach.
Bis(2-ethylhexyl) phthalate	Yes	Detected in sediments in all reaches but did not exceed benchmarks. Exceeds the Daphnid LCV and LTV in 3.03 and 4.01, but not the SAV or SCV.
Chrysene	Yes	Exceeds the REG_IV (CLP PQL) and ER_L in 3.02, 3.03, and 13. Does not exceed the ER_M in any reach. Not detected in pore water in any reach.
Dibenz(a,h)anthracene	Yes	Exceeds the ER_L in 3.03, 3.04, and 4.01, but does not exceed the REG_IV (CLP PQL) or the ER_M, which is less than the CLP PQL, in any reach. Not detected in pore water in any reach.
Fluoranthene	Yes	Exceeds the REG_IV and/or ER_L in 2.04, 3.02, 3.03, and 18. Does not exceeds the ER_M or EPASQC01 in any reach. Not detected in pore water in any reach.
Fluorene	Yes	Exceeds the ER_L in 0, 3.02, 3.03, 3.04, 4.03, and 13, but not the REG_IV (CLP PQL) or the ER_M in any reach. Not detected in pore water in any reach.
Naphthalene	Yes	Exceeds the ER_L and/or REG_IV (CLP PQL) in 0, 2.04, 3.02, 3.03, 3.04, and 13, and the EQ_PART and ER_M in 13. Not detected in pore water in any reach.

Table 6.8 (continued)

Chemical	COPEC	Reason for inclusion or rejection
PAHs, Total	Yes	Exceeds the REG_IV in reaches 0, 3, 4, 7, 8, 13, and 18 and in subreach 2.04; the ER-L in 0, 2.04, 3.02, 3.04, 4.01, 4.04, 7, 8, 13, and 18. Did not exceed the ER-M in any reach. Not detected in pore water in any reach.
Aroclor 1248	Yes	Exceeds the ER_L in subreaches 3.02 and 3.04, the ER-M in 3.02, but not the EQ_PART in any reach. Not detected in pore water in any reach.
Aroclor 1254	Yes	Exceeds the ER_L and/or the EQ_PART, in 2.02, 3.02, 3.03, 3.04, 4, and 22. Exceeds the ER_M in 2.02, 3.02, 3.03, 3.04, 4.01, 4.04, and 22. Exceeds several benchmarks, including the SAC and SCV, in pore water in reach 18.
Aroclor 1260	Yes	Exceeds the ER_L, but not the EQ_PART, in 2.02, 3.02, 3.03, 3.04, 4.01, 4.02, and 4.03. Exceeds the ER_M in 2.02, 3.02, 3.03, 3.04, and 4.01. Not detected in pore water in any reach.
Phenanthrene	Yes	Exceeds the ER_L and/or REG_IV (CLP PQL) in 0, 2.04, 3.02, 3.03, 3.04, and 13. Exceeds the ER_M in 13, but does exceed and the EPASQC01 in any reach. Not detected in pore water in any reach.
Pyrene	Yes	Exceeds the ER_L in 3.02 and 3.03, but not the ER_M in any reach. Not detected in pore water in any reach.

Maximum concentrations of COPECs in pore water (Fig. 6.12) appear to be acutely toxic to Daphnids in only four subreaches, including Upper Melton Hill (reach 0), and potentially chronically toxic to Daphnids (i.e., ΣTU>0.2) in all reaches except subreach 2.03. Manganese is the principle contributor to toxicity in most reaches. Only subreach 4.04 pore water remains acutely toxic to Daphnids when manganese is removed, but the maximum pore water concentrations in all reaches in Poplar Creek and the Lower Clinch River remain chronically toxic to Daphnids. Copper also contributes to toxicity in all reaches. Median concentrations of COPECs are also potentially chronically toxic to Daphnids in most reaches, but acute toxicity is not indicated (Fig. 6.13). Pore water toxicity generally decreases along Poplar Creek (reaches 13 and 3), but increases along reach 4.

Maximum and median concentrations of COPECs in whole sediment appear to be toxic in every reach, and every COPEC contributed to the total toxicity (i.e., maximum TU>0.01). To aid interpretation, the median ΣTUs for each of four general categories of COPECs and mercury are presented in Figure 6.14a. The PAH category is the sum of TUs for the individual PAHs, rather than the Total PAH concentration divided by the Total PAH ER-L. Median TUs for individual COPECs are presented in Figs. 6.14b-d. Individual pesticides were not presented because the only pesticides detected were 4,4'-DDD and 4,4'-DDT, which were detected in only one sample from reach 1. The magnitude of the TUs (e.g., median ΣTUs >50 in several reaches) indicates that the sediments are extremely toxic and/or the effects concentrations (i.e., the ER-Ls) are extremely conservative for these sediments. Even if the absolute magnitude is not accurate the relative magnitudes evince some interesting differences among reaches. For example, pesticides occur in reach 1 only, mercury appears to be the principle contributor to toxicity in reach 3, PAHs contribute most in Poplar Creek above East Fork (reach 13), PCBs are the principle contributors in subreaches 2.02 and 4.04 and significant contributors in Poplar Creek (subreaches 3.02 and 3.04).

Exposure effects profiles for sediment exposures. For each COPEC, the distribution of observed concentrations in whole sediment and pore water is compared to the distribution of concentrations toxic to aquatic biota. The interpretation of the relationship of these distributions is presented in Sect. 6.1.6. Toxicity information is drawn from the toxicity profiles (Appendix F1). If sufficient toxicity data exist, the empirical distribution functions are presented graphically (Appendix F3). Aqueous effects distributions (Figs. F3.1–F3.5) were derived from standard toxicity test data. Aqueous toxicity data for hardness dependant metals were adjusted to a hardness of 100 mg/l. There is relatively little standardized sediment test data available. Therefore, the distribution of observed concentrations is compared to the ER-L and ER-M from Long et al. (1995) and two effects distributions, community level effects and lethality (Figs. F3.6–F3.16). The effects distributions were derived using the marine and estuarine toxicity data from McDonald et al. (1994) and freshwater data from Long and Morgan (1991). Although data from studies of salt water sediments may not seem relevant to freshwater sediments, these data have been recommended by EPA Region IV (1994). Their use may be justified on the basis of the lack of a high quality data set for freshwater and the apparently small difference in the toxicity of many chemicals between the two media relative to the differences among sites within a medium.

Some level of effect would be expected to rarely occur at concentrations below the ER-L (i.e., a negligible risk), occasional occur at concentrations between the ER-L and ER-M (i.e., a marginal risk), and usually occur at concentrations above the ER-M (i.e., a significant risk). However, the ER-L and ER-M are derived from the Biological Effects Database for Sediments (BEDS), which is an assemblage of data from studies of marine and estuarine systems. The primary study types were spiked sediment bioassays and co-occurrence analysis. These studies evaluated benthic community structure and a wide variety of species (e.g., sandworms, polychaetes, oligochaetes, sea urchins, bivalves, shrimp, flounder, amphipods, etc.). End points included taxa richness, diversity, density, mortality, growth, respiration, behavior (e.g., avoidance, emergence, reburial, etc.) and suborganismal effects (e.g., Mixed-Function

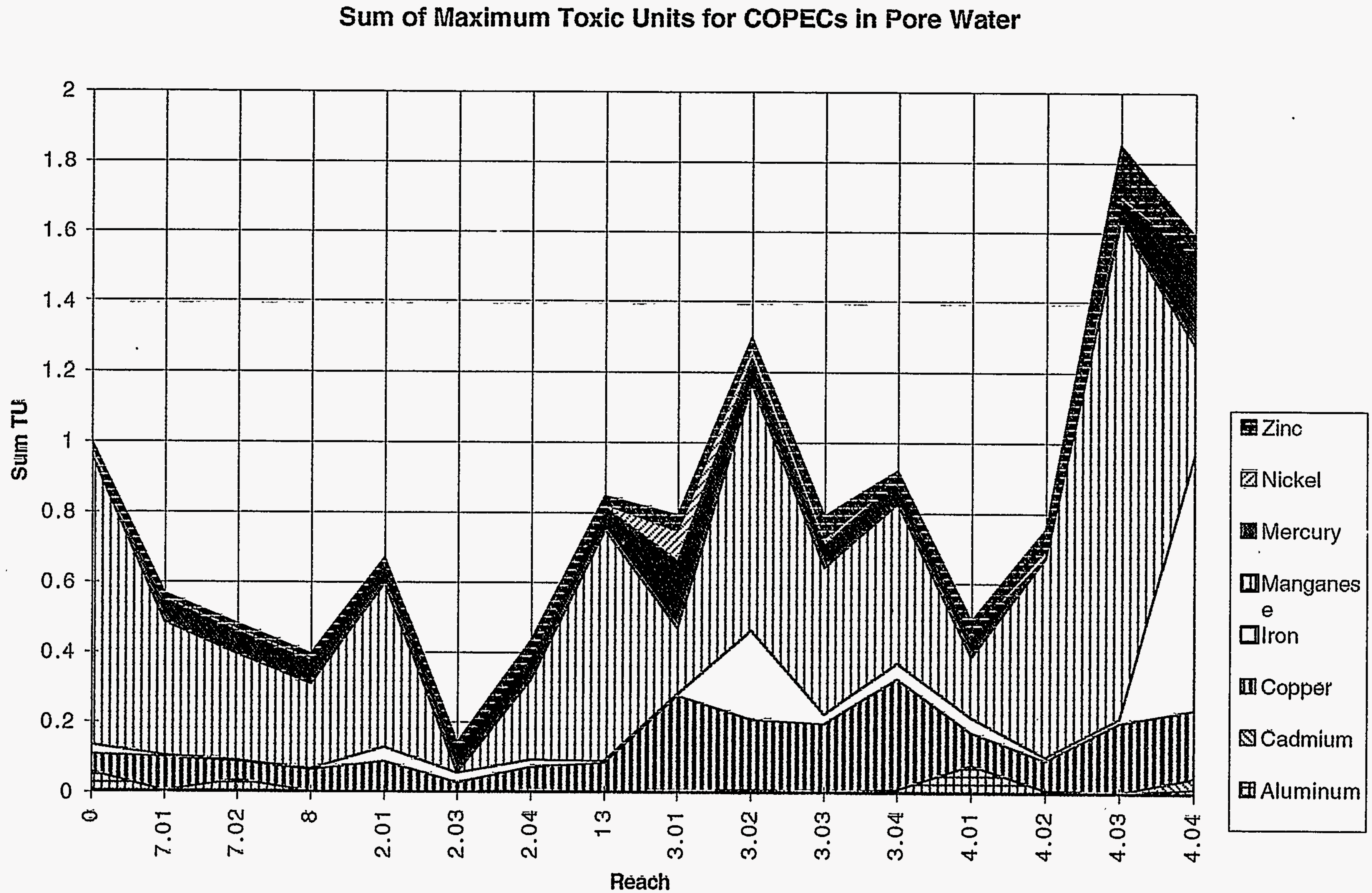


Fig. 6.12. Plot of maximum concentration of pore water contaminants of potential ecological concern normalized to toxic units (TU). The upper contour of the graph is the sum of TU for the reaches. Includes contaminants of potential ecological concern with maximum TUs >0.01.

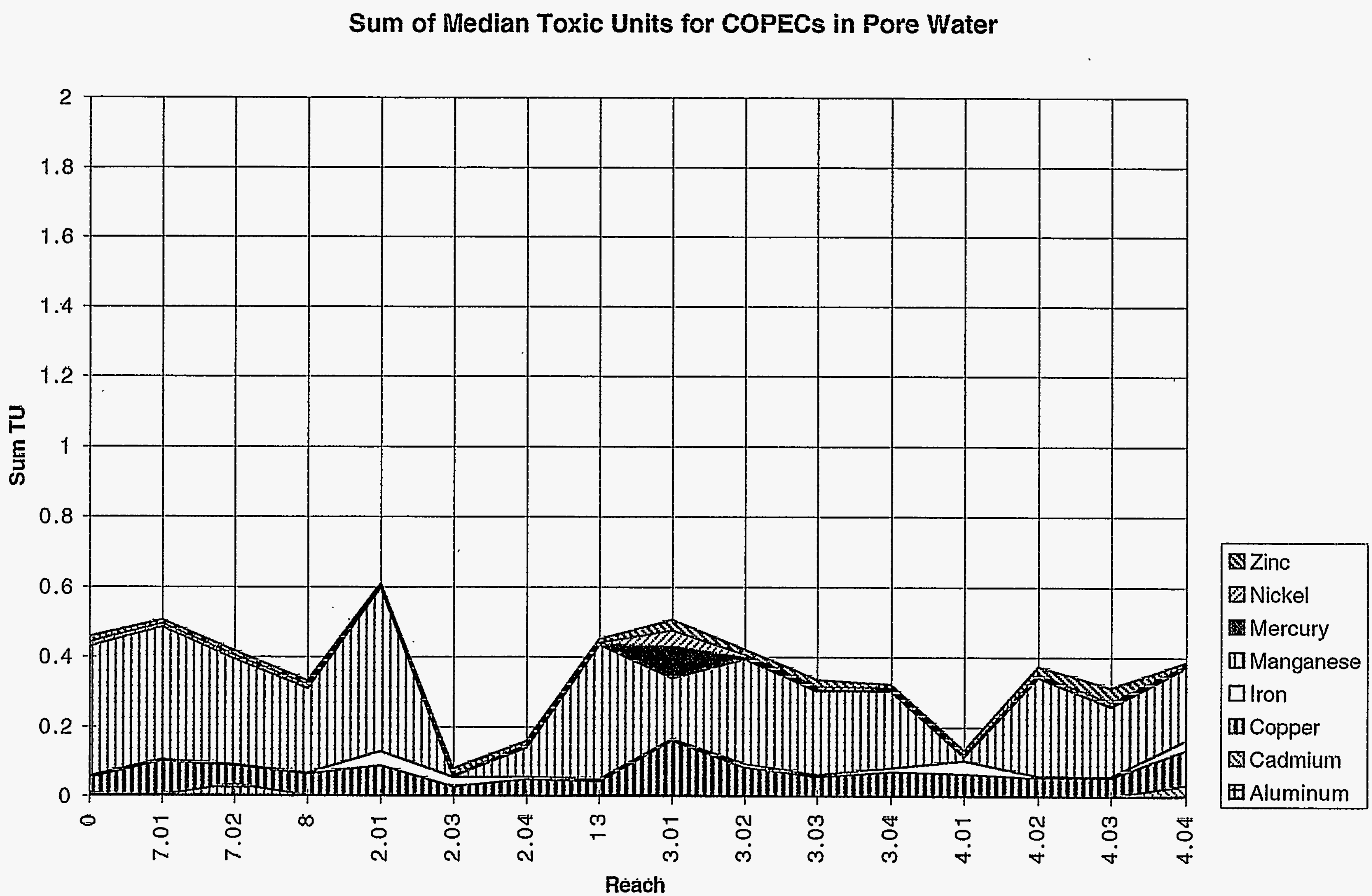


Fig. 6.13. Plot of median concentration of pore water contaminants of potential ecological concern normalized to toxic units (TU). The upper contour of the graph is the sum of TU for the reaches. Includes contaminants of potential ecological concern with maximum TUs >0.01.

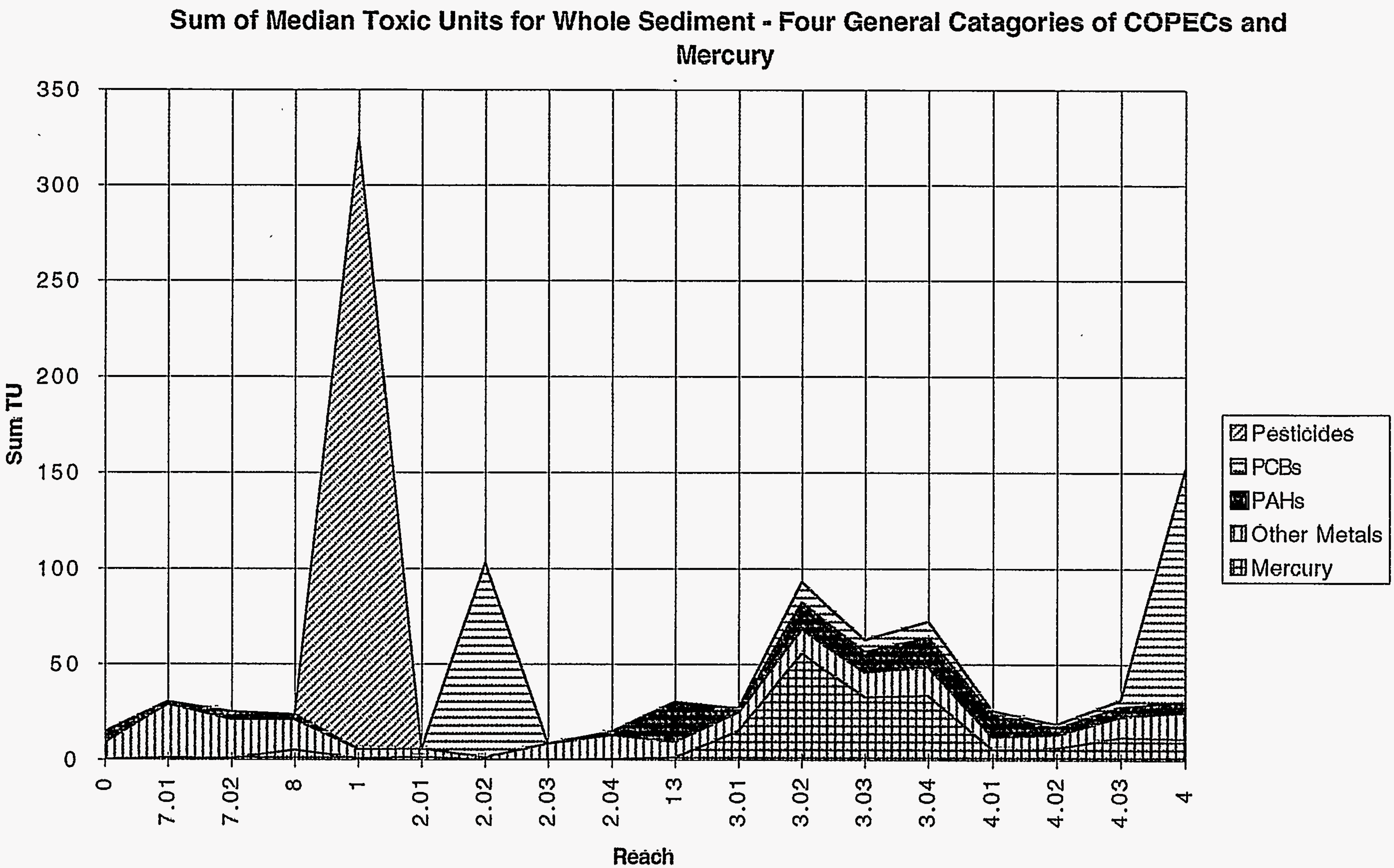


Fig. 6.14a. Plot of median concentration of whole sediment contaminants of potential ecological concern normalized to toxic units (TU). The upper contour of the graph is the sum of TU for the reaches. Includes contaminants of potential ecological concern with maximum TUs >0.01. This graph (a) is four general categories of contaminants of potential ecological concern and mercury.

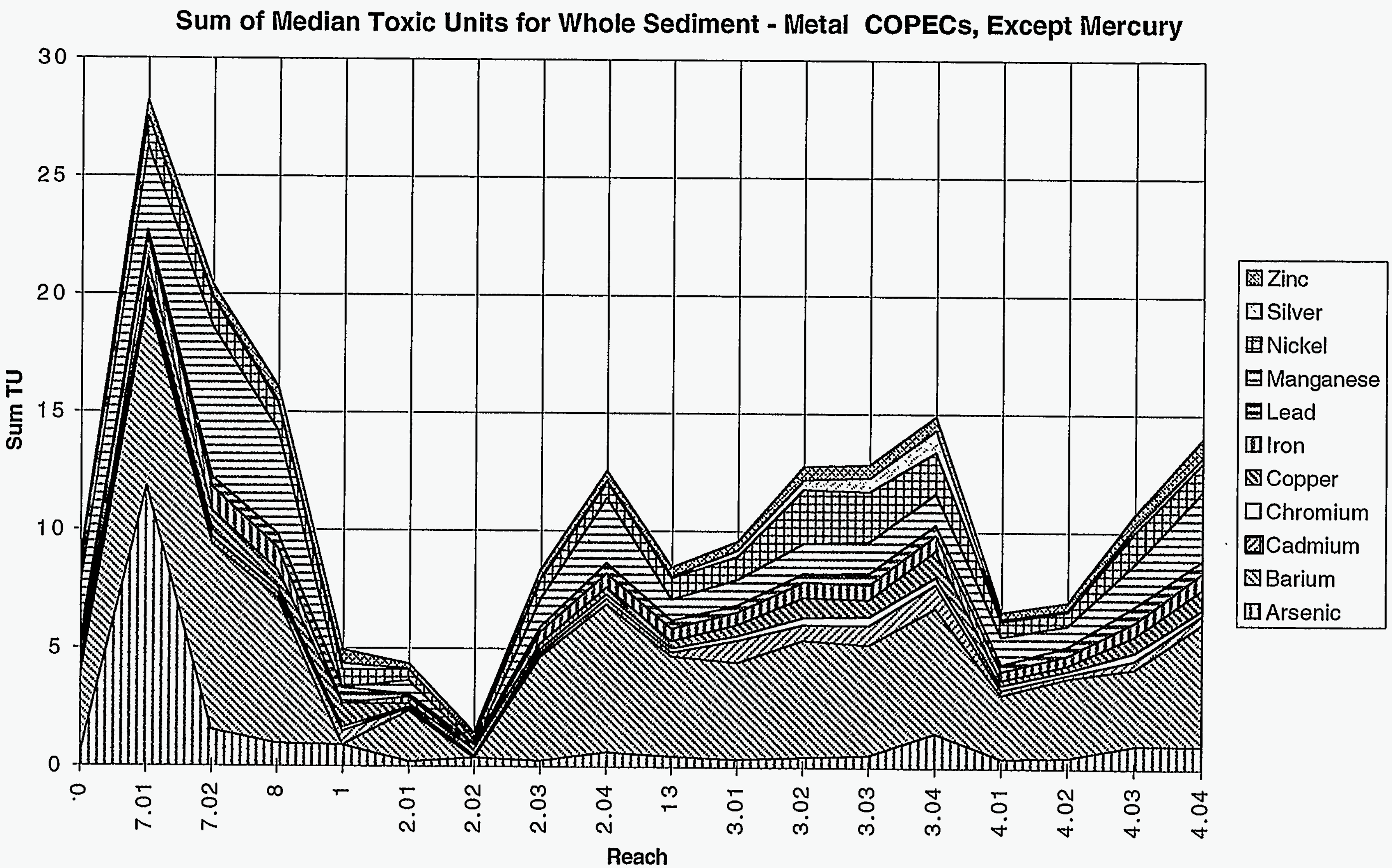


Fig. 6.14b. Plot of median concentration of whole sediment contaminants of potential ecological concern normalized to toxic units (TU). The upper contour of the graph is the sum of TU for the reaches. Includes contaminants of potential ecological concern with maximum TUs >0.01. This graph (b) is metal contaminants of potential ecological concern, except mercury.

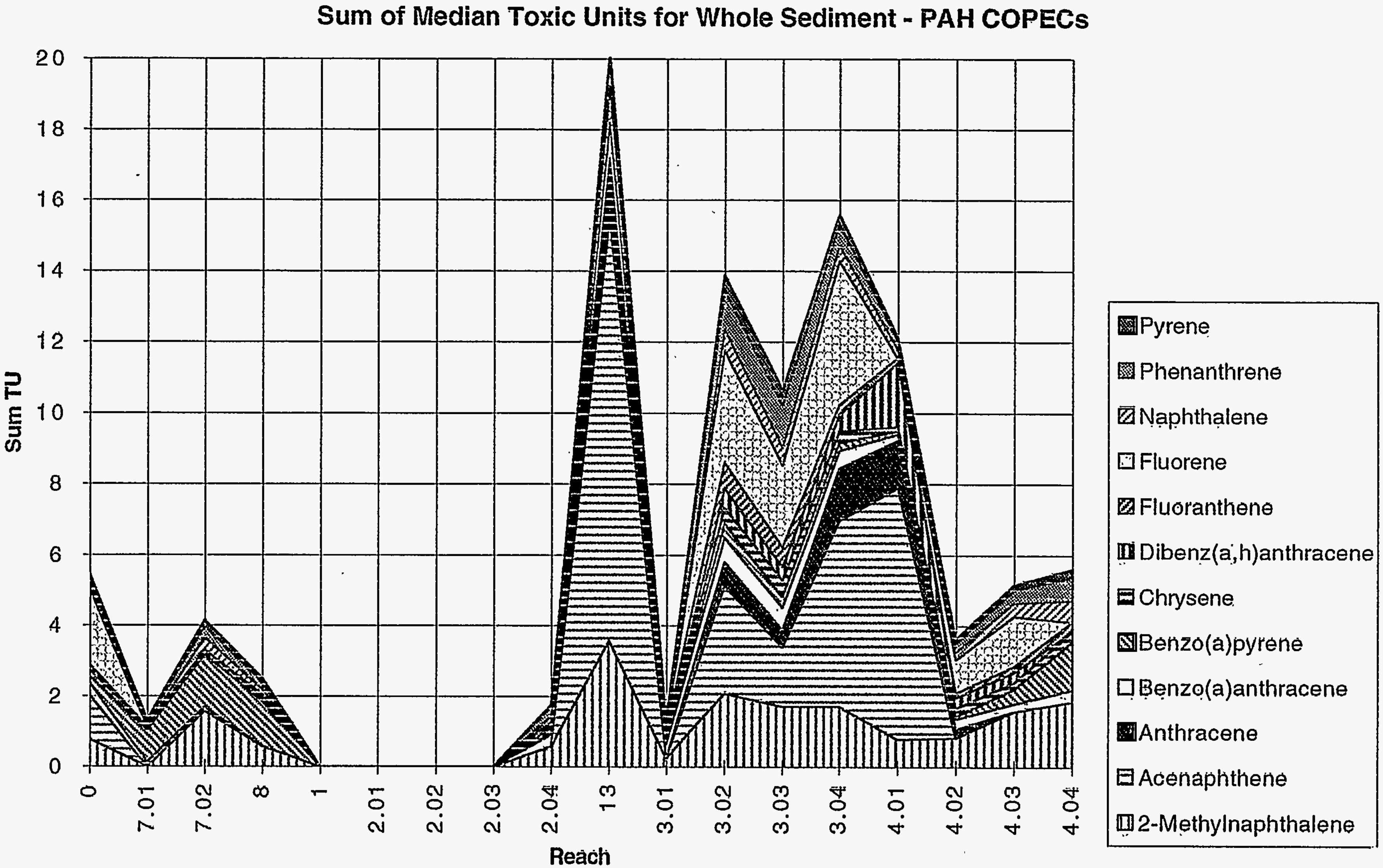


Fig. 6.14c. Plot of median concentration of whole sediment contaminants of potential ecological concern normalized to toxic units (TU). The upper contour of the graph is the sum of TU for the reaches. Includes contaminants of potential ecological concern with maximum TUs >0.01. This graph (c) is polycyclic aromatic hydrocarbon contaminants of potential ecological concern.

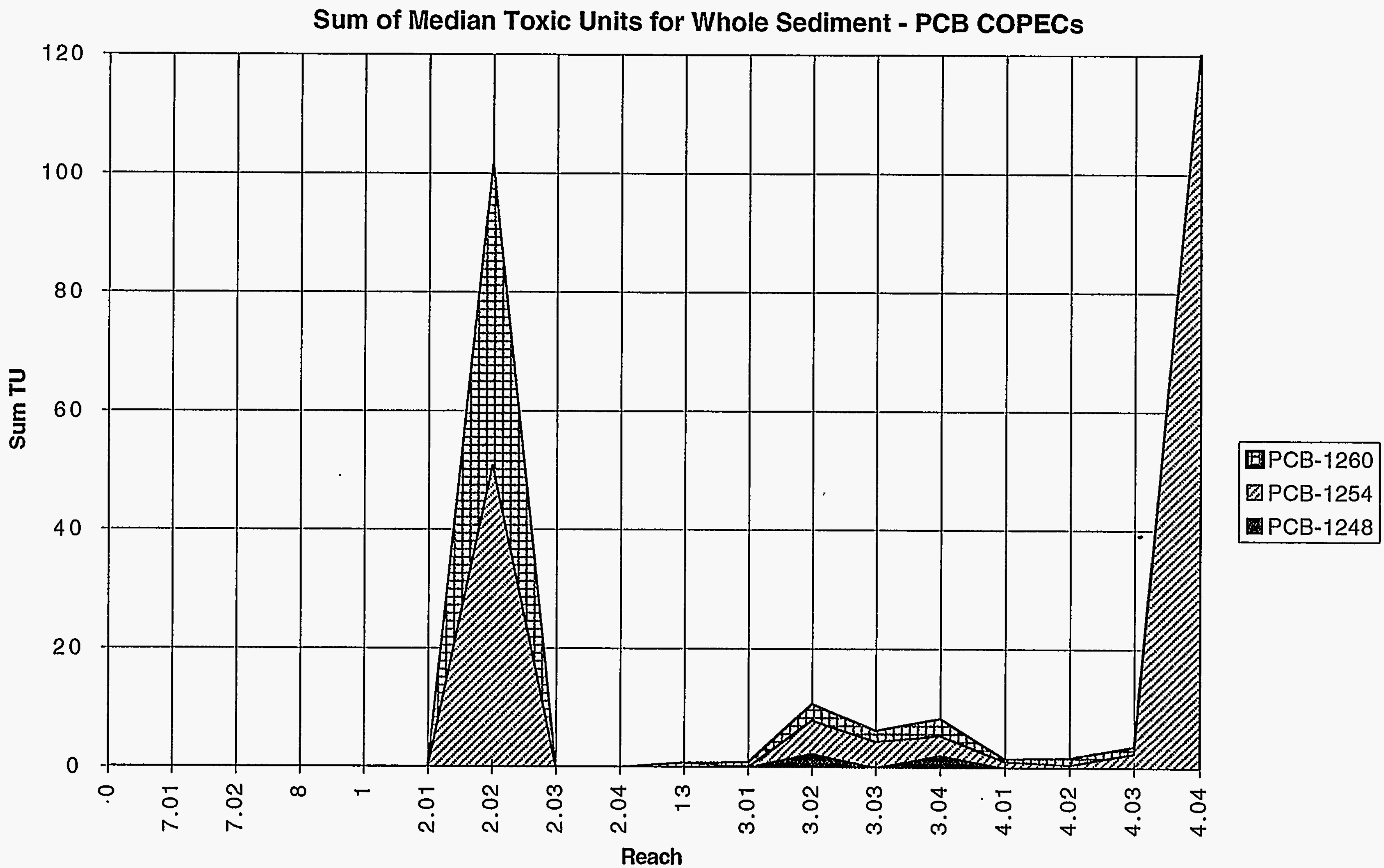


Fig. 6.14d. Plot of median concentration of whole sediment COPECs normalized to toxic units (TU). The Upper contour of the graph is the sum of TU for the reaches. Includes COPECs with Maximum TUs greater than 0.01. This graph (d) is PCB COPECs.

Oxidase, abnormal chromosomes, etc.). Hence, the ER-Ls and ER-Ms include many sources of variance as a result of the disparate sources of data and the variability inherent to the toxicity tests. Factors that differ between studies may include the physical characteristics of the sediment (i.e., percent organic matter, texture, pH, pore water hardness, acid-volatile sulfide content, redox conditions, etc.), behavior and exposure of the test organisms (i.e., filter feeders, burrowers, epibenthic organisms, ingestion of sediment, respiration of pore water, etc.), sensitivity of the test organisms, and the chemical species present. Factors inherent to the test type may include effects of other contaminants in the co-occurrence studies and the bioavailability of contaminants in spiked bioassay tests. Spiked sediment tests may over estimate bioavailablity because the chemical forms tested may dissociate more readily than the forms occurring at the site. Also, sorption to solids may bind less of the tested form, or bind it less securely.

The two effects distributions also were derived from the BEDS (MacDonald et al. 1994). Freshwater sediment data from Long and Morgan (1991) were included also and only concentrations associated with effects were used. Measures of community effects included taxa richness and abundance of benthic invertebrates. The lethal effects distributions potentially included mortality data from all study types and tested species. Thus, exceeding the lowest sediment effects concentration or the ER-L is not necessarily indicative of toxicity. The rationale and approach for interpreting the relationship of the exposure and effects distributions are presented in Sect. 6.1.6. In this assessment, risks to benthic invertebrates from COPECs are characterized as significant, marginal, or negligible based on the likelihood of effects and the evaluation of modifying factors. Significant risks are generally indicated if there is a >50% likelihood of toxic effects (i.e., the observed concentration exceeds the 50th percentile of the effects distribution). Marginal risks are generally indicated if there is a >20%, but <50%, likelihood of toxic effects. Negligible risks are generally indicated if the likelihood of toxic effects is <20%. This interpretation is consistent with the approach developed in Long and Morgan (1991), Long et al. (1995), and McDonald et al. (1994). The final characterization of risks based on single chemical concentrations will be determined by evaluating the risks suggested by the comparisons to the ER-L and ER-M, the risks suggested by the comparison to the two effects distributions, and the factors that modify toxicity. Factors that may modify the characterization of risk include concordance of effects and exposure concentrations and the relative sensitivities of freshwater and marine species. The spatial extent of the potential risk is also characterized, as discussed in Sect. 6.1.6.

Table 6.8 summarizes, for each COPEC, which benchmarks were exceeded in each medium (e.g., sediment and pore water) and subreach. Tables F3.1 and F3.2 detail the maximum observed concentration for each reach, benchmark values, and the corresponding HQs. Therefore, this section addresses only that information which aids in characterizing the risks associated with these COPECs. The metal COPECs are discussed separately and then summarized as a group. PAHs are addressed as a group with additional discussion of the three PAHs with Sediment Quality Criteria. Organic COPECs are addressed in one of the following groups: pesticides, phthalate esters, and total PCBs.

Aluminum. Aluminum in pore water presents a negligible risk at most sites in the OU. However, pore water concentrations may be a marginal risk at localized sites in Poplar Creek below the ash disposal area (subreach 3.04), in several areas of the lower Clinch River (reach 4), in Lower McCoy Branch (7.02), and in the negative reference reach (reach 0). Maximum pore water concentrations exceeded the chronic NAWQC in reach 0 and in subreaches 3.04, 4.01, 4.02, 4.04, and 7.02. Only subreach 4.01 exceeded a CV for an invertebrate species. Pore water concentrations are from filtered samples. Therefore, they represent exposure to dissolved aluminum and indicate a possibility of actual toxic effects. Al was detected in most samples from these reaches, but the median concentration was less than all chronic effects values in all reaches. Sediment concentrations of Al have not been shown to be toxic. That is, effects observed in co-occurrence analyses of contaminated sediments have not been attributed to Al and spiked sediment toxicity tests were not found.

Arsenic. Arsenic concentrations are highest in areas highly contaminated with coal ash (subreaches 7.01, 7.02, and 3.04), intermediate in areas where ash and ash-associated arsenic are diluted (reach 1 and subreaches 4.03 and 4.04), and lower in other areas, but still within the range of concentrations associated with toxicity in sediments. These lower concentrations may reflect regional arsenic levels.

The distributions of arsenic concentrations in subreaches 3.04 and 7.02 and the single arsenic concentration from subreach 7.01 lie in the range in which community level effects and lethality occur at most other sites (Fig. F3.6b and d). Hence, the coal ash disposal areas in McCoy Branch and the gooseneck bend in Poplar Creek appear to present a significant risk to benthic invertebrates in reach 7 and subreach 3.04. The intermediate sites (reach 1 and subreaches 4.03 and 4.04) fall above the community level effects distribution and at approximately the middle of the lethality distribution (Fig. F3.6a and c). These distributions suggest that risks to benthic invertebrates in these reaches may be increased by arsenic from the coal ash disposal areas in McCoy Branch and at the Kingston steamplant. However, only two studies reporting arsenic effects at the community level were found. Also, arsenic was not speciated in the effects data set and it is not clear whether the forms of arsenic at the sites used to generate the effects distribution are sufficiently similar to the forms in the coal ash-contaminated sediments to be relevant. Arsenic appears to present a marginal risk at most other sites in the OU. Except as noted previously, arsenic concentrations in most reaches in the CR/PC system appear to be relatively similar to the negative reference reaches 0 and 8. Although the data are limited, this may indicate that the reported effects concentrations are too conservative for CR/PC sediments. This is supported by the inconsistent effects reported in the sediment toxicity literature at these concentrations (Appendix F1). Also, only subreach 7.02 has sediment arsenic concentrations that exceeded the ER-M from Long et al. (1995) and arsenic was detected in pore water in all reaches but did not exceed any aquatic benchmarks.

Barium. Barium in sediment and pore water does not appear to present a risk to benthic invertebrates in the OU. The only sediment benchmark is a background value from EPA Region V, which has no toxicological basis. The Region V background level was exceeded at all sites and is apparently not representative of local background. There is some limited toxicity data for aqueous barium concentrations. The Secondary Acute Values (SAVs) and SCVs are calculated to be conservative because there is little toxicity data. The SCV was exceeded at all sites and appears to be too low for local conditions. Although not exceeded at every site, the SAV was exceeded in two negative reference reaches and none of the acute values used in its derivation were exceeded in any reach. Furthermore, the observed pore water concentrations were at least an order of magnitude less than the lowest CV found for any aquatic organism.

Boron. No data are available to evaluate the toxicity of boron in whole sediments, but boron in pore water appears to present a negligible risk to benthic invertebrates in the OU. Concentrations in pore water in all reaches, including references, exceeded the LCV for daphnids. That benchmark is apparently below the local background due to the compounding of effects across all life stages in its derivation. Boron does appear to be elevated in Poplar Creek below the ash disposal area, which is the only subreach (3.04) in which the maximum concentration exceeds any other benchmarks (SCV). The maximum concentration does not exceed the LCV for daphnids or the SAV and the median concentration is less than the SCV. Also, the maximum concentration is less than the toxicity values used in its derivation. Given that, boron is unlikely to be toxic to benthic invertebrates in the subreach 3.04.

Cadmium. Cadmium appears to present a marginal risk in Poplar Creek below East Fork and at very localized hot spots in subreach 4.04, and a negligible risk in all other reaches, including negative references. The distributions of ambient Cd concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.7(a-d). The minimum observed concentration in all reaches approximates or exceeds the lowest lethal effects concentration, but the median observed concentration is never more than the median lethal concentration. That is, the distribution of observed Cd crosses the effects distribution below the 50th percentile. None of the observed Cd concentration distributions cross the community level effects distribution. The plot of median ΣTUs in whole sediment (Fig. 6.14b) indicates that reach 3 is impacted the most by cadmium. Cd was detected in one of eleven pore water samples from subreach 4.04. This concentration and the aqueous test end points for Cd are shown in Fig. F3.1. Though the ER-M from Long et al. (1995) is not exceeded at any site, freshwater species are reported to be more sensitive to Cd than marine species (Appendix F1). However, Cd concentrations exceed the relatively conservative ER-L in reach 3 and at sites in subreach 4.04 only.

Cd concentrations in Poplar Creek sediment appear to be more toxic below EFPC (reach 3) than above the confluence (reach 13). Although the data are limited, toxicity from Cd in sediment appears to increase below the confluence with Mitchell Branch also. A consistent concentration gradient is not evident within reach 3 (Fig. F3.7b). However, these results suggest that East Fork and Mitchell branch are the primary contributors to risks from Cd in sediment. Contributions to risk from Mitchell Branch and the ash disposal area can not be distinguished from East Fork. The distribution of concentrations in reach 3 suggest that community level and lethal effects are not probable at any of the sampled sites, but are possible at roughly half of the sampled sites.

The distributions of Cd concentrations in all other reaches were more similar to each other, and indicative of a more homogenous distribution of Cd, than the distributions observed in reach 3 (Fig. F3.2a-d). The Clinch River and McCoy Branch distributions suggest that Cd toxicity is unlikely in these reaches. These may be local background concentrations, because there does not appear to be any relationship to potential sources.

Cd was observed at the detection limit in one of the eleven pore water samples from subreach 4.04. This value exceeded half the invertebrate CVs, which were the lowest aqueous toxicity values graphed, and was marginally lower than the chronic NAWQC (Fig. F3.1). The maximum sediment concentration in 4.04 was not continuous with the majority of the distribution (Fig. F3.7c). These results suggest that concentrations of Cd in sediment or pore water at localized hot-spots within subreach 4.04 may be toxic.

Chromium. Chromium appears to present marginal to significant risks in localized areas in Poplar Creek below Mitchell Branch, but negligible risks in most of Poplar Creek and the other reaches. The distributions of ambient Cr concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.8(a-d). The plot of median ΣTUs in whole sediment (Fig. 6.14b) indicates that reaches 3 and 4 are similarly impacted by Cr and that Cr is one of the least important metal COPECs. The sediment effects data are relatively inconsistent, especially below the ER-M (Long et al. 1995), which was not exceeded at any site. Cr was detected in pore water in all reaches but does not appear to be toxic.

Cr concentrations in Poplar Creek sediment above Mitchell Branch cover a very narrow range and suggest negligible toxicity in these reaches (subreach 3.01 and reach 13). The 100th percentile for each subreach shows a downstream gradient below Mitchell Branch, but the distributions as a whole are indistinguishable from each other (Fig. F3.8b). These results suggest that lethal effects of Cr are toxicity is possible to likely at localized hot-spots in Poplar Creek below Mitchell Branch. Community level effects are possible, though unlikely, at localized hot-spots in subreaches 3.02 and 3.03, but are unlikely at most sites in Poplar Creek. This also suggests that Mitchell Branch is the primary contributor to risks to benthic invertebrates from exposure to Cr in sediment.

Cr lethality is unlikely in most of the Clinch River and McCoy Branch Embayment (Figs. F3.8a,c, and d). Cr lethality is possible at <20% of the sites in lower McCoy Branch Embayment (7.02) and the

Clinch River below the Emory River (4.03). Community level effects are unlikely in the Clinch River and McCoy Branch Embayment.

Cobalt. Risks to benthic invertebrates from cobalt are uncertain. Co in pore water appears to be chronically toxic in parts of Poplar Creek and the Clinch River. Although Co concentrations in Poplar Creek increase below East Fork, a concentration gradient is not evident and Co appears to be toxic above the OU (reach 13) as well. Co concentrations in pore water in the Clinch River below the Emory River (4.03 and 4.04) also exceeded benchmarks. However, very little aqueous toxicity data, and no sediment toxicity data, were found (Appendix F1). All of the aquatic benchmarks were derived from one study using *D. magna* and fathead minnows. Hence, the precision of the toxicity values and influence of water hardness are unknown.

Copper. Copper appears to present a marginal risk at most sites in Poplar Creek below Mitchell Branch; a marginal risk in localized areas of the Clinch River below the Emory River and in Lower McCoy Branch Embayment; and a negligible risk in all other reaches. The distributions of ambient Cu concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.9(a-d). The minimum observed concentration in all reaches approximates or exceeds the lowest lethal effects concentration. Pore water concentrations for these reaches and the aqueous test end points for Cu are shown in Fig. F3.2.

Cu concentrations in Poplar Creek sediment appear to be more toxic below Mitchell Branch than in reach 13 and subreach 3.01 (Fig. F3.9b). Although a concentration gradient is not evident, these results suggest that Mitchell Branch is the primary contributor to risks to benthic invertebrates from Cu in sediment. The effects data are only moderately consistent, especially below the ER-M (Long et al. 1995), which was not exceeded at any site. These results suggest that Cu in sediment may be toxic at most sites below Mitchell Branch, but toxicity is not likely in any reach. The distributions of Cu concentrations in pore water are indicative of localized chronic toxicity: about 20% of the samples appear to be chronically toxic. However, there is no evidence that pore water is more toxic below Mitchell Branch.

Cu in sediment appears to be a negligible risk to benthic invertebrates in most of the Clinch River and McCoy Branch Embayment (Figs. F3.9a,c, and d). The distributions in sediment suggest that risks from Cu are higher below the Emory River (subreaches 4.03 and 4.04) and adjacent to the ORR (reach 1) than in adjoining reaches. Cu toxicity is possible but unlikely in reach 1 and in subreaches 4.03 and 4.04. Pore water concentrations also suggest chronic toxicity in the Clinch River below the Emory River.

Iron. Iron in sediment and pore water appears to present a marginal risk to benthic invertebrates in subreaches 3.02 and 4.04, which the plot of maximum ΣTUs in pore water (Fig. 6.12) indicates are impacted the most by Fe. Maximum sediment concentrations in subreaches 3.03 and 4.04 and in reach 7 marginally exceeded the Lowest Effect Level for Fe in Ontario sediments (Table 6.8). This indicates a slight risk in these areas. Maximum pore water concentrations of Fe exceeded toxicity benchmarks in several reaches. Most of the toxicity values are based on Fe in acidic effluents, which are not present in this OU. The maximum concentration in 4.04, however, also exceeded the one CV for *D. magna* in circum neutral water (Sect. 6.2.2.1). These are dissolved Fe concentrations and represent the dissolved fraction in the pore water.

Lead. Lead in OU sediments does not appear to contribute to risks to benthic invertebrates. There appears to be a negligible risk of Pb toxicity at most sites in the CR/PC System and a marginal risk in very localized areas of the Clinch River below Poplar Creek, subreach 2.04, and subreach 3.02. The distributions of ambient Pb concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.10(a-d). The only trend evident from these distributions is that Pb concentrations appear to increase downstream in the Clinch River below Poplar Creek (Fig. F3.10c). Hence, the apparent risks to benthic invertebrates from Pb in Clinch River sediment are not consistent with any known contaminant sources.

Manganese. There is only weak evidence of increased risks from Mn released from the OU, given the uncertainty in the toxicity data and the lack of concordance with known contaminant sources. The plots of median ΣTUs in whole sediment (Fig. 6.14b) and pore water (Fig. 6.13) indicate that Mn is potentially toxic in almost every reach, including negative reference reaches. Hence, there does not appear to be any relationship between known contaminant sources and the potentially toxic levels of Mn. Also, very little aqueous or sediment toxicity data were found (Appendix F1). All of the aquatic benchmarks were derived from one study using *D. magna* and fathead minnows, and the only sediment toxicity information is for sediments in Ontario, Canada. Given the uncertainty in the toxicity data and the lack of concordance with known contaminant sources, there is only weak evidence of increased risks from Mn released from the OU.

Mercury. Mercury in sediment appears to be a significant risk to benthic invertebrates in Poplar Creek, the lower Clinch River, and Walker Branch Embayment; a negligible risk in subreaches 2.02 and 2.03 and most of reach 0; and a marginal risk in all other reaches. The distributions of ambient Hg concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.11(a-d). The plots of maximum ΣTUs in whole sediment (Fig. 6.14a) and pore water (Fig. 6.13) indicate that reaches 3 and 4 are impacted the most by Hg. Pore water concentrations and the aqueous test end points for Hg are shown in Fig. F3.3. The sediment effects distribution may be overly conservative for Clinch River sediment, since toxicity appears to be possible and/or probable at upstream and negative reference sites (reaches 0, 8, and 13) as well. This is supported by the lack of consistency in the toxicity data: biological effects were observed in only 42% of the documented studies with concentrations greater than the ER-M (Appendix F1). However, effects were nearly always observed in freshwater studies at concentrations prevalent in reaches 3 and 4 (i.e., 1–100 mg/kg).

There are two general classes of mercury: organic mercury, including methyl mercury, and inorganic mercury, including metallic mercury, mercuric chloride, and mercuric sulfide (EPRI 1987). Methyl mercury concentrations were measured in sediment and pore water samples from the CR/PC OU. Speciation of inorganic mercury was not performed as part of this investigation. However, speciation and bioavailability studies of mercury in EFPC floodplain soils were performed as part of the Lower EFPC RI (DOE 1994a; Barnett and Turner 1995). In general, these studies indicated that the dominant form was mercuric sulfide, that mercuric chloride constituted <1 % of the total mercury, and that mercuric chloride was the most soluble, and hence most bioavailable, form of inorganic mercury. It is likely that the composition of inorganic mercury in sediments downstream of EFPC is comparable to that in the Lower EFPC floodplain soils because they both have EFPC surface water and sediment as the primary sources of mercury.

Methyl mercury is the most toxic of the mercury species (Eisler 1987 and EPRI 1987). However, pore water concentrations of methyl mercury in the CR/PC did not exceed aqueous toxicity benchmarks. Mercury was not speciated in the sediment effects data. The proportion of inorganic mercury represented by mercuric chloride increases with increasing chloride ion concentrations, but the proportion of mercuric sulfide changes little over the range of naturally occurring sulfide ion concentrations (EPRI 1987). Nearly all of the sediment effects data are from marine and estuarine systems. Hence, it is likely that the sediment benchmarks and effects distributions represent exposures to more soluble, and thus more toxic, mixtures of inorganic mercury species than the exposures received by benthos in this OU.

Mercury concentrations in Poplar Creek sediment appear to be more toxic below Mitchell Branch than in reach 13 and subreach 3.01 (Fig. F3.11b). However, there is only limited data from subreach

3.01, the distribution spans two orders of magnitude, and it overlaps the other distributions for Poplar Creek. The upper end of the distribution in subreach 3.04 (above the 60th percentile) is not continuous with the rest of the distribution. This indicates a more heterogenous distribution of Hg than is observed in subreaches 3.02 and 3.03 and suggests that the ash disposal area may contribute to risks from exposure to Hg in some sediments of subreach 3.04. These results suggest significant risks to benthic invertebrates from Hg in sediment at nearly every site below East Fork and Mitchell Branch.

Hg concentrations are likely to be toxic at nearly every site in the Clinch River below the Emory River (subreaches 4.03 and 4.04) and at most sites in subreaches 4.01 and 4.02 (Fig. F3.11c). The distributions of ambient Hg in sediment appear to be marginally toxic in all other reaches (Figs. F3.11a and d).

Nickel. In general, sediment concentrations of nickel appear to present a significant risk at most sites in Poplar Creek below Mitchell Branch, and a marginal risk at most sites in the Clinch River below the Emory River and in Lower McCoy Branch Embayment. Ni in sediment appears to present a significant risk of community level effects in Poplar Creek below Mitchell Branch, in the Clinch River below the Emory River, and in Lower McCoy Branch Embayment. Negligible to marginal risks of community level effects are indicated for all other reaches. Risks of lethal effects are significant for localized areas of Poplar Creek below Mitchell Branch; marginal for most site in Polar Creek below Mitchell branch and the Clinch River below the Emory River, and localized areas of Lower McCoy Branch Embayment; and negligible for all other reaches, including reach 0. The distributions of ambient Ni concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.12(a-d). The plots of median ΣTUs in whole sediment (Fig. 6.14b) and pore water (Fig. 6.13) indicate that Poplar Creek is impacted the most by Ni. Pore water concentrations and the aqueous test end points for Ni are shown in Fig. F3.4. Interpretation of the community level effects distribution is uncertain because of the limited data and extreme slope of the distribution. That is, a marginal effects range can not be easily discerned.

Ni concentrations in Poplar Creek sediment appear to be more toxic below Mitchell Branch than in reach 13 and in subreach 3.01 (Fig. F3.12b). Ni concentrations exceeded the 50th percentile of the community level effects distribution at most of the sites in subreaches 3.02, 3.03, and 3.04, whereas concentrations in reach 13 and in subreach 3.01 do not appear to be as toxic. The maxima below Mitchell Branch also exceeded the ER-M from Long et al (1995). However, only 17% of the studies used to derive that value indicated biological effects at concentrations greater than the ER-M, which indicates that the threshold for toxicity is highly uncertain.

Nickel in sediment appears to be a marginal risk to benthic invertebrates in most of the Clinch River and McCoy Branch Embayment (Figs. F3.12a,c, and d). The distributions suggest that risks from Ni are higher below the Emory River (subreaches 4.03 and 4.04) than in subreaches 4.01 and 4.02. With the noted exceptions, the Clinch River and McCoy Branch Embayment distributions are comparable to the negative reference sites (reaches 0 and 8). Pore water concentrations also suggest chronic toxicity in the Clinch River below the Emory River (Fig. F3.4).

Silver. Silver appears to be a significant risk in most of Poplar Creek below Mitchell Branch and in very localized areas of subreach 4.04; a marginal risk in most of reach 1 and in subreaches 4.03 and 4.04; and a negligible risk in other reaches. The distributions of ambient Ag concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.13(a-d). A high percentage (92.8%) of studies indicated an incidence of effects at concentrations greater than the ER-M, which was exceeded in subreaches 3.02 and 3.03 only.

The Ag concentrations in Poplar Creek appear to be lethal to benthic invertebrates at most sites below Mitchell Branch (Fig. F3.13b). Community level effects are indicated for localized areas of subreaches 3.02 and 3.03 only. Reference sediments (reach 13) do not appear to be as toxic. These distributions suggest that Mitchell Branch is the primary contributor of risks from Ag in sediment. However, the limited data preclude excluding East Fork as a source of Ag toxicity.

Ag appears to be a negligible risk at most sites in the Clinch River and McCoy Branch Embayment (Figs. F3.13a,c, and d). The exceptions to this are reach 1 and in subreaches 4.03 and 4.04 (Fig. F3.12c), where most sites appeared to be marginally toxic. The Emory River may contribute to the risks in the Clinch River, but there are no obvious relationships between known contaminant sources and increased risks from Ag in sediment.

Uranium. Uranium appears to be a negligible risk to benthic invertebrates in the CR/PC System. Although U in Poplar Creek pore water increases below Mitchell Branch and a concentration gradient is evident, the only benchmark exceeded was the SCV. This toxicity value is highly conservative because of the small data set used to derive it. The maximum pore water concentrations were 1 to 3 orders of magnitude less than the lowest reported toxic concentration (a fathead minnow LC50 of 2.8 mg/l).

Zinc. Zinc in sediment presents a marginal risk at a few sites in Poplar Creek below Mitchell Branch and most of subreach 4.04 and a negligible risk in all other reaches in the OU. The distributions of ambient Zn concentrations in whole sediment and sediment effects concentrations are shown in Fig. F3.14(a-d). Pore water concentrations and the aqueous test end points for Zn are shown in Fig. F3.15.

Approximately 30% of the sediment samples from subreach 3.02 have concentrations higher than the ER-L and the 20th percentile of the lethal effects distribution (Fig. F3.14b). This may indicate that Zn from Mitchell Branch increased toxicity in sediments at some locations. However, a concentration gradient is not evident in Poplar Creek, all other Poplar Creek sites were less than the 20th percentile of the lethal effects distribution, and none of the sites in reaches 3 or 13 exceeded the ER-M reported by Long et al. (1995).

Summary of metals. Four metals (As, Hg, Ni, and Ag) present a significant risk to benthic invertebrates at most sites in at least one reach. All four are a significant risk in Poplar Creek. East Fork and/or Mitchell Branch appear to be the primary contributors of these COPECs, though the ash disposal area may increase risks as well. Hg presents a wide spread significant risk in the Clinch River below Poplar Creek. McCoy Branch Embayment is the only other reach in which a metal COPEC, As, is a significant risk at most sites.

Two metals, As and Cr, in whole sediment are a significant risk to benthic invertebrates at localized hot-spots in at least one reach: arsenic in Poplar Creek below East Fork, but above the ash disposal area and Cr below Mitchell Branch. Pore water COPECs were not a localized significant risk.

Six metals (As, Cu, Hg, Fe, Ni, and Zn) in whole sediment are a marginal risk to benthic invertebrates at most sites in at least one reach. Every reach, including negative reference reaches, had concentrations of at least one metal COPEC at which toxicity is possible. Also, Al in pore water presents a marginal risk at localized areas in several reaches.

Five metal COPECs (Ba, B, Pb, Mn, and U) are a negligible risk to benthic invertebrates in the CR/PC System.

Phthalate esters. Bis(2-ethylhexyl)phthalate was the phthalate ester detected at concentrations exceeding benchmarks. Risks to benthic invertebrates from bis(2-ethylhexyl)phthalate appear to be a negligible. Bis(2-ethylhexyl)phthalate was detected in sediment in all reaches, both above and below the OU. Whole sediment toxicity data were not found for bis(2-ethylhexyl)phthalate. Pore water concentrations were estimated using the equilibrium partitioning approach and were less than all aqueous toxicity benchmarks. However, bis(2-ethylhexyl)phthalate was detected in pore water samples from the Clinch River (subreaches 2.04 and 4.01) and Poplar Creek (subreaches 3.02 and 3.03). Maximum concentrations in subreaches 3.03 and 4.01 marginally exceeded (e.g., HQs of 1.6 or 3.3) the lowest chronic and lowest test values for daphnids. These toxicity values are lower than the secondary acute and CVs, which were not exceeded. A subsequent test with daphnids observed no significant effects at concentrations higher than those found in the OU. Furthermore, the frequency of detection was relatively low, there are no known sources of plasticizers in the OU, and this is a common laboratory contaminant.

Total PAHs. PAHs as a group appear to be major contributors to risk in Poplar Creek above (reach 13) and below (reach 3) East Fork Poplar Creek, as indicated by the median ΣTU plot for the general categories of COPECs in whole sediment (Fig. 6.14a). Total PAHs appear to be a significant risk to benthic invertebrates in localized areas of Poplar Creek, above and below East Fork, and in subreach 4.04 of the Cinch River. PAHs present a negligible to marginal risk in all other reaches, including reference reaches.

PAHs were detected in whole sediment samples from all reaches except 1, 2.01, 2.02 and 2.03. PAHs were not detected in pore water in any reach. All 12 detected PAHs were COPECs. The median ΣTU plot for PAHs in whole sediment (Fig. 6.14c) suggests that PAHs as a group present, at a minimum, a marginal risk to benthic invertebrates at most sites in each reach in which they were detected. That is, the effects concentration used to calculate the TUs was the ER-L, which is the 10th percentile of Long et al.'s (1995) effects distribution. However, the risks from each individual PAH in subreaches 3.01 or 7.01 are negligible (i.e., the maximum observed concentration was less than the ER-L). At least one PAH exceeded the ER-L in the remaining reaches. The median ΣTU plot for PAHs in whole sediment (Fig. 6.14c) indicates that the 3 PAHs that contribute the most risk in the CR/PC System are 2-methylnaphthalene, acenaphthene, and fluorene.

The distributions of measured Total PAH concentrations were similar among all reaches in the OU, except subreach 4.04, which was generally an order of magnitude higher than the other distributions. The highest Total PAH concentration was in reach 13. The distributions for Poplar Creek (reaches 3 and 13) suggest a more heterogeneous distribution of PAHs than the distributions for the other reaches. The maximum concentration in all reaches, except subreaches 3.01 and 4.02, exceeds the Total PAH ER-L. None of Total PAH concentrations exceed the ER-M. The distribution of observed concentrations in reach 13 suggests that localized areas of that reach are likely to contain lethal concentration of PAHs. Localized concentrations in subreach 4.04 may be lethal also. However, the distributions of total PAH concentrations suggest that lethality is unlikely at all other reaches. Total PAH concentrations should produce significant community level effects at all sites, based on these distributions. That is observed total PAH concentrations are approximately an order of magnitude higher than the community level effects distribution. However, the effects distribution is based on very little data. Hence, local total PAH concentrations, including concentrations at reference reaches, are significantly toxic and/or the community level effects distribution overestimates the likelihood of effects.

Sediment Quality Criteria (EPA_SQC01) are available for three of the detected PAHs: acenaphthalene, fluoranthene, and phenanthrene. Although these were all detected in Poplar Creek, none occurred at concentrations exceeding the SQC. Three PAHs occurred at concentrations that exceed their respective ER-Ms: 2-methylnaphthalene in reaches 0 and 13, napthalene in reach 13, and phenanthrene in reach 13. Concentrations of all individual PAHs at all sites potentially affected by known sources were less than the individual ER-Ms.

PCBs. Total PCBs appear to be a wide spread significant risk to benthic macroinvertebrates in Poplar Creek below Mitchell Branch, and a localized significant risk in subreaches 2.02, 4.01, 4.04. The median ΣTU plot for the general categories of COPECs in whole sediment (Fig. 6.14b) indicates that PCBs contribute to risk in subreach 2.02, Poplar Creek (reaches 3 and 13), and Lower Clinch River (reach 4). The distributions of ambient total PCB concentrations in whole sediment and the sediment effects concentrations are shown in Fig. F3.16a and b. Three congeners were detected: Aroclor 1248, Aroclor 1254, and Aroclor 1260. Aroclor 1248 was detected in two subreaches (3.02 and 3.04) of Poplar Creek only, whereas Aroclor 1254 was present in each reach where PCBs were detected, except subreach 3.01. PCBs were not detected in pore water samples in these reaches. But most of the concentrations estimated using the equilibrium partitioning model were only marginally higher than the highly conservative SCV. This may suggest that PCB toxicity is more likely to be associated with the solid phase. Given that, ingestion of sediments would be a more important exposure pathway than respiration of interstitial water. PCBs appear to be significantly toxic in most reaches where they were detected: maxima exceeded the ER-M from Long et al. (1995) and/or the 50th percentile of either effects distribution (Figs. F3.16a-b). However, the toxicity data are highly uncertain. Freshwater data indicate little or no concordance between toxicity and PCB concentrations and there is a low (51%) incidence of biological effects at concentrations greater than the ER-M ("probable-effects range") reported by Long et al. (1995). Also, MacDonald et al. (1994) expressed a low degree of confidence in the Florida Sediment Quality Assessment Guidelines (SQAG). Florida SQAGs were not used in this assessment. However, the data used to derive the effects distributions presented in Appendix F3 were from the same database used to derive the Florida SQAGs. Given these uncertainties, the evidence for PCB toxicity in sediments in the OU is moderate.

PCB concentrations in Poplar Creek sediment appear to be more toxic below Mitchell Branch than in reach 13 and in subreach 3.01 (Fig. F3.1a). Although a concentration gradient is not evident below Mitchell Branch, toxicity appears to be more likely than not at most sampling sites in these reaches. However, as noted above, low confidence in the toxicity data suggests that toxicity is possible, but not probable. Even so, Mitchell Branch appears to increase the risks to benthic invertebrates from PCBs in sediment.

A gradient in PCB toxicity in sediments is not evident in the Clinch River (Figs. F3.16a-b). Toxicity appears to be more likely than not at most sampling sites in these reaches. However, as noted above, confidence in the toxicity data is only moderate.

Pesticides. 4,4'-DDT and 4,4'-DDD appear to present a marginal risk to benthic invertebrates in the Clinch River adjacent to the ORR (reach 1). These pesticides were detected in only one sample, but that concentration exceeded the ER-M from Long et al. (1995). Toxicity appears to be associated with the solid phase. Although the pore water concentration was not measured in reach 1, it was estimated using the equilibrium partitioning model. The estimated concentration was less than the SCV, which is a highly conservative screening value. Confidence in the sediment toxicity data is low. Data for freshwater organisms was not found and Long and Morgan (1991) observed that the data for marine organisms does not cluster well around the ER-L and ER-M. Long et al. (1995) found that the incidence of adverse effects did not increase consistently and markedly with increasing concentrations of p,p'-DDE and total DDT, which were the only pesticides included in that report. Given these uncertainties, it appears that toxicity is possible at isolated locations in reach 1 only.

6.3.3.2 Ambient sediment toxicity

There was very little indication of sediment toxicity in any of the tests and no clear spatial trends. Two expressions of the results of sediment toxicity tests are used, the average magnitude of the effects and the frequency of effects that are significant. Test results are deemed significant if (1) there is at least a 20% difference between the response of the organisms exposed to sediment or pore water from the OU and the organisms exposed to control water or reference sediment/pore water, or (2) the likelihood that the difference is due to chance is <5%. Test results are related to two expressions of exposure, the location of the sample relative to sources and the relationship between the test responses and the concentrations of sediment COPECs from the same location. Mean proportional reductions in survival best represent the likely impacts in a reach because of the inherent spatial heterogeneity of sediments and sediment-associated contaminants. The methods and results of the sediment tests are discussed in Sect. 3.4.2. This section analyzes their implications for risks to benthic macroinvertebrates in the OU.

Survival of the sediment-associated organism *H. azteca* was the most responsive and reliable test, but the magnitude and frequency of significant effects was very low. Less than 25% of the tests from all reaches and dates had significant reductions in survival and the mean reduction was <20% for every reach. The least responsive tests were those using pore water as the exposure medium.

Mortality of *H. azteca* in sediments from all four subreaches of Poplar Creek was statistically significantly elevated relative to the reference site (PCM 6.0, reach 13) in at least one test (Fig. 6.15). However, the sediment toxicity laboratory notes that the first test of *H. azteca* was used for method performance evaluation and technician training and advises caution when interpreting these results. Only subreaches 3.01 and 3.03 had significant mortality in subsequent tests. Furthermore, significant toxicity was not observed in confirmatory toxicity tests of sediment collected from three highly contaminated reach 3 sites in December 1995. Even with the first test included, the mean proportional reductions in survival were not biologically significant (i.e., reduced <20% relative to the reference) at any of the Poplar Creek locations.

Mortality of *H. azteca* in sediments from both of the tested Clinch River subreaches (CRM 19.0; subreach 2.03 and CRM 9.0; subreach 4.01) were significantly elevated relative to the reference site (CRM 22.0; 2.01) in three of the seven tests (Fig. 6.15). One of the significant responses for subreach 2.03 was from the first *H. azteca* test, which is caveated as noted above. Though the proportion of significant tests was higher in the Clinch River than in Poplar Creek, the mean proportional reductions in survival were still not biologically significant (i.e., reduced <20%).

Mortality of *H. azteca* in sediments from McCoy Branch Embayment (MBM 0.2; subreach 7.02 and MBM 0.4; subreach 7.01) was not significantly elevated relative to the reference site (WBM 0.4–0.6; reach 8) in either of the two tests (Fig. 6.15).

Concentrations of COPECs in whole sediment were not statistically significantly positively correlated with *H. azteca* survival ($\alpha = 0.05$). However, significant correlations may be masked by the spatial heterogeneity of sediment-associated contaminants. Sediment samples for toxicity testing and chemical analysis were collected at different times and, although the whole sediment concentrations are unlikely to vary appreciably in time, they are likely to vary in space. That is, although an effort was made to collect samples from the same location, the observed chemical concentrations may not reflect the exposures received during any given toxicity test.

In summary, these tests indicate that some of the sediments at these locations are toxic, but when evaluated as a whole, the impacts do not appear to be biologically significant. Significant impacts may, however, be obscured by impacts at the reference sites. The fact that sediment test results were more sensitive than pore water test results may indicate that sediment-associated contaminants contribute more to risks in the sediments or that the sample processing procedure significantly altered the toxicity of the pore water.

6.3.3.3 Biological surveys

Discriminant analysis as described in Appendix F8 shows no clear distinction between Poplar Creek embayment and the mainstem Clinch River. Clinch River reach 2 (above Poplar Creek) is as different from reach 4 (below Poplar Creek) as either is from Poplar Creek itself (reaches 3 and 13). The analysis of total abundance and species richness (Figs. 6.16–6.19) does, however, demonstrate some fine-scale spatial patterns within the reaches.

Figure 6.16 shows a distinct gradient in total benthic invertebrate abundance (# of individuals/m^2), within the Clinch River. Abundance is highest at the mouth of the Clinch (subreach 4.04) and declines with increasing distance from Watts Bar Reservoir. The only exception to this pattern is subreach 4.01; invertebrate abundance within this subreach is significantly lower than in either of the adjoining subreaches. The pattern of taxonomic richness (# of taxa/m^2) shows that richness is in general positively correlated with abundance. Richness is high in the lowermost subreaches (4.02–4.04), low in the uppermost subreaches (2.02 and 2.03), and low in subreach 4.01. Taxonomic richness of invertebrates is higher in subreach 2.04 than would be expected based on the abundance data. Chironomids and tubificids are frequently associated with soft substrates and are sometimes used as indicators of poor environmental quality. Figure 6.17 shows that both families are abundant in the Clinch, comprising more than 80% of the total benthic invertebrates in most subreaches. Tubficids predominate in subreaches 4.01 and above; chironomids predominate in subreaches 4.02–4.04.

Figure 6.18 shows that there is no evident spatial trend in invertebrate abundance or species richness within Poplar Creek embayment. The two uppermost stations within reach 13 (Poplar Creek mile 8.0 and mile 7.0) are low in both abundance and richness, however, the next two stations (reach 13, mile 6.0 and subreach 3.01, mile 5.1) are high in both abundance and richness. Subreaches 3.02, 3.03, and 3.04 are low in abundance and richness compared to adjoining subreaches. The differences in abundance exceed 20% and are statistically significant. Comparison of Figs. 6.16 and 6.18 shows that invertebrates are on average much more abundant in the Clinch River than in Poplar Creek, but taxonomic richness is similar. As in the Clinch, tubificides and chironomids are the dominant invertebrate taxa at all Poplar Creek stations except mile 1.0 (subreach 3.04).

These trends do not in and of themselves demonstrate a contaminant impact. Measured in terms of abundance and richness, Clinch River subreach 4.01, immediately below Poplar Creek is significantly different from subreach 4.02 downstream. However, given the obvious upstream-downstream spatial gradient present in the Clinch, this difference may be due to ecological conditions rather than to a contaminant effect. There is no such gradient observable in the Poplar Creek data. Low abundance and richness present in reach 13 and in subreaches 3.02, 3.03, and 3.04 may be attributed to local conditions, including the presence of contaminants.

The TVA (TVA 1994) qualitatively characterized the sediment in the CR/PC benthic invertebrates survey samples. The sites in subreaches 4.02, 4.03 and 4.04 were more lacustrine in nature than the other Clinch River sites. Silt and detritus, primarily in the form of decaying leaves, were the predominant substrates in samples from subreaches 4.03 and 4.04. The amount of detritus was much lower, and the amount of sandy substrates was generally higher, in samples from sites above subreach 4.02. This is consistent with the spatial pattern of particle size distribution observed in the samples for chemical analysis. Sediment from survey samples in subreach 3.04 was almost entirely composed of fine silt (TVA 1994). Poplar Creek sediments above subreach 3.04 were a mixture of fine and coarse sediments with varying amounts of silt present in nearly every survey sample (TVA 1994).

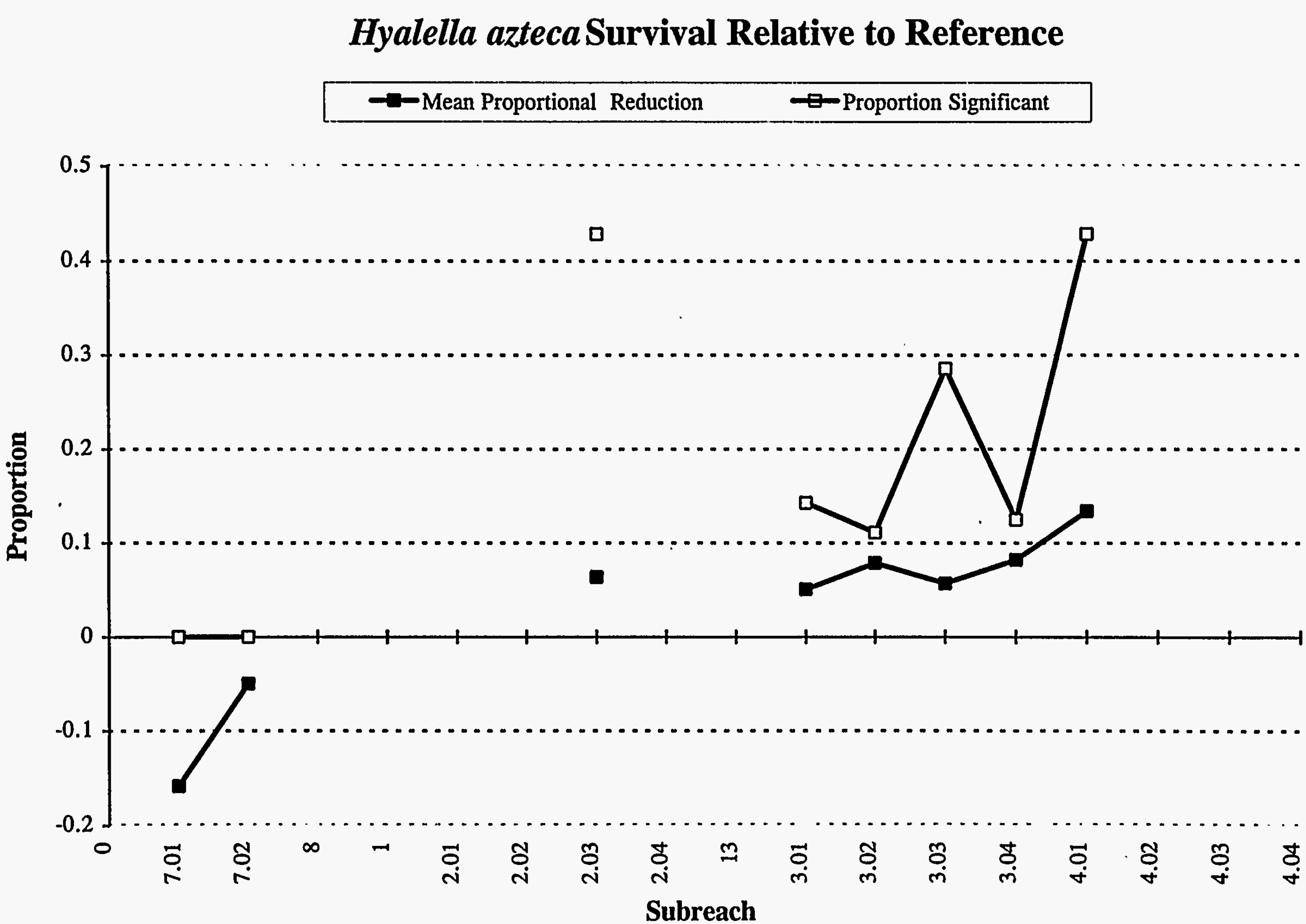
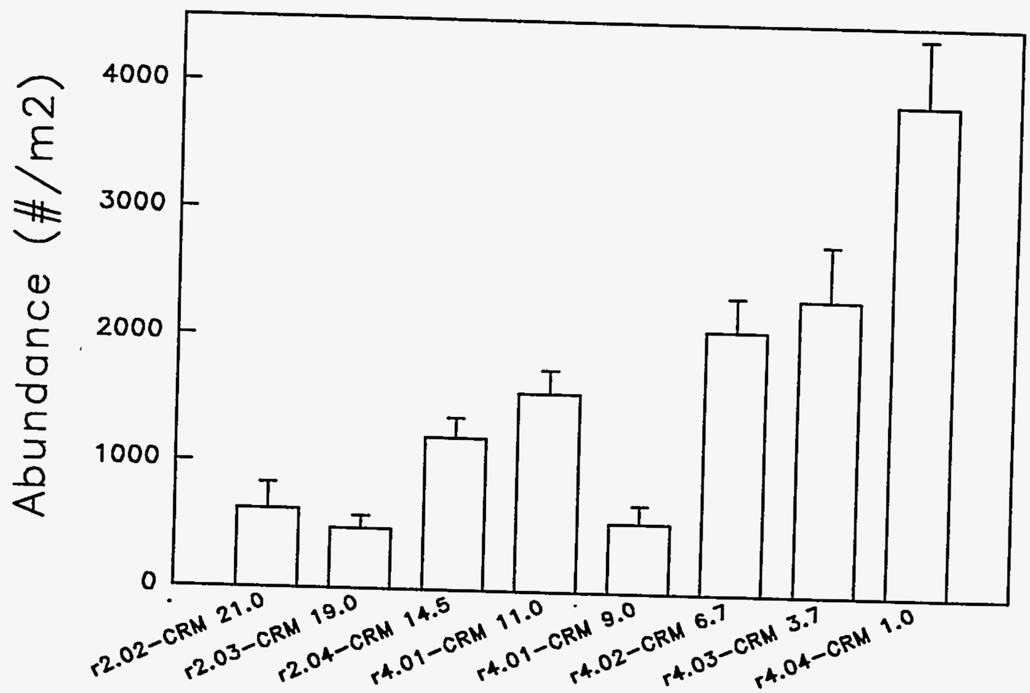


Fig. 6.15. Proportional reduction in *Hyalella azteca* survival relative to references and proportion of tests in which survival was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly decreased).



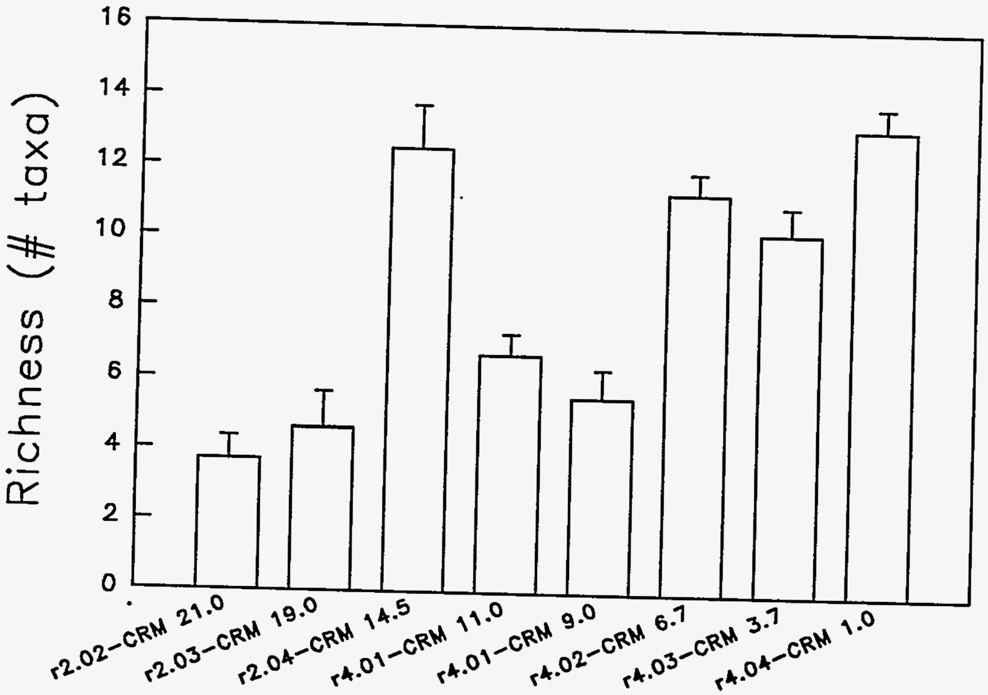


Fig. 6.16. Taxonomic richness and abundance (Mean ± SE) of benthic macroinvertebrates in reaches of the Clinch River in 1994.

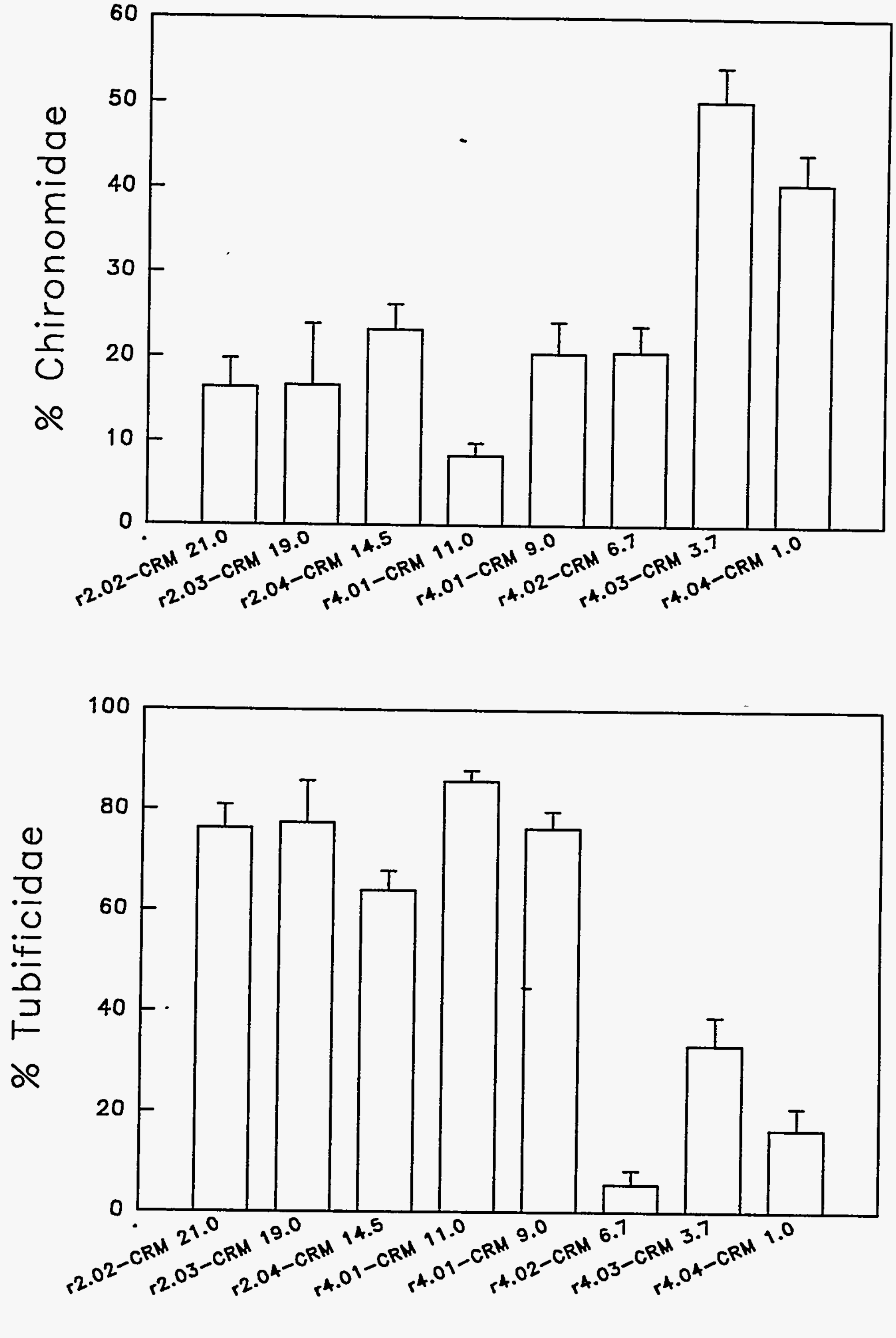


Fig. 6.17. Percentage of benthic macroinvertebrate community (Mean ± SE) represented by *Chironomidae* and *Tubificidae* in reaches of the Clinch River in 1994.

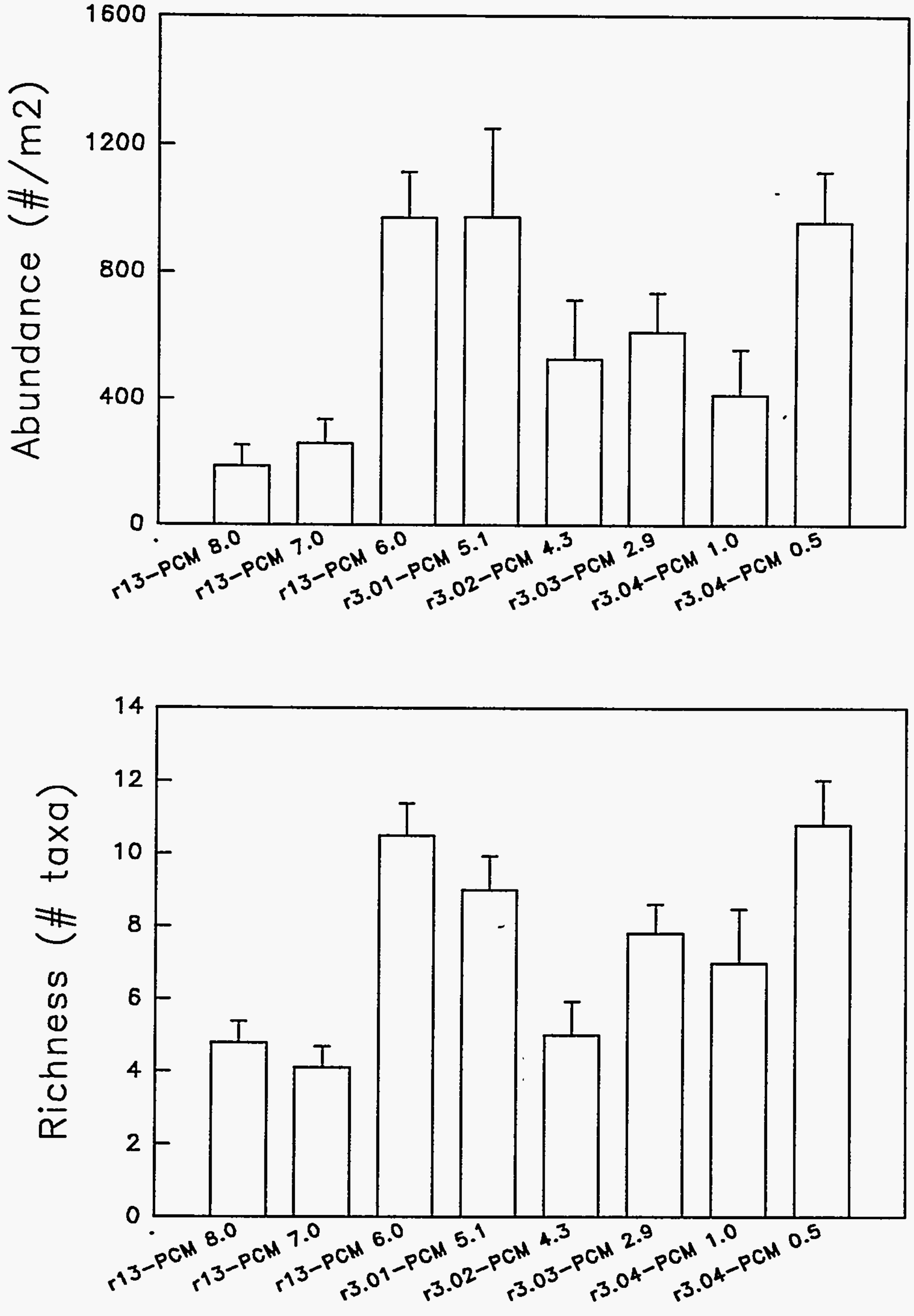
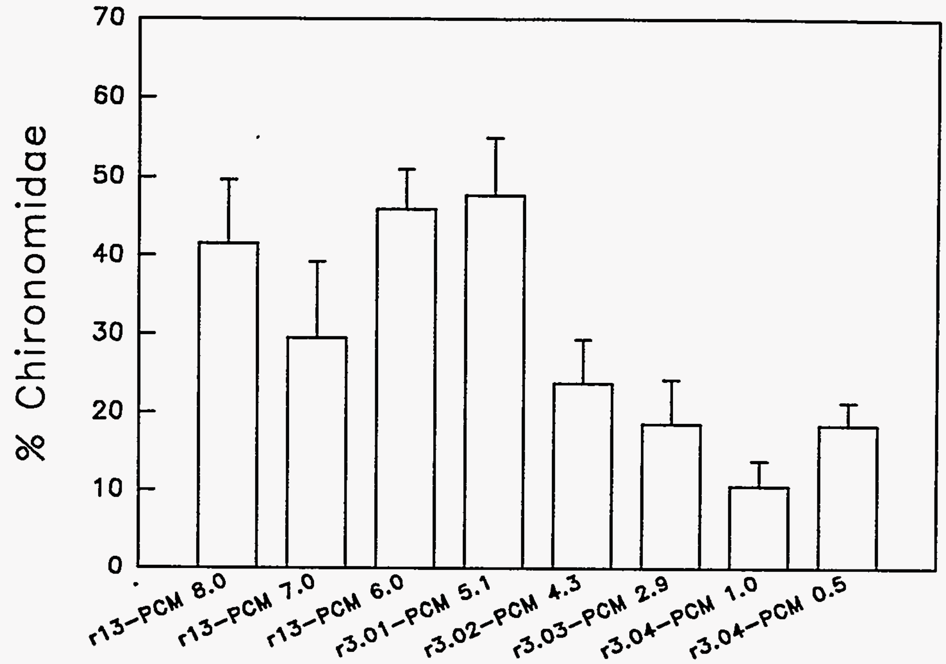


Fig. 6.18. Taxonomic richness and abundance (Mean ± SE) of benthic macroinvertebrates in reaches of Poplar Creek in 1994.



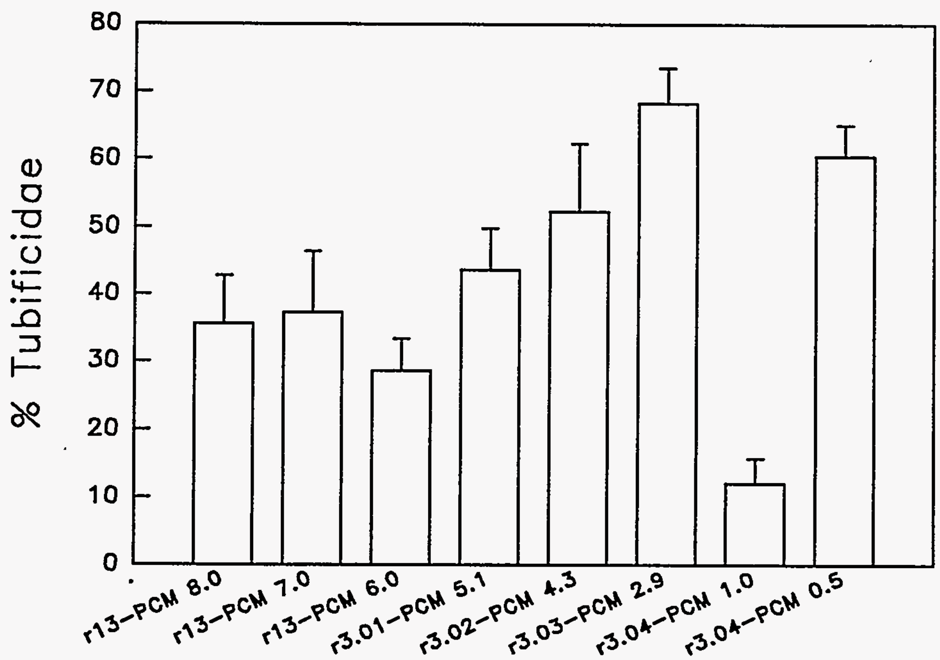


Fig. 6.19. Percentage of benthic macroinvertebrate community (Mean ± SE) represented by *Chironomidae* and *Tubificidae* in reaches of Poplar Creek in 1994.

Multiple linear regression analysis as described in Appendix F8 indicates that differences in habitat are more important than differences in contaminant concentrations, with regard to the amount of variation explained in taxa richness, dominance, and the abundance of five taxa (*Chironomidae*, other *Diptera*, *Tubificidae*, *Hexagenia limbata*, and *Veneroida*). Contaminants cannot, however, be excluded from consideration based on these results.

Table 6.9 shows the amount of variance in each of these seven response variables explained by including physical and chemical parameters (full model R^2) or only physical parameters (reduced model R^2). The additional variance explained by including contaminants is the difference between the full model R^2 and reduced model R^2. Including contaminants significantly ($p < 0.05$) increased the amount of explained variance in all benthic community measures for the Clinch River and in four benthic community measures for Poplar Creek.

The variance explained by contaminants given the habitat variables are in the model, however, was small relative to the variance explained by physical parameters alone. Habitat differences in the Clinch River explained more than half of the observed variance in five of the seven response variables, including taxa richness and dominance. In fact, 83% of the variance in *Hexegenia limbata* abundance was explained by the habitat variables alone. Contaminants in the Clinch River explained <20% of the variance above that explained by the habitat variables in all response variable except non-*Chironomidae Diptera* (23%). In general, statistical analysis of the Poplar Creek data did not explain as much of the variance in the community measures. However, habitat characteristics were still more important than contaminants. Nearly half of the variance in taxa richness (49%) was explained by physical parameters alone, but only an additional 11% was explained by including contaminant concentrations. The greater importance of habitat relative to contaminants may be partly due to the number of observations at each site. The physical parameters, except for percent TOC, were reported for each benthic invertebrate sample, whereas contaminant concentrations were reported once for each benthic sampling site.

Table 6.10 shows the number of benthic community measures significantly ($p < 0.10$) correlated with each of the contaminant variables and the consistency of the signs of these correlations with the hypothesis that contaminants negatively impact the benthic invertebrate community. The fact that chemical variables explain a significant amount of variance in a variety of community measures for the Clinch River and Poplar Creek provides some (admittedly weak) evidence that contaminants may be adversely affecting the benthic community in some locations. Of the contaminants included in the analysis, PAH would appear to be among the ones most likely to be important and toxic metals in Poplar Creek would appear to be among the ones least likely to be important. However, this is weak evidence given the high collinearity of many of the explanatory variables, especially correlations between PAH, PCB, and TOC.

6.3.3.4 Weight of evidence for benthic invertebrates

The weighing of evidence is performed by asking the following questions concerning each reach in the OU:

1. Is the benthic invertebrate community less species-rich or abundant than would be expected?
2. Is the sediment toxic to aquatic organisms?
3. Does that sediment contain chemicals in toxic amounts?
4. What factors account for apparent discrepancies in the results?

Table 6.9. Amount of variance in benthic community measures explained by the physical, physical plus chemical, and chemical variables and the significance of the chemical variables

Community measures	Amount of variance explained			P-value for chemicals
	Physical variables[a] (%)	Physical and chemical variables[b] (%)	Chemical variables given physicl variables[c] (%)	
		Clinch River		
Taxa richness	52	69	17	<0.0001
Dominance	57	67	10	0.0018
Chironomidae[d]	64	78	14	0.0001
Diptera (other)[d]	21	44	23	0.0002
Hexagenia limbata[d]	83	90	7	<0.0001
Tubificidae[d]	39	53	14	0.0032
Veneroida[d]	55	64	9	0.0140
		Poplar Creek		
Taxa Richness	49	60	11	0.0157
Dominance	23	28	5	0.5014
Chironomidae[d]	25	44	19	0.0057
Diptera (other)[d]	33	39	5	0.4412
Hexagenia limbata[d]	61	70	9	0.0122
Tubificidae[d]	47	53	6	0.2481
Veneroida[d]	45	55	11	0.0336

[a] Physical variables included depth, substrate characteristics (% sand, % silt, % clay, % gravel), and total organic carbon.
[b] Chemical variables included total PCBs, total polycyclic aromatic hydrocarbons, and toxic metals in whole sediment samples and ammonia nitrogen and aluminum in sediment pore water. Toxic metals (arsenic, cadmium, chromium, lead, mercury, nickel, silver, and zinc) were combined into toxic units prior to analysis.
[c] Variance explained by chemicals was derived by subtracting the R^2 for physical variables only from the R^2 for physical plus chemical variables.
[d] Abundance

Table 6.10. Signs of significant correlations between chemical variables and benthic invertebrate variables and the consistency[a] of signs with negative impacts on benthic communities

Chemical variables	Clinch River			Poplar Creek		
	Inconsistent	Consistent	Not significant at $\alpha = 0.05$	Inconsistent	Consistent	Not significant at $\alpha = 0.05$
Toxic metals[b]	2	2	3	3	0	4
PCBs (total)	3	2	2	3	1	3
PAHs (total)	2	3	2	1	3	3
Ammonia[b]	1	1	5	1	2	4
Aluminum[b]	2	2	3	1	3	3

PAH = polycyclic aromatic hydrocarbon.
[a]High dominance, *Tubificidae* abundance, and *Chironomidae* abundance are generally accepted as indicators of poor water quality. For these variables, a positive coefficient was interpreted as being consistent with contaminant effects. A negative coefficient for taxa richness and abundance of non-*Chironomidae Diptera*, *Hexagenia limbata*, and *Veneroida* was interpreted as being consistent with contaminant effects.
[b]Toxic metals in whole sediment included arsenic, cadmium, chromium, lead, mercury, nickel, silver, and zinc. Because all of these were found to be highly intercorrelated in initial exploratory analyses, the concentrations of the individual metals were combined into toxic units prior to analysis.

Poplar Creek Embayment (reach 3). The weight of evidence suggests that the benthic invertebrate community is significantly impacted by contaminants. This conclusion is based on the results indicating that sediment-associated organisms at most sites are exposed to levels of several contaminants that have been observed to be toxic; the biosurvey results show a >20% reduction in taxa richness and abundance; the statistical analysis of the physical, contaminant, and biosurvey data did not exclude contaminants as a causal factor; and the sediment toxicity tests are too ambiguous to definitively exclude impacts in this reach. The lines of evidence concerning risks to benthic invertebrates in Poplar Creek Embayment are summarized in Table 6.11, and are discussed below.

If significant toxic effects were occurring, then the community would have fewer species and individuals than other sites, and they do. That is, subreaches 3.02, 3.03, and 3.04 are low in abundance and richness compared to adjoining subreaches. The differences in abundance exceed 20% and are statistically significant. Although this is the most ecologically relevant line of evidence, interpretation of these results is complicated by the highly variable responses, the presence of contaminants and apparent impacts in the reference reach, the importance of habitat characteristics wit h regard to community measures, and the high collinearity of the physical and contaminant variables.

If significant toxic effects were occurring, then the sediment should be toxic, and some samples were. However, the interpretation of these results is uncertain because of the frequency and magnitude of effects. At least one, but less than half, of the sediment samples from each site were significantly toxic to *H. azteca*. The mean reduction in survival, which includes tests with higher than reference survival rates, did not meet the end point criterion for biological significance. The inconsistent responses in samples from the same sites is suggestive of a highly heterogeneous system. Hence, some, but not most, of the sediment appears to be acutely lethal to at least one test species. Some impacts at sites below East Fork may be obscured by toxic effects in the reference sediments from reach 13. That is, a control sediment was not available and reductions in survival are relative to the reference.

Table 6.11. A summary of risk characterization for the benthic invertebrate community in Poplar Creek (reach 3)

Evidence	Result[a]	Explanation
Biological surveys	+	Subreaches 3.02, 3.03, and 3.04 are low in abundance and richness compared to adjoining subreaches. The differences in abundance exceed 20% and are statistically significant. Habitat is a principle factor, but contaminants may also contribute to the observed differences.
Toxicity tests	±	At least 1 significant test in each reach, but mean response was not biologically significant.
Media analyses	+	Risks from 4 metals (As, Hg, Ni, and Ag) and PCBs are significant at most sites in several reaches, especially below Mitchell Branch and East Fork Poplar Creek. Arsenic and chromium are significant at localized hot-spots below Mitchell Branch and East Fork Poplar Creek. Polycyclic aromatic hydrocarbons may also contribute to toxicity.
Weight-of-evidence	+	Contaminant mediated impacts to the community are likely, though habitat characteristics are critical also.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
± indicates that the evidence is too ambiguous to interpret.

If significant toxic effects were occurring, then chemicals should be found in sediment at concentrations that have been reported in the literature to be toxic, and they were. Several COPECs were a significant risk at most sites in at least one reach. Although the primary sources of metals and PCBs appear to be within the OU, PAHs appear to be sourced above the OU and may be a natural constituent of the system. That is, a seam of coal fines was consistently observed in Poplar Creek sediments, especially upstream of the OU.

Upper Clinch River arm (reach 2). Although there is some indication of toxicity in sediments from the Clinch River above Poplar Creek but below the Melton Hill dam (reach 2), the invertebrate community does not appear to be significantly affected. This conclusion is based on the results indicating that the benthic community in Watts Bar is similar to other regional mainstem reservoirs, that contaminant concentrations are not a significant risk, and that the sediment toxicity tests neither support, nor refute, this conclusion. The lines of evidence concerning risks to benthic invertebrates are summarized in Table 6.12, and are discussed below.

If significant toxic effects were occurring, then the community would have fewer species and individuals than other sites. This does not appear to be the case. The relatively low richness and abundance observed in Watts Bar is typical of other mainstem reservoirs in the upper Tennessee Valley. Although these metrics are low relative to other sites in Watts Bar, habitat quality appears to be a causal factor (e.g., the amount of organic matter was also low in these reaches). Although the statistical analyses did not exclude contaminants as a source of the variability in the community measures, they did indicate that habitat characteristics are considerably more important for many of the community measures (e.g., physical variables alone explain 83% of the variability in the abundance of the mayfly *Hexagenia limbata*).

If significant toxic effects were occurring, then the sediment should be toxic. Although some samples were toxic, the biological significance of these results is uncertain because of the frequency and magnitude of effects. Less than half of the sediment samples were significantly toxic to *H. azteca* and the mean reduction in survival did not meet the end point criterion for biological significance. The inconsistent responses in samples from the same site is suggestive of a highly heterogeneous system. Also, the other sediment toxicity tests were not suggestive of significant toxicity.

If significant toxic effects were occurring, then chemicals should be found in the sediment at concentrations that have been reported in the literature to be toxic. Although some COPECs may be toxic, none are likely to be toxic. That is, several contaminants present a marginal risk at one or more sites in reach 2, but no contaminant was observed at concentrations suggesting a significant risk to benthic invertebrates and a relationship to known sources was not evident. Given the high degree of uncertainty associated with the sediment effects data, this is, at best, weak evidence of contaminant mediated impacts in this reach.

Lower Clinch River arm (reach 4). Although there is some indication of toxicity in sediments from the Clinch River below Poplar Creek, the invertebrate community does not appear to be significantly affected. This conclusion is based on the results indicating that the benthic community in Watts Bar is similar to other regional mainstem reservoirs, that reach 4 has the highest richness and abundance observed in the OU, and that contaminant concentrations which are likely to be toxic were observed only at sites where the community metrics were highest. The sediment toxicity tests neither support, nor refute, this conclusion. The lines of evidence concerning risks to benthic invertebrates are summarized in Table 6.13, and are discussed below.

If significant toxic effects were occurring, then the community would have fewer species and individuals than other sites. This is clearly not the case. The benthic invertebrate community in this reach has the highest density and species richness observed in the CR/PC system. Subreach 4.01 had reduced richness and abundance relative to subreach 4.02, but this is probably due primarily to habitat differences. That is, although the statistical analyses did not exclude contaminants as a source of the variability in the community measures, they did indicate that habitat characteristics are considerably more important for many of the community measures.

If significant toxic effects were occurring, then the sediment should be toxic. Although some samples were toxic, the biological significance of these results is uncertain because of the frequency and magnitude of effects. Less than half of the sediment samples were significantly toxic to *H. azteca* and the mean reduction in survival did not meet the end point criterion for biological significance. The inconsistent responses in samples from the same site is suggestive of a highly heterogeneous system. Also, the other sediment toxicity tests were not suggestive of significant toxicity.

Table 6.12. A summary of risk characterization for the benthic invertebrate community in the upper Clinch River arm of Watts Bar Reservoir (reach 2)

Evidence	Result[a]	Explanation
Biological surveys	-	Density and abundance are low, but habitat appears to be the principle causal factor and Watts Bar is similar to other Upper Valley Reservoirs.
Toxicity tests	±	At least 1 significant test, but mean response was not biologically significant due to high variability between tests.
Media analyses	±	Risks are marginal from metals and polycyclic aromatic hydrocarbons, but no contaminant of potential ecological concern was a significant risk.
Weight-of-evidence	-	Community does not appear to be significantly impacted by contaminants.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
± indicates that the evidence is too ambiguous to interpret.

Table 6.13. A summary of risk characterization for the benthic invertebrate community in the lower Clinch River arm of Watts Bar Reservoir (reach 4)

Evidence	Result[a]	Explanation
Biological surveys	-	Density and abundance highest in these reaches, probably due in part to habitat quality.
Toxicity tests	±	At least 1 significant test. Mean response was not biologically significant, but subreach 4.01 was the only site tested.
Media analyses	+	Risks from polycyclic aromatic hydrocarbons and mercury are significant at most sites in at least one subreach, including 4.04.
Weight-of-evidence	-	Community does not appear to be significantly impacted by contaminants.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
± indicates that the evidence is too ambiguous to interpret.

If significant toxic effects were occurring, then chemicals should be found in sediment at concentrations that have been reported in the literature to be toxic. Several COPECs were observed at concentrations that are likely to be toxic. However, these samples are from areas supporting the most diverse and abundant benthic communities. Given the high degree of uncertainty associated with the sediment effects data, this is, at best, weak evidence of contaminant mediated impacts in this reach.

McCoy Branch embayment (reach 7). The benthic invertebrate community in McCoy Branch Embayment is unlikely to be significantly impacted by contaminated sediments. Only arsenic is present at concentrations which are likely to be toxic. Community survey data were not available for McCoy Branch. However, these sediments were tested in the laboratory and there was no indication of toxicity, either acute or sub-chronic. Given the significant uncertainties associated with the sediment effects data, it appears unlikely that impacts to the community are actually occurring. The lines of evidence concerning risks to benthic invertebrates in the McCoy Branch Embayment are summarized in Table 6.14.

Table 6.14. A summary of risk characterization for the benthic invertebrate community in McCoy Branch Embayment (reach 7)

Evidence	Result[a]	Explanation
Biological surveys		Community surveys not performed
Toxicity tests	-	Sediments were not toxic.
Media analyses	+	Risks from arsenic are significant.
Weight-of-evidence	-	It is unlikely that the community is significantly impacted by contaminants.

[a] + indicates that the evidence is consistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
- indicates that the evidence is inconsistent with the occurrence of a 20% reduction in species richness of abundance of the benthic invertebrate community.
$\pm$ indicates that the evidence is too ambiguous to interpret.

6.3.4 Uncertainties Concerning Risks to Benthic Invertebrates

The following issues constitute the major sources of uncertainty in the risk assessment for the benthic invertebrate community. The primary source of most of these uncertainties is the inherent heterogeneity of the sediment system.

There is a high degree of uncertainty associated with the sediment effects data because sediments are highly heterogeneous and complex. There are few standardized sediment toxicity tests and most of the available data are from marine and estuarine systems. The end points used to define the ER-L and ER-M range from community level responses to sub-organismal effects. These sources of variance are discussed in more detail in Sect. 6.3.1.2.

The relationship between observed sediment toxicity and the chemical and physical data are uncertain. Samples for sediment toxicity testing were collected at approximately the same locations as the samples for chemical analysis. Because sediment contamination may be very heterogeneous, the exposures received by the test organisms may be different from those estimated using the chemical analyses data.

There is uncertainty associated with the biosurvey data because benthic invertebrate ecology is complex and influenced by the heterogeneity of the sediment system. The characteristics of the benthic invertebrate community at a site is a function of non-contaminant and contaminant parameters, both of which may vary considerably within and between sites. The contribution of each of these factors to the variance in observed community metrics cannot be clearly delineated with the available data. That is, statistical analyses were confounded by the high collinearity of the physical characteristics of the sediments and the contaminant concentrations.

The sediment characterization section (Sect. 3.4) identified ammonia in pore water as a non-contaminant parameter which may be ecologically important. A major source of ammonia is the decomposition of organic matter. Thus, ammonia is a natural constituent of aquatic systems and is likely to be elevated in sediments with high organic matter content. Sewage treatment plants are also a potential source. However, release of ammonia from treatment plants is controlled under the National Pollution Discharge Elimination System (NPDES). Pore water concentrations of ammonia in all reaches, except 2.01, were higher than the chronic NAWQC. Several reaches, including reference reaches 0 and 13, had pore water concentrations that exceeded the acute NAWQC. These reaches were also generally high in sediment or pore water total organic carbon. Given the measured concentrations, ammonia may be a contributing factor to the degraded benthic invertebrate communities in Poplar Creek. This is highly uncertain, however, because ammonia toxicity is dependent on pH and temperature (EPA 1985b). Field measures of these parameters in pore water were not available and laboratory measures are not representative: both pH and temperature can change rapidly. Surface water measures of pH are unlikely to be representative of pore water pH, because the slower exchange rate of pore water may result in the accumulation of ammonia and other natural products which elevate pH.

The lack of clean reference sites contributed to the uncertainties in all three lines of evidence. Some of the chemicals in the OU may be at naturally occurring levels. This determination was complicated by the lack of acceptable reference concentrations. The sediment toxicity tests and community survey results may be due to local sediment and habitat characteristics. Estimation of the importance of these factors also was complicated by the lack of acceptable reference sites and laboratory control sediments.

Furthermore, there is a general need for reliable and standard sublethal whole sediment tests. That is, the only standard test that uses whole sediment is an acute lethality test.

6.4 RISKS TO PISCIVOROUS WILDLIFE

6.4.1 Exposure Assessment for Piscivorous Wildlife

Exposure of piscivorous wildlife to contaminants may be expressed as the rate of ingestion of contaminated media (fish and water) or as the concentration of contaminants accumulated in the tissues of the piscivore itself. Contaminant exposure through ingestion was estimated for osprey (*Pandion haliaetus*), the great blue heron (*Ardea herodias*), mink (*Mustela vison*), and river otter (*Lutra canadensis*). Exposure estimates were calculated for all contaminants in CR/PC reaches 0, 1, 13, 6, 18, 5, and 15 and subreaches 2.01, 2.02, 2.04, 3.01, 3.02, 3.03, 3.04, 4.01, and 4.04. Exposure through contaminants accumulated in tissues was measured for nestling great blue herons.

An iterative approach was employed to assess the exposure of piscivores to contaminants. First, conservative exposure estimates were generated using point-estimates of exposure parameters and conservative assumptions. These conservative estimates were compared to NOAELs to identify COPECs. More realistic estimates of exposure at each subreach were then generated for the COPECs using the Monte Carlo simulation. The Monte Carlo simulations incorporate the variability in the contaminant concentrations and use more realistic foraging and life history data. Exposure distributions generated by these distributions were compared to NOAELs and LOAELs to identify subreaches where exposure is sufficiently high to present a risk. Finally, as part of the Risk Characterization (Sect 6.4.3.), the Monte Carlo simulation of exposure from adjacent subreaches was performed to address how foraging behavior and movements of piscivores influences contaminant exposure.

6.4.1.1 Exposure through oral ingestion of fish and water

Oral exposure to contaminants experienced by wildlife may come from multiple sources. They may consume contaminated food (either plant or animal), drink contaminated water, or ingest soil or sediment. Soil or sediment ingestion may be incidental while foraging or grooming or purposeful to meet nutrient needs. The total oral exposure experienced by an individual (Eq. 6.1) is the sum of the exposures attributable to each source and may be described as:

$$E_{total} \approx E_{food} + E_{water} + E_{soil} \tag{6.1}$$

Where:

E_{total} = total exposure from all pathways
E_{food} = exposure from food consumption
E_{water} = exposure from water consumption
E_{soil} = exposure from soil consumption

Because osprey dive and catch fish in open water and great blue herons forage by stalk and strike techniques while wading (Ehrlich et al. 1988), incidental ingestion of sediment by these end points is unlikely. While Hamilton (1940) observed sand in 1.3% of mink scats examined, this amount did not account for any measurable scat volume. Because river otter are more piscivorous than mink, soil ingestion be otter is likely to less than that of mink. Therefore, ingestion of soil or sediment by osprey, heron, mink, or otter was assumed to be negligible. Exposure of piscivorous wildlife (Eq. 6.2) may then be described as:

$$E_{total} \approx E_{food} + E_{water} \tag{6.2}$$

For exposure estimates to be useful in the assessment of risk to wildlife, they must be expressed in terms of a body weight-normalized daily dose or mg contaminant per kg body weight per day (mg/kg/d). Exposure estimates expressed in this manner may then be compared to toxicological benchmarks for wildlife, such as those derived by Opresko et al. (1994), or to doses reported in the toxicological literature. Estimation of the daily contaminant dose an individual may receive from a particular medium for a particular contaminant may be calculated using the following equation:

$$E_j = \sum_{i=1}^{m} \left(\frac{IR_i \times C_{ij}}{BW} \right) \tag{6.3}$$

where:

E_j = total exposure to contaminant (j) (mg/kg/d)
m = total number of ingested media (e.g., food, water, or soil)
IR_i = consumption rate for medium (I) (kg/d or L/d)
C_{ij} = concentration of contaminant (j) in medium (I) (mg/kg or mg/L)

BW = body weight of end point species (kg)

For the purposes of this assessment, the generalized model was modified to reflect the foraging characteristics of the end point species and the idiosyncracies of the available data. First, because the literature suggests that osprey, heron, mink, and otter do not consume fish of the same size, fish size was added to the model. It was assumed that all four species would forage and take fish opportunistically based upon the size of the fish. Fish species was assumed to be an unimportant factor in prey selection.

Second, while piscivorous wildlife consume whole fish, only part of the contaminant data for fish from the Clinch River was for whole fish. While all shad (both threadfin and gizzard) were analyzed as whole fish, all non-shad fish were analyzed as fillets. Both fillet and carcass (whole-body minus fillet) analyses were performed for 25 non-shad fish (15 largemouth bass and 10 channel catfish). These data were used to produce a fillet to whole-body ratio (see below) so that whole body concentrations in non-shad fish could be estimated. Non-shad fish were then combined with shad to produce summary statistics (i.e., mean, standard error, etc.) for all fish within each size class. With these additional considerations taken into account, the exposure model (Eq. 6.4) for Clinch River piscivorous wildlife is as follows:

$$E_j = \left(\frac{IR_w \times C_{w\text{-}j}}{BW}\right) + \left(P_a \times \sum_{i=1}^{n} \frac{(P_{f\text{-}I} \times IR_f) \times C_{ij}}{BW}\right) \tag{6.4}$$

where:

E_j = exposure to contaminant (j) (mg/kg/d)
IR_w = ingestion rate of water (L/d)
$C_{w\text{-}j}$ = concentration of contaminant (j) in water (mg/L)
BW = body weight (kg)
P_a = proportion of aquatic prey in diet
n = number of size categories of fish consumed
$P_{f\text{-}I}$ = proportion of fish in diet in size class (I)
IR_f = ingestion rate of fish (kg/d)
C_{ij} = concentration of contaminant (j) in fish in size class (I) (mg/kg)

Exposure estimates were calculated for all contaminants in all Clinch River subreaches for which data were available. Because wildlife are mobile, their exposure is best represented by the mean contaminant concentration in media. To be conservative, the UCL_{95} is used in exposure estimates. To prevent bias that may result from calculating UCL_{95} using data that contains values below the detection limit, product limit estimator was used to calculate the UCL_{95} for contaminants observed in fish and water. These data were used in the initial exposure estimates. Exposure estimates for contaminants that may potentially present a risk to piscivorous wildlife (based upon comparisons to NOAELs) were recalculated using Monte Carlo simulations. ***(Note: because the purpose of the initial exposure estimate is to be conservative and to identify COPECs, the UCL_{95} was used regardless of whether or not the value exceeded the maximum observed value. Overestimates of exposure that may occur at the screening level are addressed through the use of Monte Carlo simulation).***

Life history parameters for end point species. Species-specific parameters necessary to estimate exposure using the equation described above are listed in Tables F4.1, F4.2, F4.3, and F4.4.

Fillet-to-whole fish ratios. Contaminant-specific fillet-to-whole fish ratios were calculated to estimate the whole-body contaminant concentration for fish for which only fillet analyses were performed. For organic chemicals, the ratios were developed using data from 15 largemouth bass and 10 channel catfish for which both fillet and carcass (whole-body minus fillet) analyses were performed. Whole fish contaminant concentrations were determined by calculating the weighted average of the fillet and carcass contaminant concentrations (weighted by the proportion of the total fish weight represented by the carcass and fillet weights, respectively). The contaminant concentration in the whole fish was then divided by the fillet concentration to produce the fillet-to-whole fish ratio (Table F4.5). Bass and catfish were pooled to produce one ratio for each contaminant. Ratios were not available for all contaminants detected in the larger pool of Clinch River fish. Ratios for these contaminants were approximated using ratios for related contaminants. The ratios for arsenic, copper, mercury, selenium, and zinc were obtained from spotted bass collected in the vicinity of the Portsmouth Gaseous Diffusion Plant (Southworth et al. 1994). Mean ratio values were used to generate all whole-fish contaminant concentration estimates.

Contaminant concentrations in abiotic and biotic media. Contaminant concentrations in water, and fish are needed to estimate exposure. The UCL_{95} [calculated using the product limit estimator (PLE)] for contaminants detected in water and fish from the Clinch River are presented in Table F4.6. Note that data were not available for all size classes of fish in all subreaches. Contaminant concentrations in the missing size classes were assumed to be equivalent to that in the next higher size class for which contaminants were detected. If contaminants were detected in fish in size classes above and below that in which data were missing, concentrations were estimated to be the mean of the concentrations in the higher and lower size classes.

Exposure modeling using point-estimates. To estimate contaminant exposure experienced by osprey, the following assumptions were made:

1) body weight = 1.5 kg
2) food consumption = 0.3 kg/d (fresh weight)
3) water consumption = 0.077 L/d
4) soil consumption = 0 kg/d
5) diet consists 100% of fish
6) fish sizes consumed: 0-10 cm 3.3%
   11-20 cm 42.1 %
   21-30 cm 46.7 %
   31-40 cm 6.6 %
   >41 cm 1.3 %
7) all prey selected based upon size and not by species
8) individuals forage exclusively within each given subreach

To estimate contaminant exposure experienced by great blue heron, the following assumptions were made:

1) body weight = 2.39 kg
2) food consumption = 0.42 kg/d (fresh weight)
3) water consumption = 0.1058 L/d
4) soil consumption = 0 kg/d
5) diet consists 100% of fish or other aquatic prey.
6) contaminant concentration in fish is representative of that in other aquatic prey.

7) fish sizes consumed: 0-10 cm 39.2%
11-20 cm 47.1 %
21-30 cm 13.7 %
8) all prey selected based upon size and not by species
9) individuals forage exclusively within each given subreach

To estimate contaminant exposure experienced by mink, the following assumptions were made:

1) body weight = 1 kg
2) food consumption = 0.137 kg/d (fresh weight)
3) water consumption = 0.099 L/d
4) soil consumption = 0 kg/d
5) diet consists 54.6% of fish or other aquatic prey.
6) contaminant concentration in fish is representative of that in other aquatic prey.
7) fish sizes consumed: 0-10 cm 72%
11-20 cm 28 %
8) all prey selected based upon size and not by species
9) individuals forage exclusively within each given subreach
10) all non-aquatic prey consumed are uncontaminated.

To estimate contaminant exposure experienced by river otter, the following assumptions were made:

1) body weight = 8 kg
2) food consumption = 0.9 kg/d (fresh weight)
3) water consumption = 0.64 L/d
4) soil consumption = 0 kg/d
5) diet consists 100% of fish or other aquatic prey.
6) contaminant concentration in fish is representative of that in other aquatic prey.
7) fish sizes consumed: 0-10 cm 16.7%
11-20 cm 16.7%
21-30 cm 16.7 %
31-40 cm 25 %
>41 cm 25 %
(50%<30 cm, 50%>30 cm)
8) all prey selected based upon size and not by species
9) individuals forage exclusively within each given subreach

Using the Clinch River piscivore exposure model and the assumptions and data described above, exposure to contaminants was estimated for osprey, great blue heron, mink, and river otter using the CR/PC system. Exposure estimates for all analytes detected in fish samples are presented in Tables F4.7, F4.8, F4.9, and F4.10. Some contaminants were detected in water but not in fish tissues. Exposure to these contaminants is considered as part of the benchmark screening process and are presented in Sect. 6.4.3.1.

Exposure modeling using Monte Carlo simulations. Employing point estimates for the input parameters in the Clinch River Piscivore exposure model does not take into account the variation and uncertainty associated with the parameters and therefore may over or under estimate the contaminant exposure that end points may receive in any given reach. In addition, calculating the model using point estimates produces a point estimate of exposure. This estimate provides no information concerning the distribution of exposures or the likelihood that individuals within a reach will actually experience potentially hazardous exposures. To incorporate the variation in exposure parameters and to provide a better estimate of the potential exposure experienced by piscivores in each reach, the exposure model was re-calculated using Monte Carlo simulations. A detailed discussion of the theory and application of Monte Carlo simulation is presented in Sect. 5.5.

Monte Carlo simulations of exposure estimates were performed for all end point-subreach-contaminant combinations where comparison of point exposure estimates to NOAELs produced HQs$\geq$1 (NOAELs are presented in Sect. 6.4.2.1.; Screening of exposure estimates against NOAELs is presented in Sect. 6.4.3.1.). End point-subreach-contaminant combinations for which Monte Carlo simulations were performed are presented in Table F4.11.

Distributions were used for the following parameters in the Clinch River piscivore exposure model: end point body weight, average contaminant concentrations in fish and in water, and the proportion of aquatic prey in mink diet. All distributions were assumed to be normal, except osprey body weight which was assigned a triangular distribution (while a standard deviation was not available, a range was). Because these wildlife are mobile, the contaminant concentration they are exposed to on a daily basis is best represented by the average concentration instead of the entire distribution. The standard error of the mean was used to describe variation in the average contaminant concentration. All other distributions employed the calculated standard deviation of the observed data. Piscivore body weight distributions and values are listed in Table 6.15.

Table 6.15. Piscivore body weight distributions and values

End point	Distribution	Mean (kg)	Standard deviation	Range	Source
Osprey (♀)	triangular	1.568		1.25–1.9	Dunning 1984
Great blue heron (♀)	normal	2.204	0.337		Dunning 1984
Mink (♀)	normal	0.974	0.202		EPA 1993
River otter	triangular	8		5.84–10.4	EPA 1993

Sources:

Dunning, J. B. 1984. *Body Weights of 686 Species of North American Birds*. Western Bird Banding Association Monograph No. 1. Eldon Publishing Co., Cave Creek, Ariz.

EPA (U. S. Environmental Protection Agency). 1993. *Wildlife Exposure Factors Handbook*. Vol. I. EPA/600/R-93/187a. Office of Research and Development, Washington, D.C.

Exposure of female individuals was modeled because females were determined to be the most sensitive subgroup. Toxicity data used to evaluate the significance of estimated contaminant exposures generally consider effects on reproduction. Because reproductive effects are most likely to be evident among female individuals, females were chosen as the best models for contaminant exposure. In addition, adverse impacts to females are most likely to result in population-level effects.

The proportion of aquatic prey in the diets of osprey, heron, and otter were assumed to be 100%. No data suggest that non-aquatic prey constitute a significant portion of their diet (see end point discussion, above). In contrast, mink have a very variable diet. Aquatic prey (fish, amphibians, crayfish, etc.) may make up from 16% to 92%. Nine observations from five studies indicate the proportion of aquatic prey to be 0.546±0.21 (mean±standard deviation; Table F4.3). Mean and standard errors for contaminants in fish are presented in Table F4.12.

Monte Carlo simulations were performed using the @Risk software. Samples from each distribution were selected using latin hypercube sampling. The number of iterations, or recalculations, of each exposure simulation was determined by the convergence criteria set in the software. Under these criteria, iterations are performed until the between-iteration percent change in the percentiles, mean, and standard deviation are below 1.5% (i.e., the percentile, mean, and standard deviation for the latest iteration is <1.5% different than the those from the previous iteration). Using this convergence criteria, from 600 to 1000 model iterations were performed for each exposure estimate. Monte Carlo estimates of contaminant exposures are presented in Table F4.13.

6.4.1.2 Internal exposure of great blue herons to contaminants

To determine if contaminants from the ORR are being bioaccumulated by piscivorous wildlife, great blue heron eggs and chicks were collected from two colonies located within 3 km of the ORR and two colonies located >10 km from the site. Analyses were performed to determine the concentrations of arsenic, chromium, mercury, and PCBs in eggs, and feathers, liver, fat, and muscle of chicks. Elevated levels of Cr, Hg, and PCBs were observed in eggs from the ORR colonies (Tables N2 and N4). Hg concentrations in feathers and liver (Table N3) and PCB concentrations in fat (Table N5), liver (Table N6), and muscle (Table N8) were significantly elevated in samples from the ORR as compared to data from the offsite locations. A detailed discussion of these data are presented in Sect. 3.5.

6.4.2 Effects Assessment for Piscivorous Wildlife

6.4.2.1 Single chemical toxicity data

Toxicological benchmarks for piscivorous wildlife. To determine if the contaminant exposure experienced by piscivorous wildlife that use the Clinch River could produce adverse effects, exposure estimates from Sect. 6.4.1.1. were compared to NOAELs and LOAELs derived according to the methods outlined by Opresko et al. (1994) and EPA (1993d). NOAELs represent the highest exposure at which no adverse effects were observed among the animals tested. LOAELs represent the lowest exposure at which significant adverse effects are observed.

Toxicological studies of the effects of contaminants observed in the Clinch River were obtained from the open literature. Due to differences in physiology between birds and mammals, separate studies were located (if available) for each wildlife class. Only studies of the effects of chronic oral exposures, whether in food, water, or by oral intubation, were used. To make the NOAELs and LOAELs relevant to possible population effects, preference was given to studies that evaluated effects on reproductive parameters. In the absence of a reproduction end point, studies that considered effects on growth, survival, and longevity were used.

In cases where a NOAEL for a specific chemical was not available, but a LOAEL had been determined experimentally or where the NOAEL was from a subchronic study, the chronic NOAEL was estimated. EPA (1993d) suggests the use of uncertainty factors of 1 to 10 for subchronic to chronic NOAEL and LOAEL to NOAEL estimation. Because no data were available to suggest the use of lower values, uncertainty factors of 10 were used in all instances in which they were required.

Smaller animals have higher metabolic rates and are usually more resistant to toxic chemicals because of more rapid rates of detoxification. It has been shown that metabolism is proportional to body surface area which, for lack of direct measurements, can be expressed in terms of body weight (bw) raised to the 2/3 power ($bw^{2/3}$) (EPA 1980a). As shown in Eq. 6.5, if the dose (d) itself has been calculated in terms of unit body weight (i.e., mg/kg), then the dose per unit body surface area (D) equates to:

(6.5)

$$D = \frac{d \ x \ bw}{bw^{2/3}} = d \ x \ bw^{1/3}$$

The assumption is that the effective dose per body surface area for species "a" and "b" would be equivalent. Therefore, knowing the body weights of two species and the dose (d_b) producing a given effect in species "b," the dose (d_a) producing the same effect in species "a" can be determined:. Using this approach, if a NOAEL was available for the test species ($NOAEL_t$), the equivalent NOAEL for a wildlife species ($NOAEL_w$) was calculated by using the adjustment factor for differences in body size:

(6.6)

$$NOAEL_w = NOAEL_t \left(\frac{bw_t}{bw_w} \right)^{1/3}$$

This methodology is equivalent to that the EPA uses in their carcinogenicity assessments and Reportable Quantity documents for adjusting from animal data to an equivalent human dose.

NOAELs and LOAELs were derived for osprey, great blue heron, and mink. Mammalian and avian NOAELs and experimental information used to estimate wildlife NOAELs and LOAELs (e.g. test species, test end points, citation, etc.) are listed in Tables F4.14 and F4.15.

More contaminants were detected in surface water than were observed in fish tissue. To simplify exposure estimation and contaminant screening, a water benchmark (Eq. 6.7) was used for these contaminants. The water benchmark is the concentration of the contaminant in the drinking water of an animal (C_w, in mg/L) resulting in a dose equivalent to a $NOAEL_w$. It can be calculated from the daily water consumption rate (W, in L/day) and the average body weight (bw_w) for the species:

(6.7)

$$C_w = \frac{NOAEL_w \ x \ bw_w}{W}$$

Water benchmarks for osprey, great blue heron, mink, and river otter are presented in Table F4.16.

Ecotoxicological profiles. Ecotoxicological profiles of the effects of As, Cu, Hg, Se, DDT, and PCBs to wildlife are presented in Appendix F1.

6.4.2.2 Effects of contaminants on the reproductive performance of mink

Studies, conducted at Michigan State University Experimental Fur Farm, evaluated bioaccumulation of contaminants and reproductive effects in mink fed fish collected from Poplar Creek, the Clinch River (upstream of Melton Hill Dam) and the ocean. Mink were fed five diets consisting of 75% fish and 25% commercial mink diet (Table 6.16).

Table 6.16. Diet composition and contaminant concentrations for each diet

Diet	Fish Composition	Contaminant Concentration	
		Mercury	Aroclor 1260
A	75% ocean	0.02 ± 0.00	0.169 ± 0.002
B	75% Clinch River	0.05 ± 0.00	11.44 ± 0.327
C	25% Poplar Creek 50% ocean	0.09 ± 0.00	4.69 ± 0.174
D	50% Poplar Creek 25% ocean	0.15 ± 0.01	10.41 ± 0.250
E	75% Poplar Creek	0.22 ± 0.01	20.67 ± 0.458

Twenty- three PCB congeners were also present in varying amounts. Concentrations of most congeners increased progressively from diets A through E (Table N12).

Ten mink (8 females and 2 males) were fed each diet for ~7 months (3 months prior to breeding - 6 weeks postpartum). Reproductive indices measured included: number of females mated; number of females whelping; length of gestation; number of kits whelped (alive, dead); kit sex ratio; average kit body weight at birth, 3, and 6 weeks of age; and kit survival to 3 to 6 weeks of age. At 6 weeks of age, 3 kits from dietary groups A, B, C, and E were euthanized, organs (liver, spleen, and kidneys) were weighed, and tissue samples (liver, kidney, and remaining carcass) were analyzed for contaminant accumulation. (note: kits from diet D were not sampled). At the termination of the study, all adult mink were necropsied. Organs (brain, liver, kidneys, heart, lungs, gonads, and adrenal glands) were weighed and examined for histopathologies. Adipose tissue, liver, kidney, and hair were analyzed for contaminant accumulation. Liver tissue also was analyzed for ethoxyresorufin-o-deethylase (EROD) activity.

The bioaccumulation of mercury in liver, kidney, and hair (Table N13), and Aroclor 1260 (and other PCB congeners) in liver and fat (Tables N14 and N15) substantially increased in adult female mink from groups fed diet A up to diet E. Mink offspring also bioaccumulated mercury in kidney tissue and carcasses and many other PCB congeners in the liver and carcasses (Tables N16 and N17), increasing progressively from mink fed diets A through E. The lowest levels were observed for mink fed diet A and increased to a maximum observed among mink fed diet E.

Significant effects were observed only among mink fed diet E; no adverse effects were observed for any other diet. Adverse effects from diet E included: weight reduction in adult mink and their offspring, reduction in litter size, and increase in liver EROD activity in adult females. Weight reduction was observed at the end of the experimental period, increasing magnitude from diet groups A to E. At the end of the experiment, the mean whole body weights of female mink in diet group E were significantly less ($p = 0.03$) than mean weights of females in diet group A (percent reduction =20%). Mean female relative organ weights (organ weights/body weight) were not significantly different among diet groups. At 6 weeks of age, mean whole body weights were also significantly lower ($p = 0.004$) in male kits from diet group E compared to those from diet group A (percent reduction =17%). Similar trends were observed for female kits, although differences were not statistically significant. No histological lesions were attributed to any diet. Mean litter size was significantly reduced ($p = 0.01$) in diet group E compared to diet groups A, B, and C (percent reduction relative to diet A=38%); but not diet group D. Liver EROD activity was significantly increased in adult female mink from diet groups D and E compared to those from diet group A.

6.4.2.3 Great blue heron reproduction survey

To determine if contaminants from the ORR are adversely affecting piscivorous wildlife, reproductive success of great blue herons at two colonies located within 3 km of the ORR and two colonies located >10 km from the site was monitored. Data were collected from each nest colony between 1992 and 1994. The mean number of eggs/nest, number of chicks/nest, egg weight, and eggshell thickness did not differ between colonies within 3 km of the ORR and those >10 km away (Table N8) A detailed discussion of these data are presented in Sect. 3.5.

6.4.3 Risk Characterization for Piscivorous Wildlife

While three lines of evidence are available to assess risks to piscivorous wildlife along the Clinch River, not all are available for every end point. Single chemical toxicity data are available for all four end points. However, toxicity tests and field surveys are available only for mink and great blue heron respectively.

6.4.3.1 Single chemical toxicity data

Exposure estimates generated by the exposure model (see Sect. 6.4.1.1.) produced by both point estimates of parameter values and Monte Carlo simulation represent exposure at the individual level. The exposure estimates using point estimates of parameter values at each individual sampling point are used to identify COPECs and locations that contribute significantly to risk. In contrast, the watershed-level exposure distributions generated by Monte Carlo simulation represent the likelihood that an individual within the area for which exposure is modeled will experience a particular exposure.

Two types of single chemical toxicity data are available with which to evaluate piscivore contaminant exposure: NOAELs and LOAELs. NOAELs are used to screen exposure estimates generated from point-estimates of exposure parameters; if the estimate is greater than the NOAEL, adverse effects are possible and additional evaluation is necessary (i.e., exposure modeling using Monte Carlo simulation). LOAELs are compared to the exposure distribution generated by the Monte Carlo simulation. If the LOAEL is lower than the 80th percentile of the exposure distribution, there is a >20% likelihood that individuals within the modeled location are experiencing contaminant exposures that are estimated to produce adverse effects. By combining literature-derived population density data with the likelihood or probability of exceeding the LOAEL, population-level impacts may be estimated.

Screening point estimates of exposure. To determine if the contaminant exposures experienced by osprey, great blue heron, mink, and river otter along the CR/PC are potentially hazardous, the total contaminant exposure estimates (generated using point estimates of parameter values; Tables F4.7, F4.8, F4.9, and F4.10) were compared to estimated NOAELs for these species (Tables F4.14 and F4.15). To quantify the magnitude of hazard, a HQ was calculated where: HQ = exposure/NOAEL. HQs > 1 indicate that individuals may be experiencing exposures that are in excess of NOAELs and suggest that adverse effects may be occurring. HQs for osprey, great blue heron, mink, and river otter along the Clinch River are presented along with the point estimates of exposure in Tables F4.7, F4.8, F4.9, and

F4.10. It should be noted that because few data are available for specific PCB congeners, all PCBs were compared to Aroclor-1254 toxicity data.

The spatial distribution of contamination and potential risks to osprey, great blue heron, mink and river otter in the CR/PC system are illustrated in Figs. 6.20, 6.21, 6.22 and 6.23, respectively. These figures display the sum of the NOAEL-based HQs (e.g., ΣTUs) for the six most important contaminants: arsenic, copper, DDT, mercury, selenium, and total PCBs. (Total PCBs were determined by summing the exposure of all Aroclors within a given reach. This value was then compared to the Aroclor-1254 NOAEL). Importance of contaminants was determined based upon the magnitude of the HQ. River subreaches were arranged from the northern-most to the southern-most.

While the ΣTUs scale differed among end points (lowest for mink, Fig. 6.22; highest for osprey, great blue heron and river otter, Figs. 6.20, 6.21, and 6.23), the pattern of ΣTUs was similar among all four end points; with the exception of a PCB spike at subreach 2.04, the maximum ΣTUs were observed at the Poplar Creek subreaches (13, 3.01, 3.02, 3.03, and 3.04). For osprey and great blue heron, the contaminants contributing the most to total risk are mercury followed by total PCBs (Figs. 6.20 and 6.21). PCBs, mercury, and selenium account for the majority of risk to mink and river otter (Figs. 6.22 and 6.23). A summary of the subreaches where NOAEL-based HQs>1 were observed is listed in Table F4.11 by end point and contaminant.

Screening of water data. More chemicals were detected in Clinch River water than in fish flesh. To evaluate the risk that these contaminants may present, HQs were calculated for these contaminants using the UCL_{95} water concentration and the water benchmarks from Table F4.16. HQs for these contaminants are presented in Table F4.17. With the exception of thallium in reach 1, no contaminant produced an HQ>1 for any end point in any Clinch River reach. In reach 1, thallium in water exceeded the benchmark for both mink and river otter (Table F4.17). This observation must be viewed with caution, because thallium was detected in only one of 48 samples from this reach.

Screening Monte Carlo simulation estimates of exposure. To incorporate the variation present in the parameters employed in the exposure model, Monte Carlo simulations were performed for exposure of each species to contaminants where NOAEL-based HQs>1 were observed (see Table F4.11). The mean, standard deviation, and 80th percentile of the simulated exposures are presented in Table F4.13. By superimposing NOAEL and LOAEL values on these distributions, the likelihood of an individual experiencing potentially hazardous exposures can be estimated and the magnitude of risk to individuals may be determined. These comparisons are presented in Table F4.18. Interpretation of the comparison of exposure distributions to NOAELs and LOAELs is described in Table 6.17.

To evaluate the likelihood and magnitude of population-level effects on piscivores, literature-derived population density data (expressed as number of individuals/km of stream or pond shoreline) were combined with lengths of streams or pond shorelines for which risks were assessed to estimate the number of individuals of each end point species expected to be present in each watershed.

Table 6.17. Comparison of exposure distributions to NOAELs and LOAELs

Comparison	Meaning	Risk-based interpretation
NOAEL >80th percentile of exposure distribution	<20% of exposuresgreater than NOAEL	Individual- and population-level adverse effects are highly unlikely
NOAEL < 80th percentile < LOAEL	>20% of exposures greater than NOAEL,but <20% of exposures greater than LOAEL	Individuals experiencing exposures at the high end of the distribution may experience adverse effects, but those effects are unlikely to significantly contribute to effects on the Oak Ridge Reservation population.
LOAEL <80th percentile of exposure distribution	>20% of exposures greater than LOAEL	Effects on some individuals are likely and they may contribute significantly to effects on the Oak Ridge Reservation population.

NOAEL = no-observed-adverse-effect level; LOAEL = lowest-observed-advserse-effect level.

Literature-derived population densities used for each end point species were: mink: 0.6/km; river otter 0.37/km; and great blue heron; 2.3/km. Linear population density data were not available for osprey (osprey are addressed separately). It should be noted that density values for all end point species but the great blue heron represent the maximum values obtained from the literature (Tables F4.2, F4.3, and F4.4). The literature values for herons (Table F4.2) appear inflated and are not believed to accurately represent densities in the CR/PC system. Therefore, the minimum value, which is consistent with local densitities, was used. Population estimates based upon these densities are listed in Table 6.18 for the primary CR/PC subreaches.

The number of individuals within a given subreach likely to experience exposures greater than LOAELs can be estimated using cumulative binomial probability functions (Dowdy and Warden 1983). Binomial probability functions are estimated using the following equation:

$$b(y;n;p) = \binom{n}{y} p^{y} (1-p)^{n-y} \tag{6.8}$$

where:

- y = the number of individuals experiencing exposures > LOAEL
- n = total number of individuals within the subreach
- p = probability of experiencing an exposure in excess of the LOAEL
- b (y; n; p) = probability of y individuals out of a total of n, experiencing an exposure > LOAEL, given the probability of exceeding the LOAEL=p.

By solving this equation for y=0 to y=n, a cumulative binomial probability distribution may be generated that can be used to estimate the number of individuals within a subreach that are likely to experience adverse effects. Summing the number within each subreach across all subreaches and dividing by the total estimated CR/PC population, the proportion of the population potentially at risk may be estimated.

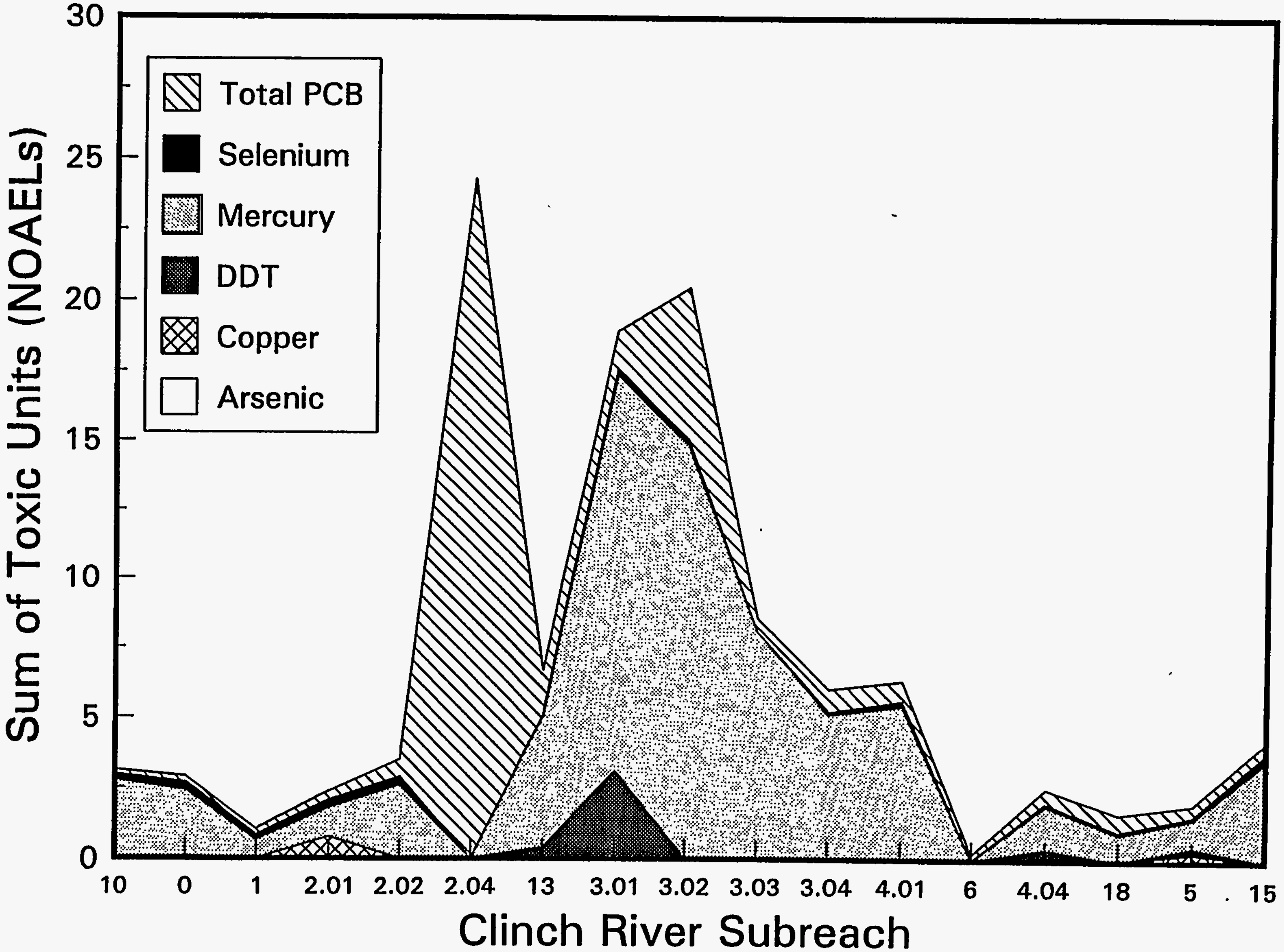


Fig. 6.20. Sum of no-observed-adverse-effect level–based toxic units for osprey in the Clinch River/Poplar Creek system.

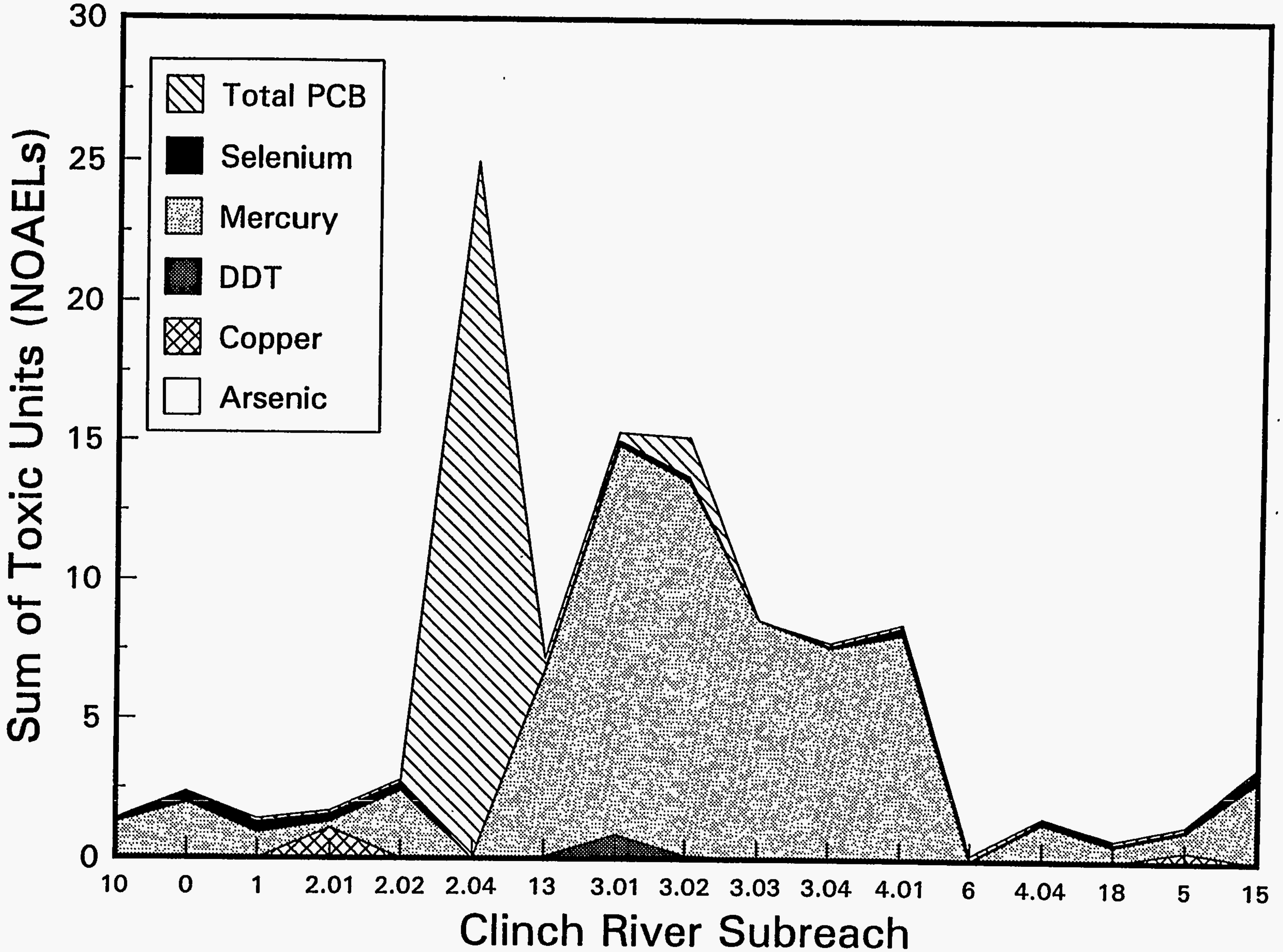


Fig. 6.21. Sum of no-observed-adverse-effect level–based toxic units for great blue heron in the Clinch River/Poplar Creek system.

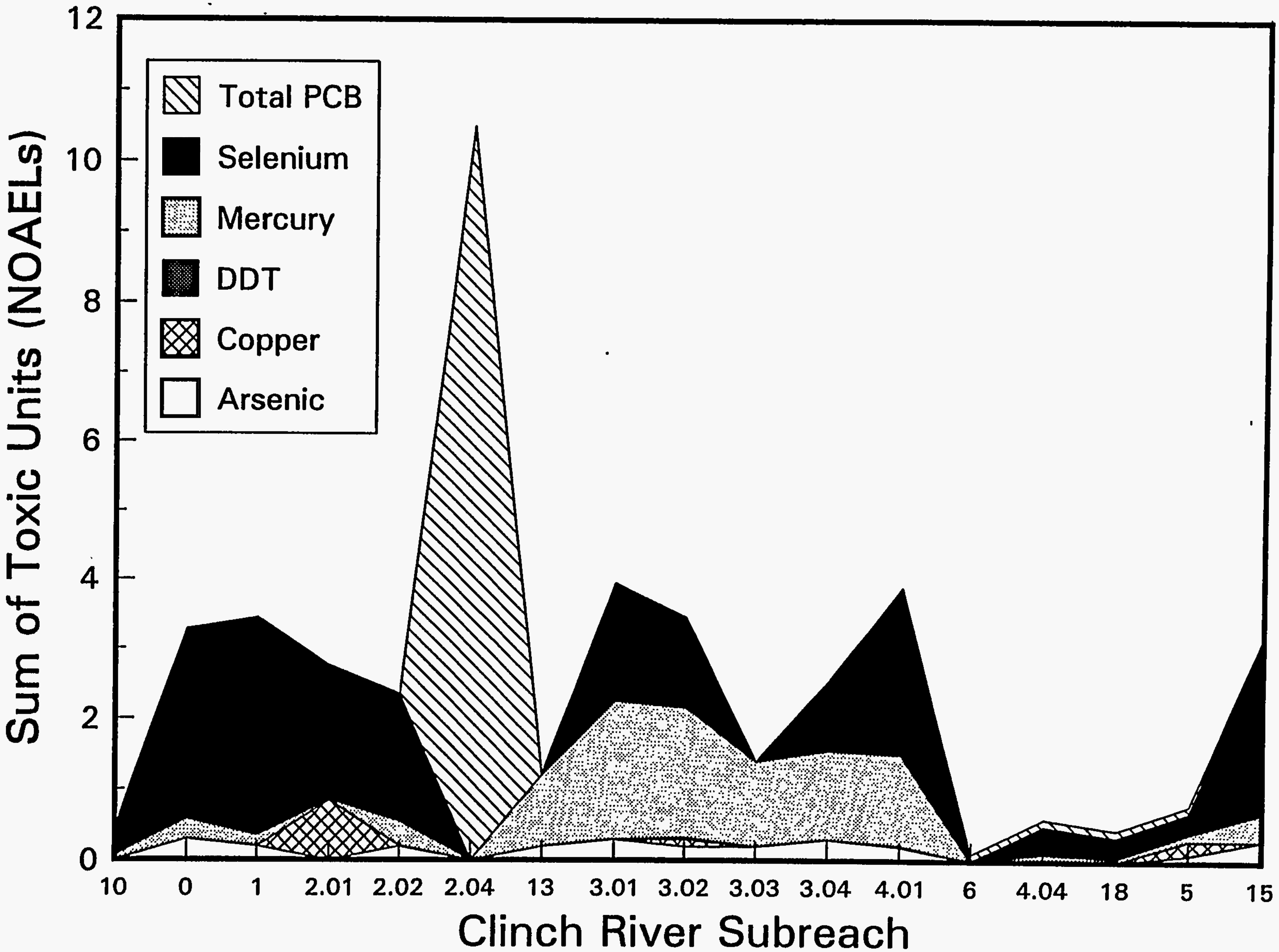


Fig. 6.22. Sum of no-observed-adverse-effect level–based toxic units for mink in the Clinch River/Poplar Creek system.

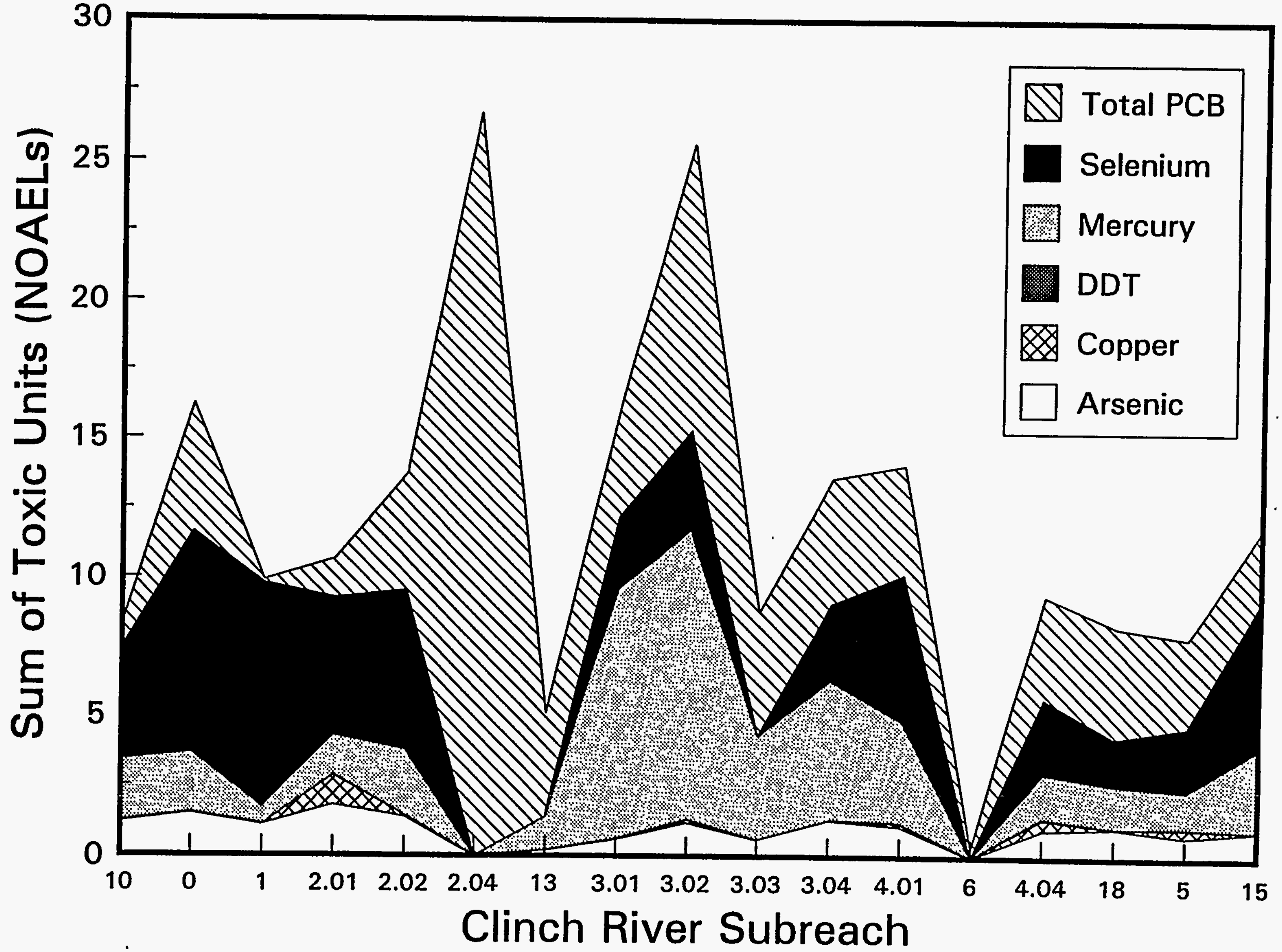


Fig. 6.23. Sum of no-observed-adverse-effect level–based toxic units for river otter in the Clinch River/Poplar Creek system.

It should be noted that population estimates focused on those reaches that are within the OU and immediately downstream of the ORR (reaches 2, 3, and 4). These reaches were selected because (1) they represent the greatest exposures, (2) using all reaches would greatly over-estimate the population immediately adjacent to the ORR and reduce the likelihood of detecting a potential population-level effect.

Because the area of this population range is small relative to the mobility of these species and because it is limited to contaminated areas, it is a conservative definition of the exposed populations of mink, otter, and herons.

Table 6.18. Estimated number of individuals by watershed

Subreach	Length (km)	Mink	River otter	Great blue heron
2.01	7.7	5	3	18
2.02	10.6	6	4	24
2.03	28.7	17	11	66
2.04	23.0	14	9	53
3.01	3.1	2	1	7
3.02	5.6	3	2	13
3.03	9.7	6	4	22
3.04	4.2	3	2	10
4.01	19.2	12	7	44
4.02	19.5	12	7	45
4.03	18.0	11	7	41
4.04	7.6	5	3	17
Total	156.9	96	60	360

Binomial probability distributions were generated only for contaminant-end point-subreach combinations where the percent of the exposure distribution exceeding the LOAEL was 20% to 80% (these values are reported in Table F4.18). If the percent of the exposure distribution exceeding the LOAEL was <20%, it was assumed that no individuals within that subreach were experiencing adverse effects. Conversely, if the percent of the exposure distribution exceeding the LOAEL was > 80%, it was assumed that all individuals within that subreach were experiencing adverse effects. The total numbers of individuals of each end point species estimated to be experiencing adverse effects within each subreach are summarized in Table F4.19. Because all exposure estimates were either >80% or <20%, calculation of cumulative binomial probability distributions was not necessary.

While all great blue heron that forage within subreaches 3.01 and 3.02 are likely to experience mercury exposures that exceed the LOAEL (Figs 6.24 and 6.25), these birds represent only 6% of the total birds expected to use the CR/PC reaches downstream of the ORR (reaches 2, 3, and 4; Table F4.19). Because the number estimated to be adversely affected is les than 20%, population-level effects to from mercury great blue herons downstream of the ORR are unlikely. PCBs and copper are also unlikely to have adverse effects on the great blue heron population.

Among mink, while estimated exposures to mercury, selenium and PCBs exceeded NOAELs in subreaches downstream of the ORR (Table F4.18), estimated exposures to these contaminants did not exceed LOAELs within any subreach (Tables F4.18 and F4.19). Therefore population-level effects of contaminants to mink are unlikely.

Among river otter, while adverse effects from arsenic, selenium, and copper are unlikely, individual otter within subreaches 3.01 and 2.04 may be adversely affected by exposure to mercury or PCBs, respectively (Figs 6.26 and 6.27; Tables F4.18 and F4.19). Because the river otter is a threatened species in Tennessee, adverse effects to individuals are a concern. Therefore, mercury in subreach 3.01 and PCBs in subreach 2.04 present a risk to river otter.

Osprey at subreaches 3.01 and 3.02, are estimated to receive exposures to mercury in excess of the LOAEL >99% and 70% of the time (Figs 6.28 and 6.29; Table F4.18). No other contaminants in any other subreach are estimated to present a risk to osprey. To evaluate the significance of mercury exposure among osprey within subreaches 3.01 and 3.02, the foraging range of osprey must be considered. Osprey are wide-ranging species, with individuals ranging 10 to 15 km from their nest sites in search of food (Van Daele and Van Daele 1982). EPA (1993g) reports the mean foraging radius for osprey to be 1.7 km with a range of 0.7 km to 2.7 km. Of the three active osprey nests in the vicinity of the ORR (B. Anderson, Pers. Comm.), two are located along Melton Hill Reservoir (reach 1) and one on Poplar Creek approximately at the border between subreaches 3.03 and 3.04 (Fig. 6.30). While the reach 1 nest sites are within 15 km of subreaches 3.01 and 3.02, due to the availability of suitable habitat nearer to their nests and that the mean foraging radius is 1.7 km, birds from these nest sites are unlikely to forage within subreaches 3.01 and 3.02. Therefore potentially deleterious mercury exposure to these birds is unlikely. In contrast, the 3.03/3.04 nest site is within 3 km of subreaches 3.01 and 3.02. Birds from this nest may therefore forage and be exposed to elevated mercury in fish from subreaches 3.01 and 3.02.

To estimate and model the potential mercury exposure for osprey at the 3.03/3.04 nest location, it was assumed that the birds would travel up to 5 km from the nest to forage. Most foraging was assumed to occur near to the nest, with ~50% of diet obtained from within 1 km of nest, 25% obtained from 1 to 2 km, 15% from 2 to 3 km, and 5% each from 3 to 4 km and 4 to 5 km from the nest, respectively. Subreaches that fell within these ranges were identified. Subreaches, approximate distance from nest and estimated contribution to total diet for the nest are listed in Table 6.19.

Table 6.19. Subreaches, approximate distance from nest and estimated contribution to total diet for the nest

Subreach	Distance from nest (km)	Contribution to diet (%)
4.01	~1–4	5
2.04	No data	No data
2.03	No data	No data
2.02	~5	5
13	4–5	5
3.01	~3–4	15
3.02	~3	20
3.03	0–2	25
3.04	0–1	25

To estimate the mercury exposure for osprey from the 3.03/3.04 nest site, Monte Carlo simulation was performed on the sum of the exposure estimates for each subreach within 5 km of the nest site. Exposure from each subreach was weighted by the proportion of the total diet it was projected to contribute (see above). Mean (±STD) mercury exposure for osprey from the 3.03/3.04 nest site was estimated to be 0.043±0.0041 mg/kg/d. While the 80th percentile (0.046 mg/kg/d) is less than the LOAEL (0.056 mg/kg/d) it is greater than the NOAEL (0.006 mg/kg/d). These results indicate that while imminent adverse effects to osprey at the 3.03/3.04 nest site are unlikely (because the LOAEL was not exceeded), adverse effects are possible if birds from this nest consistently experience exposures at the high end of the distribution (because the NOAEL was exceeded).

Effects of retained contaminants.

Arsenic. The 80th percentile for exposure of river otter to arsenic exceeded the NOAEL but not the LOAEL at 11 subreaches (Table F4.18). Both the NOAEL and LOAEL for river otter are based upon a study in which reproductive success and offspring survival was observed among mice fed arsenite for three generations (Schroeder and Mitchner 1971). One dose level administered (1.261 mg/kg/d), designated as the chronic LOAEL, resulted in declining litter size with each successive generation. A chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL correction factor of 0.1. Based on the results of Schroeder and Mitchner (1971), river otter experiencing exposures ≥ LOAEL are likely to display a decline in litter size.

Copper. The 80th percentile for exposure of great blue heron and river otter to copper at subreach 2.01 exceeded the NOAEL but not the LOAEL (Table F4.18). The great blue heron NOAEL and LOAEL for copper were derived from a study of chicks fed copper oxide for 10 wks. (Mehring et al. 1960). While consumption of 61.7 mg/kg/d copper reduced growth and increased mortality, no effects were observed at the 47 mg/kg/d exposure level. The study was considered to represent a chronic exposure, therefore a subchronic-chronic correction factor was not employed. The 47 mg/kg/d exposure was considered to be a chronic NOAEL; the 61.7 mg/kg/d exposure was considered to be a chronic LOAEL

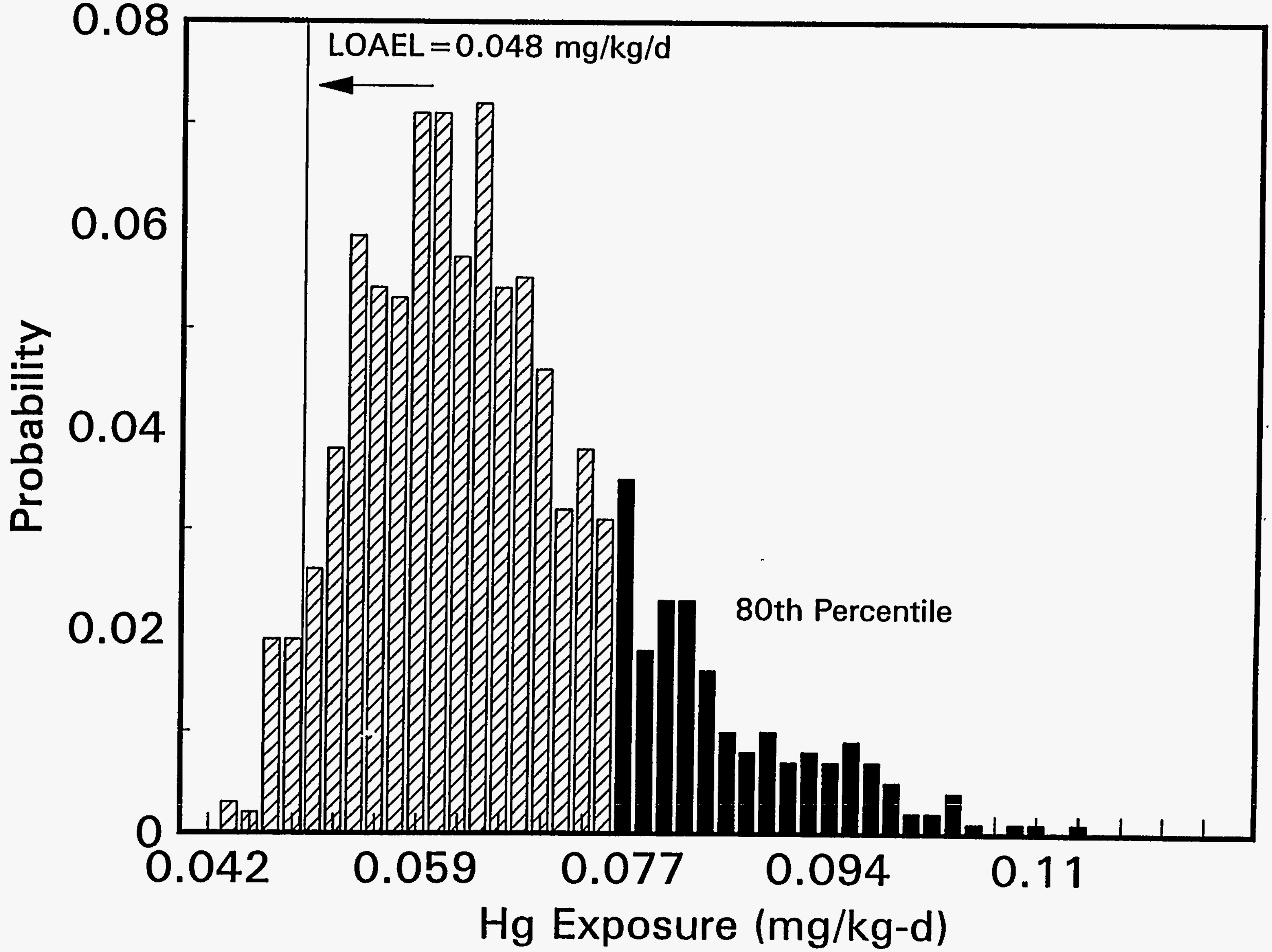


Fig. 6.24. Estimated distribution of mercury exposure for great blue heron in subreach 3.01. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution is greater than the lowest observed-adverse-effect level, adverse effects to individuals resulting from contaminant exposure are likely.

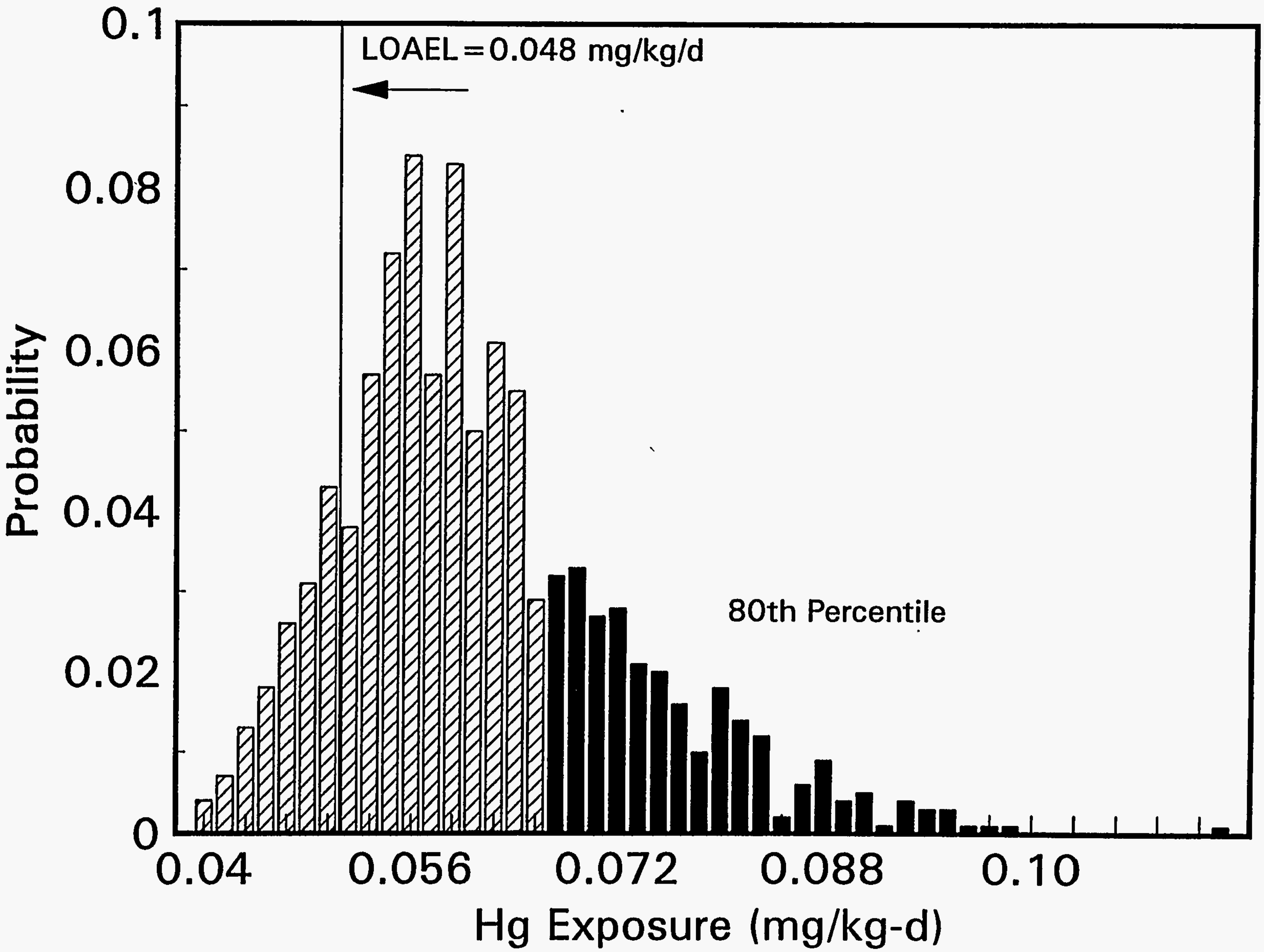


Fig. 6.25. Estimated distribution of mercury exposure for great blue heron in subreach 3.02. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution is greater than the lowest observed-adverse-effect level,, adverse effects to individuals resulting from contaminant exposure are likely.

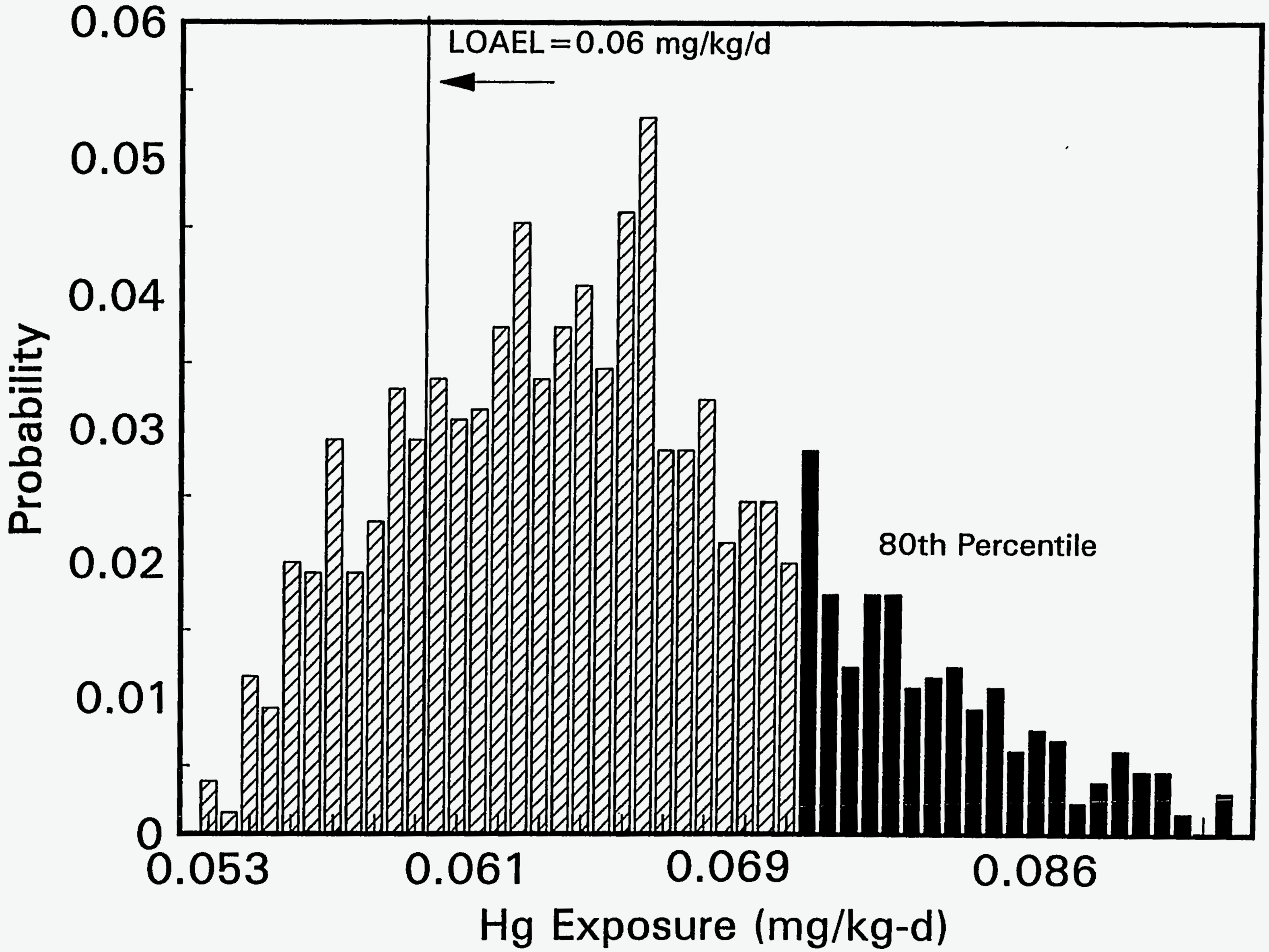


Fig. 6.26. Estimated distribution of mercury exposure for river otter in subreach 3.01. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution is greater than the lowest-observed-adverse-effect level, adverse effects to individuals resulting from contaminant exposure are likely.

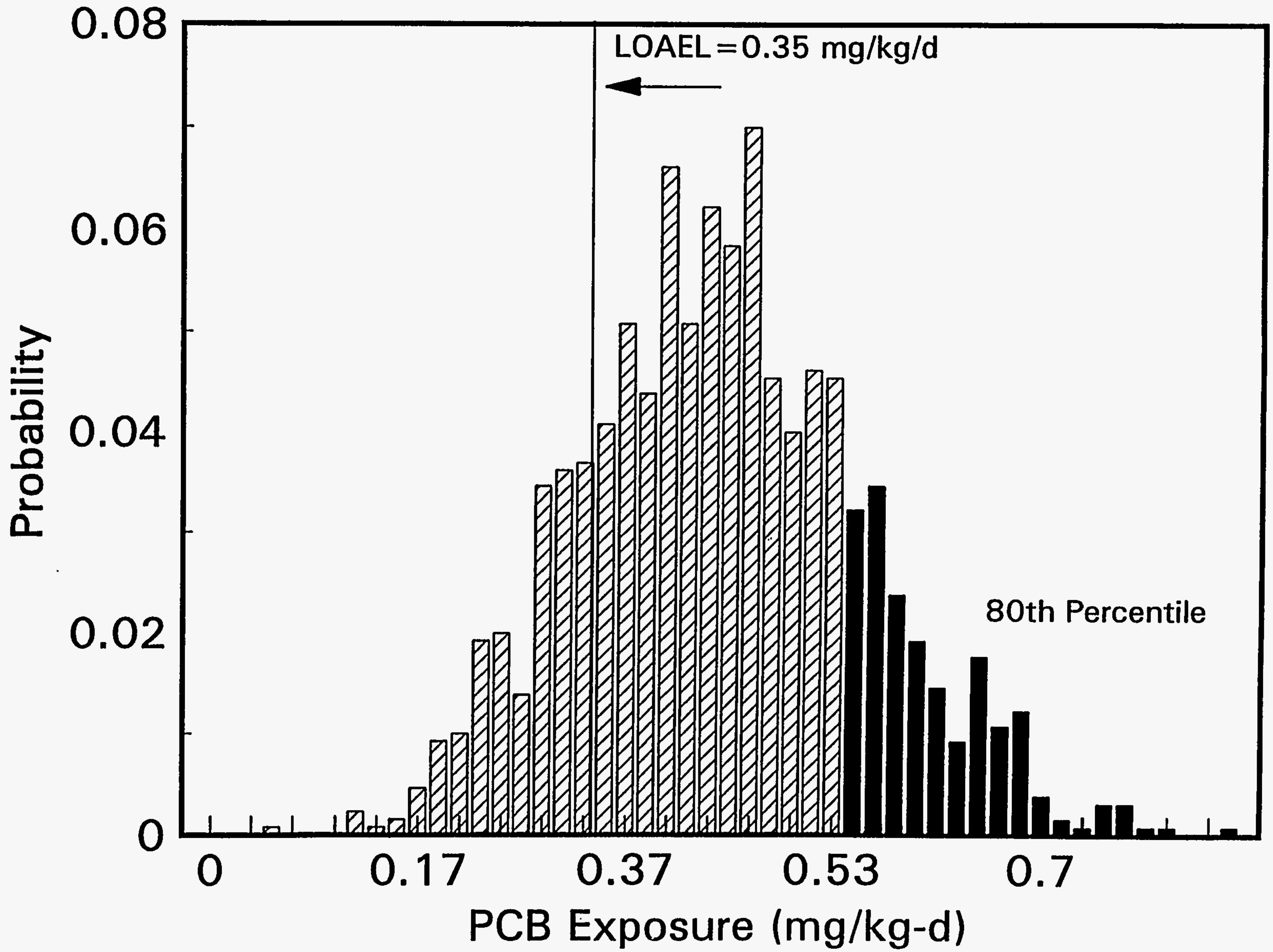


Fig. 6.27. Estimated distribution of PCB exposure for river otter in subreach 2.04. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution is greater than the lowest-observed-adverse-effect level, adverse effects to individuals resulting from contaminant exposure are likely.

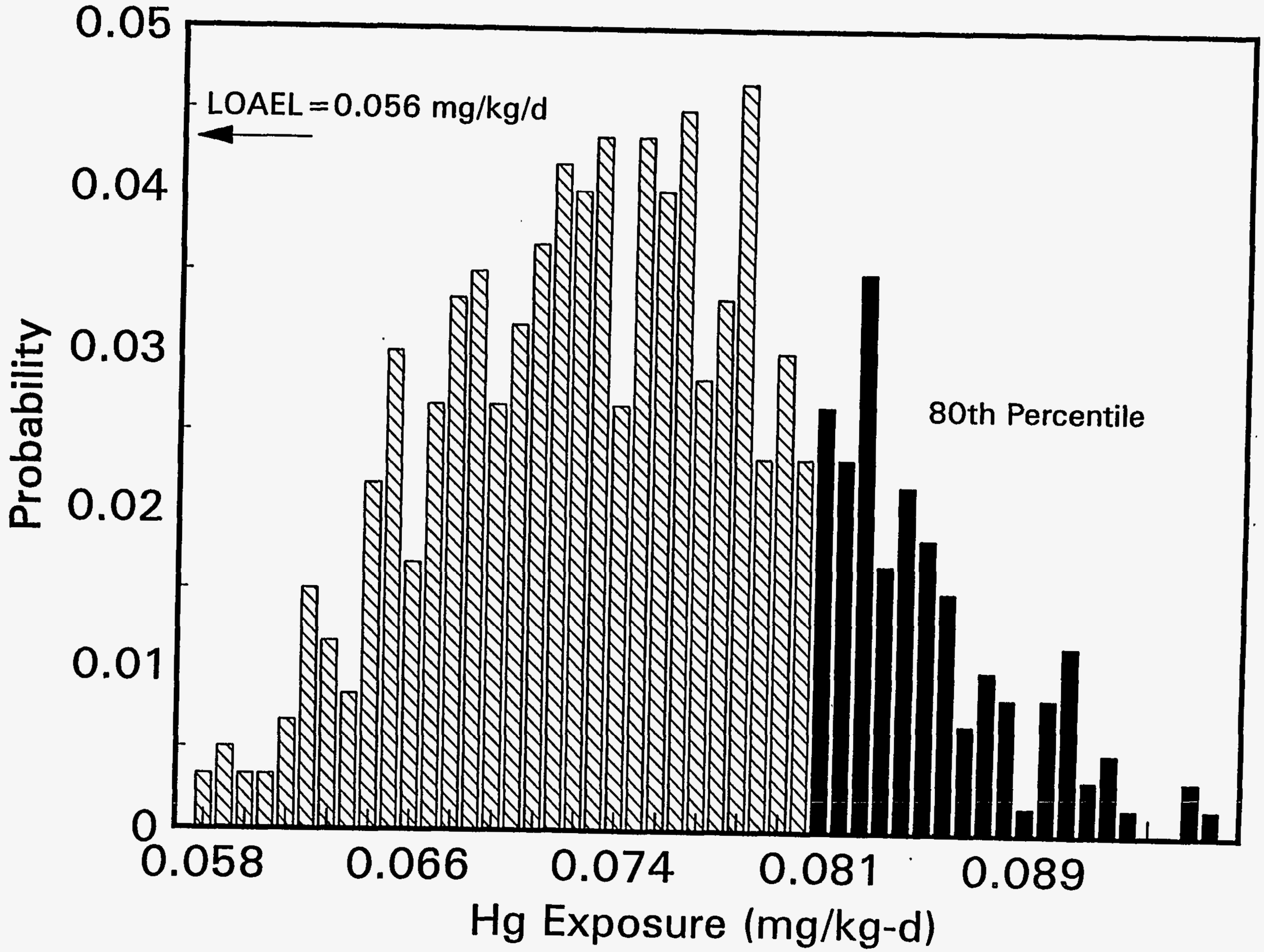


Fig. 6.28. Estimated distribution of mercury exposure for osprey in subreach 3.01. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution is greater than the lowest-observed-adverse-effect level, adverse effects to individuals resulting from contaminant exposure are likely.

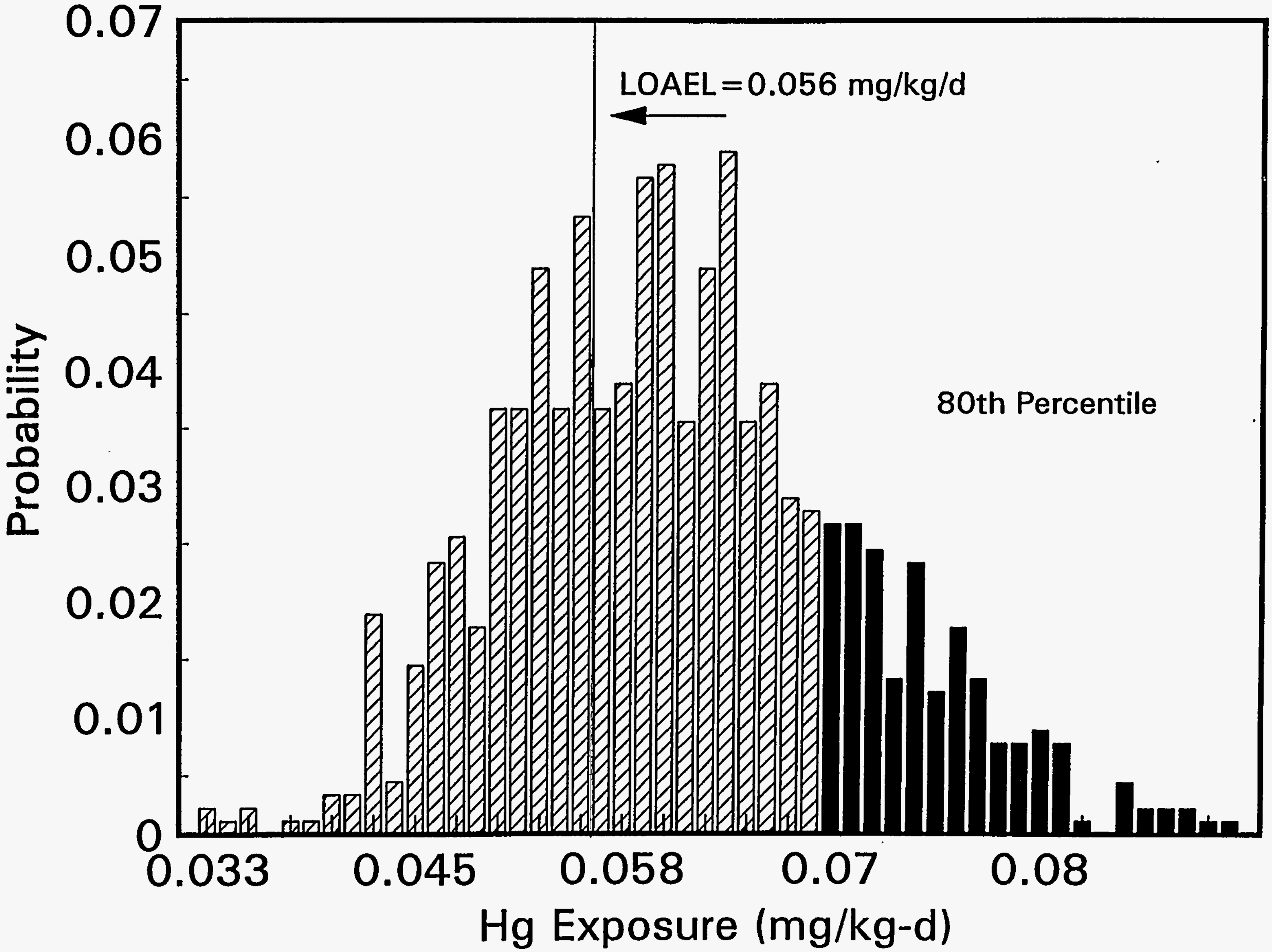


Fig. 6.29. Estimated distribution of mercury exposure for osprey in subreach 3.02. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution is greater than the lowest-observed-adverse-effect level, adverse effects to individuals resulting from contaminant exposure are likely.

Both the NOAEL and LOAEL for river otter are based on a study in which mink were fed copper sulfate for 357 days (including a critical lifestage) (Aulerich et al. 1982). While consumption of 15.14 mg/kg/d copper increased the percentage of mortality in mink kits, no adverse effects were observed at a 11.71 mg/kg/d exposure level. Based on the results of Aulerich et al. (1982), otter experiencing exposures ≥ LOAEL may display a reduction in offspring survival.

DDE. The 80th percentile for exposure of osprey to DDE at subreach 3.01 exceeded the NOAEL but not the LOAEL. The osprey NOAEL and LOAEL for DDE were derived from a study of brown pelicans exposed to DDT for 5 yrs. (Anderson et al. 1975). Because DDE is a metabolite of DDT, effects from DDE were assumed to be comparable to those observed for DDT. Chronic exposure to 0.028 mg/kg/d DDT reduced reproductive success. This dose level was considered to be a LOAEL. Because an experimental NOAEL was not established, the NOAEL was estimated using LOAEL-NOAEL correction factor of 0.1. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individual birds may become evident cannot be defined.

Mercury. For the purposes of this assessment, it is assumed that 100% of the Hg to which piscivores are exposed consists of methyl mercury.

The 80th percentile for mercury exposure experienced by osprey and great blue heron exceeded both the NOAEL and LOAEL at subreaches 3.01 and 3.02; exposure at all other modeled subreaches exceeded the NOAEL but not the LOAEL. Both the avian NOAEL and the LOAEL are based upon a study of mallard ducks fed methyl mercury for three generations (Heinz 1979). The study was considered to represent a chronic exposure and a subchronic-chronic correction factor was not employed. The only dose level administered, 0.064 mg/kg/d, caused hens to lay fewer eggs, lay more eggs outside of the nest box, and produce fewer ducklings. This dose level was considered to be a LOAEL. Because an experimental NOAEL was not established, the NOAEL was estimated using LOAEL-NOAEL correction factor of 0.1. Based on the results of Heinz (1979), birds experiencing exposure ≥ LOAEL are likely to display impaired reproduction.

The 80th percentile for mercury exposure experienced by mink exceeded the NOAEL at reach 13 and at subreaches 3.01, 3.02, 3.03, 3.04, and 4.01; the LOAEL was not exceeded at any modeled subreach. For river otter, while the 80th percentile for mercury exposure exceeded the NOAEL at 14 subreaches, the LOAEL was exceeded only at subreach 3.01 (Table F4.18). The mink and otter NOAELs for mercury was derived from a study of mink fed methyl mercury for 93 days (Wobeser et al. 1976). While consumption of 0.247 mg/kg/d methyl mercury resulted in significant mortality, weight loss, and behavioral impairment, no effects were observed at the 0.15 mg/kg/d exposure level. The 0.15 mg/kg/d exposure was considered to be a NOAEL. Because the study was subchronic in duration (<1 yr), a subchronic-chronic correction factor was applied (NOAEL=0.015). The mink LOAEL for mercury was derived from a study of rats fed methyl mercury for 3 generations (Verschuuren et al. 1976). While consumption of 0.16 mg/kg/d methyl mercury resulted in reduced pup viability, no effects were observed among rats consuming 0.032 mg/kg/d. Because the data were derived from a multigeneration study, the 0.16 mg/kg/d exposure was considered to be a chronic LOAEL. Based on the results of Verschuuren et al. (1976), mink experiencing exposure ≥ LOAEL are likely to display impaired reproduction. Because the NOAEL for mink was derived using a subchronic-chronic correction factor, it is not reasonable to assume that mink experiencing exposure greater than the NOAEL but less than the LOAEL will experience increased mortality. It is more likely that mink whose exposures are at the high end of the distribution will display impaired reproduction (effects based on Verschuuren et al. 1976). The effects are likely to be less pronounced than that displayed among individuals where exposure is greater than the LOAEL.

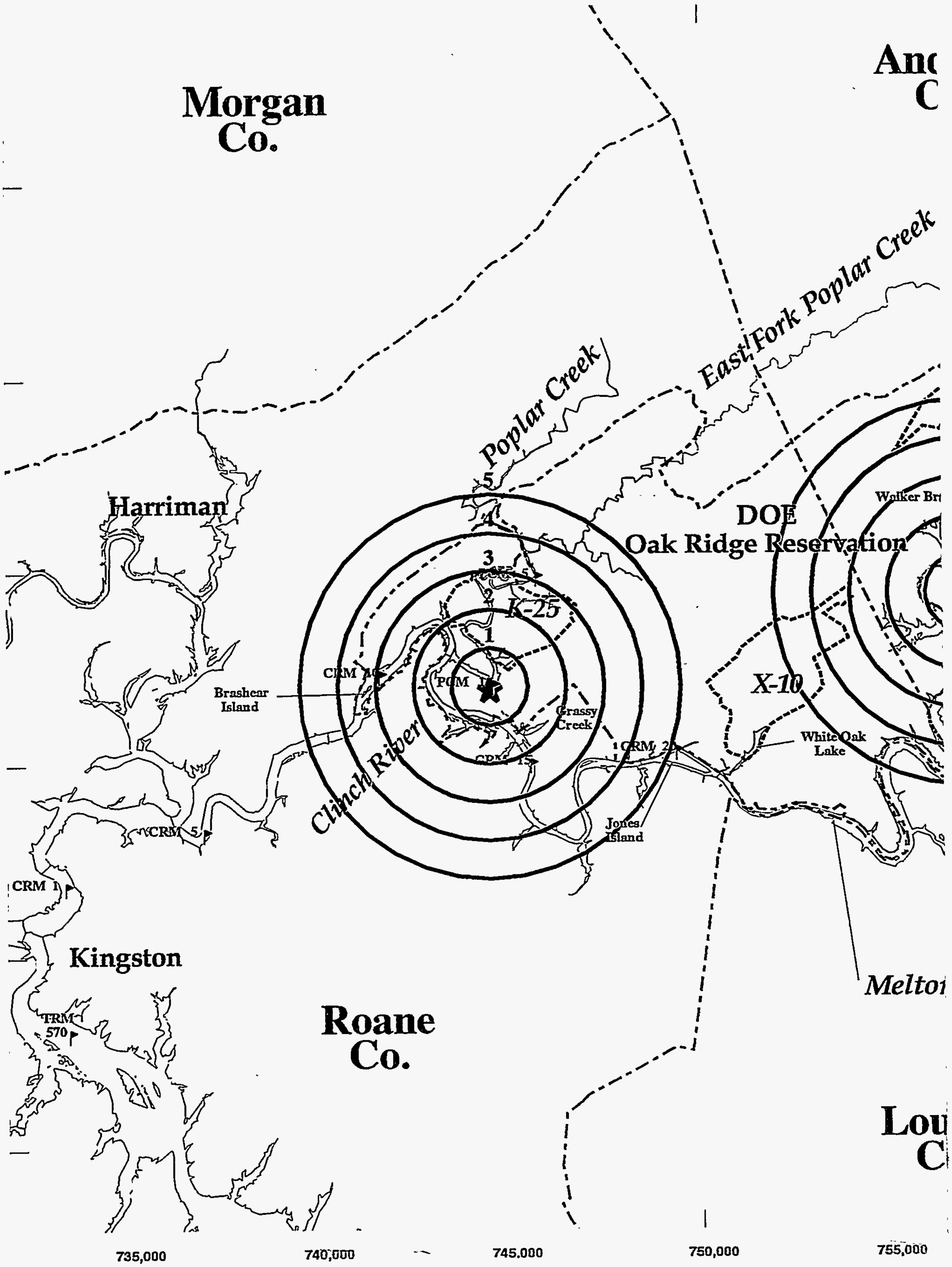
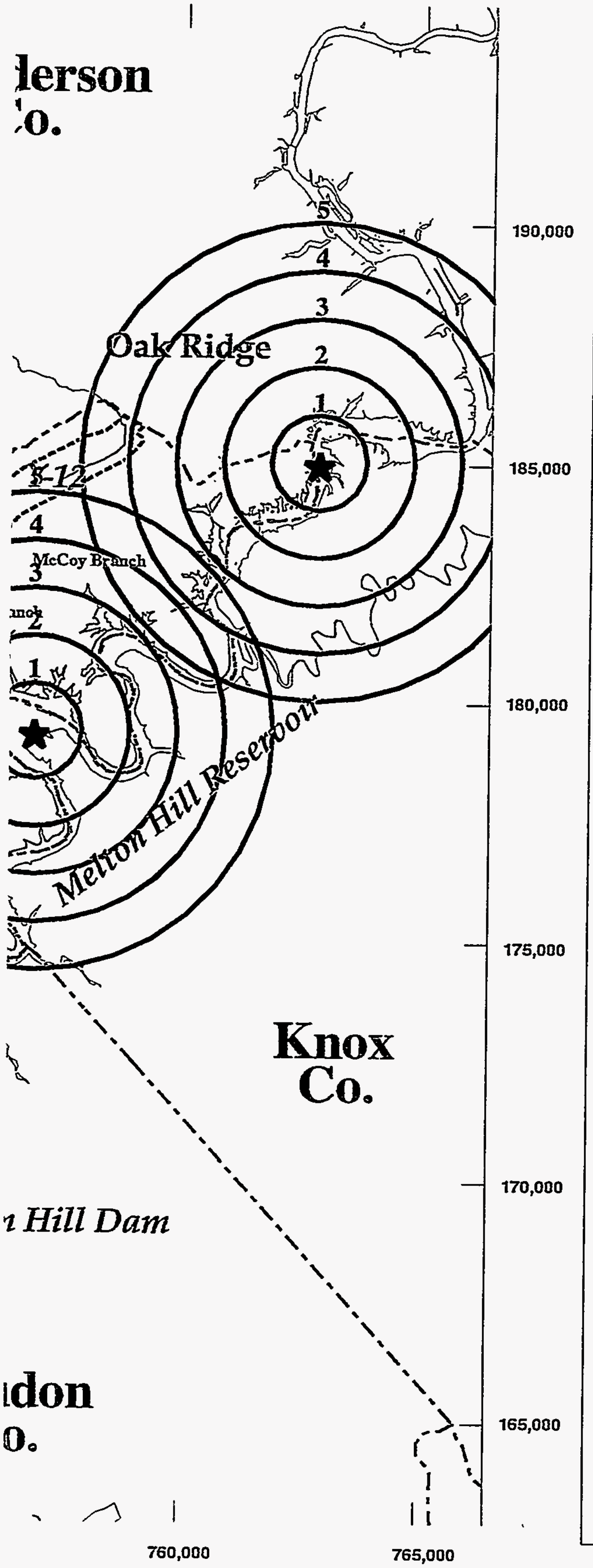
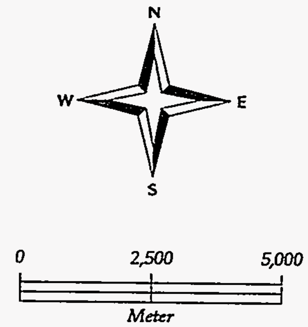


Fig. 6.30. Location of active osprey nesting locations adjacent to the Oak Ridge Reservation. Circles represent 1 km radi



Clinch River Environmental Restoration Program

Osprey Nests and Approximate Ranges



LEGEND

★	Nest Location
	River Mile Marker
	DOE Boundary
	Plant Boundary
	County Boundary

Tennessee State Plane Coordinate System, NAD 83
Source: Lake boundary provided by TVA
Sample locations provided by
Environmental Science Division

Prepared by GeoSpatial Support (GSS)
Enironmental Information Management
Enviromental Restoration Division

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i from nest.

PCBs. The 80th percentile for PCB exposure experienced by osprey exceeded the NOAEL but not the LOAEL at reach 13 and at subreaches 2.04, 3.01, and 3.02; a similar relationship (exposure exceeding the NOAEL but not the LOAEL) was observed for great blue heron at subreaches 2.04 and 3.02. Both the avian NOAEL and LOAEL are based upon a study in which reduced egg hatchability was observed among ring-necked pheasants fed two dose levels, 1.8 and 3.6 mg/kg/d Aroclor 1254 for 17 weeks (Dahlgren et al. 1972). The study was considered to represent a chronic exposure, therefore a subchronic-chronic correction factor was not employed. Effects were observed at both dose levels, therefore the 1.8 mg/kg/d dose level was considered to be a LOAEL. Because an experimental NOAEL was not established, the NOAEL was estimated using LOAEL-NOAEL correction factor of 0.1. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individual birds may become evident cannot be defined.

The 80th percentile for PCB exposure experienced by mink exceeded the NOAEL but not the LOAEL at subreach 2.04. For river otter, while the 80th percentile for PCB exposure exceeded the NOAEL at 14 subreaches, the LOAEL was exceeded only at subreach 2.04 (Table F4.18). The mink and otter NOAELs and LOAELs for PCBs was derived from a study of mink fed Aroclor 1254 for 4.5 mo. (Aulerich and Ringer 1977). While consumption of 0.69 mg/kg/d Aroclor 1254 reduced kit survivorship, no effects were observed at the 0.14 mg/kg/d exposure level. The study was considered to represent a chronic exposure, therefore a subchronic-chronic correction factor was not employed. The 0.14 mg/kg/d exposure was considered to be a chronic NOAEL; the 0.69 mg/kg/d exposure was considered to be a chronic LOAEL Based on the results of Aulerich and Ringer (1977), mink experiencing exposure ≥ LOAEL are likely to display reduced kit survivorship.

Selenium. The 80th percentile for selenium exposure experienced by mink and river otter exceeded the NOAEL but not the LOAEL at 9 (mink) and 13 (otter) subreaches. Both the NOAEL and the LOAEL are based upon a study in which reduced reproductive success and offspring survival was observed among mice fed one dose level, 0.75 mg/kg/d Se in drinking water, for three generations (Schroeder and Mitchner 1971). The study was considered to represent chronic exposure, therefore a subchronic-chronic correction factor was not employed. This dose level was considered to be a LOAEL. Because an experimental NOAEL was not established, the NOAEL was estimated using LOAEL-NOAEL correction factor of 0.1. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individual mink may become evident cannot be defined. Based on the results of Schroeder and Mitchner (1971), mink experiencing exposure ≥ LOAEL are likely to display reduced reproductive success and offspring survival.

6.4.3.2 Mink toxicity tests

To evaluate the nature and magnitude of toxicity of contaminants in fish from the Clinch River to mink, fish were collected from the Poplar Creek embayment, formulated into mink diets, and fed to mink. Mink were fed five different diets:

Diet A: 75% ocean fish, 25% mink chow
Diet B: 75% fish from upstream of ORR, 25% mink chow
Diet C: 25% Poplar Creek fish, 50% ocean fish, 25% mink chow
Diet D: 50% Poplar Creek fish, 25% ocean fish, 25% mink chow
Diet E: 75% Poplar Creek fish, 25% mink chow

Ten mink (2 males, 8 females) were fed each diet for 7 months; starting ~3 months prior to breeding, extending to 6 weeks post-partum. Bioaccumulation, growth, histopathology, and reproduction were recorded. Significant effects were observed only among mink fed diet E. These effects included

statistically significant reductions in body weights of adult females and male kits and in litter size. Percent reductions were 20% and 17% for adult female and male kit weights, respectively, and 37.7% for litter size. A detailed discussion of the methods and results of the mink toxicity test is presented in Sect. 3.5.

To evaluate how the exposures experienced by mink in the toxicity test compare to those modeled for mink along the Clinch River, Monte Carlo simulations of mink exposure were performed using the concentrations of mercury and Aroclor 1260 measured in the five diets (Tables N10 and N12). Parameter values in the exposure model were those used in estimates using Clinch River data (body weight=0.974±0.202 kg; food ingestion rate = 0.137 kg/d). Results of the exposure simulation are presented in Table F4.20. Estimated exposures to mercury and Aroclor 1260 in diet A were below both the NOAEL and LOAEL. For diets B, C, D. and E, mercury exposures exceeded the NOAEL (only marginally for diet B) but not the LOAEL, suggesting that it is unlikely that toxicity observed in diet E is attributable to mercury. In contrast, exposures to Aroclor 1260 in diets B, C, D, and E were greater than both the NOAEL and LOAEL (Table F4.20). These data suggest that impaired reproduction should have been evident in all four diets, not just diet E.

Estimating that toxicity should be observed in four diets but actually observing it only in the highest concentration suggests that the LOAEL for PCBs used in this assessment is too low and is not representative of the toxicity of the PCBs present on the ORR. ORR-specific NOAEL and LOAEL for PCBs (represented by PCB 1260) of 1.7 mg/kg/d and 3 mg/kg/d can be derived from the toxicity test exposure estimate for diets B and E (Table F4.20). While an ORR-specific LOAEL for mercury cannot be derived from these toxicity test results, a site-specific NOAEL may be estimated. The ORR-specific NOAEL for mercury would be 0.022 mg/kg/d (diet D) or 0.033 mg/kg/d (diet E, assuming that mercury is not contributing significantly to toxicity observed from this diet).

The mercury exposure estimate for mink at the Clinch River subreach where the highest exposure estimate was obtained (subreach 3.01; mean = 0.027±0.013 mg/kg/d) is higher than that estimated for test diets A, B, C, and D, but less than that in diet E (Table F4.16). The estimated total PCB exposure at subreach 2.04 (mean = 0.311±0.222 mg/kg/d) is less than that in all test diets except the control diet (diet A; Table F4.20).

Several conclusions may be drawn from these toxicity test data.

1) Comparisons of exposure estimates to NOAELS and LOAELs suggest that effects observed in diet E are attributable to PCBs. It is unlikely that mercury is significantly contributing to adverse effects observed in the toxicity test. A significant contribution of mercury to the toxic effects would have resulted in a LOAEL that was lower than literature values, not higher.

2) Given the difference between predicted and observed toxicity from the test diets, the PCB LOAEL used in this assessment is too low and does not reflect toxicity observed among mink exposed to Poplar Creek fish.

3) Consumption of a diet consisting of 75% fish from the Poplar Creek produces reproductive impairment in mink.

4) Assuming that toxicity in the test is entirely due to PCBs, a LOAEL for mink on the ORR fish of 3 mg/kg/d can be derived. Using the ORR-specific value rather than the literature value, PCBs would not be expected to cause toxic effects on survival, growth, or reproduction of mink in any ORR watershed.

Differences between the results of the toxicity tests and modeled exposures for mink on the ORR may result for several reasons.

1) Differences in fish size. Exposure estimates for mink on the ORR were based solely on contaminant concentrations in fish most likely to be consumed by mink (i.e., ≤30 cm in length). Due to the large volume of fish needed to formulate the test diets and to feed mink for 7 months, the majority of fish used in the toxicity test were large (mean=39 cm, STD=17 cm). Because body burdens of bioaccumulative contaminants like mercury and PCBs are generally greater in older, larger individuals, concentrations in the toxicity test diets were higher than that in fish expected to be consumed by mink on the ORR.

2) Differences in fish species. Over 50% of the fish used in the test diets were sucker, carp, or buffalo (Table C20). In contrast, only 0.1% of fish used to estimate exposure for Clinch River piscivores were carp; no data from sucker and buffalo were used in the exposure assessment. Because fish species accumulate contaminants differently, variation in species included in test diets and modeled diets may have contributed to the differences in results.

3) Differences in the PCB congener composition on the ORR vs. that used in the literature toxicity test. PCBs measured in environmental samples are not Aroclors. Aroclors are specific mixtures of PCB congeners as manufactured. The environmental measurements of PCBs used in the Poplar Creek toxicity test are called Aroclor 1254 or Aroclor 1260 because they have ~54% or 60% Cl. The congener makeup of Aroclor 1254 or 1260 from the Poplar Creek fish is likely to be very different from the congener makeup of Aroclor 1254 or 1260. More importantly, PCB toxicity is generally correlated with individual congeners, not with Aroclors.

6.4.3.3 Biomonitoring data

Great blue heron reproduction study. To determine if contaminants from the ORR are adversely affecting piscivorous wildlife, bioaccumulation of contaminants and reproductive success of great blue herons at two colonies located within 3 km of the ORR and two colonies located >10 km from the site was monitored. Data were collected from each nest colony between 1992 and 1994. A discussion of these data are presented in Sect. 3.5.

Analyses indicated significantly elevated levels of Cr, Hg, and PCBs in eggs (Tables N2 and N14), Hg in feathers and liver of chicks (Table N3) and PCBs in fat (Table N5), liver (Table N6), and muscle (Table N7) of chicks from samples from the ORR as compared to data from the offsite locations. King et al. (1991) report that 0.5 to 1.5 mg/kg mercury concentrations in bird eggs may are associated with reproductive failure; Harris et al. (1993) report a NOAEL for hatching success of Forster's Tern eggs to be 7 mg/kg. Mean concentrations of mercury (0.17 mg/kg) and PCBs (1.68 mg/kg) in great blue heron eggs from within 3 km of the ORR are substantially below both levels, suggesting that reproductive effects from mercury or PCBs in eggs are unlikely.

Despite elevated contaminant burdens, the mean number of eggs/nest, number of chicks/nest, egg weight, and eggshell thickness did not differ between colonies within 3 km of the ORR and those >10 km away (Table N8). In addition, the number of eggs/nest observed at the colonies within 3 km of the ORR (3.5 eggs/nest) and at the colonies >10 km away (3.2 eggs/nest) are comparable to those reported in EPA (1993g) (3.16 to 4.37 eggs/nest)

The results of the great blue heron reproduction survey indicate that herons are experiencing higher contaminant exposures at the colonies adjacent to the ORR. However, this exposure is not sufficiently high to result in adverse effects to the populations at the colonies within 3 km of the reservation.

Osprey reproduction. While an osprey monitoring study was not performed as part of the CRRI, an ongoing osprey reintroduction program is being conducted by the TWRA in the Clinch/Tennessee River system. Osprey are currently nesting at three locations adjacent to the reservation: Gallaher and Solway bends (both located in reach 1) and in Poplar Creek approximately at the boundary of subreaches 3.03 and 3.04. Mean reproductive success at these three osprey nests was 3 young/nest (B. Anderson, pers. comm). For comparison, mean reproductive success of osprey in N. America ranges from 1.7 to 2.14 young/nest (EPA 1993a).

6.4.3.4 Weight of evidence

Osprey. Two lines of evidence, literature toxicity data and biomonitoring data, were available to evaluate ecological risk to osprey. Comparison of exposure estimates to LOAELs indicates that within only two subreaches (Poplar Creek 3.01 and 3.02) are significant risks present. These risks are attributable solely to mercury. Risk from mercury is not retained when exposure is recalculated taking into account the spatial component of osprey foraging behavior. Reproductive success of osprey adjacent to the ORR is high relative to that observed among other osprey populations in N. America. The weight of evidence suggests that contaminants from the ORR do not present a risk to osprey.

Great blue heron. Two lines of evidence, literature toxicity data and biomonitoring data, were available to evaluate ecological risk to great blue heron. Comparison of exposure estimates to LOAELs indicates that exposure to Hg within the subreaches 3.01 and 3.02 may be sufficiently high to result in impaired reproduction of individual herons that forage exclusively within these areas. However, because these subreaches account for only 6% of the heron population downstream of the ORR, adverse population-level effects are unlikely. Biomonitoring data indicate that PCBs and mercury are being accumulated in heron eggs and chicks, but levels are below those reported to adversely affect other bird species. Because heron reproduction adjacent to the ORR does not appear to be impaired, the weight of evidence suggests that contaminants from the ORR do not present a risk to great blue heron.

Mink. Two lines of evidence, literature toxicity data and toxicity test data, were available to evaluate ecological risk to mink. Comparison of exposure estimates to LOAELs indicates no significant risk in any subreach. Toxicity test results indicate that consumption of a diet consisting primarily of fish from the Poplar Creek embayment adversely affects reproduction. The maximum exposure estimated for mink along the Clinch River however is significantly lower than the toxicity test exposure level that produce effects. The weight of evidence suggests that contaminants from the ORR do not present a risk to mink.

River Otter. Two lines of evidence, literature toxicity data and the PCB NOAEL and LOAEL and mercury NOAEL derived from the Poplar Creek mink toxicity test, were available to evaluate potential risk to river otter. Comparison of exposure estimates to literature-derived LOAELs indicates a significant risk from mercury in subreach 3.01 and from PCBs in subreach 2.04.

Using approach outlined in Sect.6.4.2.1. and the ORR-specific PCB NOAEL and LOAEL and mercury NOAEL for mink (Sect. 6.4.3.2), ORR-specific values for otter were estimated (Table 6.20).

Table 6.20. Oak Ridge Reservation–specific values for otter

Analyte	Estimated NOAEL (mg/kg/d)	Estimated LOAEL (mg/kg/d)
PCBs	0.85	1.5
Mercury	0.011[a]	no data

NOAEL = no-observed-adverse-effect level; LOAEL = lowest-observed-advserse-effect level.
[a]To be conservative, based on the lower of two possible NOAELs.

Comparison of the ORR-specific PCB LOAEL to the exposure distributions presented in Table F4.13 indicate that there is a <1% likelihood of individual otter in subreach 2.04 experiencing PCB exposure greater than the ORR-specific LOAEL. Therefore, based upon the results of the Poplar Creek mink toxicity test, PCBs are unlikely to present a significant risk to otter in the CR/PC system.

The ORR-specific mercury NOAEL, while greater than the literature-derived NOAEL (0.008 mg/kg/d; Table F4.15) is still less than the literature-derived LOAEL (0.06 mg/kg/d; Table F4.15). Therefore, the results of the Poplar Creek mink toxicity test do not significantly alter the conclusions derived from evaluation of the literature-based toxicity data. The weight-of-evidence indicates that individual otter using subreach 3.01 may be adversely affected by mercury exposure. Because the river otter is a state threatened species, effects to any individual is significant.

6.4.4 Uncertainties Concerning Risks to Piscivorous Wildlife

6.4.4.1 Estimation of contaminant concentrations for missing fish size classes

Contaminant concentrations in fish size classes for which no data were available were assumed to be comparable to that observed larger fish. Because larger fish generally have higher contaminant concentrations than smaller fish, concentrations in smaller fish are likely to be overestimated. Consequently, exposure estimates that use these estimated fish values are likely to overestimate the actual contaminant exposure experienced.

6.4.4.2 Bioavailability of contaminants

It was assumed that 100% if the contaminant concentration reported in fish and water was bioavailable. Much of the biotic media are bioavailable; however, the uptake efficiencies for wildlife may not be 100%. Therefore, exposure estimates based upon the contaminant concentrations in media are conservative and are likely to overestimate the actual contaminant exposure experienced.

6.4.4.3 Extrapolation from published toxicity data

While published toxicity studies are available for mink, there are no published data for osprey or great blue heron. To estimate toxicity of contaminants at the site, it was necessary to extrapolate from studies performed on test species (i.e., mallard ducks, ring-necked pheasant, and rats). While it was assumed that toxicity could be estimated as a function of body size, the accuracy of the estimate is not known. For example, osprey or herons may be more or less sensitive to contaminants than ducks or pheasants.

Additional extrapolation uncertainty exists for those contaminants for which data consisted of either LOAELs or was subchronic in duration. For either case, an uncertainty factor of 10 was employed to estimate NOAELs or chronic data. The uncertainty factor of 10 may either over- or underestimate the actual LOAEL-NOAEL or subchronic-chronic relationship.

Toxicity of PCBs to piscivorous wildlife was evaluated using toxicity data from studies on Aroclor 1254. Because toxicity of PCB congeners can vary dramatically, the applicability of data for Aroclor 1254 is unknown. Comparison of the results of the mink toxicity test results and the estimated LOAELs for mink, suggests the Aroclor 1254 data do not accurately reflect (i.e., overestimate) the toxicity of the PCB mixture present in Clinch River fish.

6.4.4.4 Variable food and water consumption

While food consumption by piscivorous wildlife was assumed to be similar to that reported for the same or related species in other locations, the validity of this assumption cannot be determined. Food consumption by wildlife along the Clinch River may be greater or less than that reported in the literature, resulting in either an increase or decrease in contaminant exposure. Similarly, water consumption for all species was estimated according to the allometric equations of Calder and Braun (1983). The accuracy with which the estimated water consumption represents actual water consumption is unknown.

6.4.4.5 Single contaminant tests vs exposure to multiple contaminants in the field

While piscivores along the Clinch River are exposed to multiple contaminants concurrently, published toxicological values only consider effects experienced by exposures to single contaminants. Because some contaminants to which wildlife are exposed can interact antagonistically, single contaminant studies may overestimate their toxic potential. Similarly, for those contaminants that interact synergistically, single contaminant studies may underestimate their toxic potential.

6.4.4.6 Inorganic constituents or species present in the environment

Toxicity of metal species varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. For example, Arsenic (III) and methyl mercury are more toxic than arsenic (V) and inorganic mercury, respectively. The available data on the contaminant concentrations in media do not report which species or form of contaminant was observed. Because benchmarks used for comparison represented the more toxic species/forms of the metals (particularly for arsenic and mercury), if the less toxic species/form of the metal was actually present in fish from the Clinch River or Poplar Creek, potential toxicity at the sites may be overestimated.

6.4.4.7 Contaminant concentrations in aquatic prey

While fish are the primary prey of piscivores, other aquatic prey are also consumed. It was assumed that the contaminant concentration in fish was representative of that in other aquatic prey. Due to the different life histories of other aquatic prey (i.e., amphibians, crayfish, benthic invertebrates), their contaminant burdens are likely to differ from that in fish. Therefore, assuming comparability to fish may either over or underestimate exposure.

6.4.4.8 Fish size selection

Data concerning the sizes of fish consumed by piscivores were obtained from the literature. Because fish sizes consumed by Clinch River piscivores may differ from that reported in the literature, exposure may be overestimated or underestimated. In addition, while Alexander (1977) report that mink only consume fish ≤20 cm in length, few data from the Clinch River were available in this size range. To provide data with which to assess exposure, the size range of fish consumed by mink was expanded to include fish up to 30 cm in length. Including these larger fish may overestimate the exposure experienced by mink along the Clinch River.

6.4.4.9 Monte Carlo simulation

To perform Monte Carlo simulations, distributions must be assigned to parameters. Because wildlife are mobile, the mean of the contaminant concentration is likely to best represent their exposure. For this reason the contaminant concentrations in fish were assumed to be normally distributed.

The literature values used for body weights of each end point are nationwide values which may overestimate or underestimate the body weight of species found at the site. Similarly the proportion of fish and aquatic prey in mink diet were derived from data from northern locations (i.e., MI, Canada, etc.). The applicability of these data to the percentage of fish and aquatic prey consumed by mink in Tennessee is unknown.

6.4.4.10 Estimated whole fish concentrations

Contaminant concentrations in whole no-shad fish were estimated using contaminant specific fillet to whole fish ratios. Data to generate ratios were only available for PCBs in largemouth bass and channel catfish from the Clinch River. Ratios for metals were obtained from spotted bass samples from near the PORTS facility in Ohio. Applicability of these ratios to species other than those from which they were developed is unknown. Similarly, applicability of metal ratios from Ohio spotted bass to Clinch River fish is unknown.

6.5 RISKS TO INSECTIVOROUS WILDLIFE

6.5.1 Exposure Assessment for Insectivorous Wildlife

Insectivorous wildlife may be exposed to contaminants through ingestion of contaminated media (aquatic insects and water) associated with the Clinch River. Rough-winged swallows (*Stelgidopteryx serripennis*) and little brown bats (*Myotis lucifugus*) forage regularly within the Clinch River system. During the breeding season, from April to July, the swallows reside in burrows located in the high vertical banks. Although no burrows were located within Poplar Creek, swallows may also reside in crevices in rock cliffs, bridges, and tunnels. The swallows continue to forage in the area until they migrate south in the fall. The little brown bats are nocturnal foragers along the waterways; living in caves, trees, and buildings during the day (Harvey 1992). Little brown bats and gray bats hibernate throughout the winter. Colonies of the federally endangered gray bat (*Myotis grisescens*) have been found within the Tennessee river basin, specifically at Norris and Chickamauga lakes. Although no colonies have been found in Roane or Anderson counties, gray bats have been observed foraging in these areas. The exposure parameters for both little brown bats and the gray bats are similar. Therefore, this assessment will address both species of bats simultaneously. The following sections describe the modes of exposure that occur at the Clinch River system for these end points, the ways in which exposure was estimated, and the exposure data available for ecological risk assessment. Exposure estimates were calculated for all contaminants in all CR/PC reaches for which data were available (reaches 0, 1, 5, 6, 10, 13, 15, and 18 and subreaches 2.01, 2.02, 2.04, 3.01, 3.02, 3.03, 3.04, 4.01, and 4.04).

6.5.1.1 Exposure through oral ingestion of aquatic insects and water

Potential routes of exposure for rough-winged swallows, little brown bats, and gray bats include ingestion of contaminated food and surface water associated with the CR/PC. All three of the end point species have diets consisting almost exclusively of aerial insects which are abundant on site. The total exposure experienced by insectivorous wildlife is represented by the sum of the exposures from each individual source (e.g., food and water).

The generalized wildlife exposure model, as described in Sect. 6.4.1.1, was used to calculate contaminant exposure to insectivorous wildlife. In general, wildlife may be exposed to contamination via three pathways: oral, dermal, or inhalation. The primary route of exposure for these end points is through oral exposure. The feathers and fur of swallows and bats reduce the likelihood of significant dermal exposure by limiting the contact of skin with contaminated media. Therefore, dermal exposure is assumed to be negligible. Inhalation of contaminants is also assumed to be negligible since volatile organic carbons are present in extremely low amounts in the surface water. Thus, the exposure model for Clinch River insectivorous wildlife is as follows:

$$E_j = \left(\frac{IR_w \times C_{w\text{-}j}}{BW}\right) + \left(\frac{(C_{i\text{-}j})(P_i) \times IR_i}{BW}\right) \tag{6.9}$$

where

E_j = exposure to contaminant (j) (mg/kg/d)
IR_w = ingestion rate of water (L/d)
$C_{w\text{-}j}$ = contaminant concentration (j) in water (mg/L)
BW = body weight (kg)
$C_{i\text{-}j}$ = contaminant concentration (j) in aquatic insects (mg/kg)
P_i = proportion of aquatic insects (mayflies) in diet
IR_i = ingestion rate of insects (kg/d)

Exposure estimates were calculated for all contaminants in all Clinch River subreaches for which data were available. Because wildlife are mobile, their exposure is best represented by the mean contaminant concentration in the media. The UCL_{95} is a conservative estimate of exposure for wildlife. The PLE was used to calculate the UCL_{95} in order to prevent bias from values below the detection limit. Exposure estimates for contaminants that may potentially present a risk to insectivorous wildlife (based upon comparisons to NOAELs) were recalculated using Monte Carlo techniques.

Life history parameters for end point species. Species-specific parameters necessary to estimate exposure using the equation described above are listed in Tables F5.1 and F5.2.

Contaminant concentrations in abiotic and biotic media. Contaminant concentrations in surface water, sediment, and food are needed to estimate exposure for insectivorous wildlife. The PLE UCL_{95} surface water concentrations within each subreach were used to estimate exposure from drinking water. The only surface water contaminants that were used in the exposure model were those present in the mayflies. The contaminants in surface water which were not detected in mayflies were compared to the NOAELs for surface water consumption for swallows and bats. If the concentration is below the water consumption benchmark, the contaminant need not be considered further.

Concentrations of contaminants in food (mayflies) were measured at three locations in Clinch River subreaches 3.03 and 4.04 and reach 18 (Table F5.3). Contaminant concentrations in mayflies at other locations were estimated using sediment-mayfly contaminant uptake factors developed for the Clinch River System (Table F5.4). Contaminant uptake factors were calculated for arsenic, lead, mercury, zinc, and PCBs by dividing the contaminant concentration found in mayflies by the concentration found in the sediment closest to the point of mayfly collection. The average uptake factor from the three locations for each contaminant was used to estimate the mayfly contaminant concentrations using the sediment data at all other reaches. However, the mercury uptake factor was calculated using sediment and mayfly concentrations collected within subreaches 3.03 and 3.04 only. Data from reach 18 was excluded due to potential differences in mercury speciation off the ORR. The estimated and measured mayfly contaminant concentrations, and surface water concentrations for each subreach were used in the insectivorous exposure model to estimate exposure for swallows and bats.

The resultant estimated mayfly concentrations were compared to the measured concentrations (at three locations) by performing a Student's t-test. This comparison was performed to determine the quality of the uptake factors. There were no significant differences between the estimated and the actual mayfly contaminant concentrations (Table F5.5). Therefore, this model appears to be a valid predictor of mayfly concentrations for each subreach.

Exposure modeling using point-estimates. To estimate contaminant exposure experienced by rough-winged swallows, the following assumptions were made:

1) body weight = 0.0159 kg
2) food consumption = 0.012 kg/d
3) water consumption = 0.0037 L/d
4) soil consumption = 0 kg/d
5) diet consists 37.93% aquatic insects (32.89% dipterans and 5.04% ephemeropterans: represented by mayflies) (Beal 1918)
6) individuals forage exclusively within each given subreach

To estimate contaminant exposure experienced by little brown and gray bats, the following assumptions were made:

1) body weight = 0.0075 kg
2) food consumption = 0.0025 kg/d
3) water consumption = 0.0012 L/d
4) soil consumption = 0 kg/d
5) diet consists 100% aquatic insects (represented by mayflies)
6) individuals forage exclusively within each given subreach

Using the Clinch River insectivorous exposure model and the assumptions and data described above, exposure to contaminants was estimated for rough-winged swallows and little brown and gray bats foraging in the Clinch River. Exposure estimates for analytes detected in mayfly samples (arsenic, mercury, lead, zinc, and PCBs) are presented in Tables F5.6 through F5.15.

Exposure modeling using Monte Carlo simulations. Employing point estimates for the input parameters in the Clinch River insectivore exposure model does not take into account the variation and uncertainty associated with the parameters and may over or under estimate the contaminant exposure that end points may receive in any given reach. In addition, calculating the model using point estimates produced a point estimate of exposure. This estimate provides no information concerning the distribution of exposures or the likelihood that individuals within a reach will actually experience potentially hazardous exposures. To incorporate the variation in exposure parameters and to provide a better estimate of the potential exposure experienced by insectivores in each subreach, the exposure distribution was simulated using Monte Carlo techniques. For a detailed discussion of Monte Carlo simulation (Sect. 6.4).

Monte Carlo simulations of exposure estimates were performed for all end point-subreach-contaminant combinations where comparison of point exposure estimates to NOAELs produced HQs ≥ 1 (NOAELs are presented in Sect 6.5.2.1; Screening of exposure estimates against NOAELs is presented in Sect. 6.5.3.1). Monte Carlo simulations were performed for the subreach-contaminant combinations presented in Table F5.16.

Distributions were used for the following parameters in the Clinch River insectivore exposure model: 1) end point body weight; 2) sediment-mayfly uptake factors (Table F5.4); and 3) contaminant concentrations in sediment and mayflies (Table F5.17) and water (Table F5.18). All distributions were assumed to be normal, except for the body weight distribution of the little brown bat which was assigned a triangular distribution and the log normal sediment distribution. Because wildlife are mobile and are more likely to be exposed to average concentrations and not to the extreme values, the standard error of the mean was used to describe variation for contaminant concentrations. All other distributions employed the standard deviation. The food and water consumption rates used in the exposure model were average values.

Rough-winged swallows also feed on a variety of other types of insects (i.e., Hymenoptera, Coleoptera, Hemiptera) which do not come in contact with sediments. Therefore, the percentage of aquatic insects (dipterans 32.89% and ephemeropterans 5.04%; Beal, 1918) in the diet of rough-winged swallows was assumed to be 37.93%.

The little brown and gray bat diet was assumed to consist of 100% aquatic insects. Belwood and Fenton (1976) found that only 80.4% of the diet consisted of chironimids (39.5%), tricopterans (31.5%), and miscellaneous insects (9.4%); while the remaining portions of the diet were lepidopterans (11.0%), coleopterans (5.5%), and neuropterans (3.1%). Additionally, Rabinowitz (1978) found that mayflies are the choice food item of gray bats within eastern Tennessee. Since the bat's diet is highly variable and opportunistic (Fenton and Barclay 1980), exposure was conservatively estimated using contaminant concentrations from a 100% aquatic insect diet. This assessment assumed that contaminant concentrations found in mayflies are representative of aquatic insects.

The initial comparison of the exposure distributions with the estimated LOAELs was conducted under the assumption that the population would be feeding exclusively from a single subreach. This assumption is appropriate for rough-winged swallows since their feeding territory does not extend over 1/2 mile from the nesting site (Stoner and Stoner 1941). Therefore, contaminant exposure within a subreach could potentially impact a colony if present within that subreach. In contrast, little brown and gray bats have a larger home range and may forage within multiple subreaches. LaVal et al. (1977) documented that gray bats were observed foraging 8.5 km upstream or downstream of their roost. Therefore, if the contaminant exposure from foraging in a single subreach exceeded the LOAEL, the exposure distribution would be reassessed for bats foraging in adjacent subreaches (e.g., 3.01 and 3.02) or within the entire reach.

Table 6.21. Insectivorous body weight distributions

End point	Distribution	Mean (kg)	Standard deviation	Range	Source
Rough-winged swallow	normal	0.0159	0.00058		Dunning 1984
Little brown bat	triangular	0.0075		0.007-0.009	Gould 1955 Burt and Grossenheider 1976

Sources:

Burt, W. H., and R .P. Grossenheider. 1976. *A Field Guide to the Mammals of America North of Mexico.* 3rd ed. Houghton Mifflin Co., Boston.

Dunning, J. B. 1984. *Body Weights of 686 Species of North American Birds.* Western Bird Banding Association Monograph No. 1. Eldon Publishing Co., Cave Creek, Ariz.

Gould, Ed. 1955. "The Feeding Efficiency of Insectivorous Bats." *J. Mammal.* 36:399–407.

Monte Carlo Simulation could not be performed for exposure to mercury (both rough-winged swallows and bats) and PCBs (bats only) at reach 22 (White Oak Creek Embayment). Only one sediment sample was collected from each subreach, therefore, an appropriate contaminant distribution found in mayflies could not be estimated. Consequently, the exposure estimate for each end point for Hg and PCBs within each subreach was calculated as a point exposure estimate and compared directly to a LOAEL to obtain a HQ.

6.5.2 Effects Assessment for Insectivorous Wildlife

Effects assessment involves the identification of known effects of contaminants on swallows and bats using ecotoxicological benchmarks. The types, development, and interpretation of appropriate toxicological benchmarks are discussed and available toxicity data relevant to swallows and bats are summarized in toxicity profiles for COPEC. Conventional toxicity data consists of published values for toxicity of contaminants to test species. These data are used in the development of toxicological benchmarks which are used to determine if biological effects are likely. Contaminant exposures are compared to benchmarks to obtain HQs used in the risk characterization.

6.5.2.1 Ecotoxicological benchmarks

To determine if the contaminant exposures experienced by swallows and bats that use the Clinch River or Poplar Creek could produce adverse effects, exposure estimates are compared to NOAELs and LOAELs derived according to the methods outlined by Opresko et al. (1994) and EPA (1993i). NOAELs represent the highest exposure at which no adverse effects were observed among the animals tested. LOAELs represent the lowest exposure at which significant adverse effects are observed.

Toxicological studies of the effects of contaminants observed within the Clinch River were obtained from the open literature. Only studies of long-term, chronic oral exposures were used to estimate the NOAEL. To make the NOAELs relevant to possible population effects, preference was given to studies that evaluated effects on reproductive parameters. In the absence of a reproduction end point, studies that considered effects on growth, survival, and longevity were used. The NOAELs and LOAELs for rough-winged swallows and little brown bats and the studies which derived them are located in Tables F5.19

and F5.20. Specific details on development of the NOAELs for all wildlife end points are discussed in Sect. 6.4.2.1.

6.5.2.2 Ecotoxicological profiles

The mercury and PCB toxicity profiles can be found in Appendix F1.

6.5.3 Risk Characterization for Insectivorous Wildlife

Risk characterization integrates the results of the exposure assessment (Sect. 6.5.1) and effects assessment (Sect. 6.5.2) to estimate risks (the likelihood of effects given the exposure). Procedurally, the risk characterization in this assessment is performed for each assessment end point by (1) screening all measured contaminants against toxicological benchmarks, (2) estimating the effects of the contaminants retained by the screening analysis, (3) listing and discussing the uncertainties in the assessment. There is only one line of evidence, single chemical toxicity data, available to evaluate the risks to rough-winged swallows and bats foraging at the Clinch River.

6.5.3.1 Chemical screening for insectivorous wildlife

Two types of single chemical toxicity data are available with which to evaluate insectivore contaminant exposure: NOAELs and LOAELs. The total contaminant exposure estimates (Tables F5.6–F5.15) for rough-winged swallows and little brown/gray bats foraging within each subreach of the Clinch River were compared to estimated NOAELs to determine if adverse effects are possible. The comparison of estimated exposure and the NOAEL acts as a screening tool to identify contaminants of concern which will be further evaluated for possible estimated effects (i.e., exposure modeling using Monte Carlo simulation). LOAELs are compared to the exposure distribution generated by the Monte Carlo simulation. If the LOAEL is lower than the 80th percentile of the exposure distribution, >20% of the end point population is experiencing contaminant exposures that are likely to produce adverse effects. Consequently, population-level effects to swallows and bats are likely.

Screening point estimates of exposure. To determine if the contaminant exposures experienced by rough-winged swallows and little brown/gray bats feeding within the Clinch River are potentially hazardous, the total exposure estimates were compared to estimated NOAELs. HQs were calculated to quantify the magnitude of the hazard where: HQ = estimated exposure/NOAEL. HQs >1 indicate that individuals may be experiencing exposures that are in excess of NOAELs, and may suggest that adverse effects may be occurring. HQs for rough-winged swallows and bats are presented along with exposure estimates in Tables F5.6–F5.15.

The spatial distribution of contamination and potential risks to rough-winged swallows and little brown/gray bats for each Clinch River subreach are illustrated in Figs. 6.31 and 6.32. These figures display the sum of the NOAEL-based HQs (e.g., sum of toxic units or ΣTUs) for five contaminants: arsenic, mercury, lead, zinc and Aroclor 1254. The importance of each contaminant, relative to an end point, was determined based on the magnitude of the HQs. River reaches were arranged from upstream to downstream locations.

The magnitude of the ΣTUs was much greater (at least 3 times) for rough-winged swallows compared to little brown and gray bats. The maximum ΣTUs for both end points was observed at reach 22 (White Oak Creek Embayment) followed by subreaches 3.01, 3.02, and 3.04 in Poplar Creek (below Mitchell Branch, East Fork Poplar Creek, and Poplar Creek Mouth). However, the magnitude of risk to bats is minimal (HQ<1) at Poplar Creek 3.01 and 3.02. The total risk at subreaches 3.01, 3.02, and 3.04

and at reach 22 is attributable primarily to mercury. Additionally, zinc contributes to the total risk to swallows, while PCBs contribute slightly to risk to little brown and gray bats.

A summary of the reaches where HQs > 1 were observed is listed for each contaminant in Table 6.22. HQs > 1 were not observed for any contaminant or insectivorous end point in Clinch River reaches 0, 1, 8, 13, 18, and 5, or subreaches 7.01, 7.02, 2.01, 2.02, 2.03, 2.04, 3.03, 4.01, 4.02, 4.03, or 4.04.

Table 6.22. Reaches where hazard quotients greater than one were observed

Subreach or reach	Name	Contaminants with a hazard quotient >1	
		Rough-winged swallow	Little brown or gray bats
3.01	Poplar Creek: East Fork Poplar Creek	Mercury	
3.02	Poplar Creek: Mitchell Branch	Mercury	
3.04	Poplar Creek Mouth	Mercury	Mercury
22	White Oak Creek Embayment	Mercury	Mercury, PCBs

Screening of water data. More chemicals were detected in Clinch River surface water than in mayflies. To evaluate the risk that these contaminants may present, HQs were calculated for these contaminants using the UCL_{95} water concentration and the water consumption benchmarks from Table F5.21. HQs for these contaminants are presented in Table F5.22. No contaminant produced a HQ > 1 for any end point in any Clinch River subreach.

Screening Monte Carlo simulation estimates of exposure. To incorporate the variation present in the parameters employed in the Clinch River insectivore exposure model, Monte Carlo simulations were performed for all end point-subreach-contaminant combinations where HQs > 1 were observed. The mean, standard deviation, and 80th percentile of the simulated exposures are presented in Table 6.23. Graphical representations of the exposure distributions are presented in Figs 6.33(a-d) and 6.34(a-d). By superimposing NOAEL and LOAEL values on these distributions, the proportion of the population experiencing potentially hazardous exposures, can be identified and the magnitude of risk may be determined. Interpretation of the comparison of exposure distributions to NOAELs and LOAELs is described in Table 6.24.

The interpretations described above apply to exposure estimates for populations within a given subreach. In order to determine population-level effects for each end point and adequately evaluate risk to the population, home range must be considered. The feeding territory of the rough-winged swallow does not extend over 1/2 mile from the nesting site (Stoner and Stoner 1941). Therefore, it was assumed that there is a colony, equivalent to a distinct population, of rough-winged swallows within each subreach. Consequently, the exposure and effects experienced by individuals may be assumed to be representative of that experienced by the population. In contrast, distinct populations of little brown and gray bats are not likely to be found within single subreaches. Bats have a larger home range and may forage up to 8.5 km upstream or downstream of their caves (LaVal et al. 1977). Therefore, the

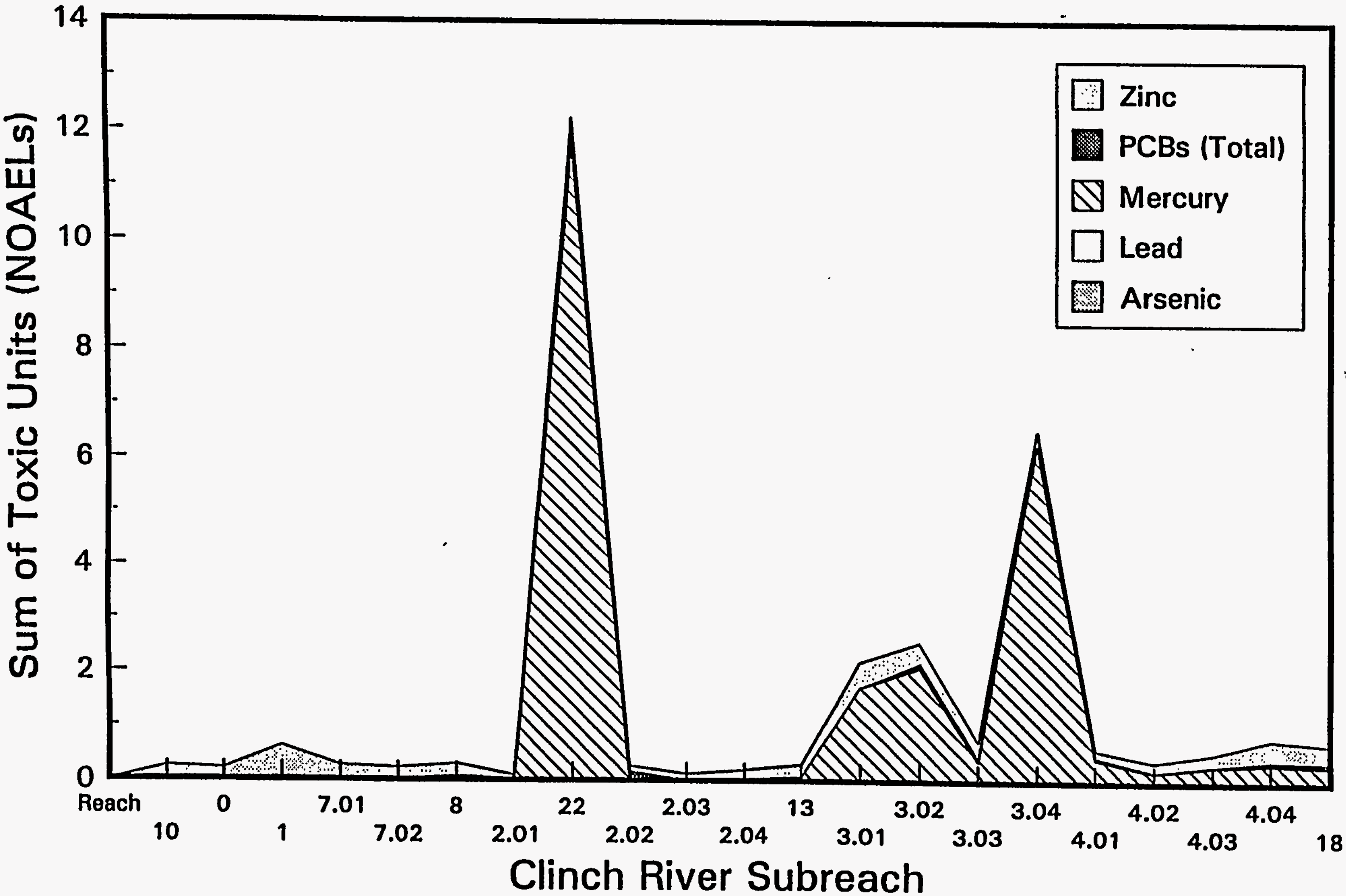

Fig. 6.31. Sum of no-observed-adverse-effect level–based toxic units for rough-winged swallows.

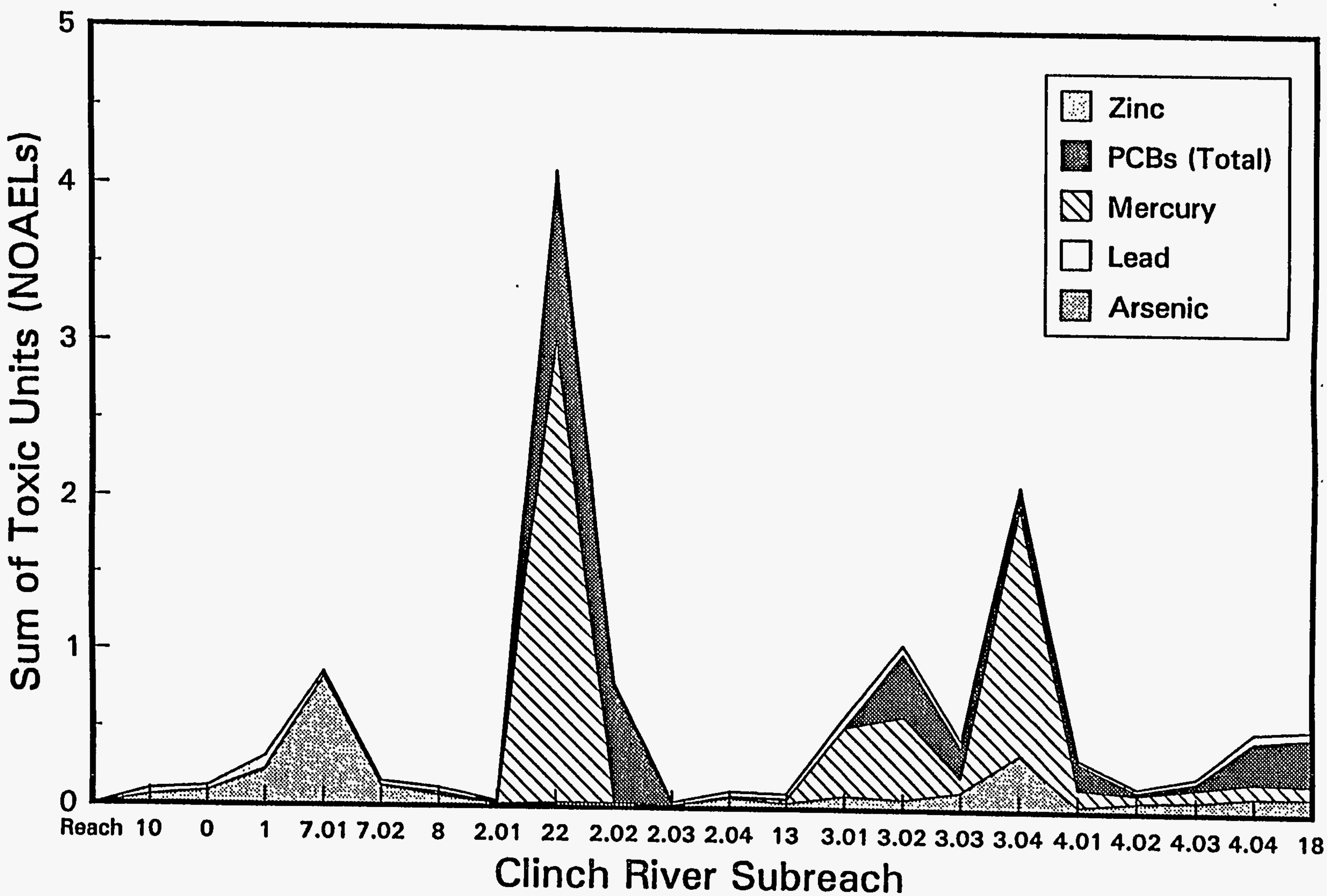

Fig. 6.32. Sum of no-observed-adverse-effects level–based toxic units for little brown bats and gray bats.

Table 6.23. Results of Monte Carlo simulation of insectivore contaminant exposure estimates in the Clinch River

Analyte	Subreach	End point	Simulation results (mg/kg/d)		
			Mean	Standard deviation	80th percentile
Mercury	3.01	Rough-winged swallow	0.314	2.409	0.418
Mercury	3.02	Rough-winged swallow	0.048	0.345	0.058
Mercury	3.03	Rough-winged swallow	0.009	0.064	0.011
Mercury	3.04	Rough-winged swallow	0.100	0.706	0.121
Mercury	3.01	Little brown/gray bat	0.308	0.073	0.368
Mercury	3.02	Little brown/gray bat	0.045	0.003	0.048
Mercury	3.03	Little brown/gray bat	0.009	0.0006	0.009
Mercury	3.04	Little brown/gray bat	0.094	0.007	0.099

Mercury lowest-observed-adverse-effect level for rough-winged swallows = 0.25 mg/kg/d
Mercury lowest-observed-adverse-effect level for little brown/gray bats = 0.57 mg/kg/d

Table 6.24. Comparison of exposure distributions to NOAELs and LOAELs

Comparison	Meaning	Risk-based interpretation
NOAEL>80th percentile of exposure distribution	<20% of population is experiencing exposures greater than NOAEL	Individual- and population-level adverse effects are highly unlikely
NOAEL < 80th percentile < LOAEL	>20% of population is experiencing exposures greater than NOAEL, but <20% is experiencing exposures greater than LOAEL	While population-level effects are unlikely, individuals exposed to exposures at the high end of the distribution may experience adverse effects.
LOAEL <80th percentile of exposure distribution	>20% of population is experiencing exposures greater than LOAEL	Individual- and population-level adverse effects are likely.

NOAEL = no-observed-adverse-effect level; LOAEL = lowest-observed-adverse-effect level.

population level effects must be evaluated based on exposures estimated from foraging within adjacent subreaches or within an entire reach (the entire Poplar Creek). Because the home range of each end point is considered in this assessment, population-level effects can be evaluated.

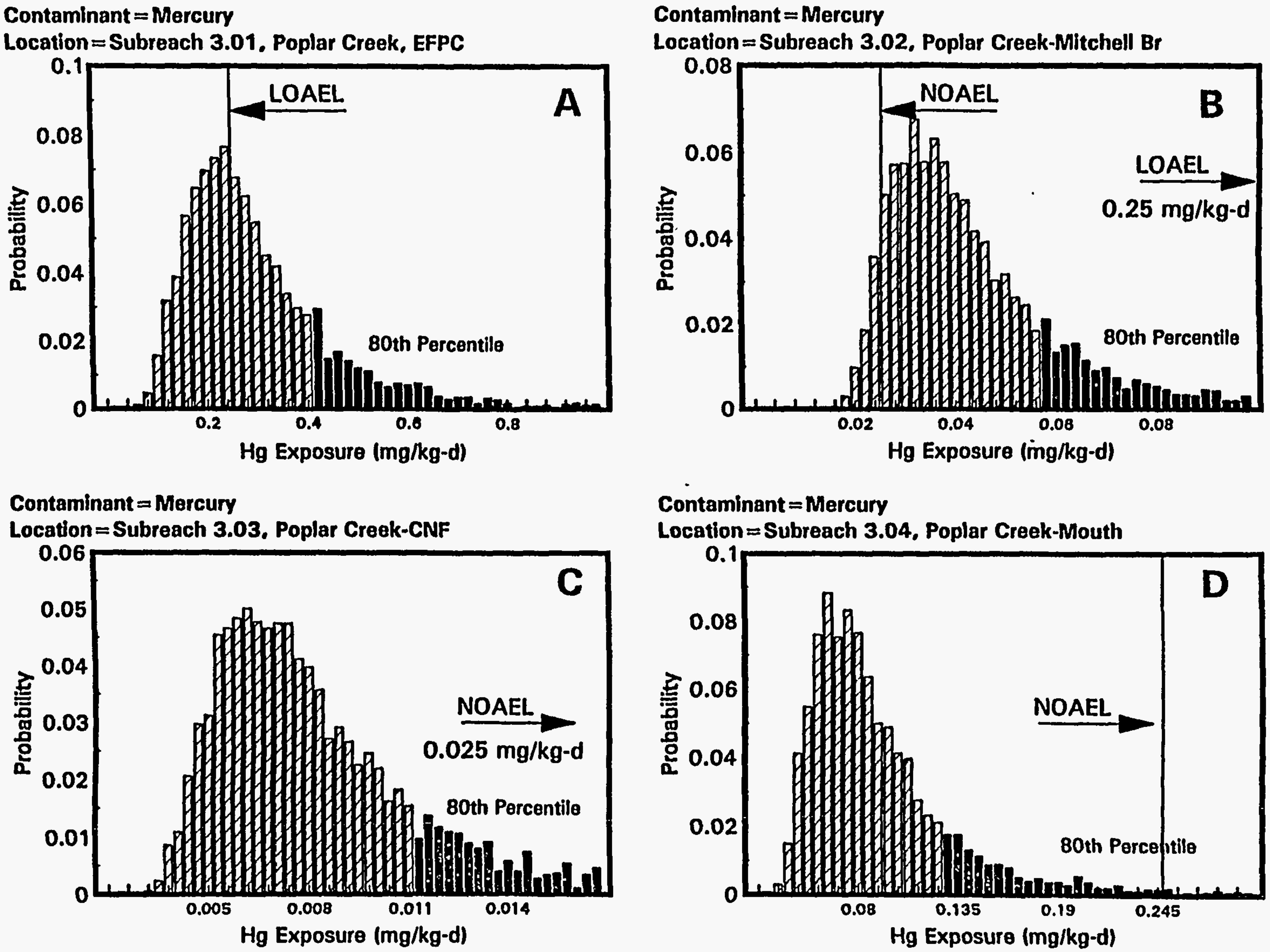

Fig. 6.33. Distribution of estimated mercury exposures for rough-winged swallows.

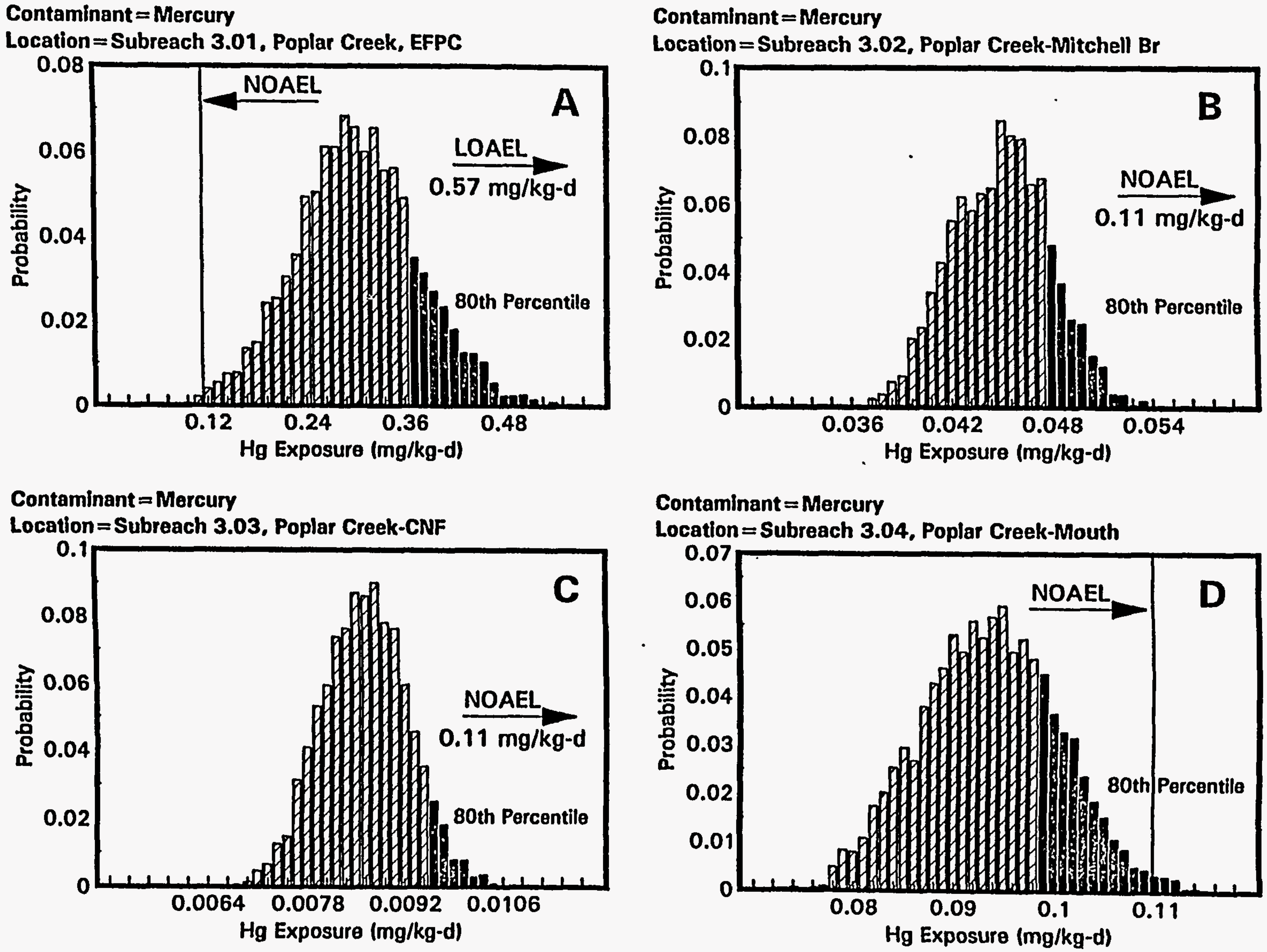

Fig. 6.34. Distribution of estimated mercury exposures for little brown and gray bats.

The results from the comparison of exposure distributions with NOAELs and LOAELs are presented in Table 6.25. More than ~60% of the population of rough-winged swallows at subreaches 3.01 may experience mercury exposures in excess of estimated LOAELs (Fig. 6.33a). Consequently, both individual and population-level effects are likely among rough-winged swallows foraging within this subreach. Mercury exposures to swallows foraging from subreach 3.02 exceeded the NOAEL, however, more than 99% of the population experienced exposures less than the LOAEL (Fig. 6.33b). Therefore, it is unlikely that population-level effects would occur. Although unlikely, individual swallows at the upper portion of the distribution could experience adverse effects. Rough-winged swallows foraging from downstream subreaches 3.03 and 3.04 are experiencing mercury exposures less than the NOAEL; thus, adverse effects are unlikely for individuals (Fig. 6.33c and d). The average mercury point estimate exposure (Table F5.8) for swallows foraging in reach 22 (White Oak Creek Embayment) exceeded the estimated LOAEL by ~1.17 times. Therefore, population-level effects could potentially occur if a colony of swallows is foraging in this area.

Table 6.25. Comparison of NOAEL and LOAEL with the exposure distributions of rough-winged swallows and little brown/gray bats

Subreach	Analyte	Percentile greater than NOAEL	Percentile greater than LOAEL
		Rough-winged swallows	
3.01	Mercury	>99%	~60%
3.02	Mercury	~90%	<1%
3.03	Mercury	<1%	<1%
3.04	Mercury	<1%	<1%
		Little brown/gray bats	
3.01	Mercury	>99%	<1%
3.02	Mercury	<1%	<1%
3.03	Mercury	<1%	<1%
3.04	Mercury	<5%	<1%

NOAEL = no-observed-adverse-effect level; LOAEL = lowest-observed-adverse-effect level.

The little brown or gray bat population will unlikely experience adverse effects due to foraging within Poplar Creek. More than 99% of bats feeding in subreach 3.01, may experience mercury exposures greater than the NOAELs. However, >99% of the exposure distribution was less than the LOAEL (Fig. 6.34a). Therefore, individuals may experience adverse effects if bats forage exclusively from Poplar Creek 3.01. Mercury exposures for bats foraging within subreaches 3.02, 3.03, and 3.04 were less than the NOAEL. Therefore, it is highly unlikely that bats feeding within multiple subreaches are at risk from mercury exposures.

The average mercury and PCB point estimate exposure for little brown or gray bats foraging within reach 22 was only 60% and 10% of the LOAEL, respectively (Tables F5.13 and F5.14). Therefore, population-level effects are unlikely to occur for bats feeding from reach 22. However, individuals which are maximally exposed to mercury could potentially experience adverse effects. In contrast, it is unlikely that individuals will experience adverse effects due to PCBs, since the average PCB exposure was equivalent to the NOAEL (HQ=1.01).

Populations of the federally endangered gray bat are generally small and restricted. Therefore, effects which may be experienced by individuals is of particular importance. If more than 20% of the population is experiencing exposures greater than the NOAEL, individuals within that portion of the population could potentially be impacted. Therefore, the local population of gray bats could experience adverse effects from mercury only if foraging occurred exclusively from Poplar Creek 3.01 and White Oak Creek Embayment. Since the bat's home range is much greater than the size of a single subreach, it is highly unlikely that gray bats are at risk.

6.5.3.2 Effects estimation of retained contaminants of concern

Following screening and exposure simulation, mercury was retained as a contaminant of concern for both rough-winged swallows and little brown or gray bats. Additionally, PCBs were retained for gray bats since there is a potential for individuals to experience adverse effects.

Mercury. The NOAEL and LOAEL for rough-winged swallows are based upon a study of the reproductive success and offspring survival of mallard ducks fed methyl mercury for three generations (Heinz 1979). The study was considered to represent chronic exposure; therefore, a subchronic-chronic correction factor was not employed. A single dose level of 0.064 mg/kg/d was administered causing hens to lay fewer eggs, lay more eggs outside the nest box, and produce fewer ducklings. This dose level was selected as the LOAEL. Because an experimental NOAEL was not established, the NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL correction factor of 0.1. The NOAEL and LOAEL for rough-winged swallows are 0.025 and 0.25 mg/kg/d, respectively.

The reproduction of a colony of rough-winged swallows is likely to be impaired due to mercury exposures if they feed exclusively from subreach 3.01 in Poplar Creek (80th percentile=0.418 mg/kg/d; Fig. 6.33a), and reach 22 in White Oak Creek Embayment (mean=0.293 mg/kg/d). However, the average mercury exposure experienced by swallows feeding within reach 22 was only based on one sediment sample collected within White Oak Creek Embayment. Therefore, the potential magnitude of risk estimated within this reach may not be representative of the entire embayment. Maximally exposed individuals may experience adverse effects as a result of feeding exclusively within subreach 3.02 (80th percentile=0.058 mg/kg/d; Fig. 6.33b). However, since an experimental NOAEL was not established, the nature and exposure level at which effects to individuals may become evident cannot be defined. Also, the effects which may occur, are likely to be less pronounced than that displayed by individuals where exposure greater than the LOAEL.

Both the NOAEL and LOAEL for bats are based upon a study in which reproductive success and offspring survival was observed among rats fed methyl mercury for three generations (Verschuuren et al. 1976a, 1976b, 1976c). The highest dose administered (0.16 mg/kg/d), designated as the LOAEL, resulted in reduction in offspring viability. This exposure also resulted in reduction in growth, increased kidney weight, and altered kidney histochemistry (Verschuuren et al. 1976b). No effects were observed at a dose of 0.032 mg/kg/d. The study was considered to represent chronic exposure; therefore, a subchronic-chronic correction factor was not employed. The NOAEL and LOAEL for little brown bats are 0.114 and 0.56 mg/kg/d, respectively.

The reproduction of individual little brown and gray bats may be impaired due to mercury exposures if bats feed exclusively from subreach 3.01 in Poplar Creek (80th percentile=0.368 mg/kg/d: Fig. 6.34a) reach and 22 in White Oak Creek Embayment (mean=0.342 mg/kg/d). However, the average mercury exposure experienced by little brown and gray bats feeding within reach 22 was only based on one sediment sample collected within White Oak Creek Embayment. Therefore, the potential magnitude of risk estimated within this subreach may not be representative of entire embayment. Additionally, since bats are likely to forage over an area larger than a single subreach, the likelihood that individual bats are at risk from mercury exposure in Poplar Creek and White Oak Creek Embayment is minimal.

PCBs. Both the NOAEL and LOAEL for the little brown bat are based on a study in which reproduction was observed among white-footed mice for 18 months (Linzey 1987). One dose administered, 1.35 mg Aroclor 1254/kg/d in the diet, caused a reduction in the number of offspring per litter and increased the duration between litters. This dosage was considered the chronic LOAEL. A chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL correction factor of 0.1. The NOAEL and LOAEL for little brown bats were 0.187 and 1.87 mg/kg/d, respectively.

The average point estimates of exposure from PCBs experienced by bats within reach 22 was only 10% of the LOAEL. Therefore, impacts to the little brown or gray bat populations are unlikely. Furthermore, the average exposure estimate (0.188 mg/kg/d) slightly exceeded the NOAEL having a HQ of only 1.007. Therefore, it is unlikely (although possible) that individuals within the population are experiencing adverse effects. Since an experimental NOAEL was not established, the nature and exposure level at which effects to individuals may become evident cannot be defined. Additionally, other studies have shown that no effects have occurred at much higher levels of exposure for little brown and big brown bats. Clark and Stafford (1981) exposed little brown bats to 15 ppm and 1,000 ppm Aroclor 1260 (5 mg/kg/d and 330 mg/kg/d, respectively) for 40 days. Mortality was only observed within the higher dosage group. Clark (1978) also exposed pregnant big brown bats to 6.36 ppm Aroclor 1260 (4.77 mg/kg/d) for 28 days during a critical life stage. This exposure did not effect the reproductive success (e.g., stillbirths, litter size, litter weight, parturition data) of the bats. Since big brown bats are much larger, and considered more sensitive to toxicants, similar results would be expected for little brown bats. Therefore, it is highly unlikely that adverse effects would result to individuals from exposures experienced within reach 22.

6.5.3.3 Weight of evidence

Rough-winged swallows. Only one line of evidence, literature toxicity data, was available to evaluate ecological risk to rough-winged swallows. Using only comparisons to benchmarks, the data suggest that reproduction within a colony of rough-winged swallows may be adversely impacted by mercury concentrations in aquatic insects (i.e., mayflies) in subreach 3.01 in Poplar Creek and reach 22 in White Oak Creek Embayment. Individuals may also be adversely affected from foraging in subreach 3.02 in Poplar Creek. However the magnitude of adverse effects which may occur can not be evaluated. Potential adverse effects which may occur include reduced egg production and hatchability, thinner eggshells, or shell-less eggs.

Little brown and gray bats. Only one line of evidence, literature toxicity data, was available to evaluate ecological risk to bats. Using only comparison to benchmarks, the data suggest that populations of little brown and gray bats are not at risk from contaminant exposure. Although, individuals of little brown and gray bats may be adversely affected by mercury concentrations in aquatic insects (i.e., mayflies) within subreach 3.01 and reach 22. Individuals may be at risk only if foraging takes place exclusively within those areas. This is highly unlikely since the foraging range of bats is ~8.5 km from the roost (LaVal et al. 1977).

6.5.4 Uncertainties Concerning Risks to Insectivorous Wildlife

6.5.4.1 Bioavailability of contaminants

It was assumed that 100% if the contaminant concentration reported in mayflies and water was bioavailable. Much of the biotic media are bioavailable; however, the uptake efficiencies for wildlife may not be 100%. Therefore, exposure estimates based upon the contaminant concentrations in media are conservative and are likely to overestimate the actual contaminant exposure experienced.

6.5.4.2 Extrapolation from published toxicity data

There are no published NOAELs for rough-winged swallows and little brown or gray bats. To estimate toxicity of contaminants at the site, it was necessary to extrapolate from NOAELs observed for test species (i.e., mallard ducks and rats). While it was assumed that toxicity could be estimated as a function of body size, the accuracy of the estimate is not known. For example, rough-winged swallows and bats may be more or less sensitive than mallard ducks and rats.

Additional extrapolation uncertainty exists for those contaminants for which data consisted of either LOAELs or was subchronic in duration. For either case, an uncertainty factor of 10 was employed to estimate NOAELs or chronic data. The uncertainty factor of 10 may either over- or underestimate the actual LOAEL-NOAEL or subchronic-chronic relationship.

6.5.4.3 Variable food and water consumption

While food consumption by wildlife was assumed to be similar to that reported for the same species in other locations (Nagy 1987 - swallows; Anthony and Kunz 1977 - little brown/gray bats), the validity of this assumption cannot be determined. Food consumption at the CR/PC may be greater or less than that reported in the literature, resulting in either an increase or decrease in contaminant exposure. Similarly, water consumption was estimated according to the allometric equations of Calder and Braun (1983). The accuracy with which the estimated water consumption represents actual water consumption is unknown.

6.5.4.4 S ingle contaminant tests vs exposure to multiple contaminants in the field

While swallows and bats at the Clinch River are exposed to multiple contaminants concurrently, published toxicological values only consider effects experienced by exposures to single contaminants. Because some contaminants to which wildlife are exposed can interact antagonistically, single contaminant studies may overestimate their toxic potential. Similarly, for those contaminants that interact synergistically, single contaminant studies may underestimate their toxic potential.

6.5.4.5 Inorganic constituents or species present in the environment

Toxicity of metal species varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. For example, Arsenic (III) and methyl mercury are more toxic than arsenic (V) and inorganic mercury, respectively. The available data on the contaminant concentrations in media do not report which species or form of contaminant was observed. Because benchmarks used for comparison represented the more toxic species/forms of the metals (particularly for arsenic and mercury), if the less toxic species/form of the metal was actually present in mayflies from the Clinch River or Poplar Creek, potential toxicity at the sites may be overestimated.

6.5.4.6 Dietary composition of insectivorous wildlife

This assessment assumed that the diet of the little brown and gray bat was composed of 100% mayflies or similar aquatic insects. The concentration found in mayflies was assumed to be representative of other aquatic aerial insects at the site. In contrast, rough-winged swallows feed on a variety of aquatic and terrestrial insects. Beal (1918) documented that only 37.93% of the diet of rough-winged swallows consists of aquatic insects, specifically dipterans and emphemeropterans. Therefore, it was assumed that the contaminant concentrations found in mayflies was representative of the 37.93% of the diet and the remainder of the diet was considered uncontaminated (since terrestrial insects would not have contact with the sediment). Thus, the exposure calculations for both bats and swallows may over or underestimate the contaminant exposure obtained feeding from aquatic insects.

6.5.4.7 Food source availability

Since the abundance of mayflies vary temporally, it was assumed that concentrations found in mayflies are similar to other aquatic insects at the site. The concentrations in the aquatic insect diet may vary from those used to calculate exposure. Therefore, the exposure may be overestimated or underestimated for swallows and bats.

6.5.4.8 Monte Carlo simulation

To perform Monte Carlo simulations, distributions must be assigned to parameters. The sediment distribution was assumed to be log normal; this is a standard accepted distribution for abiotic media concentrations. In cases where a distribution was calculated based on only a few sediment samples, the variance within the model was enhanced.

The literature values used for body weights of each end point are nationwide values which may overestimate or underestimate the body weight of species found at the site.

6.5.4.9 Estimated mayfly concentrations

The concentrations predicted in mayflies with the use of sediment-mayfly uptake factors may underestimate or overestimate exposure depending on the representativeness of the sediment concentrations within each subreach. The uptake factors are only based on an average of measured contaminant concentrations in four mayfly samples from four locations. Additionally, a small sample size for sediments may not be representative of the contaminant concentrations found within an entire subreach. Therefore, the risk predicted may not be representative for that entire foraging area.

6.5.4.10 Presence of a rough-winged swallow colonies within each subreach

This risk assessment assumed that there was a swallow colony (equivalent to a distinct population) present and foraging exclusively within each subreach. This assumes that there is suitable habitat present for swallows to nest. Although, there may not be adequate burrow locations present within the majority of Poplar Creek or White Oak Creek Embayment, swallows will also utilize bridges and crevices in rock cliffs. Moreover, colonies of swallows have been observed frequently on most locations within these areas.

6.6 RISKS FROM DREDGED SPOIL

6.6.1 Exposure Assessment of Dredged Spoil

This assessment evaluates the hypothetical scenario where sediments within the Clinch River or Poplar Creek are dredged and deposited on land. The dredged spoil would then provide a potential pathway for ecological receptors. Any terrestrial ecosystem that developed on the dredged spoils would be exposed to contaminants in those sediments from which the spoil was derived. The spoil is assumed to be derived from mixed deep water sediments where irrigation will not take place. Vegetation can become established on the spoil, bioaccumulate contaminants, and then be consumed by herbivorous wildlife (e.g., eastern cottontail) (Fig. 6.2). For additional information on this scenario see the human health risk assessment (Sect. 5.2.3.6).

6.6.1.1 Characterization of exposure environment and routes

Plants. The main exposure route for plants is uptake of contaminants from the spoil through the roots. This may occur in a passive mode as the plant takes up water for respiration or by active uptake mechanisms. Volatile organic contaminants in the spoil may potentially enter the plant through leaf stomata or the plant cuticle. Another potential route of exposure is through association of contaminants with airborne spoil particles (dust).

Herbivorous wildlife. Potential routes of exposure for herbivorous wildlife include ingestion of contaminated vegetation and surface water. In addition, some species may ingest soil incidentally while foraging or purposefully to meet nutrient needs. The total exposure experienced by terrestrial wildlife is represented by the sum of the exposure from each individual source (e.g., vegetation, spoil, water).

6.6.1.2 Quantification of exposure

Plants. The contaminant concentrations found in the sediments is assumed to represent the total potential reservoir of elements and compounds available for plant uptake in the future dredge scenario. The future bioavailability of these contaminants which is dependent upon the chemical (e.g., pH, organic carbon) and physical (e.g., clay, moisture content) nature of the spoil can not be addressed for this assessment. This assessment assumes that the measured concentrations in the deep sediments are estimates of exposure equivalent to the soil concentrations causing toxic effects in laboratory studies.

Herbivorous wildlife. The primary pathway of contaminant exposure is through oral ingestion of food and soil. Consumption of surface water, in most cases, is considered a pathway which contributes minimal contaminant exposure. Additionally, it is unlikely that rabbits will be ingesting water from the Clinch River due to inaccessibility from the presence of high river banks. For the purpose of this scenario, it is assumed that wildlife will be obtaining uncontaminated drinking water from smaller bodies of water offsite. Therefore, the total oral exposure model for herbivorous wildlife is as follows:

$$E_j = \left(\frac{IR_v \times C_{v\text{-}j}}{BW}\right) + \left(\frac{IR_s \times C_{s\text{-}j}}{BW}\right) \tag{6.10}$$

Where:

E_j = exposure to contaminant (j) (mg/kg/d)
IR_v = ingestion rate of vegetation (mg/d)

$C_{v\text{-}j}$ = contaminant concentration (j) in vegetation (mg/kg)
BW = body weight (kg)
$C_{s\text{-}j}$ = contaminant concentration (j) in spoil (mg/kg)
IR_s = ingestion rate of spoil (mg/d)

The eastern cottontail was identified as the herbivorous wildlife end point. Species-specific parameters necessary to estimate exposure using the above equation arp-18therbivTablesF6ildlife end point. Spec

Contaminant concentrations in biotic and abiotic media. Contaminant concentrations in vegetation and sediment are needed to estimate exposure. Deep water sediment data originated from cores during the CRRI Phase 1 (Cook et al. 1992) and Phase 2 (Table F6.2). Individual core concentrations were calculated from mass-weighted averages over the entire length of the core. The UCL_{95} or maximum concentration (whichever was lower) from all cores within each reach and subreach (reaches 1 and 7 and subreaches 2.04, 3.01, 3.02, 3.03, 3.04, 4.01, 4.02, 4.03, and 4.04) were used to estimate exposure for incidental ingestion. Initially, the maximum sediment concentration was compared to background concentrations (Table F6.3). Reach 13, an upstream reference site, was designated as representive background levels uninfluenced by DOE effluents. For contaminants which exceeded background, the UCL_{95} or maximum contaminant concentrations in sediment and vegetation was used to calculate exposure from food ingestion. Leafy vegetables [modeled for the human health risk assessment (Sect. 5.2.3.6)], or soil-plant uptake factors were used to estimate exposure from ingestion of vegetation. Soil-plant uptake factors calculated for flouranthene, dibenzofuran, naphtalene, phenathrene, and acenaptheylene were derived from the log octanol-water partition coefficient (log K_{ow}) using the following equation (Travis and Arms 1988):

$$\text{Log } B_v = 1.588 - 0.578 \log K_{ow} \qquad (6.11)$$

where:
B_v = Bioaccumulation factor for vegetation
K_{ow} = Octanol Water Partitioning Coefficient (Source: Hull and Suter 1994).

Copper and lead soil-plant uptake factors were derived from Baes et al. (1984) and the uranium soil-plant uptake factor was calculated based on concentrations found in collocated soil and vegetation samples on Lower East Fork Poplar Creek (DOE 1994a).

Exposure modeling using point-estimates. To estimate contaminant exposure experienced by eastern cottontails feeding on the spoil in each reach, the following assumptions were made:

(1) body weight = 1.2 kg
(2) food consumption = 0.237 kg/d
(3) spoil consumption = 0.015 kg/d
(4) diet consists 100% of vegetation
(5) 100% of the habitat available is suitable and
(6) home range is smaller than the size of the contaminated area (i.e., 100% of time is spent within the contaminated area.

Using the herbivorous exposure model, assumptions, and data described above, exposure of contaminants was estimated for the eastern cottontail foraging on the future spoil created within each subreach. Exposure estimates for analytes in sediments above background concentrations are presented in Table F6.4.

Exposure modeling using Monte Carlo simulations. Employing point estimates for the input parameters in the Clinch River herbivore exposure model does not take into account the variation and uncertainty associated with the parameters and may over or under estimate the contaminant exposure that end points may receive in any given subreach. In addition, calculating the model using point estimates produced a point estimate of exposure. This estimate provides no information concerning the distribution of exposures or the likelihood that individuals within a subreach will actually experience potentially hazardous exposures. To incorporate the variation in exposure parameters and to provide a better estimate of the potential exposure experienced by herbivores in each subreach, the exposure model was re-calculated using Monte Carlo simulations.

Monte Carlo simulations of exposure estimates were performed for all subreach-contaminant combinations where comparison of point exposure estimates to NOAELs produced HQs ≥ 1 (Table 6.26) (NOAELs are presented in Sect 6.6.2.1; Screening of exposure estimates against NOAELs is presented in Sect. 6.6.3.1).

Distributions were used for the following parameters in the spoil herbivore exposure model: (1) eastern cottontail body weight and (2) contaminant concentrations in sediment (spoil). The body weight distributions of the eastern cottontail was assigned a triangular distribution and the sediment distribution was normal (Table 6.27). The vegetation concentrations were either the PLE UCL_{95} or the maximum values. Because wildlife are mobile and are more likely to be exposed to average concentrations and not to the extreme values, the standard error of the mean was used to describe variation for contaminant concentrations. All other distributions employed the standard deviation. The food consumption rate and incidental ingestion of soil rate used in the exposure model were average values.

Table 6.26. Monte Carlo simulations of exposure estimates

Reach or subreach	Contaminants
1: Lower Melton Hill	As, Cd
7: McCoy Branch	As, Cd, Se, Tl, V
2.04: Clinch River (Grassy Creek to Poplar Creek)	As, Cd, Mn, Se,
3.02: Poplar Creek below Mitchell Branch	Cd, Hg
3.03: Poplar Creek below CNF outfall	Cd, Hg
3.04: Poplar Creek Mouth (below ash disposal)	Ba, Cd, Hg, Se, Tl, V
4.01: Clinch River (Poplar Creek to below Brashear Island)	As, Cd, Hg, Aroclor 1260
4.02: Clinch River (below Brashear Island to Emory River)	As
4.03: Clinch River (Emory River to Kingston City Park)	As, Cd, Hg
4.04: Mouth of Clinch River	As, Cd, Hg

Table 6.27. Herbivorous body weight distribution

End point	Distribution	Mean (kg)	Standard deviation	Range (kg)	Source
Eastern Cottontail	triangular	1.244		0.842-1.533 based on ♀	Chapman et al. 1980

Source: Chapman, G. A., S. Ota, and F. Recht. 1980. *Effects of Water Hardness on the Toxicity of Metals to Daphnia magna* (Status Report—Sept. 1980). U.S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oreg.

6.6.2 Effects Assessment of the Dredged Spoil

Effects assessment involves the identification of known effects of contaminants on plants and herbivorous wildlife using ecotoxicological benchmarks. The types, development, and interpretation of appropriate toxicological benchmarks are discussed and available toxicity data relevant to plants and wildlife are summarized in toxicity profiles for COPECs. Conventional toxicity data consists of published values for toxicity of contaminants to test species. These data are used in the development of toxicological benchmarks which are used to determine if biological effects are likely. Contaminant exposures are compared to benchmarks to obtain HQs used in the risk characterization.

6.6.2.1 Toxicological benchmarks

Plants. Contaminant concentrations in the future spoil within each subreach were compared to toxicological benchmarks in order to determine if contaminants could produce adverse effects on the plant community. Tests conducted in natural soils are assumed to be representative of the exposure of plants to contaminants measured in the spoil.

Growth and yield parameters were used from phytotoxicity studies for the derivation of the benchmarks. These parameters are direct estimates of the assessment end point for plants. Twenty percent reduction in growth or yield was used as the threshold for significant effects to be consistent with other screening benchmarks for ecological risk assessment. Additionally, the FFA parties adopted this level of effects for ecological end points (Suter et al. 1994).

The method used for the derivation of soil benchmarks (Will and Suter 1994) is based on the National Oceanographic and Atmospheric Administration's method for deriving the ER-L (Long and Morgan 1991) which has been recommended as a sediment screening benchmark by EPA Region IV. The ER-L is the tenth percentile of the distribution of various toxic effects thresholds for various organisms in sediments.

The phytotoxicity benchmarks were derived by rank ordering EC_{20} values for plant growth or production and then identifying a number that approximated the tenth percentile (Will and Suter 1994). As with the ER-Ls, statistical fitting was not used because there was seldom sufficient data and because these benchmarks are to be used as screening values and do not require the consistency and precision of regulatory criteria. Screening benchmarks for phytotoxic effects of contaminants present in the future spoils are presented in Table F6.5.

Herbivorous wildlife. To determine if the contaminant exposures experienced by eastern cottontail foraging on future spoils could produce adverse effects, exposure estimates are compared to NOAELs and LOAELs derived according to the methods outlined by Opresko et al. (1994) and EPA (1993b).

NOAELs represent the highest exposure at which no adverse effects were observed among the animals tested. LOAELs represent the lowest exposure at which significant adverse effects are observed.

Toxicological studies of the effects of contaminants observed in the spoil were obtained from the open literature. Only studies of long-term, chronic oral exposures were used to estimate the NOAEL. To make the NOAELs relevant to possible population effects, preference was given to studies that evaluated effects on reproductive parameters. In the absence of a reproduction end point, studies that considered effects on growth, survival, and longevity were used. The NOAELs and LOAELs for eastern cottontails and the studies which derived them are located in Table F6.6. This table presents only the NOAELs and LOAELs for contaminants which had HQs >1. Specific details on development of the NOAELs and LOAELs for all wildlife end points are discussed in Sect. 6.4.2.1.

6.6.2.2 Ecotoxicological profiles for plants

Several elements for which analytical data are available are plant macronutrients and there is no evidence of potential plant toxicity. Therefore, no benchmarks have been established for these elements and no toxicity profiles will be given. These macronutrients include calcium, magnesium, and potassium. Toxicity profiles are presented for chemicals occurring in soil at concentrations exceeding benchmarks and background for toxicity to plants (Table F6.5). There are many organic contaminants which were detected in sediment for which we have no information to offer. These contaminants include: 4,4'-DDD, 4,4'-DDT, 1,4-dichlorobenzene, 2-methylnapthalene, 4-methylphenol, 4-nitrophenol, 4-chloro-3-methylphenol, acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(ghi)perylene, benzo(k)fluoranthene, benzoic acid, bis(2-ethylhexyl)phthalate, carbazole, chrysene, delta-BHC, di-n-octylphthalate, dibenzofuran, dibenz(a,h)anthracene, dimethylphthalate, diethylphthalate, fluoranthene, fluorene, heptachlor, indeno(1,2,3-cd)pyrene, methoxychlor, napthalene, N-nitrosodiphenylamine, pentachlorophenol, phenathrene, phenol, pyrene, and toxaphene.

The ecotoxicological profiles for contaminants of potential ecological concern for plants may be found in Appendix F1. Contaminants include arsenic, boron, cadmium, chromium, manganese, mercury, nickel, selenium, silver, uranium, vanadium, and zinc.

6.6.2.3 Ecotoxicological profiles for herbivorous wildlife

The ecotoxicological profiles for contaminants of potential ecological concern for herbivorous wildlife may be found in Appendix F1. Contaminants include arsenic, barium, cadmium, manganese, mercury, Aroclor 1260, selenium, thallium, and vanadium. There are many organic contaminants which were detected in sediment and modeled in vegetation for which we have no information to offer. These contaminants include: 1,4-dichlorobenzene, 2-methylnapthalene, 4-methylphenol, 4-nitrophenol, 4-chloro-3-methylphenol, acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(b)fluoranthene, benzo(ghi)perylene, benzo(k)fluoranthene, benzoic acid, carbazole, chrysene, delta-BHC, di-n-octylphthalate, dibenzofuran, dibenz(a,h)anthracene, dimethylphthalate, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, methoxychlor, napthalene, N-nitrosodiphenylamine, pentachlorophenol, phenathrene, phenol, pyrene, and toxaphene. Therefore, potential adverse effects from exposure to these contaminants can not be evaluated.

6.6.3 Risk Characterization for the Dredged Spoil

Risk characterization integrates the results of the exposure assessment (Sect. 6.6.1) and effects assessment (Sect. 6.6.2) to estimate risks (the likelihood of effects given the exposure). Procedurally, the risk characterization in this assessment is performed for each assessment end point by (1) screening all measured contaminants against toxicological benchmarks, (2) estimating the effects of the contaminants retained by the screening analysis, (3) listing and discussing the uncertainties in the assessment. There is only one line of evidence, single chemical toxicity data, available to evaluate the risks to plants and eastern cottontails inhabiting the future spoil within each subreach.

6.6.3.1 Single chemical toxicity data for plants

COPECs in the spoil were identified by comparing the maximum chemical concentrations measured in sediments to phytotoxicity benchmarks to derive HQs by the formula

HQ = media concentration / toxicological benchmark.

HQs >1 suggest that the chemical may be hazardous to the plants that may grow on the spoil. HQs <1 suggest that the chemical is nonhazardous and need not be considered further. Benchmarks for plants growing in soils, contaminant concentrations found in the future spoil, and HQs for these contaminants are given in Table F6.5. The COPECs which may pose a risk to plants within each subreach are listed in Table 6.28.

The contaminants in sediment within each subreach listed in Table 6.28 exceeded background concentrations and phytotoxicity benchmarks (Will and Suter 1994). These analyte concentrations have caused toxic effects when added to surface soils. The specific effects for each contaminant may be found in the ecotoxicological profiles found in Appendix F1. The magnitude of the COPEC HQ indicates the contaminants which would contribute the most risk to plants inhabiting the spoil.

6.6.3.2 Single chemical toxicity data for herbivorous wildlife

Two types of single chemical toxicity data are available with which to evaluate herbivore contaminant exposure: NOAELs and LOAELs. The total contaminant exposure estimates (Table F6.4) for eastern cottontails feeding on the spoils were compared to estimated NOAELs to determine if adverse effects are possible. The comparison of estimated exposure and the NOAEL acts as a screening tool to identify contaminants of concern which will be further evaluated for possible estimated effects (i.e., exposure modeling using Monte Carlo simulation). LOAELs are compared to the exposure distribution generated by the Monte Carlo simulation. If the LOAEL is lower than the 80th percentile of the exposure distribution, more than 20% of the end point population is experiencing contaminant exposures that are likely to produce adverse effects. Consequently, population-level effects to eastern cottontails are likely.

Screening point estimates of exposure. To determine if the contaminant exposures experienced by cottontails feeding on the future spoils along the Clinch River and Poplar Creek are potentially hazardous, the total exposure estimates (Table F6.4) were compared to estimated NOAELs. HQs were calculated to quantify the magnitude of the hazard where: HQ = estimated exposure/NOAEL. HQs >1 indicate that individuals may be experiencing exposures that are in excess of NOAELs, and may suggest that adverse effects may be occurring. HQs for cottontails are presented along with exposure estimates in Table F6.4.

Table 6.28. Contaminants of potential ecological concern which may pose a risk to plants within each subreach

Reach or subreach	Contaminant of potential ecological concern	Hazard quotient
1	Arsenic	1.1
7	Arsenic	6.4
	Chromium	58.6
	Manganese	3.2
	Selenium	9.5
	Vanadium	21.3
2.04	Arsenic	1.02
	Manganese	9
	Selenium	2.4
3.02	Mercury	20.3
	Nickel	2.4
	Selenium	1.4
	Uranium	3.3
	Zinc	3
3.03	Cadmium	1.1
	Chromium	44.4
	Mercury	22.2
	Nickel	2.1
	Silver	1.7
	Uranium	1.7
	Zinc	3.2
3.04	Arsenic	9.4
	Boron	87.7
	Cadmium	1.9
	Chromium	39.2
	Mercury	85.6
	Nickel	1.7
	Selenium	1.6
	Uranium	3.2
	Vanadium	29.6
4.01	Mercury	40.5
4.02	Arsenic	1.1
4.03	Chromium	36.9
	Manganese	3.1
	Mercury	11.7
4.04	Arsenic	1.2
	Chromium	33.4
	Manganese	4
	Mercury	18.5
	Zinc	3.1

The spatial distribution of contamination and potential risks to cottontails at spoils for each Clinch River subreach are illustrated in Fig. 6.35. This figure displays the sum of the NOAEL-based HQs (e.g., sum of toxic units or Σ TUs) for the most important contaminants. Arsenic, cadmium, and mercury are the primary contaminants contributing to risk. However, barium, manganese, Aroclor 1260, selenium, thallium, and vanadium also contribute to overall risk at several locations. Importance of contaminants was determined based on the magnitude of the HQs. River subreaches were arranged from upstream to downstream locations. A summary of the subreaches where HQs >1 were observed is listed in Table 6.29 for each contaminant.

Screening Monte Carlo simulation estimates of exposure. To incorporate the variation present in the parameters employed in the Clinch River herbivorous exposure model, Monte Carlo simulations were performed for subreach-contaminant combinations where HQs >1 were observed. The mean, standard deviation, and 80th percentile of the simulated exposures are presented in Table 6.30. By superimposing NOAEL and LOAEL values on these distributions, the proportion of the population experiencing potentially hazardous exposures, can be identified and the magnitude of risk may be determined. Interpretation of the comparison of exposure distributions to NOAELs and LOAELs is described in Table 6.31. The results from the comparison of exposure distributions with NOAELs and LOAELs are presented in Table 6.32.

The eastern cottontail population may be adversely impacted if foraging occurred on the future spoil within reaches 1 and 7 and subreaches 3.02 3.03, 3.04, 4.01, 4.03, and 4.04. More than 99% of the population is projected to be exposed to mercury (subreaches 3.03, 3.04, and 4.01) and cadmium (reaches 1 and 7 and subreaches 3.02, 3.03, 3.04, 4.03, and 4.04) in excess of the estimated LOAELs (Figures 6.36–6.44). Conversely, <1% of the population of cottontails feeding at the spoil within reach 7 may experience thallium exposures in excess of the NOAEL. Additionally, <10% of the population of cottontails feeding at the spoil in subreach 3.04 may experience barium exposure in excess of the NOAEL. Therefore, adverse individual and population-level effects from thallium and barium exposure in reach 7 and subreach 3.04, respectively, are highly unlikely. The exposures for all other contaminants within reaches 1 and 7 and subreaches 2.04, 3.02, 3.03, 3.04, 4.01, 4.02, 4.03, and 4.04, exceeded NOAELs while not exceeding LOAELs. Therefore, while adverse effects to the population within each subreach is unlikely, adverse effects may be displayed among individuals that experience exposures at the upper extremes of the distributions.

6.6.3.3 Estimation of effects for herbivorous wildlife

Arsenic. Both the NOAEL and LOAEL for cottontails are based upon a study in which reproductive success and offspring survival was observed among mice fed arsenic for three generations (Schroeder and Mitchner 1971). One dose level administered (1.261 mg/kg/d), designated as the chronic LOAEL, resulted in declining litter size with each successive generation. A chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL correction factor of 0.1. The NOAEL and LOAEL for cottontails are 0.037 and 0.37 mg/kg/d, respectively. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individuals may become evident cannot be defined. However, adverse effects suggested by Schroeder and Mitchner (1971) are possible for the higher extremes of exposure on the distribution at reaches 1 and 7 and at subreaches 2.04, 4.01, 4.02, 4.03 and 4.04.

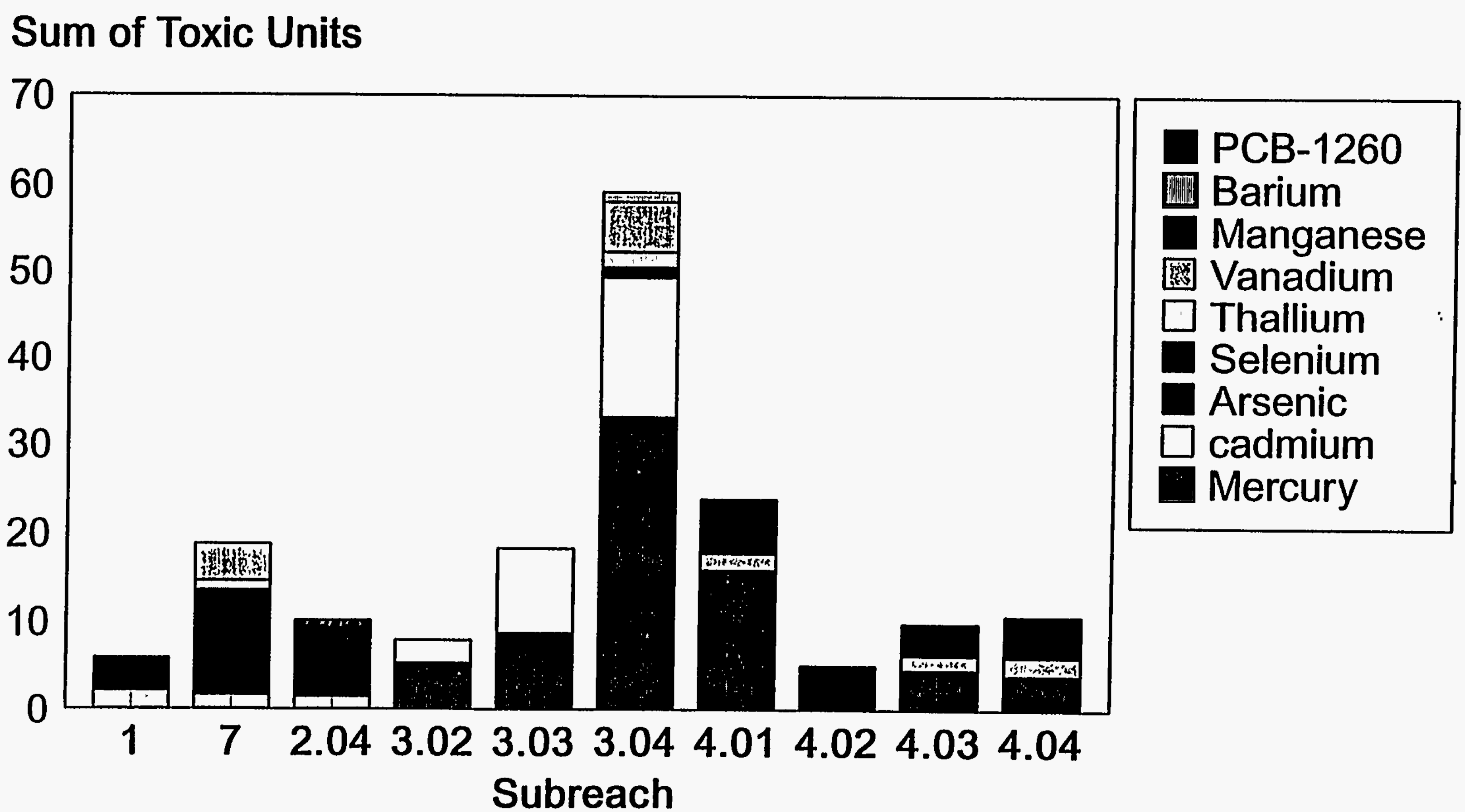

Fig. 6.35. Sum of no-observed-adverse-effect level–based toxic units for evaluation of risks to eastern cottontails foraging on dredge spoils within each subreach.

Table 6.29. Summary of subreaches where hazard quotients >1 were observed for each contaminant

Reach or subreach	Contaminant of potential ecological concern	Hazard quotient
1	Arsenic	3.8
	Cadmium	2.1
7	Arsenic	9.1
	Cadmium	1.6
	Selenium	2.8
	Thallium	1.1
	Vanadium	4.2
2.04	Arsenic	4.4
	Cadmium	1.4
	Manganese	2.5
	Selenium	1.9
3.02	Cadmium	2.64
	Mercury	5.3
3.03	Cadmium	9.6
	Mercury	8.7
3.04	Barium	1.1
	Cadmium	16.2
	Mercury	33.2
	Selenium	1.2
	Thallium	1.8
	Vanadium	5.8
4.01	Arsenic	4.5
	Cadmium	1.9
	Mercury	15.9
	Aroclor 1260	1.7
4.02	Arsenic	5.1
4.03	Arsenic	3.6
	Cadmium	1.7
	Mercury	4.6
4.04	Arsenic	4.7
	Cadmium	2.1
	Mercury	4

Table 6.30. Summary of Monte Carlo simulation for eastern cottontails exposed to contaminants in dredge spoils

Reach or subreach	Analyte	Media concentrations (mg/kg)			Simulation results (mg/kg/d)			LOAEL benchmark (mg/kg/d)
		Mean spoil	Spoil (standard deviation)	Vegetation (mean)	Mean	Standard deviation	80th percentile	
1	As	8.02	3.014	0.013	0.104	0.040	0.139	0.37
	Cd	0.572	0.155	0.007	0.009	0.002	0.010	0.007
7	As	16.67	5.374	0.03	0.214	0.073	0.273	0.37
	Cd	0.46	0.057	0.005	0.007	0.001	0.008	0.007
	Se	2.03	0.88	0.095	0.045	0.013	0.055	0.22
	Tl	0.04	0.03	0	0.000	0.000	0.001	0.05
	V	36.75	5.903	0.047	0.472	0.093	0.547	1.3
2.04	As	6.42	1.41	0.011	0.083	0.021	0.100	0.37
	Cd	0.34	0.06	0.005	0.005	0.001	0.006	0.007
	Mn	2716	1067	450	124.121	20.309	140.181	189
	Se	1.06	0.69	0.063	0.026	0.009	0.033	0.22
3.02	Hg	1.68	1.12	0.31	0.083	0.017	0.097	0.11
	Cd	0.51	0.19	0.009	0.008	0.003	0.010	0.007
3.03	Hg	6.67		0.5	0.184	0.023	0.202	0.11
	Cd	3.31	-	0.034	0.049	0.006	0.054	0.007
3.04	Ba	195.4	53.7	0.95	2.679	0.764	3.307	13.2
	Cd	2.39	1.6	0.057	0.042	0.021	0.058	0.007
	Hg	12.45	6.66	1.9	0.534	0.108	0.621	0.11
	Se	1.03	0.27	0.041	0.021	0.004	0.025	0.22
	Tl	0.41	0.16	0	0.005	0.002	0.007	0.05
	V	37.79	11.83	0.065	0.490	0.158	0.616	1.3
4.01	As	5.38	2.34	0.011	0.070	0.031	0.095	0.37
	Cd	0.36	0.15	0.007	0.006	0.002	0.008	0.007
	Hg	4.56	3.81	0.92	0.241	0.057	0.286	0.11

Table 6.30 (continued)

Reach or subreach	Analyte	Media concentrations (mg/kg)			Simulation results (mg/kg/d)			LOAEL benchmark (mg/kg/d)
		Mean spoil	Spoil (standard deviation)	Vegetation (mean)	Mean	Standard deviation	80th percentile	
	Aroclor 1260	0.7	0.68	0.003	0.009	0.009	0.016	0.16
4.02	As	11.2		0.013	0.144	0.018	0.158	0.37
4.03	As	6.56	1.28	0.009	0.084	0.019	0.100	0.37
	Cd	0.58	0.02	0.006	0.009	0.001	0.009	0.007
	Hg	3.36	0.16	0.27	0.096	0.012	0.106	0.11
4.04	As	9.36	0.536	0.012	0.120	0.016	0.134	0.37
	Cd	0.58	0.078	0.007	0.009	0.001	0.010	0.007
	Hg	2.47	0.33	0.23	0.077	0.010	0.085	0.11

LOAEL = lowest-observed-adverse-effect level

Table 6.31. Comparison of exposure distribution to NOAELs and LOAELs

Comparison	Meaning	Risk-based interpretation
NOAEL>80th percentile of exposure distribution	<20% of population is experiencing exposures greater than NOAEL	Individual- and population-level adverse effects are highly unlikely
NOAEL< 80th percentile <LOAEL	>20% of population is experiencing exposures greater than NOAEL, but <20% is experiencing exposures greater than LOAEL	While population-level effects are unlikely, individuals exposed to exposures at the high end of the distribution may experience adverse effects.
LOAEL <80th percentile of exposure distribution	>20% of population is experiencing exposures less than LOAEL	Individual- and population-level adverse effects are likely.

NOAEL = no-observed-adverse-effect level; LOAEL = lowest-observed-adverse-effect level.

Table 6.32. Results from the comparison of exposure distributions with NOAELs and LOAELS

Reach or subreach	Analyte	Percentile greater than NOAEL	Percentile greater than LOAEL
1	As	>95%	<1%
	Cd	>95%	~75%
7	As	>99%	<1%
	Cd	>95%	>35%
	Se	>95%	<1%
	Tl	<1%	<1%
	V	>99%	<1%
2.04	As	>99%	<1%
	Cd	>55%	<5%
	Mn	>99%	<1%
	Se	>60%	<1%
3.02	Cd	>90%	>65%
	Hg	>99%	<5%
3.03	Cd	>99%	>99%
	Hg	>99%	>99%
3.04	Ba	<10%	<1%
	Cd	>99%	>99%

Table 6.32 (continued)

Reach or subreach	Analyte	Percentile greater than NOAEL	Percentile greater than LOAEL
	Hg	>99%	>99%
	Se	>30%	<1%
	Tl	>50%	<1%
	V	>99%	<1%
4.01	As	>85%	<1%
	Cd	>65%	>25%
	Hg	>99%	>99%
	Aroclor 1260	~20%	<1%
4.02	As	>99%	<1%
4.03	As	>99%	<1%
	Cd	>99%	>99%
	Hg	>99%	<15%
4.04	As	>99%	<1%
4.04	Cd	>99%	>85%
	Hg	>99%	<1%

NOAEL = no-observed-adverse-effect level; LOAEL = lowest-observed-adverse-effect level.

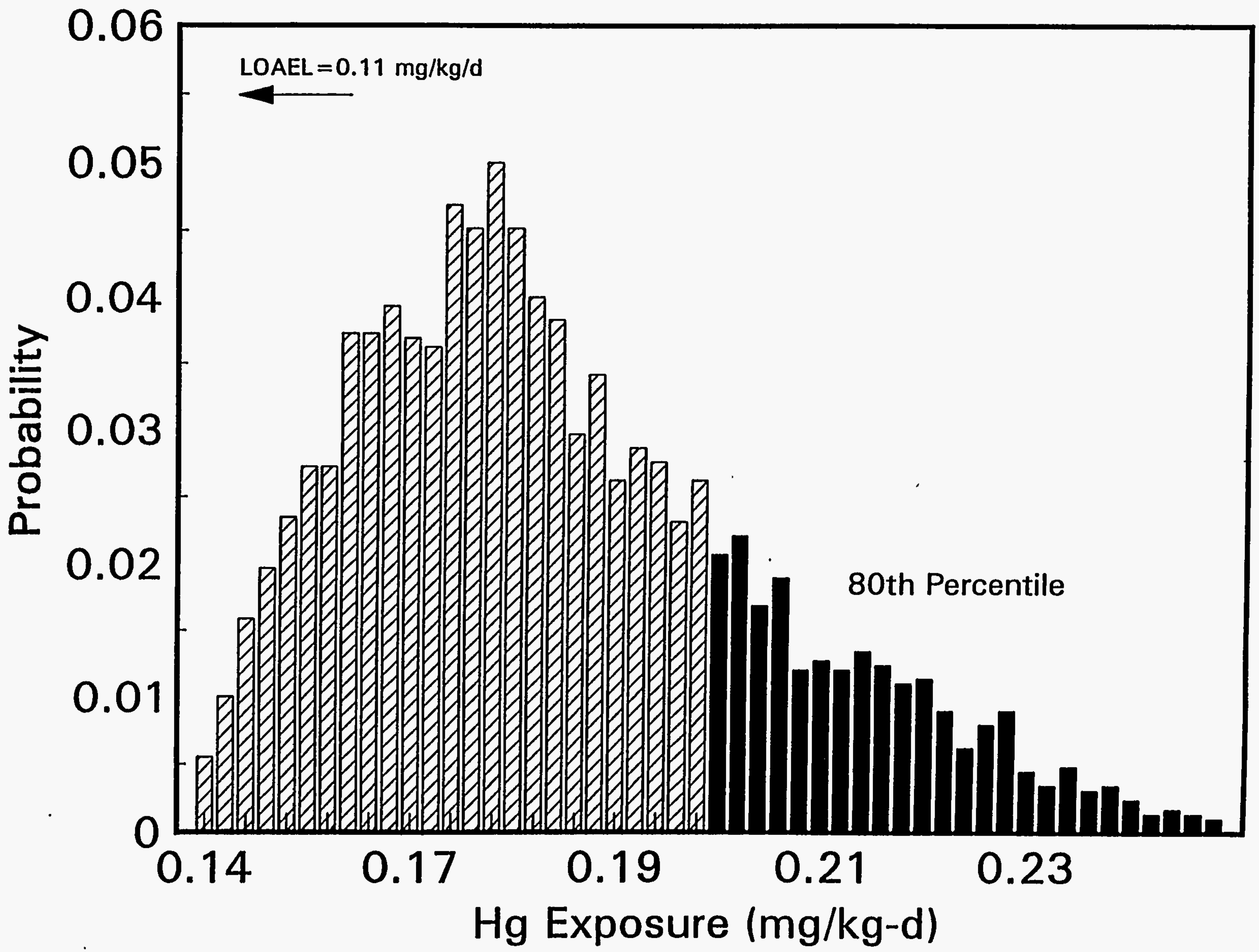

Fig. 6.36. Mercury exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.03 using Monte Carlo simulation.

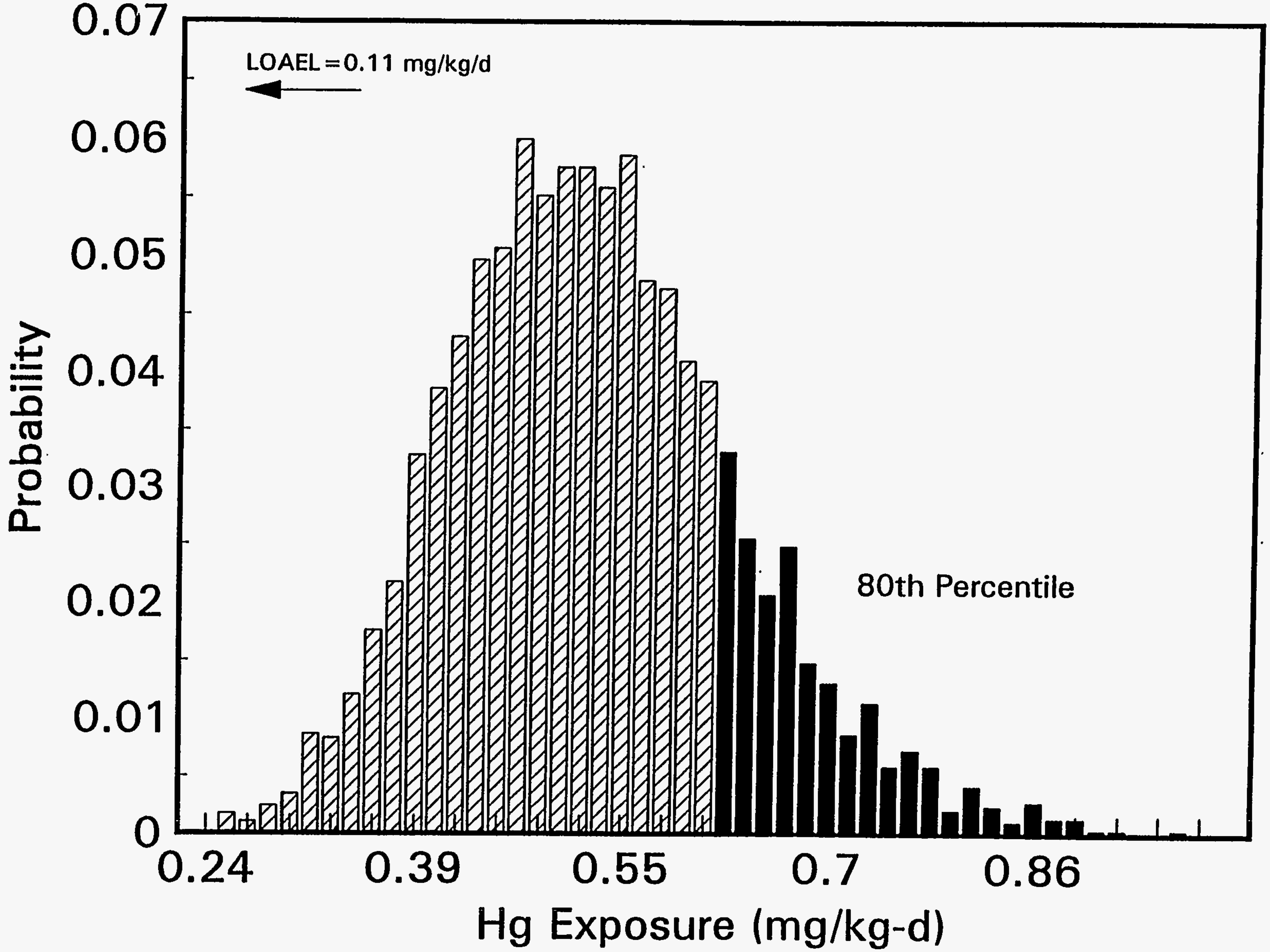

Fig. 6.37. Mercury exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.04 using Monte Carlo simulation.

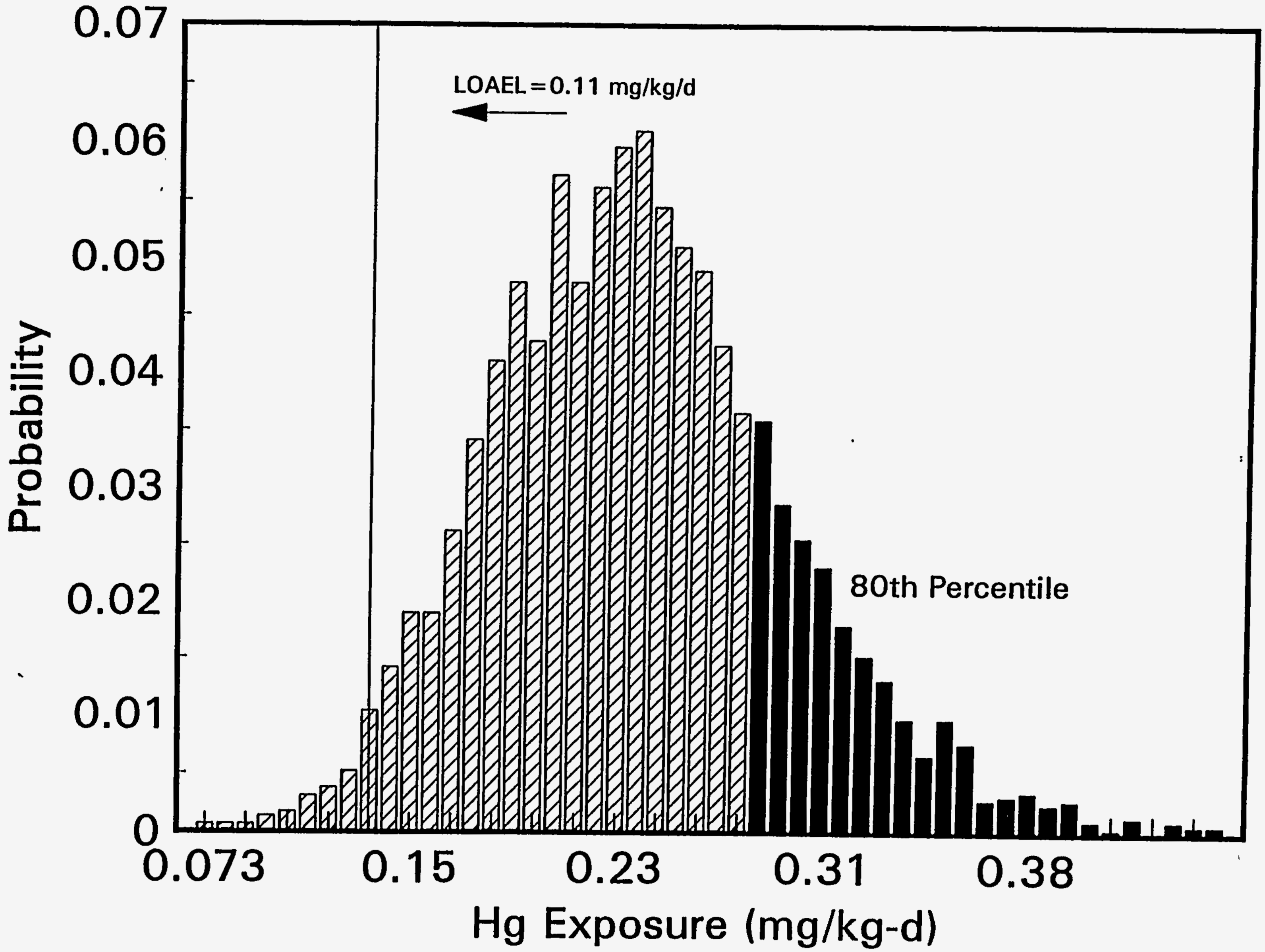

Fig. 6.38. Mercury exposure distribution for eastern cottontails foraging on a future spoil within subreach 4.01 using Monte Carlo simulation.

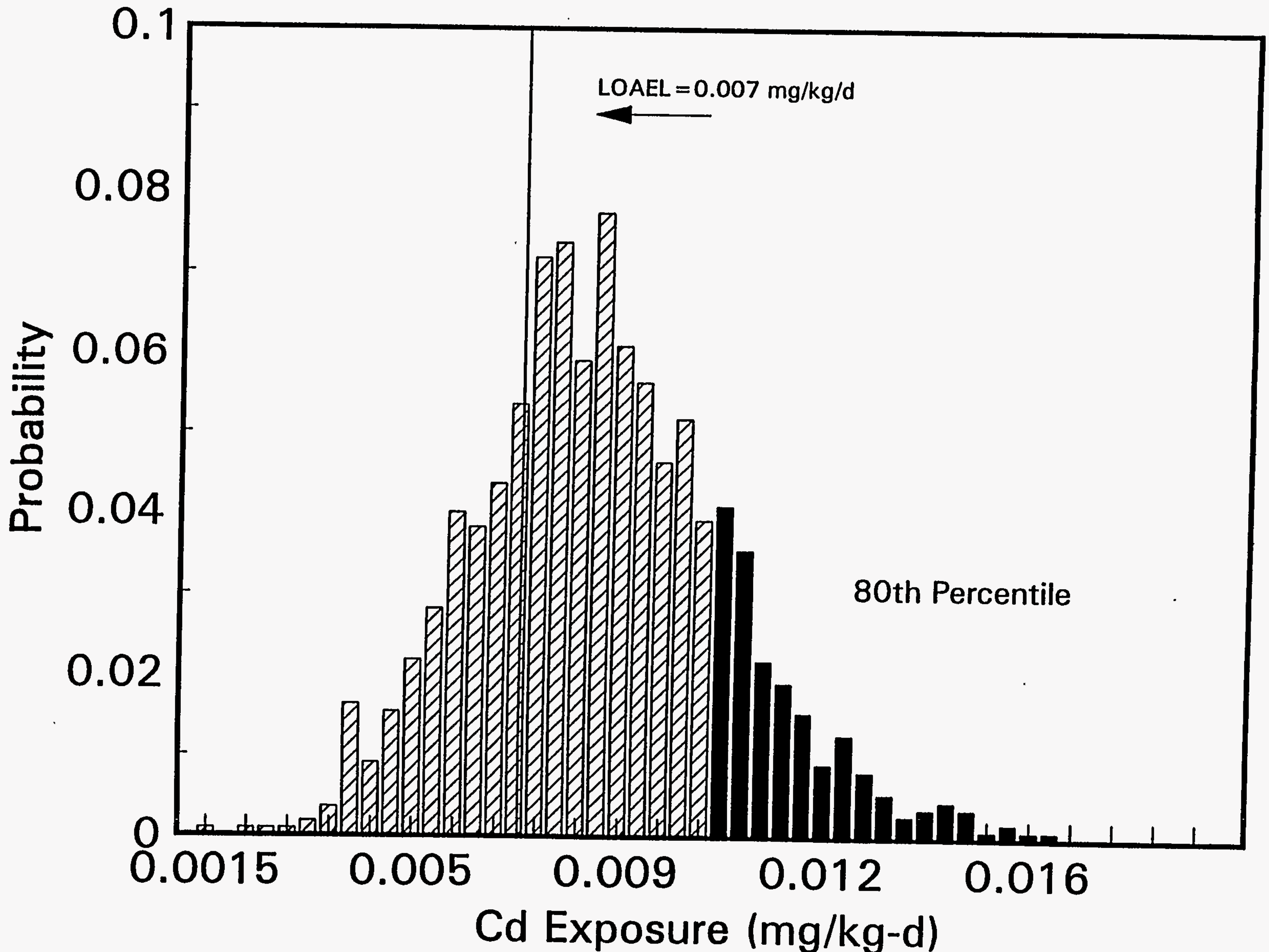

Fig. 6.39. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within reach 1 using Monte Carlo simulation.

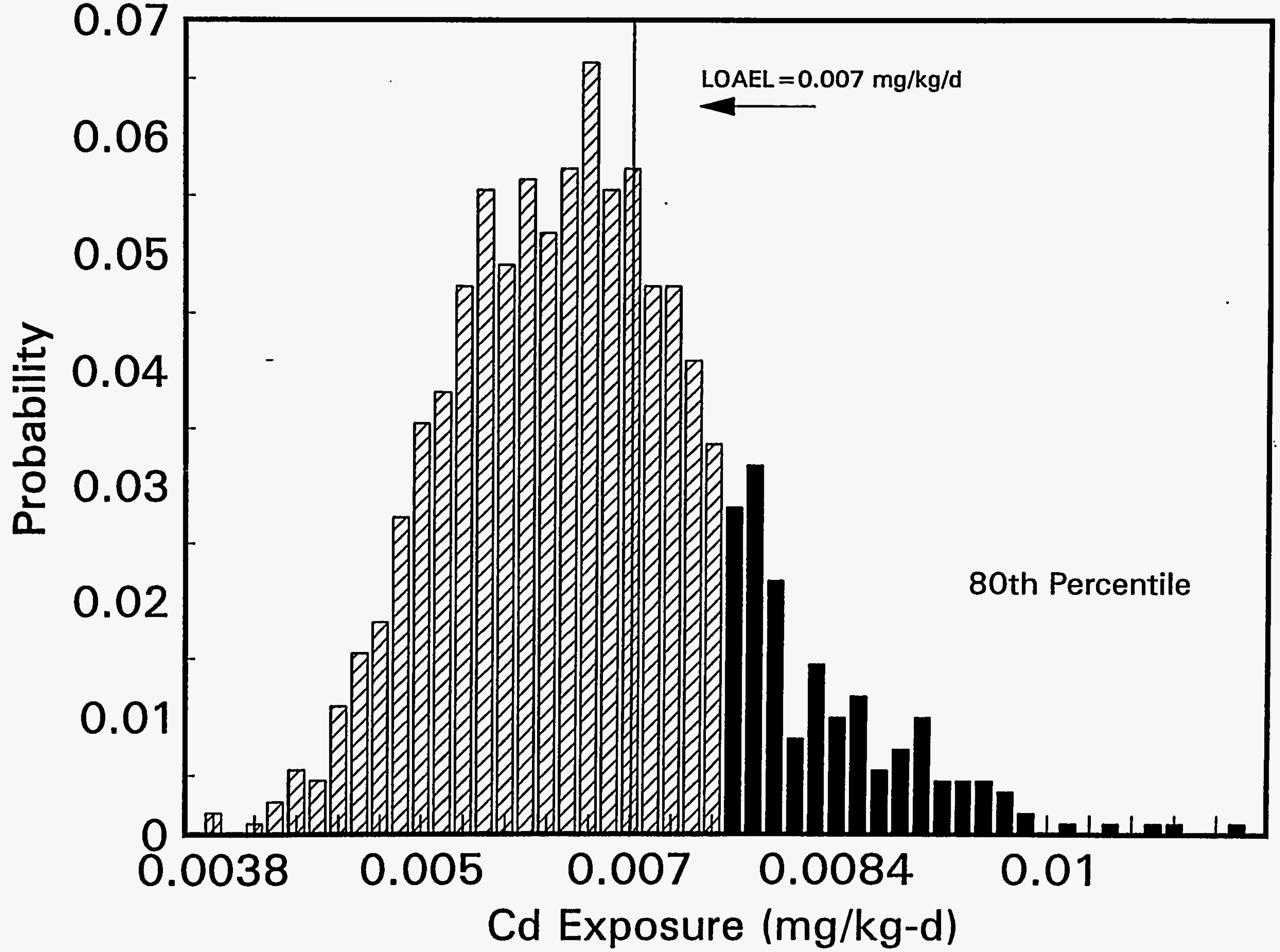

Fig. 6.40. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within reach 7 using Monte Carlo simulation.

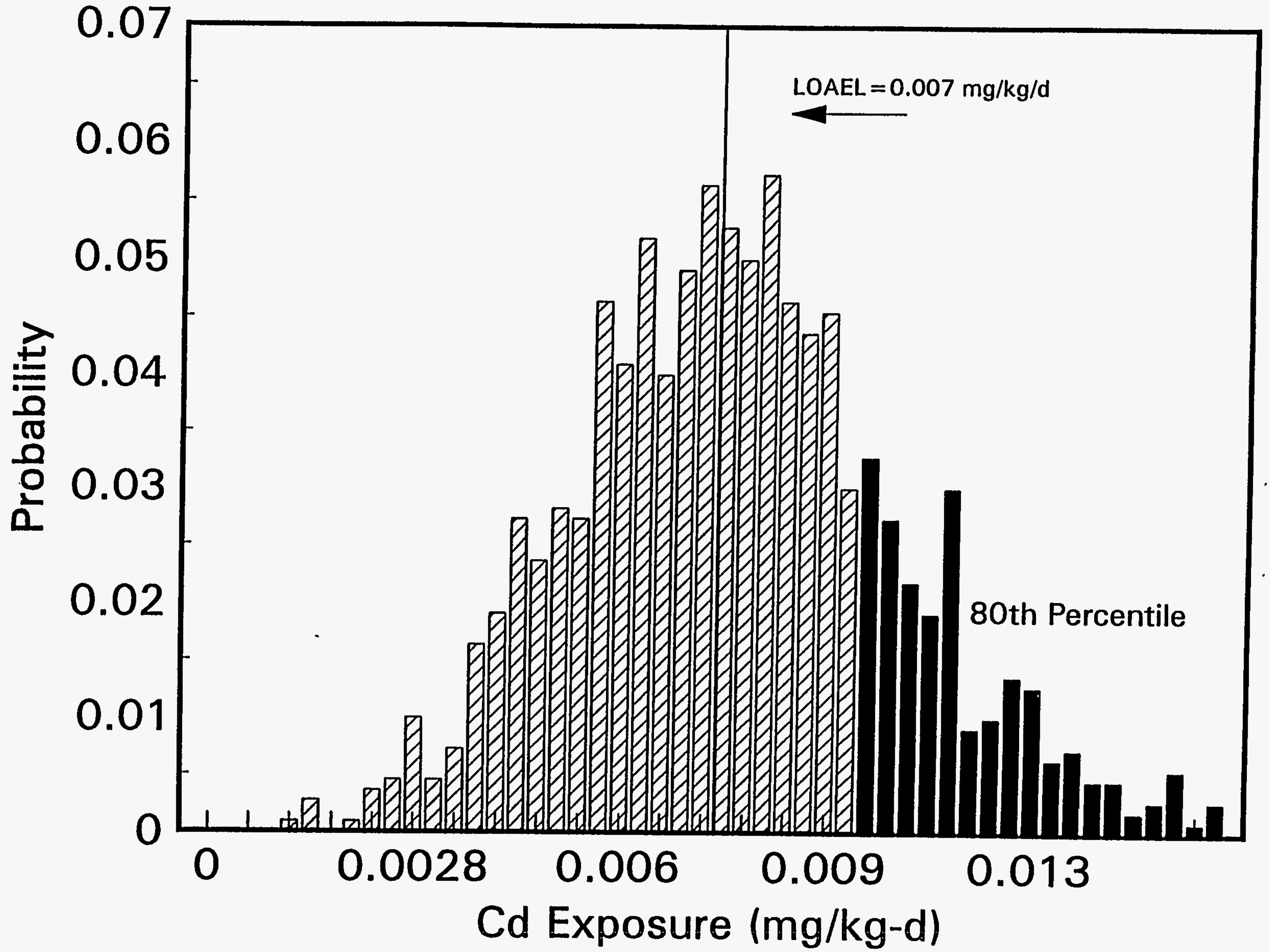

Fig. 6.41. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.02 using Monte Carlo simulation.

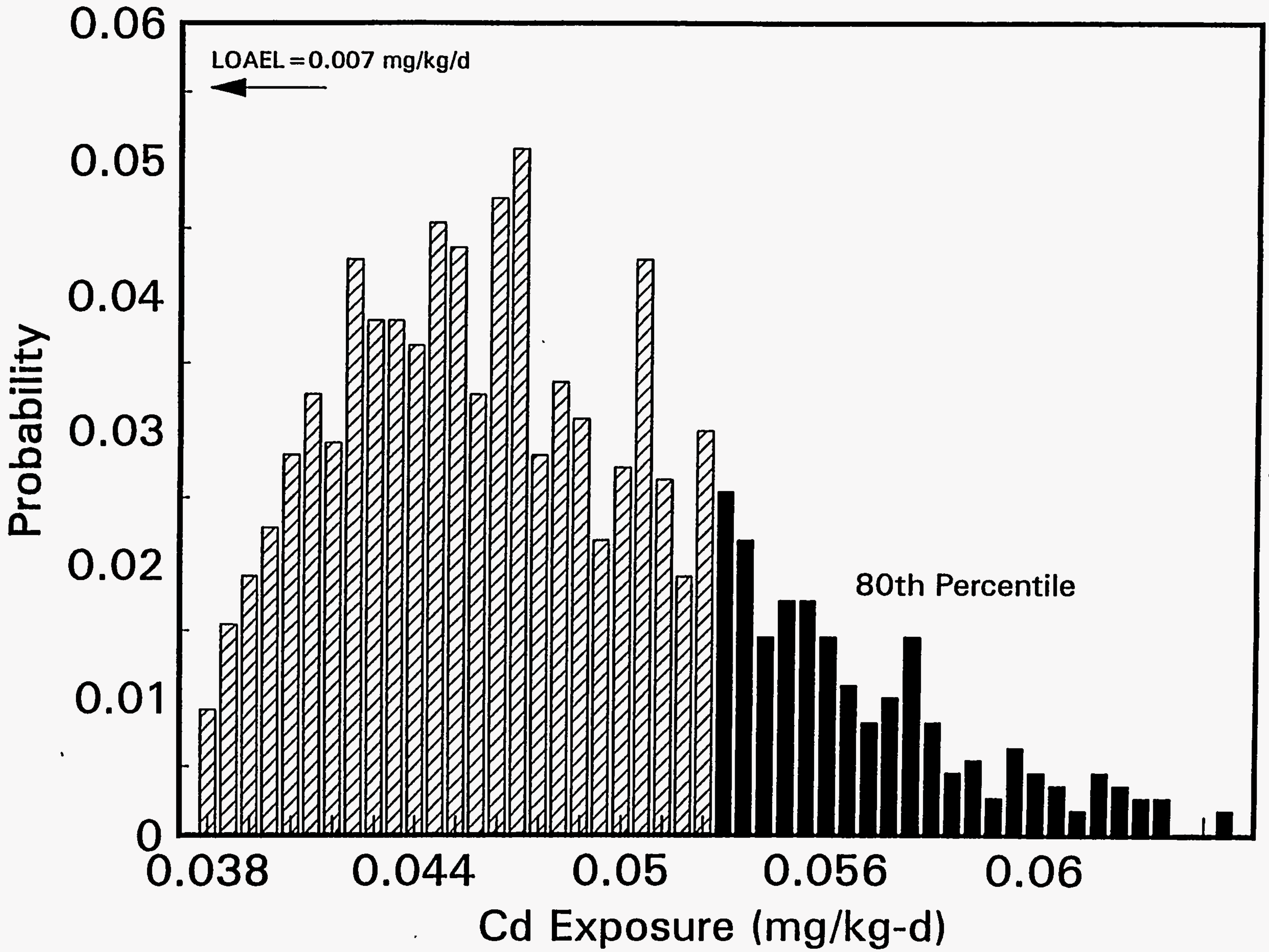

Fig. 6.42. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.03 using Monte Carlo simulation.

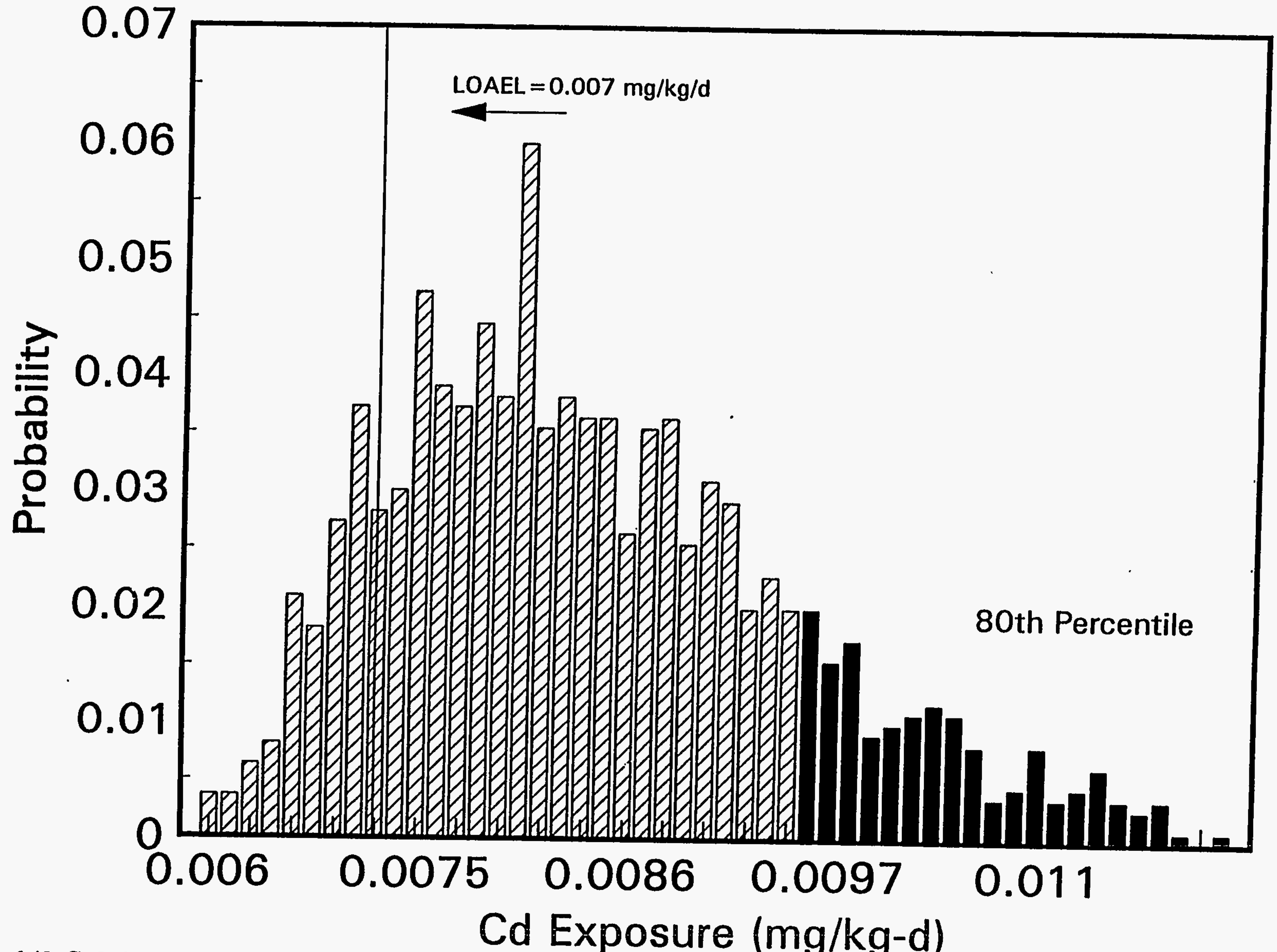

Fig. 6.43. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 4.03 using Monte Carlo simulation.

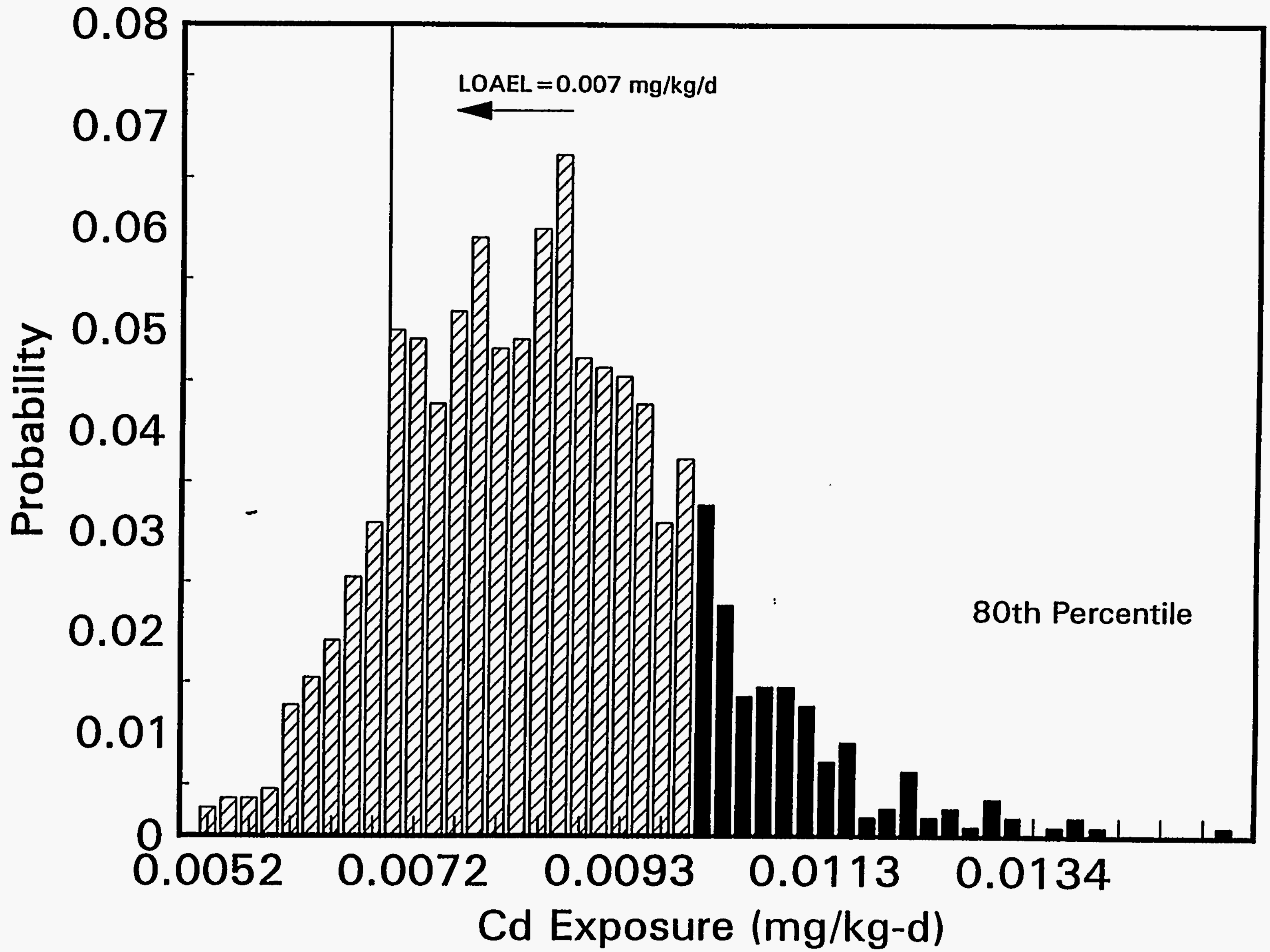

Fig. 6.44. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 4.04 using Monte Carlo simulation.

If there is a risk from arsenic exposure to individuals within the population, it would be primarily attributed to direct ingestion of the spoil. Arsenic bioaccumulates in vegetation at relatively low levels (Bioaccumulation factor = 0.004; SAIC 1995). Therefore, the presence of arsenic within the spoil itself would pose a risk through direct ingestion.

Barium. The NOAEL for eastern cottontails is based on a study in which growth, food and water consumption, and hypertension was observed among rats fed barium chloride for 16 months (Perry et al. 1983). Three dose levels were administered in this study. The maximum dose (5.1 mg/kg/d) did not affect growth, food or water consumption and was therefore considered to be a chronic NOAEL. The LOAEL was based on a study which observed mortality in rats fed barium for 10 days (Borzelleca et al. 1988). Four doses were administered and exposure of rats to the highest dose (300 mg/kg/d) resulted in 30% mortality to female rats. The 300 mg/kg/d dose is considered to be a subchronic LOAEL; therefore a chronic LOAEL was estimated by multiplying the subchronic LOAEL by a subchronic to chronic uncertainty factor of 0.1. Less than 10% of the population of cottontails feeding within subreach 3.04 may experience exposures in excess of the NOAEL. Therefore, adverse effects from barium at this location is highly unlikely.

Cadmium. Both the NOAEL and LOAEL for cottontails are based upon a study in which reproductive success was observed among rats fed cadmium for four generations (Wills et al. 1981). Three dose levels were administered in the study. A dose of 0.01 mg/kg/d, designated as the chronic LOAEL, resulted in a 63% reduction in fertility (number litters/number of females). The lowest dose (0.008 mg/kg/d) resulted in no adverse effects and was thus designated the chronic NOAEL. The NOAEL and LOAEL for cottontails are 0.005 and 0.007 mg/kg/d, respectively. Based on results of Wills et al. (1981), cottontails foraging at a spoil within reaches 1 and 7 or subreaches 3.02, 3.03, 3.04, 4.03, and 4.04 may experience a reduction in fertility. The exposure experienced by cottontails foraging within subreach 2.04 is between the NOAEL and LOAEL. Therefore, there is a potential that *individuals* foraging within this subreach may experience impaired reproduction.

Manganese. Both the NOAEL and LOAEL for eastern cottontails are based upon a study in which reproductive success was observed among rats fed manganese for 224 days through gestation (Laskey et al. 1982). Three doses were administered (28, 88, and 280 mg/kg/d) in the study. Pregnancy percentage and fertility among rats exposed to 280 mg/kg/d was significantly reduced; therefore, this dosage was considered the chronic LOAEL. No effects on other reproductive parameters (e.g., litter size, ovulations, resorptions, preimplantation death, fetal weights) were observed at lower dosages. Therefore, the 88 mg/kg/d dosage level was considered the NOAEL. The NOAEL and LOAEL for eastern cottontails are 58.6 and 189 mg/kg/d, respectively. The exposure experienced by cottontails foraging within subreach 2.04 is between the NOAEL and LOAEL. Therefore, there is a potential that *individuals* foraging within this subreach may experience impaired reproduction.

Mercury. Both the NOAEL and LOAEL for eastern cottontails are based upon a study in which reproductive success and offspring survival was observed among rats fed methyl mercury for three generations (Verschuuren et al. 1976a, 1976b, 1976c). The highest dose administered (0.16 mg/kg/d), designated as the LOAEL, resulted in reduction in offspring viability. This exposure also resulted in reduction in growth, increased kidney weight, and altered kidney histochemistry (Verschuuren et al. 1976b). No effects were observed at a dose of 0.032 mg/kg/d. The study was considered to represent chronic exposure; therefore, a subchronic-chronic correction factor was not employed. The NOAEL and LOAEL for eastern cottontails are 0.021 and 0.110 mg methyl Hg/kg/d, respectively. Based on the results of Verschuuren et al (1976a,b,c), eastern cottontails experiencing exposure at spoils within subreach 3.03, 3.04, and 4.01 are likely to display impaired reproduction. Additionally, because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individuals may become evident from foraging within subreaches 3.02, 4.03, and 4.04 cannot be defined.

PCBs. Both the NOAEL and LOAEL for eastern cottontails are based on a study in which oldfield mice were fed Aroclor-1254 for 12 months (McCoy et al. 1995). A dose level of 0.68 mg/kg/d, designated as the chronic LOAEL, caused a reduction in the number of litters, offspring weights, and offspring survival. Because an experimental NOAEL was not established, the chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL uncertainty factor of 0.1. The NOAEL and LOAEL for cottontails are 0.016 and 0.16 mg/kg/d, respectively. Approximately 20% of the cottontail population may experience exposures in excess of the NOAEL. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individuals may become evident cannot be defined. However, population-level adverse effects from foraging on a spoil in subreach 4.01 are highly unlikely.

Selenium. The eastern cottontail NOAEL and LOAEL are based on a study in which selenate was fed to mice for 3 generations (Schroeder and Mitchner 1971). An administered dose level of 0.76 mg/kg/d, designated as the chronic LOAEL, caused reduced reproductive success with a high incidence of runts and failure to breed. Because an experimental NOAEL was not established, the chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL uncertainty factor of 0.1. The NOAEL and LOAEL for cottontails are 0.023 and 0.22 mg/kg/d, respectively. Cottontails foraging at spoils in reach 7 and in subreaches 2.04 and 3.04 may experience exposures between the NOAEL and LOAEL. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individuals may become evident cannot be defined. However, population-level adverse effects from foraging on these future spoils in subreach 4.01 are highly unlikely.

Thallium. The eastern cottontail NOAEL and LOAEL are based on a study in which rats were fed thallium sulfate for 60 days (Formigli et al. 1986). This study represents subchronic exposures since the duration of the study did not include a critical lifestage. Rats exposed to a single dose, 0.074 mg/kg/d, displayed reduced sperm motility. Since this is a subchronic exposure, a subchronic-chronic uncertainty factor of 0.1 was applied to obtain a chronic LOAEL. To estimate the chronic NOAEL, the chronic LOAEL was multiplied by a LOAEL-NOAEL uncertainty factor of 0.1. The NOAEL and LOAELs for cottontails are 0.005 and 0.05 mg/kg/d, respectively. Cottontails foraging at a spoil in subreach 3.04 may experience exposures between the NOAEL and LOAEL. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individuals may become evident cannot be defined. However, population-level adverse effects from foraging at subreach 3.04 is highly unlikely.

Vanadium. The eastern cottontail NOAEL and LOAEL are based on a study in which rats were fed sodium metavanadate for 60 days prior to gestation, through gestation, delivery and lactation (Domingo et al. 1986). This study represents chronic exposures since it took place during the rat's critical lifestage. Significant effects on reproduction including increased number dead young/litter and reduction in size and weight of offspring were observed at the lowest dose administered, 5 mg/kg/d. Therefore, this dose was considered the chronic LOAEL. To estimate the chronic NOAEL, the chronic LOAEL was multiplied by a LOAEL-NOAEL uncertainty factor of 0.1. The NOAEL and LOAEL for eastern cottontail are 0.13 and 1.3 mg/kg/d, respectively. Cottontails foraging at spoils in reach 7 and subreach 3.04 may experience exposures between the NOAEL and LOAEL. Because an experimental NOAEL was not established, the nature and exposure level at which adverse effects to individuals may become evident cannot be defined. However, population-level adverse effects from foraging in these areas is highly unlikely.

6.6.3.4 Weight of evidence for the dredged spoil

Plants. Only one line of evidence, literature toxicity data, was available to evaluate ecological risk to plants. Many contaminants found within sediments potentially available in future spoils exceeded background concentrations and phytotoxicity benchmarks within all reaches and subreaches (reaches 1 and 7 and subreaches 2.04, 3.02, 3.03, 3.04, 4.01, 4.02, 4.03, and 4.04) with the exception of sediment within subreach 3.01. Therefore, if sediments were dredged and placed along the Clinch River, 1 contaminant within reach 1 (As); 5 contaminants within reach 7 (As, Cr, Mn, Se, and V); 3 contaminants within subreach 2.04 (As, Mn, and Se); 5 contaminants within subreach 3.02 (Hg, Ni, Se, U, and Zn); 7 contaminants within subreach 3.03 (Cd, Cr, Hg, Ni, Ag, U, and Zn); 9 contaminants within subreach 3.04 (As, B, Cd, Cr, Hg, Ni, Se, U, and V); 1 contaminant within subreach 4.01 (Hg); 1 contaminant in subreach 4.02 (As); 3 contaminants within subreach 4.03 (Cr, Mn, and Hg); and 5 contaminants within subreach 4.04 (As, Cr, Mn, Hg, and Zn) could potentially pose a risk to plants growing in these spoils. There are also 37 organic contaminants present in the sediment for which no phytotoxicity benchmarks are available. Therefore, the risks from those contaminants cannot be assessed.

This scenario is an extremely conservative estimation of risk for plants in the future. It is assumed that the concentrations found in the current deep sediments would be equivalent to spoil concentrations. Additionally, the contaminants are assumed to be 100% bioavailable for plant uptake in the future spoil. Since contaminants may not be totally bioavailable to plants, the risk to plant growth posed by these contaminants may be lower than suggested.

Herbivorous wildlife. Only one line of evidence, literature toxicity data, was available to evaluate risks to eastern cottontails foraging on spoils produced within each subreach. The comparison of exposure estimates to LOAELs indicate that reaches 1 and 7 and that subreaches 3.02, 3.03, 3.04, 4.01, 4.03, and 4.04 would be a significant risk to eastern cottontail populations foraging on future spoils. The risk is attributable to cadmium and mercury (at subreaches 3.03, 3.04 and 4.01 only), which may be ingested directly from the spoil or from ingestion of contaminated vegetation. The eastern cottontail population feeding on these spoils may be adversely impacted through impaired reproduction.

The remaining contaminants found in the spoil and vegetation within reaches 1 (As) and 7 (As, Se, Tl, and V) and subreaches 2.04 (as, Mn, and Se), 3.02 (Hg), 3.04 (Ba, Se, Tl, and V), 4.01 (As and Aroclor 1260), 4.02 (As), 4.03 (As and Hg), and 4.04 (As and Hg) did not produce exposures in excess of the LOAELs. Therefore, *population-level* effects on the eastern cottontails are not likely to be observed from ingestion of these contaminants on the above spoils. However, *individuals* may be adversely affected at the uppermost exposure levels of the distributions.

6.6.4 Uncertainties Concerning Risks to Plants and Herbivorous Wildlife

Factors that create uncertainty in assessing the risk posed by the COPCs in the future spoils of CR/PC sediments are discussed in the following subsections.

6.6.4.1 Equivalent sediment and spoil concentrations

The scenario assumes that the current contaminant concentrations found in sediments in the CR/PC will be equivalent to the potential spoil deposited on land. The contaminant concentrations will remain present within the spoil over time and be taken up by vegetation. This is an extremely conservative scenario which may overestimate the level of exposure and risk to the flora and fauna living at the future spoil.

International Commission on Radiological Protection and the National Council on Radiation Protection and Measurements (IAEA 1976, 1979, NCRP 1991). For terrestrial organisms the methodology developed by Soldat et al. (1974) and Baker and Soldat (1992) was modified to calculate radiation dose to terrestrial biota. Equations used to estimate doses to biota in this investigation are listed in Appendix F7.

Since dose calculations are species-specific, representative species were selected for the end point communities. They are: the mayfly *Hexagenia bilineata*, the epibenthic fish, which represents fish such as carp (*Cyprinus*), redhorse and suckers (*Moxastoma*), and catfish, (*Icthlurus*) the great blue heron (*Ardea herodias*), and the eastern cottontail (*Sylvilagus floridanus*).

6.7.1.2 Radiation dose to benthic invertebrates

Sediments are the environmental medium with the highest radionuclide concentrations. Invertebrate biota which live most of their life cycle in sediments probably represent a maximum exposure scenario. The mayfly, *Hexagenia bilineata*, is chosen as a representative benthic invertebrate. This large sediment-burrowing mayfly is common to the Clinch River, Poplar Creek, and Watts Bar Reservoir. In these waters, it usually takes one year for mayflies to complete their life cycle, which extends from deposition of eggs to hatching of adults. Nymphs are an important food source for fish. Adults hatch during summer months, but almost all adults die within one day after hatching and leaving the water. During that short time they attract fish, birds and mammals which gorge on the hatch.

Annual radiation doses were calculated for mayfly larvae completely submerged in sediments from each reach of the OU (Table 6.33). The equations for the calculation appear in Appendix F7. Sediment concentrations of radionuclides appear in Tables B16–B21. Continual, complete submergence was chosen as a conservative scenario, maximizing the estimated dose. Wright et al. (1982) reported that nymphs from Watts Bar Reservoir stock range in size from 20 to 25 mm when they have reached the large wing-pad stage of development. This size range was taken into consideration in the dosimetry calculations. The highest calculated dose rates are 490 and 280 mrad/year for larvae living in sediments sampled from reaches 2 and 4, respectively. The principal radionuclide contributing to dose rate at these locations is ^{137}Cs and its daughter product, ^{137m}Ba. The next highest dose rate was 200 mrad/year for larvae living in sediments at reach 1, with 92 percent of the dose coming from ^{60}Co. The dose rate to larvae at reach 3 was slightly less at 180 mrad/year and was due mostly to the presence of ^{137}Cs in sediments. Larvae at reach 7 were estimated to receive an annual dose of 31 mrad, less than for any other reach.

Table 6.33. Radiation dose rates (mrad/year) to mayfly larvae (*Hexagenia bilineata*) living in sediments

Reach	Number of radionuclides detected	Principal radionuclide and contribution to dose rate (%)	Dose rate (mrad/year)
1	2	^{60}Co (92%), $^{137}Cs/^{137m}Ba$ (8%)	200
2	14	$^{137}Cs/^{137m}Ba$ (93%), $^{90}Sr/^{90}Y$ (5%)	490
3	13	$^{137}Cs/^{137m}Ba$ (89%), ^{235}U (5%)	180
4	13	$^{137}Cs/^{137m}Ba$ (91%), ^{60}Co (5%)	280
7	3	$^{90}Sr/^{90}Y$ (79%), $^{137}Cs/^{137m}Ba$ (31%)	31

6.7.1.3 Radiation dose to epibenthic fish

A model epibenthic fish was used to estimate the total dose received from exposure to radionuclides in sediments, water, and internal tissues. Radionuclide concentration data are found in Tables B16–B21 (sediment) and Appendix A (surface water). For internal tissue concentrations, general bioaccumulation factors were used instead of actual measurements. Representative epibenthic fish include carp (*Cyprinus*), suckers and redhorse (*Moxastoma*), and catfish (*Ictalurus* and *Ameiurus*) which are common at all sampling locations. Detailed dosimetry equations applicable to a model, large (length = 45 cm) epibenthic fish are listed as equations 3 to 7 in Appendix F7. These equations were adapted from Blaylock et al. (1993), and used to calculate the annual dose rate to a model epibenthic fish as the sum of gamma dose rates from water immersion and sediment surface, and beta and gamma dose rates from internally deposited radionuclides. The beta component of immersion in water and exposure to sediment surfaces was neglected because of self absorption and shielding.

The calculated dose rates (mrad/year) are listed in Table 6.34 and indicate that the doses to fish are <100 mrad/year, or approximately a factor of ten less than the doses calculated for mayflies. The principal radionuclides contributing to dose rate are $^{137}Cs/^{137m}Ba$, $^{90}Sr/^{90}Y$, and ^{60}Co.

Table 6.34. Radiation dose rates (mrad/year) to epibenthic fish from exposure to radioactive sediments, water, and food

Reach	Number of radionuclides detected	Principal radionuclides and contribution to dose rate (%)	Dose rate (mrad/year)
1	6	^{60}Co (91%), $^{90}Sr/^{90}Y$ (7%)	32
2	16	$^{90}Sr/^{90}Y$ (52%), $^{137}Cs/^{137m}Ba$ (36%)	33
3	14	$^{137}Cs/^{137m}Ba$ (84%), $^{90}Sr/^{90}Y$ (10%)	56
4	15	$^{137}Cs/^{137m}Ba$ (76%), ^{60}Co (13%)	22
7	3	$^{90}Sr/^{90}Y$ (61%), $^{137}Cs/^{137m}Ba$ (26%)	5

6.7.1.4 Radiation dose to great blue heron

The great blue heron (*Ardea herodias*) was chosen to represent fish-eating birds. Biological and dosimetric parameters describing the model heron are given in Appendix F7. The total dose received by a heron would be the sum of external exposures from contaminated shoreline, external exposure from wading in contaminated water, and internal exposure from eating contaminated fish. Measurements of radionuclide concentrations in fish tissues were used from Phase 1 of the Clinch River Remedial Investigation (Cook et al. 1992). The model assumes that 100 percent of the great blue heron's diet is composed of fish. Equations 9 to 11 in Appendix F7 were used to perform the dosimetry calculations.

Although up to 15 parent radionuclides contribute to radiological exposures, ^{137}Cs and ^{90}Sr (including daughter products) contribute almost all of the dose rate at any one reach (Table 6.35). The highest dose rate of 5400 mrad/year occurs for a heron feeding at reach 2. This was the highest dose determined for any of the representative biota. Dose rates for reaches 3 and 4 are more than a factor of ten times less, and ^{90}Sr is the principal radionuclide.

Table 6.35. Radiation dose rates (mrad/year) to great blue heron from radioactive contamination in fish, water, and sediments

Reach	Number of radionuclides detected	Principal radionuclides and contribution to dose rate (%)	Dose rate (mrad/year)
1	6	^{137}Cs/^{137m}Ba (~100%)	23
2	15	^{137}Cs/^{137m}Ba (~100%)	5400
3	14	^{90}Sr/^{90}Y (79%), ^{137}Cs/^{137m}Ba (21%)	350
4	15	^{90}Sr/^{90}Y (77%), ^{137}Cs/^{137m}Ba (23%)	320
7	3	^{137}Cs/^{137m}Ba (60%), ^{60}Co (33%)	<1

6.7.1.5 Radiation dose to rabbit

A model eastern cottontail rabbit (*Sylvilagus floridanus*) weighing 1.2 kg was assumed to be a resident of a hypothetical terrestrial habitat established on a soil base of dredged sediments at each river reach. The scenario is described in detail in the human health risk assessment (Sect. 5.2.3.6) As in the assessment of nonradiological contaminants in Sect. 6.6, the spoil is assumed to be derived from mixed deep sediments where irrigation will not take place. If deep sediment is taken from the river bed and spread on shore, the organisms are potentially exposed to high concentrations of ^{137}Cs. Vegetation can bioaccumulate contaminants and be consumed by herbivores, such as rabbits. In this assessment actual radio-analytical results for sediments sampled at each reach were used in the calculations. It was assumed that each reclaimed area was large enough for a rabbit to live its entire life cycle.

The exposure scenario of dosimetric and biological assumptions for calculating rabbit exposures should lead to conservative dose estimates (over estimation of actual doses if the sediments were used to establish new terrestrial habitats). Conservative assumptions include: (1) the residence of the rabbit in a burrow for half of the time and (2) high inhalation and ingestion rates. Both internal and external exposures were assumed to occur for the model rabbit which resides in a burrow one-half of the time and spends the other half of its time above ground. Both of these situations result in external exposure from the contaminated soil and sediment mixture. All vegetation which is consumed as food becomes radioactive through soil-root uptake. Additional internal exposure is assumed to occur through inhalation of sediment/soil-derived dust which has the same radionuclide content as the dredged sediments. Assumed intake rates for ingestion of fresh vegetation and inhalation of air are 0.24 kg/day and 0.63 m^3/day, respectively (EPA 1993g). A mass loading rate of 0.0001 kg/m^3 was used for the inhalation pathway. Equations for the dose rate calculations and further assumptions are given in Appendix F7.

Dose rates computed for the four pathways of exposure are listed in Table 6.36 and range from a minimum of 11 mrad/year at reach 1, to maxima of 1500 mrad/year and 1400 mrad/year at reaches 2 and 4, respectively. At reach 3 the dose rate is 530 mrad/year, and the dose rate to the rabbit at reach 7 is 90 mrad/year. The principal radionuclides contributing to the estimated dose rates include ^{137}Cs/^{137m}Ba and ^{90}Sr/^{90}Y.

Table 6.36. Radiation dose rates (mrad/year) to rabbit from vegetation consumed, air breathed, soil surface, and burrow

Reach	Number of radionuclides detected	Principal radionuclides and contribution to dose rate (%)	Dose rate (mrad/year)
1	2	$^{137}Cs/^{137m}Ba$ (96%), ^{60}Co (4%)	11
2	5	$^{137}Cs/^{137m}Ba$ (90%), $^{90}Sr/^{90}Y$ (8%)	1500
3	7	$^{137}Cs/^{137m}Ba$ (78%), $^{90}Sr/^{90}Y$ (17%)	530
4	6	$^{137}Cs/^{137m}Ba$ (98%), $^{90}Sr/^{90}Y$ (2%)	1400
7	2	$^{90}Sr/^{90}Y$ (80%), $^{137}Cs/^{137m}Ba$ (20%)	90

6.7.2 Effects Levels for Radionuclides

The discharge of radioactive waste into the environment results in long-term, low dose exposure to organisms. In most cases, acute mortality can be discounted. Any potential increase in morbidity and mortality that might result from the exposure to chronic irradiation above background is unlikely to be detected because of natural fluctuations in the size of populations. The DOE's official guideline for radiation received from environmental sources limits the dose rate to 1.0 rad per day (NCRP 1991) for aquatic organisms. The International Atomic Energy Agency (IAEA) recommends limiting the dose for terrestrial organisms to 0.1 rad per day (IAEA 1992).

6.7.3 Risk Characterization for Organisms Exposed to Radionuclides

A brief description of the life cycle, size of the organism, and methodology used in the dose calculations is given in Sect. 6.7.1. Based on the relatively low doses calculated for representative biota (Sect. 6.7.1) and the recommended limit of 1.0 rad per day for aquatic biota (NCRP 1991) and 0.1 rad per day for terrestrial biota (IAEA 19922), detectable radiation effects on individuals or populations of organisms are not expected in the biota of the CR/PC.

More specifically, (1) the highest radiation dose rate for benthic invertebrates is over three orders of magnitude lower than the DOE guideline of 1.0 rad per day (Table 6.33). Therefore, no significant effects would be expected to result to the abundant populations of this important benthic invertebrate species. (2) All dose rates for epibenthic fish (Table 6.34) are far below any level that would be expected to produce bioeffects and are therefore of no radiological concern. (3) Even at the highest estimated dose rate of 5400 mrad/year (5.4 rad/year) for herons, no bioeffects would be expected (Table 6.35). (4) Sincethe maximuml annual dose rates for rabbits is calculated to be only 1.5 rad/year, no biological effects would be expected (Table 6.36).

Results of the specific dose calculations for these representative organisms are discussed in Sect. 6.7.1. Several details are worth noting. A dose of 5.4 rad per year which was calculated for the great blue heron in reach 2 was the highest dose determined for any of the representative biota. For all representative biota, ^{137}Cs and its daughter radionuclide ^{137m}Ba were the major dose-contributing radionuclides. All representative biota in reach 2 received the highest calculated radiation doses, except for the epibenthic fish which received the highest calculated dose from reach 3.

6.7.4 Uncertainties Concerning Risks from Radionuclides

The methodology used to estimate radiation doses to biota is believed to overestimate the doses that would be received if the exposure scenarios actually existed. Whereas some information necessary to implement the methodology is well known, so much is unknown or unspecified statistically. This paucity of information dictates a conservative, but reasonable, approach to model assumptions and radiological exposure scenarios in order not to underestimate the risk to biota.

Factors used in the dosimetry methodology can be divided into three categories: (1) biological information including life history parameters of the organism and characteristics of the environment in which it lives, (2) analytical measurements of radionuclides in media which expose the organism, and (3) dosimetry parameters used to translate the radionuclide concentrations in organism and environment into the quantity of energy absorbed (dose) by the biota. These factors are discussed below in the context of their major uncertainties as they apply to the four representative biota for which radiation doses were calculated.

6.7.4.1 Radiation dose to mayflies

The dosimetry exposure model for benthic invertebrates assumed that mayfly nymphs spend one year developing in radioactive sediments before before leaving sediments and water to become free-flying adults. Unusually high water temperatures could hasten development and thus reduce exposure to radioactive sediments. Unfavorable conditions might, instead, prolong exposure to sediments if growth and development are retarded. A reasonable upper limit might be two years of exposure in radioactive sediments and a resultant doubling of the dose compared to the model assumptions. Radionuclide concentrations are known to vary appreciably in the Clinch River with highest concentrations in the main channel and lower concentrations in shallower and shoreline areas. Edmonds et al. (1976) stated that mayflies are most abundant in shallow sediments but are found in lesser numbers down to maximum depths of about eighteen meters. Since the radionuclide concentrations used in the calculations are not for deep water sediments, the assumed exposures would probably not be exceeded for any appreciable part of the population. However, a number of simplifying assumptions for the mayfly dosimetry model are not easily verified and thus there could exist actual conditions that could lead to higher doses than calculated.

6.7.4.2 Radiation dose to epibenthic fish

Generic bioaccumulation factors were used instead of actual measured concentrations in fish flesh. It is unknown how the generic factors compare to actual bioaccumulation factors for the different fish. Each radionuclide represents an independent statistic, so some degree of uncertainty is associated with the overall effect on total dose rate. This uncertainty could be quantified by comparing results from the use of bioaccumulation factors to radioanalytical results for the flesh of epibenthic fish from the reference locations. A dominating conservative assumption in the dosimetry model is that the epibenthic fish is always in close proximity to the contaminated sediments. In reality, whenever the fish is swimming more than 60 to 100 cm above the bottom sediments the relatively non-contaminated water would act as a shield in reducing or eliminating the radiation dose from bottom sediments. The extent to which this dose-reducing swimming behavior occurs is unknown.

6.7.4.3 Radiation dose to great blue heron

Almost all of the radiation dose received by the model great blue heron comes from consumption of contaminated fish. A fish consumption rate of 0.42 kg/day (EPA 1993g) was assumed for this important exposure pathway. Also, it was assumed that 100 percent of the diet was composed of radioactively-contaminated fish. The calculated annual dose rate to the great blue heron would increase or decrease proportionally for any deviation from this food intake rate, and any consumption of non-contaminated fish would decrease the radiation dose rate proportionally. Modelling assumptions required that the daily time used to carry out normal activities be apportioned in order to estimate external radiation doses from different activities. Thus, one-third of the time was assumed for exposure to contaminated sediments, one-third of the time for external exposure to contaminated water, and one-third of the time for no external exposure. These assumptions were also used by Baker and Soldat (1992), but their choice of partitioning of time was arbitrary also.

6.7.4.4 Radiation dose to eastern cottontail rabbit

In the absence of documented observations, it was assumed that the model rabbit spent one-half of the time in a burrow and the other one-half of the time above ground. The radiological dosimetry assumptions result in a dose rate from irradiation in a burrow constructed in radioactive deep dredge spoils that is double the dose when the rabbit is on the land surface. Any deviations from these simplifying exposure assumptions would change the dose rates proportionately. The dose rate calculations revealed that the predominant pathway of exposure to the model rabbit was from direct intake of contaminated sediments, especially through inhalation of spoils-derived dust. Because of the lack of actual measurements on the quantities of spoils-derived dust expected to be inhaled, a conservative estimate of mass loading was made for the air breathed by the model rabbit. A mass loading rate of 0.0001 kg/m^3 was chosen for spoils-derived contamination of air breathed by the rabbit. While considerable uncertainty exists as to what the actual value might be for a rabbit living under the scenario developed for the dosimetry model, it is probably not reasonable to assume a larger sustained mass loading factor because of physical limitations. The fact that the inhalation pathway dominates estimated radiological exposures to the rabbit is most likely an indication of the conservative nature of the mass loading rate assumption.

6.8 SUMMARY AND CONCLUSIONS

Significant ecological risks were identified in the Poplar Creek embayment (reach 3) but not in the Clinch River. Although potentially toxic concentrations of chemicals, significant toxicity test results, or elevated levels of bioindicators of contaminant exposure in fish were observed in other reaches, only in Poplar Creek was the weight of evidence consistent with significant risks.

The weight of evidence indicates that toxic effects are causing a significant risk of a 20% reduction in fish species richness and abundance. Reach 3 water was toxic to fish embryos and concentations of dissolved metals episodically reached toxic levels. In addition, although PCBs were detected on only one occasion in reach 3, PCB concentrations in whole catfish were estimated to reach concentrations that are toxic to that species in the laboratory. That result is consistent with the discovery of liver histopathologies in largemouth bass that are characteristic of PCB exposure. Fish species richness and abundance were relatively low in upper reach 3, which is consistent with toxic effects, but may also be due to habitat factors.

Although risks to benthic invertebrates in Poplar Creek are not high and the evidence is not consistent, the weight of evidence indicates that toxic effects are causing a significant risk of a 20% reduction in benthic invertebrate species richness and abundance. Reach 3 sediments contained several metals and organic chemicals at concentrations that have been reported to be toxic in most studies at other sites. Sediment pore water and water above sediments were not found to be toxic, but some whole sediment samples from all subreaches of reach 3 were acutely lethal to an amphipod. That result suggests that sediment toxicity is highly heterogeneous and is due to chemicals that are highly associated with the solid phase. Benthic community characteristics were highly variable among sites and could not be used to imply or refute the occurrence of a 20% reduction in species richness or abundance.

Risks to fish-eating wildlife inhabiting Poplar Creek embayment are estimated to be insignificant. Great blue heron and osprey reproduction are high and mink fed 50% Poplar Creek fish or less displayed no toxic effects. Modelled exposures and effects were consistent with these results.

Risks to bats inhabiting Poplar Creek embayment are estimated to be insignificant, but swallows were estimated to be significantly at risk of a 20% reduction in population production. The risks to swallows are based on exposure of a colony on subreach 3.01 to mercury in emergent aquatic insects.

The risk to cottontail rabbits foraging on a hypothetical area consisting of dredge spoil from Poplar Creek was estimated to be significant. The risk was due entirely to mercury.

6.9 REMEDIAL GOAL OPTIONS FOR ECOLOGICAL END POINTS

6.9.1 Remedial Goal Options for the Fish Community

Fish are exposed to contaminants primarily by direct exposure to contaminated water. The CR/PC OU is not a potential source of aqueous contaminants unless contaminants are being remobilized from sediments. However, that does not appear to be occurring to a significant extent. Therefore, no actions would be taken in the CR/PC OU to remediate water. However, since significant toxic effects are occurring in Poplar Creek (reach 3) and since the ORR is a unitary CERCLA site, it is necessary to consider what actions are needed in upstream OUs to remediate the risks to the fish community in that reach. No RGOs are proposed for other reaches because the risks to the fish communities in those reaches were judged to be acceptable.

Episodically high concentrations of chemicals were observed in Poplar Creek that exceeded National Ambient Water Quality Criteria or other ecotoxicological benchmarks. Actions should be taken to ensure that dissolved-phase concentrations do not exceed acutely lethal levels at any time or chronically toxic levels during periods when fish are spawning in the embayment. These levels are listed in Table 6.37 for those chemicals that were observed at excessive concentrations in reach 3 during the RI.

The clearest indications of toxic effects on fish are the results of the toxicity tests with medaka and redbreast sunfish. However, because of limitations of the water sampling and analysis and of the available toxicity data base for fish embryos, it is not clear what chemicals are causing the toxicity. Before taking remedial actions, it would be useful to perform a toxicity identification and evaluation (TIE) for the medaka test following the EPA methodology. The TIE should indicate which chemicals are causing the toxicity to fish embryos and at what concentrations, thereby indicating what remedial actions are needed. Although the specific chemicals causing toxicity are not known, the relatively high toxicity to fish embryos of water from lower Mitchel Branch and East Fork Poplar Creek suggests that they are contributing sources (Hinzman 1996; Kszos et al. 1994; Kszos et al. 1995; Peterson et al. 1995).

Table 6.37. Remedial goal options for reducing risks to fish from contaminants in water (mg/L)

Chemical	Acute remedial goal options	Chronic remedial goal options
Copper	0.018[a]	0.012[b]
Mercury	0.0024[a]	0.0002[c]
Nickel	1.4[a]	0.011[d]
Silver	0.0041[a]	0.00012[c]

[a]Acute National ambient water quality criteria.
[b]Chronic National ambient water quality criteria.
[c]Chronic value for fathead minnow.
[d]Chronic value for *Ceriodaphnia dubia*.

6.9.2 Remedial Goal Options for the Benthic Invertebrate Community

Benthic invertebrates are exposed to contaminants primarily be direct exposure to contaminated sediment and sediment pore water. The weight of evidence indicates that Poplar Creek (reach 3) is the only reach in which contaminated sediments are a significant risk to the benthic invertebrate community. Therefore, RGOs are proposed for reach 3 only.

RGOs were developed for chemicals that were characterized as presenting a significant risk in reach 3. However, RGOs were not developed for chemicals that present a greater risk above the OU (reach 13) than in reach 3 (e.g., 2-methylnaphthalene, acenaphthene, and total PAHs). Actions should be taken to ensure that chemicals in whole sediment do not occur above concentrations at which effects are expected (Table 6.38).

Two estimates of whole sediment concentrations at which effects are expected are the generic probable effects levels from the literature (e.g., ER-Ms) and the site-specific no apparent effects levels (SSNAEL). The generic probable effects level is the lesser of the ER_M and the 50th percentile of the effects distributions presented in Appendix F3. Uncertainties associated with the generic probable effects levels are detailed in Sect. 6.3.3.1 and include the following: the available data are almost entirely from marine and estuarine systems, a wide variety of organisms and effect levels are included in the ER-Ms, and most of the data are from co-occurrence studies which may over estimate the toxicity of individual chemicals. The SSNAEL is the highest measured concentration at which toxicity was never observed at the site. This concentration was determined for those chemicals presenting a significant risk in reach 3 by compiling the measured concentrations of each chemical for all sediment sites in the OU where tocicity tests were conducted (Table F3.3). Only data for Poplar Creek were used for chemicals that may vary between reaches in chemical speciation (i.e., As, Cr, and Hg) or mixture composition (total PCBs). If significant toxicity to *H. azteca* was observed in one or more tests for a site, the individual chemical concentrations at that site were considered potentially toxic. If significant toxicity was never observed at a site, the associated individual chemical concentrations were considered nontoxic. If the highest measureed concentration was not toxic, that concentration is the SSNAEL (e.g., 67 mg/kg chromium). If the highest measureed concentrations are potentially toxic, it must be assumed for these purposes that toxicity is due to that chemical (e.g., 64 and 54 mg/kg arsenic). Thus, the SSNAEL is the next lowest concentration that is nontoxic (e.g., 25 mg/kg arsenic). Toxicity associated with concentrations of an individual chemical that are less than the SSNAEL are assumed to be, at least in part, the result of other meaured or unmeasured contributors to toxicity (e.g., PAHs). The principal uncertainties in the SSNAEL are the same uncertainties associated with the toxicity tests (Sect. 6.3.3.2). Specifically, the sediment samples for toxicity testing and chemical analysis were collected at different times. Thus, the measured concentrations may not reflect the exposures received during toxicity testing. Also, three of the sites were tested only once, during the confirmatory sampling in December of 1995. The recommended RGO for whole sediment is the higher of the generic probable effects level and the SSNAEL. That is, the RGO should not be lower than the site-specific concentrations which were tested and shown to be nontoxic.

Table 6.38. Remedial goal options for reducing risks to benthic invertebrates from contaminants in sediment (mg/kg)

Chemical	Generic probable effect level[a]	Site-specific no apparent effects level[b]	Site-specific remedial goal option
Arsenic	6.2[d]	25	25
Chromium	90	67	90
Mercury	0.15	139	139
Nickel	22	228	228
Silver	0.8	1.3	1.3
PCBs (total)	0.18	0.550	0.550

[a]The generic probable effects level is the lesser of the ER_M and the 50th percentile of the effects distributions presented in Appendix F3.
[b]The site-specific no apparent effects level (SSNAEL) is the highest measured concentration at which toxicity was never observed at the site (Table F3.3)
[c]The site-specific remedial goal option for whole sediment is the higher of the generic probable effects level and the SSNAEL.
[d]Although the 50th percentile of the community level effects distribution was the lowest generic probable effects level for arsenic, it was not used because it was based on only two data points. Rather, the next lowest available generic probable effects level, the 50th percentile of the lethal effects distribution, was used for arsenic.

7. PURPOSE AND ORGANIZATION OF THE FEASIBILITY STUDY

This feasibility study (FS) identifies Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) remedial actions that eliminate, reduce, or control risks to human health and the environment and complies with the National Environmental Policy Act in accordance with DOE Policy. The FS process, as defined in the National Contingency Plan (NCP) [40 *Code of Federal Regulations* (CFR) 300], develops remedies that protect human health and the environment, maintains protection over time, and, to the extent practical, treats waste to reduce mobility, toxicity, or volume. U.S. Environmental Protection Agency (EPA) guidance [*Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA* (Interim Final) EPA 540/5-89/004, Office of Solid Waste and Emergency Response Directive 9355.3-01, October 1988] and the NCP provide criteria for evaluating remedial technologies and alternatives. The primary requirements for the final remedy are protection of human health and the environment and compliance with applicable or relevant and appropriate requirements (ARARs).

Following EPA and Tennessee Department of Environment and Conservation (TDEC) concurrence with the FS, the U.S. Department of Energy (DOE) will prepare a proposed plan for public comment. After consideration of input from the public and regulators, DOE will issue a record of decision (ROD) in which decision makers consider site problems, potential alternatives, and outcomes and select the remedial action that best satisfies statutory goals.

This FS follows the information presented in the remedial investigation (RI), which includes site conditions, nature and extent of contamination, and risks from contaminants. The RI divides the operable unit (OU) into reaches along certain physical boundary points. Because of variability in the contaminant profile, river conditions, and/or risks, subreaches are used to identify areas with consistent characteristics. This FS assembles remedial alternatives that address conditions sitewide, when possible, and address conditions by reach and subreach, as necessary. Section 3.1 describes the study areas in detail and Figure 3.1 shows the location of each reach and subreach.

Based on baseline assessments of human health and ecological risks provided in Chapters 5 and 6, Chapter 8 defines remedial action objectives (RAOs) for the site and identifies pathways of concern and contaminants of concern (COCs). General response actions, potential remedial technologies, and process options are developed and screened for technical applicability. Those general response actions, technologies, and process options are not applicable to site or contaminant-specific conditions are described briefly, but not discussed any further. Those that meet the criteria are carried forward into an evaluation of effectiveness, implementability, and cost, and representative process options are selected.

Chapter 9 combines the retained technologies and process options to develop a range of remedial alternatives consistent with RAOs.

Chapter 10 provides a detailed analysis of each alternative to enable decision makers to choose an appropriate site remedy. Each alternative is evaluated individually on the basis of seven of the nine CERCLA criteria: (1) overall protection of human health and the environment; (2) compliance with ARARs; (3) long-term effectiveness and permanence; (4) reduction of toxicity, mobility, and/or volume of wastes or contaminants through treatment; (5) short-term effectiveness; (6) implementability; and (7) cost. State acceptance and community acceptance will be evaluated as part of the ROD. A comparison among the alternatives, using the same criteria, follows the individual evaluation.

8. IDENTIFICATION AND SCREENING OF TECHNOLOGIES

This chapter identifies, screens, and evaluates technologies and process options. Based on information presented in the RI, RAOs for the OU are developed (Sect. 8.1) and potential response actions, used either independently or in combination with other response actions, are identified.

8.1 REMEDIAL ACTION OBJECTIVES

This section develops RAOs for the CR/PC OU. RAOs are media-specific goals for protecting human health and the environment. Objectives outlined in this section are as specific as possible while retaining flexibility in the development of remedial alternatives.

RAOs are generally expressed in terms of a combination of contaminant levels and exposure routes. However, as presented later, RAOs are expressed in terms of exposures routes only. This approach is consistent with *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 1988f) and recognizes that protection of human health and the environment can be achieved by reducing or eliminating human exposure to a given medium (through implementation of a selected remedial alternative) in addition to reducing contaminant levels in that medium, if applicable. For example, if ingestion of contaminated near-shore sediment represents a significant risk to human health, this risk can be reduced by (1) reducing the potential for ingestion of sediment by restricting access to the contamination or (2) implementing a remedial alternative that reduces contaminant levels in near-shore sediments. The importance of protecting human health and the environment through preserving or restoring environmental media (i.e., reducing contaminant levels) serves as the main objective where technically feasible.

Risks to human health and the environment (assessed in Chapters 5 and 6, respectively) are evaluated for all chemicals detected in the CR/PC OU. These assessments are based on the operational history of the Oak Ridge Reservation (ORR), results of previous investigations, and results of the RI. Risk quantification is based on defensible laboratory data generated during the RI.

The primary driver, in terms of human health, for conducting an FS for the CR/PC OU is whether the sediment largely contained beneath the river and additional layers of deposited sediment are considered a medium of potential human exposure. As noted in the human health baseline risk assessment, two types of sediment access were evaluated for several areas (subreaches) along the OU river system: (1) dredging of deep sediments for land application and farming and (2) direct access to near-shore sediments during periods of low pool. Of these, only direct access to near-shore sediments is considered a viable current exposure pathway because dredging for agricultural land application is not generally practiced, and all sediment-disturbing activities are regulated. Dredging and other sediment-disturbing activities in the Clinch River/Poplar Creek system are tightly controlled by the Tennessee Valley Authority (TVA), the U.S. Army Corps of Engineers (USACE), EPA, TDEC, and DOE. Unauthorized dredging is prohibited. While the baseline human health risk assessment assumed a residential land use (i.e., contact is considered to occur throughout the entire period of low pool exposing near-shore sediments) when evaluating exposures, all the subreaches identified in the RI as having unacceptable near-shore risk are within ORR and are controlled by DOE; thus, no current residential exposures are occurring. DOE land-use controls should prevent residential development of the areas identified.

Ingestion of fish caught within the CR/PC OU is also a driver for the FS because risks associated with fish consumption (largemouth bass and catfish) exceed risk thresholds.

For the purposes of this FS, it is realistically assumed that the residential exposure scenario evaluated in the baseline human health risk assessment is not viable. The fish ingestion scenario is considered viable, although fish advisories are currently in place. The applicable COCs include As, Hg, Be, Se, polychlorinated biphenyls (PCBs), ^{137}Cs, ^{90}Sr, aldrin, chlordane, and 4,4'-DDT.

Results of the baseline ecological risk assessment suggest that sediment in seven reaches in CR/PC OU pose unacceptable risk to benthic organisms: PCM 0.0, 1.0, 1.4, 2.3, 3.01, 4.01, and 4.03.

The remedial action alternatives developed and analyzed in Chapters 9 and 10 are designed to meet the following RAOs.

- If deep sediments are a likely exposure medium under a future dredging/agricultural application scenario, prevent active dredging and land application of the contaminated portions. This will eliminate ingestion, inhalation, external exposure, dermal contact, and ingestion of meat, milk, and vegetable routes of exposure.

- Prevent or limit human consumption of fish potentially contaminated with COCs.

- For the benthic organisms at risk, prevent sediment-disturbing activities that may bring the more contaminated sediments to the surface; contain the contaminated sediments; or remove the contaminated sediments.

- Continue monitoring for potential impacts in conjunction with actions taken in upstream OUs to prevent ongoing releases to CR/PC OU.

Even though Tennessee AWQC are exceeded, the remedial action alternatives in this FS will not address the elevated water concentrations of mercury or arsenic for several reasons:

- the main source for this contaminant is upstream and is a continuing source,

- treatment at the source is preferred and is less expensive, and

- treatment of the entire flux of Poplar Creek or Clinch River is neither practical nor reasonable.

8.2 IDENTIFICATION, SCREENING, AND EVALUATION OF TECHNOLOGIES AND PROCESS OPTIONS

Technology groups and process options can be identified from several sources. The following were tools used to develop a list of technologies and process options that apply to the media of interest:

- Y-12 Plant Technology Logic Diagram (Energy Systems 1994);

- ReOpt version 2.1 (Battelle Memorial Institute);

- the *Remedial Investigation/Feasibility Study Report for Lower Watts Bar Reservoir Operable Unit*;

- *Dredging '94, Proceedings of the Second International Conference on Dredging and Dredged Material Placement*, Vols. 1 and 2;

- *Review of Removal, Containment, and Treatment Technologies for Remediation of Contaminated Sediment in the Great Lakes*; and

- the *Y-12 Plant-Bear Creek Valley OU 2 FS, Oak Ridge, Tennessee.*

Identified technologies and process options have progressed beyond the laboratory research and development stages and are potentially applicable to the source areas of the OU. As shown in Figure 8.1, the general response actions are broad categories that address similar problems. The general response action of containment, for example, includes technology groups such as capping and vertical barriers. Each of these technology groups contains several process options that may be used (e.g., capping includes Armorform, bentonite/soil cap, clean sediment, and geomembrane).

General response actions have been developed to protect human health and the environment from site contaminants. These include:

- no action,
- institutional controls and advisories,
- source containment,
- removal,
- turbidity minimization,
- ex situ treatment, and
- disposal.

These actions address the environmental and public health impacts of contaminants, possible migration pathways, and exposure routes for sources. A combination of general response actions may prove more effective in meeting the RAOs than a single action.

As specified in the RI/FS guidance (EPA 1988f), two steps are taken to reduce the number of technology groups and process options that are developed into alternatives. First, each process option is evaluated for technical applicability by comparing capabilities against the characteristics of the OU and its contaminants. Table 8.1shows site conditions, against which each process option is assessed in this screening step and the following screening step. Some options may not be applicable to site conditions (depth of contamination or types of contaminants). Process options that are not technically applicable are eliminated from further consideration. In some cases, none of the process options for a given technology is considered technically applicable at the OU, and the entire technology will be eliminated. Eliminated options or technology groups will not be further discussed in this document. Each technology that passes the first screening step will have one or more process options. Figure 8.1 indicates eliminated process options with cross-hatched boxes. A brief description is given to indicate why the option is not considered technically applicable to the site.

During the second step, the remaining process options are evaluated more closely to determine which process options and technologies will be developed into remedial alternatives. This evaluation selects a single process option or combination of process options to represent each technology group so an estimated cost can be developed for each alternative. The single process option or combination of

Table 8.1. Site physical and contaminant characteristics for the Clinch River/Poplar Creek Operable Unit, Oak Ridge, Tennessee

Parameter	Characteristics
	Physical site characteristics
Climate	Humid, temperate, minor freeze/thaw
Sediment properties	Typically soft silts and clays, with some sand; discontinuous depositional areas; bedrock exposure and/or boulders present in some areas
Reservoir pool	Summer (ft msl) / Winter (ft msl)
(above Melton Hill Dam)	795 / 790
(below Melton Hill Dam)	741 / 735
Flow rate	Controlled reservoir, 0–1215 m^3/s at Melton Hill Dam
Terrain	Variable, mostly riverine with some lacustrine characteristics in downstream areas near the confluence with the Tennessee River
Accessibility	Varied by combination of terrain, water depth and speed, and point of entry
	Contaminant characteristics
Contaminants of concern	As, Be, ^{60}Co, Cr, ^{137}Cs, Hg, Mn, Se, ^{90}Sr, ^{234}U, ^{235}U, ^{238}U, PCBs, pesticides
Waste form	Sediment: particulate association Biota: tissue Surface water: in solution and particulate association
Previous actions	Source controls under CERCLA and institutional controls by other statutory authorities
Persistence	Metals, high PCBs, high-bioconcentrating Radioisotopes, half-life dependent Organics biodegrade naturally
Mobility	Low in sediment under normal conditions; high in surface water; moderate in fish, which tend to inhabit ranges approximately within a 2-mile radius up or downstream
Radioactivity	Negligible in surface sediments, fish, and surface water; low in deep sediments

CERCLA = Comprehensive Environmental Response, Compensation, and Liability Actof 1980
DDT = dichlorodiphenyltrichloroethane
msl = mean sea level

options that appears to offer the best blend of effectiveness, implementability, and cost is carried forward for development into remedial alternatives in Chapter 9. Figure 8.1 indicates these process options with double-outlined boxes.

Process options or technology groups not carried forward to development of alternatives remain technically applicable to the site, but under current conditions do not appear to offer the best blend of effectiveness, implementability, and/or cost. Although these will not be carried forward for remedial alternative development, they can be reevaluated during the proposed plan, the ROD, or the remedial design. Figure 8.1 indicates these process options in single-outlined boxes.

Appendix I provides a comprehensive discussion of all process options remaining after the first screening step for technical applicability. The discussion includes a technical description of the option, effectiveness, implementability, cost, and the rationale for determining the representative process options carried forward into remedial alternative development.

General Response Action	Remedial Technology Type	Process Options	Description	Discussion
1.1 No Action	1.1.1 None	None	National Contingency Plan (NCP) requires the no action alternative to be carried through the detailed analysis	Required for comparative purposes by NCP
1.2 Institutional Controls and Advisories	1.2.1 Access and Use Restrictions	Deed Restrictions	Restrictions and notifications placed in the deeds of impacted properties	Potentially viable; effective in preventing improper development
		Public Advisories	Post and publish advisories describing fish consumption limits and/or water contact warnings	Viable and in place
		Permit Programs	Require permits for all sediment dredging or similar activities that may liberate contaminated sediments	Viable and in place
		Land Purchase/Condemnation	Contaminated land may be acquired by purchase or condemnation	Potentially viable; not representative because no near-shore sediment contamination identified on, private property
		Physical Barriers	Barriers such as fencing, signs, and other access-limiting measures	Potentially viable for shallow areas
	1.2.2 Maintenance and Monitoring	Monitoring	Monitoring of contaminant levels in water, sediments, and biota	Potentially viable; useful for detecting changes in OU conditions
		Surveillance and Maintenance	Routine physical site surveys and maintenance	Potentially viable; cost-effective means to ensure and verify integrity of engineered controls or devices

1.2 Representative Process Option

Reference Section Number in Appendix I

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Fig. 8.1

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Results of process option screening

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General Response Action	Remedial Technology Type	Process Options	Description	Discussion
1.3 Source Containment	1.3.1 Capping (Horizontal Barriers)	Bentonite and Soil Cap	Clay- and soil-based cap is a single-layer unit that relies on low permeability clay to inhibit infiltration of precipitation through contaminated soil	Not applicable because unsuitable for saturated sediment conditions
		Armorform	Double-layer woven fabrics engineered exclusively to serve as forms for casting concrete erosion control revetments and linings	Potentially viable; long-term protection against erosion control for shallow channels and embayments
		Riprap	Durable, nontoxic, nonacid forming rockfill used for erosion protection	Potentially viable; hand placement required for embankment applications
		Geomembrane	An impermeable synthetic liner placed over sediments to isolate contaminants	Potentially viable for near-shore sediments
		Clean Sediment	Clean sediments are placed over contaminated sediments to isolate contaminants	Potentially viable; not representative because river currents and biotic action may compromise cap integrity or reliability
	Vertical Subsurface Barrier	Soil-Bentonite Slurry Wall	Trench excavated above and/or around contaminated area and filled with bentonite or soil-bentonite slurry to create a low-permeability barrier to shallow subsurface flow	Not necessary for sediments
		Vibrating Beam-Sheet Pilings	Chemical or cement grout injected between parallel sheet pilings to create low-permeability barrier to shallow subsurface flow	Not necessary for sediments
		Grout Curtains	Low-permeability barrier of chemical or cement-based grout injection in a series of overlapping vertical wells to create a low-permeability barrier to shallow subsurface flow	Not necessary for sediments

1.3 Representative Process Option — Screened Out

Reference Section Number in Appendix I

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Fig. 8.1

Results of process option screening

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General Response Action	Remedial Technology Type	Process Options	Description	Discussion
1.4 Removal	1.4.1 Excavation	Mechanical	Removal of soil or sediments with mechanically operated equipment such as backhoe, bulldozer, etc.	Mechanical excavation will be necessary for relocation of any surface sediment
	1.4 2 Dredging	Mechanical	Removal of soft, cohesive sediment	Potentially viable; proven applicable to contaminated sediments. Not representative because of extreme difficulty in implementability and cost effectiveness for near shore sediments
		Hydraulic	Removal of bulk materials, sediments,and granular materials using barge- or surface-mounted hydraulic pumps	
		Pneumatic	Removal of liquids by using compressed air to pump slurry through a pipeline	
1.5 Turbidity Minimization	1.5.1 Water Turbidity Minimization	Caissons, Dikes, and Cofferdams	Physical barriers are installed around the area of sediment removal to intercept the resuspended sediments not captured by the process itself	Not needed for the expected site conditions
		Silt Curtains	Impervious barriers that extend vertically from the water surface to a specified depth	Potentially viable; not representative because effectiveness limited to submerged applications
		Silt Screens	A synthetic geotextile fabric that allows water to pass through small openings in the fabric but retains suspended solids	Potentially viable; not representative because effectiveness limited to submerged applications
	Surface Water Collection	Collection Sumps, Dams, Levees, and Basins	Surface water collectors for capture of sediments	Not applicable because not suitable for expected site conditions

1.5 Representative Process Option — Screened Out

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Fig. 8.1

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General Response Action	Remedial Technology Type	Process Options	Description	Discussion
1.6 Ex Situ Treatment	1.6.1 Physiochemical	Chemical Extraction/Soilwashing	Contaminants which tend to associate with finer-grained particles are separated out during this process	Potentially necessary following a removal of sediments; not representative due to minimal reduction of waste material to be treated
		Solidification/Stabilization	Sediments are mixed with cement-like matrix that immobilizes contaminated sediments	Potentially necessary following a removal of sediments
		Dehalogenation	A chemical reaction that breaks the carbon halogen bond	Not applicable to aqueous wastes, soils, sludges, or sediments; not applicable because of the reagents reactivity with water
		Chemical-Precipitation/ Flocculation	Addition of chemicals to precipitate metals from solution and mechanical techniques to enhance flocculation and settlement	Potentially viable for water remediation; not representative due to inadequate treatment facilities
		Ion Exchange	A volume and toxicity reducer that removes ionic species, principally inorganics, from aqueous waste streams	Potentially viable in treating inorganics in solutions; not representative due to inadequate treatment facilities
	Biodegradation	Biopiles	Biodegradation in a lined treatment bed	Unsuitable; not applicable because organics in near- shore sediment are not COCs
		Landfarming	Natural biodegradation is promoted by spreading contaminated material on the ground surface	
		Bioreactor	A bioreactor or treatment vessel where contaminated material can be rapidly biodegraded in a controlled environment	
		Composting	Organic carbon sources and bulking agents; woodchips, straw, etc., are mixed with contaminated soil and optimum condition maintained to promote biodegradation	

1.6 Representative Process Option — Reference Section Number in Appendix I

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Fig. 8.1

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General Response Action	Remedial Technology Type	Process Options	Description	Discussion
1.6 Ex Situ Treatment *(continued)*	1.6.3 Dewatering	Belt Filter Press	Compresses sediment slurry to remove free water-producing sediment cakes	Potentially viable; not representative because sediment and/or water may require additional treatment and/or disposal
		Settling Ponds	Uses large areas constructed to allow for gravitational settling and liquid/solid separation	
		Stabilizing Agents	Use of stabilizing agent to physically combine with slurry to absorb free water	Potentially viable
		Sloping Dewatering Pad	Constructed sloping concrete pad to place dredged sediments onto and allowing water to drain out of sediment	Potentially viable; effective in dewatering sediments passively using natural processes
		Low-Temperature Thermal Heating	A dewatering method used to dry or vaporize water and/or moisture within saturated sediments	Potentially viable; effective in dewatering sediments by use of mobile equipment

1.6 Representative Process Option — Reference Section Number in Appendix I

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Fig. 8.1

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Results of process option screening

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General Response Action	Remedial Technology Type	Process Options	Description	Discussion
1.7 Disposal	1.7.1 On-Site Sediment Disposal	Y-12 Plant Landfill V or VII	Disposal of "special waste"	Must comply with waste acceptance criteria and TDEC approval
		Beneficial Re-Use	Backfill in areas remote from human occupation	Must comply with activities limits
	1.7.2 Off-Site Sediment Disposal	Envirocare	Disposal of solid low-level mixed waste at off-site facility with adequate capacity that meet applicable waste acceptance criteria	Potentially difficult due to costs of stabilizing/ transporting prior to disposing solid low-level mixed waste
	1.7.3 Water Disposal	Y-12 Plant West End Treatment Facility	Treatment and disposal facility for potentially contaminated water removed from dewatered sediment	Must comply with acceptance criteria and TDEC approval
	1.7.4 Waste Transportation	Trucks	Vehicles used to haul material (sediment/water) for treatment or disposal	Potentially viable for hauling contaminated material
		Rail	Transport of dewatered sediment for long distance, solid low-level/mixed waste disposal	Potentially viable; representative for shipments to Envirocare

1.7 Representative Process Option
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Fig. 8.1

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9. DEVELOPMENT OF ALTERNATIVES

This chapter presents the development of remedial alternatives assembled from combinations of technologies and representative process options evaluated in Chapter 8. Section 9.1 presents the rationale for the development approach, waste type criteria, common factors to the OU, and media-specific criteria. Section 9.2 describes the alternatives, including outlines of process options and technologies used to assemble each alternative.

Technologies or response actions not currently selected for detailed description and subsequent detailed analysis may be reevaluated in the future for potential implementation.

9.1 DEVELOPMENT OF REMEDIAL ALTERNATIVES

9.1.1 Approach for Developing Remedial Alternatives

EPA guidance (EPA 1988f) establishes an approach to develop appropriate remedial action alternatives. Using this approach, the scope, characteristics, and complexity of the specific conditions at the CR/PC OU are considered in developing a range of alternatives that protect human health and the environment. Protection may be achieved by eliminating, reducing, or controlling risks posed by each pathway at a site.

The NCP lists the following preferences in developing and screening remedial action alternatives:

- use of treatment to address the principal threats posed by a site, wherever practical;
- use of engineering controls, such as containment, for waste that poses a relatively low, long-term threat and for which treatment is not practical;
- implementation of a combination of actions, as appropriate, to achieve protection of human health and the environment (e.g., in appropriate site situations, treatment of principal threats would be combined with engineering controls, such as containment, and institutional controls);
- use of institutional controls, such as drinking water supply controls and deed restrictions, to supplement engineering controls for short- and long-term management to prevent or limit exposures to hazardous substances; and
- selection of an innovative technology when the technology offers the following: the potential for comparable or better treatment performance or implementability, fewer, or lesser magnitude, adverse impacts than other available approaches, or lower costs than demonstrated technologies for similar levels of performance.

9.1.1.1 Common factors

Naturally occurring processes can affect, directly or indirectly, the effectiveness of implemented alternatives. Natural sedimentation will further the buildup of clean (noncontaminated) sediments over contaminated deep sediments and initially cover contaminated shallow sediment. With enough clean sediment cover, benthic organisms will no longer feed on contaminated sediment, thus reducing contaminants available in the food chain. This process will continue to isolate and further reduce availability of the contaminants. In spite of the benefit natural sedimentation gives contaminant isolation, too much sedimentation can eventually result in a sediment-clogged reservoir and an unnavigable waterway. Excess sediments must be periodically cleared from the channel to maintain the waterway. Moreover, because contributing sources continue to add contaminants to the system, some remedial options designed to reduce risks from contaminants presently in place may become ineffective over time. Protection from (currently) contaminated sediments may be short-lived, depending on the sedimentation rate and contaminant influx. Alternative assembly considers the good and bad aspects of natural sedimentation.

The second naturally occurring process common throughout the OU is the natural degradation of the contaminants, including biodegradation and radioactive decay. The toxicity of some metals and organics diminishes with time as naturally occurring bacteria in the sediment consume the material, producing less toxic by-products. Natural processes reduce the chemical mobility of metal ions and radioactive isotopes. The combination of natural sedimentation and natural degradation can ultimately reduce risk from ingestion of fish and exposure to shallow and deep sediments.

9.1.1.2 Media-specific criteria

Contaminated media targeted for alternative development include: (1) sediments of Poplar Creek at PCM 0.0, 1.0, 1.4, 2.3, 3.1, 4.1, and 4.3 and (2) various fish species in Reaches 1, 2, 3, and 4.

The following criteria discussion is intended to tailor alternatives to best fit the needs of the various media, pathways, and contaminants.

Alternative assembly for the reduction of exposure to contaminated biota was based on the following:

- Time, preventing contaminant migration, and natural degradation of contaminants are reasonable solutions to an overall biotic remediation in this OU. Containment or removal of contaminated sediments may also be reasonable for small areas (hot spots), although either action would destroy the existing benthic community. To expedite the biological recovery in the ecosystem, elimination or reduction of contaminant input from ORR and other sources to the river system would lower bioavailability of contaminants and reduce ecological risk levels.

- The ingestion of contaminated fish pathway is of concern for humans. If release of contaminants to the reservoir and its tributaries cease, contaminant levels in the fish population will decline. For the near term, interruption of this pathway is the primary focus for each remedial alternative for human health risk reduction.

Alternative assembly for the remediation of contaminated, shallow, near-shore sediments was based on the following:

- Near-shore sediments are submerged at summer pool under as much as 1.8 m (6 ft) of water. Some portion of the sediment profile is exposed during drawdown of the reservoir until its lowest drawdown elevation, winter pool, when all of the shallow sediments are without water coverage. Certain portions of the OU, like Poplar Creek, may have lead and lag times in response to precipitation events and discharge events from upstream tributaries and downstream dams. It is assumed that water elevations may exceed summer pool elevations (i.e., floods), but will not be less than winter pool elevation (i.e., dried up).

Alternative assembly for the remediation of contaminated deep sediments was based on the following:

- Deep sediments are relatively immobile under current daily river conditions and fluctuations, and the higher contaminant concentrations are buried under a layer of cleaner sediments in most areas.

- Deep sediments could pose unacceptable risk to humans and biota when dredged and used untreated, exclusive of surrounding clean sediments/soils.

9.2 DETAILED DESCRIPTION OF ALTERNATIVES

The following sections present detailed descriptions of all alternatives developed for CR/PC OU. The descriptions are based on preliminary designs developed for this FS. For detailed descriptions of process options used in the following alternatives see Appendix I.

9.2.1 Alternative 1—No Action

As required by the NCP, a no action alternative provides a comparative baseline against which other alternatives can be evaluated. Under this alternative, no action would be implemented and the contaminated sediments in CR/PC OU would be left "as is," without implementing any control, containment, removal, treatment, or other mitigating actions.

This alternative would be effective if all existing controls could be assumed to remain in place. Some reaches are essentially upstream of DOE source areas, and sediment depositions in these areas are not known to exceed unacceptable risk levels. Any DOE-derived contaminated sediments upstream of DOE source areas are likely the result of backflow conditions during raising or lowering of the reservoir. Sampling efforts and the baseline risk assessment results indicate no sediment contamination at unacceptable levels in the upstream reaches. The baseline risk assessment does, however, indicate that some fish species in Reaches 1, 2, 3, and 4 present unacceptable risk to human health. Sediments in portions of Poplar Creek and Clinch River may be a threat to ecological health. Discontinuing fish consumption advisories may allow humans to consume fish in sufficient quantities to cause harm. Unregulated, sediment-disturbing activities may cause fugitive sediment to deposit in clean areas unnoticed and pose unacceptable risk to human and ecological receptors.

9.2.2 Alternative 2—Institutional Controls and Advisories

Alternative 2 is proposed as a means to manage contaminants in place and without treatment. Long-term management of these contaminants must continue because the contaminants would remain in place. This alternative would address the entire area of CR/PC OU. Components of Alternative 2 include:

- public advisories,
- permit programs, and
- monitoring.

Institutional controls and advisories would be protective of human health by managing contaminants left in place. Sediment disturbance would be regulated with other statutory authorities to keep risks from potential contaminant migration to humans and other biota low. Fish consumption advisories would continue under the state of Tennessee to protect the public from contaminated fish.

DOE will work with TDEC and will bear the responsibility for ensuring that the public is protected from DOE-related contaminants. Monitoring will confirm the effectiveness of the other institutional controls by gaging the control benefits and yielding the necessary information to modify the controls, as needed. Surveillance and maintenance can be used to patrol effected areas, postings, etc., for proper maintenance. Should a larger area of contamination be identified, the advisories and controls (at small additional cost) would be expanded to address the new threat.

9.2.2.1 Public advisories

Public advisories for fish consumption throughout the OU and water contact advisories for Poplar Creek are proposed to remain in effect until future monitoring indicates that contaminant levels in affected fish populations have abated. TDEC currently administers these advisories. This FS assumes that TDEC management will continue. DOE will work with TDEC to ensure that the advisories reflect recent data and cover all species needed to adequately protect human health. Following the approval of the ROD, DOE will bear the ultimate responsibility for the advisories in DOE-impacted waters for species with DOE contaminants.

9.2.2.2 Permit program

TVA, USACE, TDEC, EPA, and DOE jointly manage restrictions on sediment-disturbing activities. For this FS, sediment-disturbing activities apply to the deep sediments in the main channel and the shallow sediments. Statutory programs that restrict sediment-disturbing activities are in effect for Clinch River and Poplar Creek, limiting the locations and depths to which sediments can be dredged or disturbed. Permitting is proposed to remain in effect indefinitely for CR/PC OU.

9.2.2.3 Monitoring

Monitoring options proposed for all areas of Clinch River and Poplar Creek include additional sediment, surface water, and fish sampling. Monitoring of sediments at the CR/PC OU would be performed annually to observe system conditions.

Fish monitoring would be conducted semiannually. Analysis of fish data will allow proper warnings to be applied to the OUs. In the event that input sources increase or existing sources become more bioavailable causing contaminants in fish to increase risk levels, modifications in advisories can be issued. Conversely, decreases in contaminants in fish may result in the lifting or lessening of the restrictions on fish consumption.

9.2.3 Alternative 3—Source Containment, Removal, and Disposal

Alternative 3 is proposed as a means to provide an additional component of source containment for shallow sediments where contaminants exceed risk thresholds for human health. Also included in Alternative 3 are containment and removal of sediments that exceed the risk threshold for benthic organisms. Areas above 223 m (733 ft) above msl will be contained and those below that level would be removed. Institutional controls and advisories as described in Alternative 2 are also included in this alternative.

Human health risk estimates developed in the RI are based on exposures to contaminated sediments under a residential scenario during periods of surface water drawdown. Therefore, infrequent visitors are probably not at risk from exposure. The sediments are under as much as 1.8m (6 ft) of water for a portion of the year and daily exposures are not likely to occur. Containment of the sediments above 223 m (733 ft) may be necessary to lower the ecological risk potential by isolating the contaminants and removing the pathway. Sediments below 223m (733 ft) will be removed for protection of benthic organisms. Alternative 4 provides the detailed description of removal activities.

The containment portion of Alternative 3 would be implemented above 223 m (733 ft), and dredging would be implemented below 223 m (733 ft) for PCM 0.0, 1.0, 1.4, 2.3, 3.1, 4.1, and 4.3 for ecological health. The institutional controls and advisories as outlined in Alternative 2 would be implemented for the entire area of CR/PC OU. The containment and removal areas and volumes associated with this alternative are provided in Table 9.1.

The geomembrane cap would place a physical barrier between the contaminated sediment and the receptor, and removing the pathway of concern. Removal of sediment eliminates the risks of human or animal contact with the most contaminated sediments. Risks presented by the shallow sediments would be diminished sharply because no complete exposure route would remain. Alternative 3 is composed of the same institutional control elements as outlined in Alternative 2, plus shallow sediment containment and deep sediment removal. Public fish consumption advisories are also planned to protect the public.

Before design of this alternative can begin, some further sampling would need to occur to delineate the exact boundaries of the areas to be removed or contained. Current estimates are based on single core samples approximately every 0.4 to 0.8 km (0.25 to 0.5 mile). This further sampling could decrease or increase the scope and cost of this alternative significantly. A small treatability study on the sediments will also need to be performed to determine the waste characteristics of the sediment and effluent after dewatering occurs. Surveys for wetlands and archaeological resources would be necessary before construction.

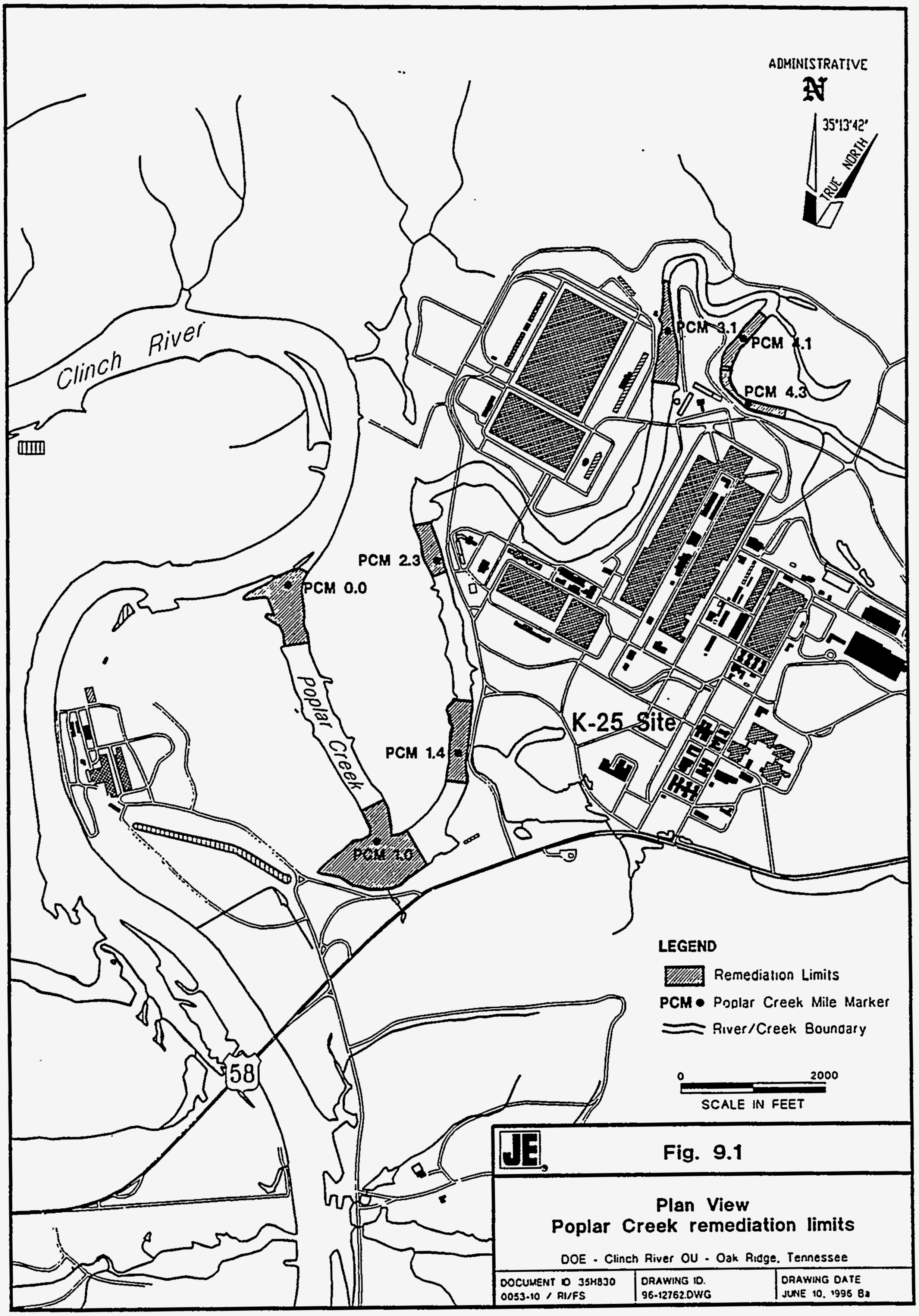
Physical barriers, such as fencing, may be used for areas where capping is technically infeasible (e.g., shore slopes too steep to install a cap).

Routine inspections of installed devices such as signs or physical barriers will be conducted annually. Additionally, inspections of the shoreline geomembrane cap would be performed annually. The most appropriate inspection time would be when the winter pool drawdown is near 100 percent complete, which will expose as much of the cap area as possible and allow easier visual inspections for erosion.

For shallow sediment containment, the geomembrane cap combines the low permeability of a synthetic geomembrane layer and the erosion resistance of riprap to cover and isolate shallow contaminants in place and reduce risk of either direct exposure or redistribution of shallow sediment contaminants by erosion. The areas to be capped, as shown in Figure 9.1, will be cleared of vegetation to allow a smooth surface on which to place the geomembrane. An anchor trench 0.3 m (1 ft) wide and 0.6 m (2 ft) deep will be cut into the top of the cap area at least 0.9 m (3 ft) beyond the top of the creek bank and at the toe of the area to be capped. The ground surface will be stripped and smoothly graded to remove any objects that may cause breaks or punctures to the geomembrane. However, no grading to significantly alter the degree of creek bank slope is proposed. As shown in Figure 9.2, a geotextile fabric liner will first be laid over the prepared surface to give the geomembrane bottom protection; second, the geomembrane will be placed; third, a geotextile fabric liner will be placed on top of the geomembrane for upper surface protection. Each layer of material will be overlapped into the anchor trenches. Riprap will then be placed on the geotextile surface overlapping the anchor trenches. Riprap thickness will be 0.6-m (2 ft) minimum. Any remaining uncapped disturbed areas will be mulched and seeded to minimize erosion.

Table 9.1. Alternative 3—Containment and removal areas and volumes, Clinch River/Poplar Creek Operable Unit, Oak Ridge, Tennessee

Location (Poplar Creek mile)	Near-shore containment	Removal of deep sediments		Total area of remediation (ft^2/acre)
	Area of remediation (ft^2/acre)	Area of remediation (ft^2/acre)	Removal quantity (yd^3)	
0.0	38,400/1.0	47,700/1.0	5,800	86,100/2.0
1.0	86,400/2.0	850,100/19.5	103,300	936,500/21.5
1.4	57,600/1.5	189,900/4.5	23,100	247,500/5.7
2.3	33,600/0.5	106,400/2.5	13,000	140,000/3.2
3.1	76,800/1.5	127,700/3.0	15,500	204,500/4.7
4.1	38,400/1.0	80,100/2.0	9,750	118,500/2.7
4.3	57,600/1.5	72,400/1.5	8,800	130,000/3.0
Total	388,800/9.0	1,474,300/34.0	179,250	1,863,100/42.8


ADMINISTRATIVE
N
35°13'42"
TRUE NORTH
Clinch River
Poplar Creek
PCM 0.0
PCM 1.0
PCM 1.4
PCM 2.3
PCM 3.1
PCM 4.1
PCM 4.3
K-25 Site
58
LEGEND
Remediation Limits
PCM ● Poplar Creek Mile Marker
River/Creek Boundary
0
2000
SCALE IN FEET
JE
Fig. 9.1
Plan View
Poplar Creek remediation limits
DOE - Clinch River OU - Oak Ridge, Tennessee
DOCUMENT ID 35H830
0053-10 / RI/FS
DRAWING ID.
96-12762.DWG
DRAWING DATE
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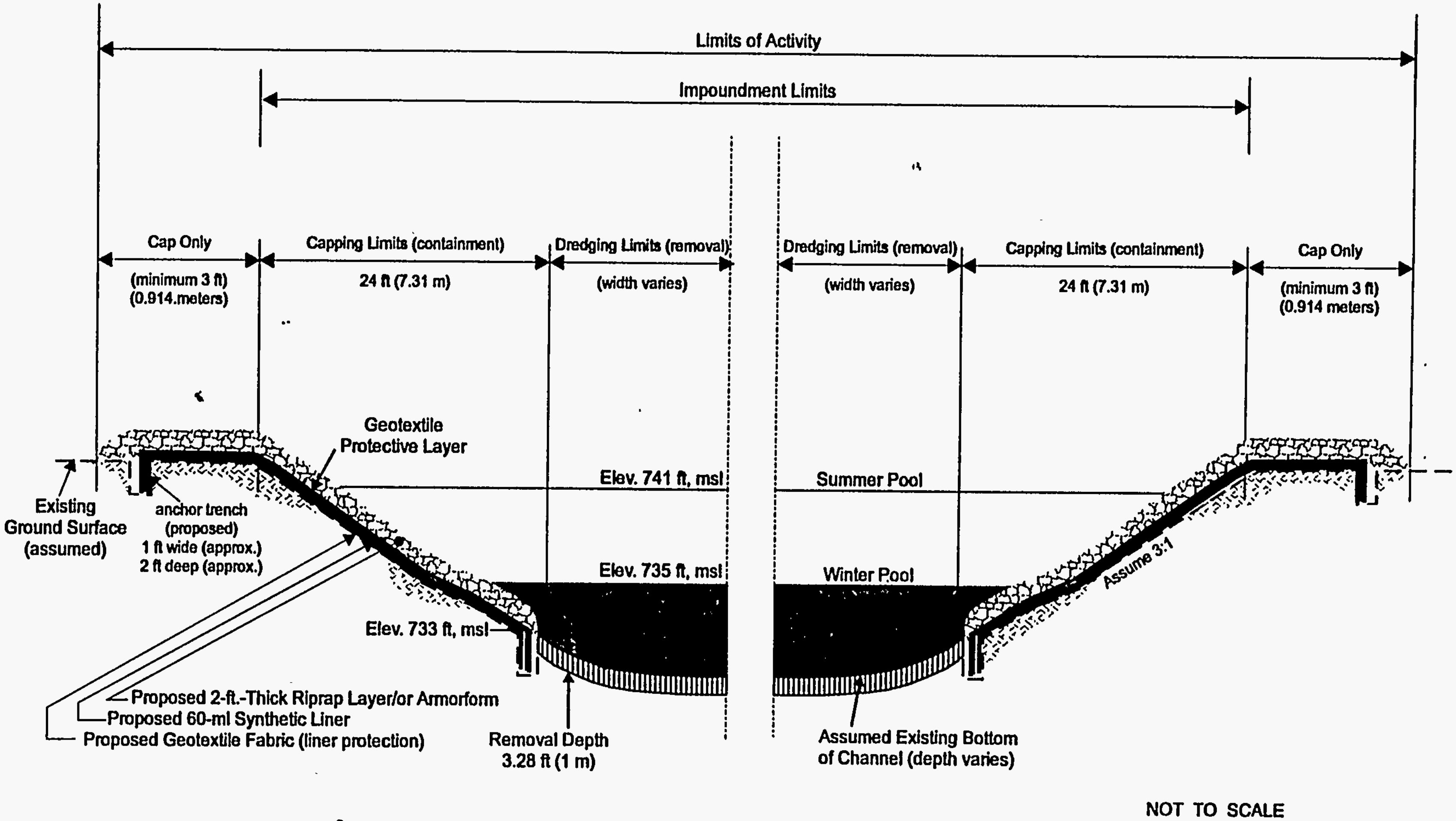


JE	Fig. 9.2	Poplar Creek - Alternative 3 (containment/removal detail) DOE - Clinch River OU - Oak Ridge, Tennessee	DOCUMENT ID: 35H830 0053-10 / RI/FS	DRAWING ID: 96-12761.CDR	DRAWING DATE: JUNE 10, 1996 Ba

The conceptual sequence of construction activities for Alternative 3's capping of shallow sediments is presented in the following list. Dredging would occur after capping was complete.

1. Install erosion and turbidity control material (e.g., silt fencing and straw bales).

2. Install the shoreline construction stability mat over areas where heavy equipment will be used adjacent to the shore.

3. Clear vegetation and vegetative root systems where cap and trench will be installed.

4. Proofroll the site (repair soft spots and create smooth surface).

5. Excavate anchor trenches top and bottom.

6. Install a bottom-protective geotextile layer over the smoothed surface.

7. Install a geomembrane layer over the area.

8. Install an upper-protective geotextile layer over the geomembrane.

9. Place 0.6 m (2 ft) of riprap over the geotextile.

10. Seed and mulch any areas disturbed by construction that extend beyond capping limits.

Postclosure monitoring would be conducted annually to ensure the cap remains reliable and performing as designed. Any repairs to the cap should be made as soon as possible. It is anticipated that this cap can last at least 30 years before maintenance is necessary. Monitoring would also include monitoring for sediments accumulating over the cap that might be contaminated and might raise risk levels.

Dredging. The dredging of deep sediments below 223 m (733 ft) will take place as described in Alternative 3. Any of the dredging technologies (mechanical, hydraulic, and pneumatic) and their selected options (clamshell, Mudcat, and pneuma, respectively) have been proven effective at removing contaminated sediments with minimal turbidity. Dredging is perhaps the most effective technology to remove deep sediments [below 233 m (733 ft)] where a backhoe would be ineffective.

Dredging sediments is typically conducted from a barge that is either stationary with the equipment moving along the bottom removing sediment, or the equipment is fixed and the barge is moved from one location to the next, or a combination of these. For shallow, restricted areas, perhaps (although not exclusively) along Poplar Creek, an all terrain/amphibious vehicle mounted with dredging equipment is also available where a barge will not fit.

The preferred dredging technique for deep sediment is the pneuma pump. It can remove sediments with minimal turbidity. Shallow water dredging would be best accomplished by a clamshell. Silt screens or curtains can be used to contain the drifting sediment within the area being dredged. The object of the chosen dredging method is to dredge the material with the least amount of water pickup as possible.

Most dredge methods require water to dislodge the sediment and carry it off. While dredging operations are being performed, considerable volumes of water are removed with the target sediment. See Figure 9.2 for dredging details.

Handling, treatment, and disposal of the excavated/dredged sediment along with its water will be main contributors to cost for Alternatives 3 and 4.

The following are operational components involved with excavating and dredging:

- Turbid Water Containment—There are basically two options to turbid water containment. First is containment of resuspended sediments during dredging activities. Dredging as described above involves the physical removal of sediments by actively dislodging the sediment by either mechanical, hydraulic, or pneumatic means. The sediments are covered with a variable amount of water. The options selected for each dredging technology are very good at recovering most of the sediment that is dislodged, but inevitably a small amount of sediment can escape the process. For this, silt curtains installed around the perimeter of the work site is proposed to fully contain the operation and prohibit migration of sediments and any associated contaminants. Recovery of the turbid water is essential to prevent redistribution of contaminated sediments. Treatment and/or disposal of the water may be required.

- Waste Transport—Dredged material (sediment slurry) must be transported to shore. The sediment can be dredged and temporarily staged on a barge pending transport and subsequent sediment off-loading. Currently the preferred scenario to dredge deep sediments in Poplar Creek is to use the pneuma pump dredge and clamshell dredge (which recover high density slurry) and continuously place the material on barges running to the dewatering area.

  For near-shore or deep sediments (in either the Clinch River or Poplar Creek), dredged sediment slurry can also be transported to shore by continuously pumping the material as it is being removed. This type of transport will apply if a hydraulic dredge is selected for sediment removal. Hydraulic types use considerable amounts of water to create a pumpable slurry, while clamshells and pneuma pumps do not. The slurry would be pumped from the barge-mounted dredge to an on-shore facility set up to process the material. This would allow dewatering of the slurry to take place with limited interruption and without additional staging and handling of hazardous material, which can be a benefit. Further treatment and/or disposal, as needed, would take place elsewhere.

  The trucks should use DOE roads mainly to create as little interference as possible on public roads. The material would be hauled to a central location much like the one previously described.

  The water component carried along in the sediment slurry will be collected and analyzed at the time of dewatering. Due to the low solubility of contaminants in the sediments, the water could potentially be discharged back to the reservoir pending confirmatory analysis. Because the water is assumed to be safe for return to the reservoir, an on-site treatment facility would not be cost-effective or necessary. A limited number of storage tanks can be used to hold the water in the interim. Once the analysis is returned that the water is safe for emptying, the tank will be emptied of its contents and reused for additional waterstaging capacity. Rotation of tanks (filling and emptying) will eliminate the need for a tank farm.

  In the event the water is contaminated, it will not be discharged into the reservoir. Transport of the contaminated water to an appropriate facility will be required for proper treatment and subsequent disposal. The water can be pumped to a tanker trailer which, when full, will be hauled to the off-site facility. The holding tank in which the contaminated water was held will be decontaminated before reuse.

Some form of transportation to its final point of disposition is required for the dredged and dewatered sediments. Method of transport is relative to volume of material, amount and type of contaminants, and destination. Local disposal can be achieved by truck transport to the chosen facility. Large volumes, especially requiring final disposition far away (e.g., Envirocare, Inc.), will be transported by rail cars because bulk rail shipments are more cost-effective (i.e., reduced risks, less trips, etc.) than trucks.

- Dewatering—Dewatering applies to only excavated or dredged sediments. Sediments removed by backhoe will have minimal water to manage (mainly water in the mud itself, not free standing water). Dewatering of dredged sediments is basically a waste stream separation of water and solids. Water volumes will depend on the percent solids removed as a slurry. Assume 60 percent solids and 40 percent water for hydraulically dredged material; assume approximately 80 percent solids for clamshell or pneuma pump dredging. After dewatering, the sediment may be treated as necessary with stabilizing agents to bind the remaining water and render the sediment suitable for disposal.

- Sediment Disposal—If remediation of the site requires removal of the sediment, then federal and state regulations governing transportation and disposal will be adhered to. Radioactivity exceeding background as well as RCRA constituents above regulatory limits will determine appropriate disposal options. The concentrations of radioisotopes in the reservoir in some of the deep sediments do pose risk when dredged. Where the radioisotopes do not present risk, it is possible that they may meet the ORR Industrial Landfill waste acceptance criteria (WACs) of less that 35 pCi/g of total uranium, or they may be used as backfill under Category I or II soils scenario (Energy Systems 1993c) on ORR. These disposition options would be more cost-effective and more easily implemented than sending the sediment off site for disposal.

  If confirmatory sampling after dredging and dewatering, identifies the sediments as RCRA-characteristic hazardous waste after failing Toxicity Characteristics Leaching Procedure (TCLP) testing, or radioactivity levels are above ORR disposal WACs, the waste is considered a mixed waste. Based on the most current analytical data received for CR/PC OU and TCLP characterization results from Lower EFPC floodplain soils (Jacobs ER Team 1995), it is likely that the sediments will be LLW and will not require treatment as a RCRA or mixed waste. The only currently approved location for disposal of these sediments is Envirocare.

9.2.4 Alternative 4—Removal and Disposal

Alternative 4 would physically remove contaminated sediments so potential receptors could not be exposed to them.

Removal of shallow and deep sediments would protect benthic organisms and human health. Alternative 4 contains the same institutional control elements as Alternative 2.

For ecological health, sediments above 223 m (733 ft) and dredging below 223 m (733 ft) for PCM 0.0, 1.0, 1.4, 2.3, 3.1, 4.1, and 4.3 would be excavated. The institutional controls and advisories in Alternative 2 would be implemented for the entire area of CR/PC OU. The removal areas and volumes associated with this alternative are provided in Table 9.2.

Table 9.2. Alternative 4—Removal areas and volumes, Clinch River/Poplar Creek Operable Unit, Oak Ridge, Tennessee

Location (Poplar Creek mile)	Near-shore removal		Removal of deep sediments		Total area of remediation (ft^2/acre)
	Area of remediation (ft^2/acre)	Removal quantities (yd^3)	Area of remediation (ft^2/acre)	Removal quantities (yd^3)	
0.0	38,400/1.0	4,650	47,700/1.0	5,800	86,100/2.0
1.0	86,400/2.0	10,500	850,100/19.5	103,300	936,500/21.5
1.4	57,600/1.5	7,000	189,900/4.5	23,100	247,500/5.7
2.3	33,600/0.5	4,100	106,400/2.5	13,000	140,000/3.2
3.1	76,800/1.5	9,350	127,700/3.0	15,500	204,500/4.7
4.1	38,400/1.0	4,650	80,100/2.0	9,750	118,500/2.7
4.3	57,600/1.5	7,000	72,400/1.5	8,800	130,000/3.0
Total	388,800/9.0	47,250	1,474,300/34.0	179,250	1,863,100/42.8

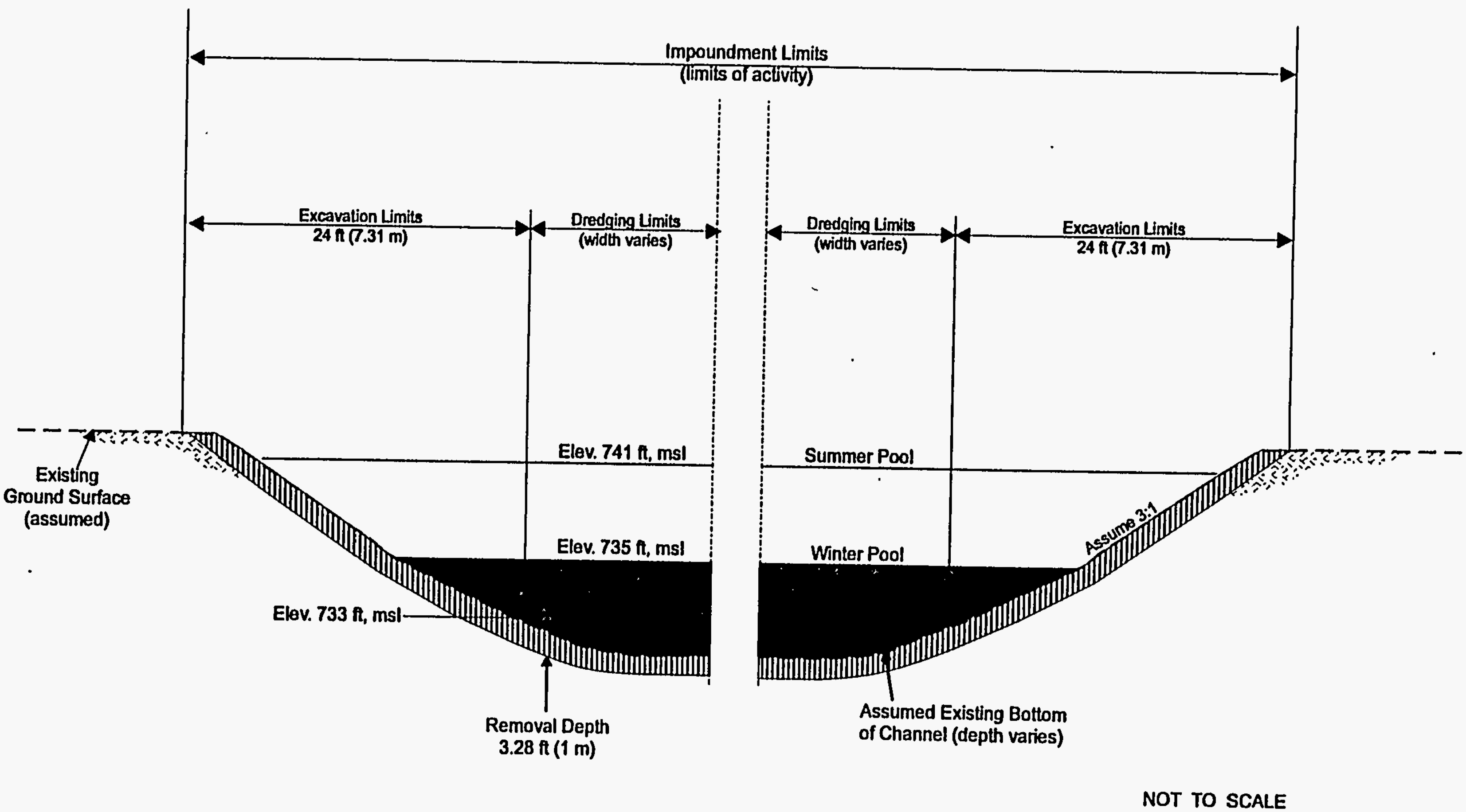
Before design of this alternative can begin, some further sampling would need to occur to delineate the exact boundaries of the areas to be removed. Current estimates are based on single core samples approximately every 0.4 to 0.8 km (0.25 to 0.5 mile). This further sampling could decrease or increase the scope and cost of this alternative significantly. A small treatability study on the sediments will also need to be performed to determine the waste characteristics of the sediment and effluent after dewatering occurs. Before construction, surveys for wetlands or archaeological resources would need to take place.

Excavation. Shoreline areas are suited for using a backhoe operated either from the shore or from a barge. Backhoe excavation of shallow sediments (between elevation 741 and elevation 733) will require removing the material during the season when the impounded water is at winter pool. Backhoe removal would have a very high rate of sediment resuspension if conducted while the target sediment is covered with water. Backhoe removal is also limited by the arm reach of the backhoe model used. Relatively steep slopes will have a short distance from shore to water and would be technically feasible to excavate. Relatively flat slopes will have a long distance from shore to water. In this case the size of the equipment may need to be larger with a longer reach to fulfill excavation needs. If the absolute distance from shore to water is too great for the necessary equipment to fully reach the target sediment, removal by dredging may prove technically feasible. During winter pool, the impounded water in the reservoir is below some of the area of sediments targeted for removal, and resuspension of contaminated sediments in the water column cannot occur. However, areas that are currently being excavated are exposed to precipitation and run-on. This water can potentially become contaminated by coming into contact with uncovered sediments and may require treatment and/or disposal, depending on the level of contamination encountered and leached. The intensity of the precipitation can also cause uncontrolled erosion and potential migration of contaminated sediments. To control erosion, a daily cover such as a tarp is recommended to divert direct precipitation from entering the work site. Silt fencing should be tied to staked straw bales installed at the top of the slope to divert run-on. As a precautionary measure, silt fencing anchored at the toe of the slope adjacent to the water will catch any sediments not otherwise contained. See Figure 9.3 for excavation details.

The following assumptions were made for excavation:

- physical access is not a problem either from the land side or from the water side, whichever is necessary;
- most shorelines are 3:1 slope or flatter; and
- operations would be performed during winter pool.

Dredging. The dredging of the deep sediments [below 223 m (733 ft)] will take place as described in Alternative 3.



JE	Fig. 9.3	Poplar Creek - Alternative 4 (removal detail) DOE - Clinch River OU - Oak Ridge, Tennessee	DOCUMENT ID: 35H830 0053-10 / RI/FS	DRAWING ID: 96-11760 CDR	DRAWING DATE: JUNE 10, 1996 Ba

10. DETAILED ANALYSIS OF ALTERNATIVES

This chapter presents information used for selecting an alternative and preparing a ROD. Section 10.1 describes the NCP criteria for each alternative. Section 10.2 presents results of the analysis of each alternative. Following the individual analysis, a comparative analysis of the alternatives is presented in Section 10.3. This analysis highlights key advantages, disadvantages, and trade-offs of each alternative.

10.1 CRITERIA FOR ANALYSIS

The CERCLA criteria used to compare the alternatives are described as follows.

10.1.1 Overall Protection of Human Health and the Environment

Overall protection of human health and the environment addresses whether an alternative eliminates, reduces, or controls threats to public health and the environment.

10.1.2 Compliance with ARARs

Compliance with ARARs addresses whether an alternative meets federal and state environmental laws and regulations. The ARARs for the CR/PC OU are provided in Appendix D.

10.1.3 Long-Term Effectiveness and Permanence

Long-term effectiveness considers the ability of an alternative to protect public health and the environment long after remedial action is complete.

10.1.4 Reduction of Toxicity, Mobility, or Volume Through Treatment

Reduction of toxicity, mobility, and volume through treatment evaluates an alternative's use of treatment to reduce the harmful nature of contaminants, the contaminants' ability to move in the environment, and the amount, or volume, of contamination present.

10.1.5 Short-Term Effectiveness

Short-term effectiveness considers the time needed for an alternative to achieve remedial response objectives and the risks posed to workers, residents, and the environment during the implementation of the remedial action.

10.1.6 Implementability

Implementability addresses the feasibility of an alternative from a technical and administrative standpoint.

10.1.7 Cost

Cost considers the amount of money it takes to design, construct, operate, and maintain the alternative.

10.1.8 State Acceptance

State acceptance addresses TDEC comments concerning the alternatives considered.

10.1.9 Community Acceptance

Community acceptance addresses written and oral public comments on the alternatives being considered. A "Responsiveness Summary" to public comments will be included in the ROD.

10.2 INDIVIDUAL ANALYSIS

10.2.1 Analysis of Alternative 1—No Action

Under Alternative 1, no further action would be taken at the site. The contaminated sediment would remain as is without any removal, treatment, containment, or mitigating technologies being implemented. Institutional controls are assumed not to exist, and no provision is made to monitor sediments, surface water, or biota. This alternative provides a baseline for comparison purposes.

10.2.1.1 Overall protection of human health and the environment

The no action alternative does not meet the RAOs or adequately protect human health and the environment. Specifically, no action does not provide the controls necessary to protect (1) benthic organisms from exposures to some sediment contamination, (2) humans (under a residential scenario) from exposures to some shallow sediments, (3) humans from fish consumption risks, or (4) human and ecological receptors from future exposures to potential contaminant disturbance (e.g., dredging for land application) or migration.

There are no short-term risks associated with implementing this alternative because no actions would occur.

10.2.1.2 Compliance with ARARs

This alternative is not protective of human health or the environment.

10.2.1.3 Long-term effectiveness and permanence

The no action alternative would not provide an effective or permanent long-term solution for the OU. The no action alternative would not provide controls for existing contamination and would not reduce existing site risks.

Investigations have shown current potential adverse effects to ecological receptors. Under the no action alternative, currently identified risks to benthic organisms from shallow and deep sediments in seven areas would continue.

Alternative 1 is not protective to humans who may catch and eat unacceptable amounts of fish from the OU. Exposure of humans and piscivorous wildlife to contaminated fish would remain at current risk levels under current conditions. Should concentrations of existing contaminants increase or new contaminants be introduced to the system, risks to humans and piscivorous wildlife may increase as a result of either new contaminants in fish populations or fish populations in uncontrolled areas being consumed at unacceptable quantities.

In the absence of monitoring, no action would not protect humans or the environment should sediments be disturbed and redistributed in the OU or perhaps elsewhere downstream. Exposure of contaminated sediment and biotic populations may increase in response to increases of currently known contaminants or new contaminants introduced to the system going undetected, thus potentially increasing human health and/or ecological risks.

10.2.1.4 Reduction of toxicity, mobility, or volume through treatment

There would be no reduction of toxicity, mobility, or volume of contaminants through implementation of Alternative 1 because this alternative does not include treatment.

10.2.1.5 Short-term effectiveness

Under the no action alternative, no remedial action would be taken; therefore, short-term effectiveness criteria do not apply. Risk to humans and the environment would remain at current baseline risk assessment and baseline ecological risk assessment levels.

10.2.1.6 Implementability

Because no remedial action would be taken under Alternative 1, it is implementable. There would be no difficulties nor uncertainties with construction. The no action alternative is included in the individual analysis of alternatives as a baseline comparison.

10.2.1.7 Cost

There are no costs associated with Alternative 1.

10.2.2 Analysis of Alternative 2—Institutional Controls and Advisories

The implementation of this alternative would include, continuance of public advisories, permit programs, monitoring, and surveillance and maintenance. Engineering detail in Chapter 9, Section 9.4.3, provides the basis for cost-estimating purposes. If this alternative is selected, the substantive components of the monitoring plan would be finalized during the design phase, consistent with the requirements of the ROD.

10.2.2.1 Overall protection of human health and the environment

Alternative 2 meets the human health RAOs listed in Chapter 8 and provides some minimal protection to the environment. Alternative 2 maintains the existing institutional controls that provide this protection.

Continued federal ownership of the site would restrict future on-site residential and agricultural land uses that could result in direct exposure to near-shore sediments through intrusive actions, including dredging.

No risks to workers are expected during implementation of this alternative. Over time, samples would be taken from the surface water, sediments, and biota. With appropriate health and safety controls, none of these activities would result in elevated health risks to workers.

Environmental risks would remain at current levels for sediment contamination, and the environment would be protected from undetected increases in contaminant input with long-term monitoring of site conditions. The restrictions on sediment disturbance will protect ecological receptors by minimizing contaminant migration while natural sedimentation decreases the bioavailability of the peak contamination sediment layer.

10.2.2.2 Compliance with ARARs

This alternative would comply with all ARARs and to be considered (TBC) guidance. TBCs for Alternative 2 are the long-term management plans for the residual radioactive material and for the management of contamination left in place, as found in DOE Order 5400.5(IV)(6)(c) and 40 CFR 300.430(e)(3), respectively. Identified TBCs (Appendix D) would be met by this alternative. Compliance with AWQC for mercury and arsenic will be accomplished at a later time.

10.2.2.3 Long-term effectiveness and permanence

Alternative 2 is not effective at reducing current exposures of benthic organisms to sediment contaminants. Seven areas were identified as having current ecological risks that exceed the RGOs for the organisms. These particular ecological risks would continue unabated until mitigated by naturally occurring processes (see Sect. 9.1.1.1). The time frame for mitigation by natural processes has not been quantified or modelled, but can be significant.

Alternative 2 meets the RAOs of preventing future residential exposure to sediments. Long-term protection of human health would be provided by sediment-disturbing permit requirements, and long-term monitoring. Unacceptable future risks identified by monitoring at the site would be reduced by eliminating the exposure pathway. For instance, in an area previously below risk levels found to have contaminants in excess of risk thresholds, the exposure pathway can be removed by implementing additional access restrictive controls. Environmental protection would be enhanced by monitoring the system for changes that could trigger actions or other controls such as those in Alternative 3 or 4 to protect biota (see Table 9.1). Limitations set forth in sediment-disturbing restrictions would also protect the environment from exposure to deep sediments that could be released and become bioavailable. Natural sedimentation would continue to cover the most contaminated sediments and would provide some protection for human and ecological receptors.

Under Alternative 2, institutional controls and advisories would be reliable in the long term if coordination of all concerned regulating bodies, in a cooperative effort to keep the deep sediments in place and undisturbed, is maintained. These sediments provide little risk to humans because there are no exposure pathways under current conditions.

Institutional controls would be reliable for preventing unauthorized disturbance of sediments and potential future exposure. Posted warnings would be reliable for advising people of the potential health hazards of fish consumption. Periodic monitoring of surface sediment samples, water, and biota will provide a reliable means to identify changes in the reservoir system, such as new sources or sediment migration. This alternative is only as effective as the implementing agency's ability to enforce the alternative or the residents' willingness to comply with the options outlined in the alternative.

Consistent with the required 5-year CERCLA reviews conducted by DOE and the regulatory agencies, performance of the controls would be monitored. The 5-year reviews may indicate the need for components of this alternative to be maintained, modified, or replaced.

10.2.2.4 Reduction of toxicity, mobility, or volume

There is no treatment as part of Alternative 2; therefore, there would be no reduction of toxicity, mobility, or volume of contaminants.

10.2.2.5 Short-term effectiveness

Alternative 2 would protect the community during implementation through the use of administrative controls. The contaminated sediments would not be disturbed; therefore, there would be no increased potential for off-site migration of contaminants as a result of remedial activities.

Risk of occupational injury is considered negligible. It is anticipated that workers conducting monitoring would use standard monitoring procedures as necessary to provide worker safety.

This alternative prevents increased risk to environmental receptors in the short term by not disturbing the contaminants. Environmental risk would remain at current levels. As stated earlier, no reduction of toxicity, mobility, or volume, except by natural attenuation, would occur under this alternative. No environmentally sensitive resources would be affected by this alternative. The benthic organisms would not be destroyed and their habitat would not be disturbed by this alternative.

The duration of sign posting activities will vary according to the extent of the area selected for implementation. This alternative could be in place and fully implemented within 6 months.

10.2.2.6 Implementability

Alternative 2 is technically feasible to implement. All activities would be performed using conventional methods. Maintaining public advisories and permit programs, along with implementation of long-term monitoring would be easily implementable.

Alternative 2 would be conducted on site and would not require any permits. There are no known administrative barriers or difficulties associated with implementation of Alternative 2.

Materials and services required for implementation of Alternative 2 are readily available.

10.2.2.7 Cost

The total project present worth cost of Alternative 2 is approximately 3.6 million. Table 10.1 provides a detailed breakdown of the escalated project costs. More detailed information is provided in Appendix B.

Direct capital cost. The total escalated cost for the direct capital cost is estimated at $69,000. Direct capital cost includes public advisories and permit programs.

Indirect capital cost. Total escalated indirect capital cost is estimated to be $71,000. Indirect capital cost includes remedial action (RA) integration and RA work plan.

Table 10.1 Cost estimate for Alternative 2 for the Clinch River/Poplar Creek, Operable Unit, Oak Ridge Tennessee

Project cost item		Cost ($thousands)[a]
	Capital cost	
Direct cost		
Public advisories		63
Permit programs		6
	Direct cost (rounded)	69
Indirect cost		
Remedial action integration		40
Remedial action work plan		31
	Indirect cost total (rounded)	71
	Total capital cost	140
	Monitoring and maintenance cost	
Monitoring		7,926
	Combined cost	8,066
Contingency–25 percent		2,017
	Total project escalated cost	10,083
	Total project present worth[b]	3,611

[a]Escalated
[b]Present worth cost based on 30-year present value, 7 percent discount rate.

Note: Costs presented in table are rounded.

Monitoring and maintenance. Total escalated cost for monitoring and maintenance (M&M) is estimated to be $7.9 million. M&M cost includes monitoring of fish, water, and sediment for approximately 30 years. Monitoring would support the required CERCLA 5-year reviews conducted by EPA.

Contingency. Total escalated cost allowed for contingency is $2.0 million.

10.2.3 Analysis of Alternative 3—Source Containment, Removal, and Disposal

Under Alternative 3:

- Shallow or near-shore sediments are capped in those areas with contamination exceeding human health or ecological RGOs. If the slope of the remediation area is too steep for capping, physical barriers will prevent access to the area.
- Deep sediments are dredged in those areas with contamination exceeding ecological RGOs.
- The remaining sediments are monitored and protected from disturbance with the use of institutional controls (e.g., deed restrictions, public advisories, permit programs) similar to those outlined for Alternative 2. Fish consumption would also be controlled.

Capping and removal would be implemented in the seven areas identified in Figures 9.1 and 9.2.

The geomembrane cap is expected to eliminate releases of contaminants by wind or water erosion and direct contact with surface and subsurface contaminants by humans and biota. The cap would consist of an impervious geomembrane layer protected by a geosynthetic fabric. The outermost layer, consisting of riprap, will serve to weight the geomembrane and protect it from wind and water erosion and biotic intrusion.

During removal of the deep sediments, the potential for surface water run-on, wind and water erosion, and resuspension of sediments is mitigated using engineering and administrative controls. Once the sediments are dredged, the sediments are dewatered, treated as necessary to meet RCRA requirements or disposal WAC, containerized, and then placed in an on-site or off-site disposal facility.

As noted above, Alternative 3 consists of containment, deep sediment removal, and institutional components; the containment is unique to Alternative 3, but the deep sediment dredging is common to both Alternatives 3 and 4, and the institutional controls are common to Alternatives 2, 3, and 4. The presentation of the detailed evaluation for Alternative 3 in this section takes advantage of the similarities between Alternative 3 and the other alternatives. As appropriate, the reader is referred to Alternative 4 for evaluations dealing with deep sediment, and to Alternative 2 for evaluations dealing with institutional controls. This means that the CERCLA criteria evaluations in this section deal primarily with those aspects unique to containment.

10.2.3.1 Overall protection of human health and the environment

Alternative 3 meets the RAOs developed for CR/PC OU by ultimately minimizing direct contact of humans with the contaminants in shallow near-shore sediments by physically isolating contaminated sediments, rendering them unavailable for exposure. The ecological RAOs are met by the isolation of shallow sediments and removal of deep sediments. These sediments would no longer be in a pathway for biota to contact, thus reducing long-term ecological risks. The existing benthic populations would be destroyed, but they may recover in 5–10 years. Recovery will likely take much longer in the capped areas because of an absence of substrate to burrow into.

10.2.3.2 Compliance with ARARs

Although DOE Orders are not ARARs, the radiation exposure limit of 100 mrem/year for exposure of the public defined in DOE Order 5400.5 would be met. This Order is the primary contaminant-specific TBC for this alternative.

Under USACE authority, Nationwide Permit 38, "Cleanup of Hazardous and Toxic Waste," requires wetlands delineation as part of notification for discharges into special aquatic sites. Because source containment will occur in an area of Poplar Creek designated as "navigable" waters, the substantive requirements may be applicable to all activities related to capping and removal activities. If adverse effects to the jurisdictional wetlands are determined, consultation with USACE and mitigation may be required (e.g., banking) under 33 CFR 330, Appendix A (C)(13). The substantive requirements for nationwide Permits 13 (Bank Stabilization), 18 (Minor Discharge), and possibly 19 (Minor Dredging), if USACE assumes authority over these activities, would need to be met for dredging.

All remedial activities for Alternative 3 would comply with the Tennessee Water Quality Control Act by being as conservative as possible in disturbing soil, sediment, and vegetation. TDEC AWQC are not met by this action but will be adressed at a later time.

No historic sites or sites eligible for listing on the National Register of Historic Places are known to occur in the area. Cemeteries and known archaeological sites are located near the Poplar Creek containment areas. All containment and removal activities would allow for at least a 15-m (50-ft) buffer around the cemeteries. If additional archaeological resources are discovered during installation of fences or access roads or during remediation, appropriate measures would be taken to comply with the Archaeological Resources Protection Act. The state archaeologist would be contacted and an assessment performed to determine the proper course of action to take.

Wetland and floodplain areas would be affected by containment and removal activities. Discussions of such are found in the "Environmental Impacts" portion of Section 10.2.3.5. No other environmentally sensitive resource has been identified in or near the remedial areas.

Reasonable precautions would be implemented to prevent particulate matter from becoming airborne during soil disturbance activities. Fugitive dust, as a visible emission beyond property boundary lines, would be limited to no more than 5 minute/hour or a total of 20 minute/day (TDEC 1200-5-8-.010). Control of sediment erosion and turbidity would comply with 40 CFR 122 and TDEC 1200-4-10-.05 through the use of silt fences, straw bales, or temporary perimeter drainage ditches.

If excavated or removed soil or sediment is determined to be a hazardous, mixed, and/or PCB-contaminated waste, storage, treatment, and/or disposal regulations under the RCRA and/or the Toxic Substances Control Act (TSCA) would be triggered. Portions of DOE Order 5820.2A would be TBCs for disposal of any LLW sediment. The U.S. Department of Transportation (DOT) regulations govern shipping, packaging, and transport of hazardous materials.

Further details of ARARs and TBCs, along with the complete list of ARARs and TBCs, are found in Appendix I.

10.2.3.3 Long-term effectiveness and permanence

Magnitude of residual risks. Shallow sediment risks are significantly reduced as all pathways are eliminated both currently and for the potential on-site resident. Deep sediment removal eliminates the pathway for ecological receptors. Fish consumption risks remain the same as in Alternative 2, institutional controls and advisories.

Adequacy and reliability of controls. Alternative 3 is adequate in meeting RAOs. The cap would be designed so that all slopes would be of proper grade for optimum wear and long-term structural stability. Lack of periodic inspections and timely repairs could result in the failure of the geomembrane cap and associated uncontrolled access to contaminants from McCoy Branch Embayment and/or Poplar Creek. The reliability of containment is very good as long as repairs are made and cap integrity remains intact.

Alternative 3 is similar to Alternative 4 in that institutional controls and monitoring are needed for long-term protection by preventing mechanical disturbance of river sediments, by controlling fish consumption, and by monitoring for contaminant migration or contributions from sources of contamination on ORR upstream of the remediation areas. Upstream contaminant sources (if not addressed by other actions) would continue to provide a source of risk to human and ecological receptors.

10.2.3.4 Reduction of toxicity, mobility, or volume through treatment

Alternative 3 would reduce mobility of the waste through containment and removal. The process of dewatering may result in a decrease of volume and /or toxicity.

10.2.3.5 Short-term effectiveness

Protection of the community during remedial action. Alternative 3 would protect the community during implementation through access controls (e.g., physical barriers, administrative controls) and engineering controls during construction. Engineering controls for run-on, direct precipitation, and runoff would be used to minimize sediment erosion; therefore, minimal potential for off-site migration of contaminated sediments during remedial activities would occur. There is the potential for dust migration from earth-moving activities that would occur during placement of the cap. In addition, considering the moisture content of the sediments, fugitive dust emissions by construction activities is expected to be minimal. However, appropriate dust control methods (e.g., wetting of the sediment) would be implemented to minimize the off-site migration of dust to ensure protection of the community.

Traffic is expected to increase during implementation as access road construction materials are brought into the site. Most of the traffic would occur on DOE properties and not in the surrounding communities. Traffic would increase slightly as dump trucks report to the site. There would be no change in the worker population of the Oak Ridge area during implementation because the existing work force could be used.

Protection of workers during remedial action. The short-term risks associated with Alternative 3 are higher than those associated with Alternative 2 because of greater potential for direct contact with contaminated sediments. However, this contact would be minimized by the implementation of a site-specific health and safety plan. Fugitive dust emissions from heavy equipment movement would be controlled by using surface wetting or dust suppressants to minimize inhalation risks.

Environmental impacts. Short-term impacts to ecological receptors would be higher under Alternative 3 than for Alternative 2 during cap installation and deep sediment removal. The environmental effects would be disruption of habitat, death of most or all benthic organisms in the area, and potential sediment resuspension during construction and dredging activities.

Disruption of habitat would be permanent for aquatic plants and most benthic organisms in shallow sediments. The riprap or rock layer on the cap surface will not be suitable habitat for the benthic organisms that used to inhabit the remediation areas. It may take several decades before the area regains suitable habitat through natural sedimentation. Deep water plants and animals should eventually recolonize the area following the completion of sediment removal. Waterfowl, amphibians, reptiles, and piscivorous animals are expected to recover and return to remediated areas. Avoiding destruction of habitat would be the first consideration in remediating these areas, followed by relocation of affected species to a suitable habitat elsewhere. Relocation may only be possible for a very limited number of species and is not an option for the benthic organisms that are currently at risk.

The effects of containment on the floodplain would be minimal. Some grading and slope alteration will be inevitable; however, existing shoreline slopes would remain close to the current natural slope and cap construction should conform to the lay of the slope wherever possible. If areas are encountered that cannot be capped without significantly altering the floodplain, then other remedial options would be identified in a floodplain analysis.

Wetland areas would be encountered in the area intended for implementation of capping and removal. Construction in a wetland to accommodate the cap construction, associated road improvements, etc., would be avoided to the extent possible. However, wetland areas throughout the subreach could be destroyed, causing adverse effects. A wetland analysis will be required if this alternative is chosen as the preferred alternative. Potentially, constructing a wetland elsewhere to replace the loss of habitat at the capped site could be required for mitigation.

Because the contaminated sediments would be disturbed to some extent, there is potential for contaminants to migrate from the site during implementation from erosion and fugitive dust. Appropriate engineering controls will minimize this effect. Appropriate uses of silt fences, staked straw bales, and tarps during construction would control the effects on surface water quality from fugitive silt caused by construction. Dust suppression measures, such as surface wetting, will minimize fugitive dust. During the dredging of deep sediments, silt screens and curtains will be used to control release of sediments.

Accidental spills or leakage of engine fuels, oils, hydraulic fluids, and coolants from equipment could result in minor soil contamination. Equipment exhaust is anticipated to have negligible effects on air quality. Standard construction management practices such as equipment maintenance and timely removal of spills would minimize the environmental effect of implementation activities.

Duration of remedial activities. Construction activities under Alternative 3 are anticipated to last 2.5 years. Administrative duration is the same as Alternative 2 for institutional controls.

10.2.3.6 Implementability

Alternative 3 is technically feasible to implement. All activities would be performed by conventional methods. Construction of the geomembrane cap and appropriate anchor trenches would be straightforward, but only during the period of the year when the reservoir water level is lowered to winter pool. Construction at any other time will be substantially more difficult, if at all possible. Thus preparation and material staging will be very important to meet this critical timing. The dredging and dewatering activities are implementable, although the dredging apparatus may need to be transported from another state by river and lock.

Although Alternative 3 would be conducted on DOE property, waterways adjacent to the ORR are co-managed under the statutory authority of other federal and state authorities. Permits would be needed to regulate waterway traffic as the areas of concern for capping are in the areas managed by TVA, USACE, and TDEC. Dredging activities may impede navigation for several months.

Materials required for implementation of Alternative 3 include geomembrane, geotextile, and riprap; earth-moving and compaction equipment; silt fences; dredging equipment; dewatering equipment; water tanks; tanker trucks; and sampling equipment. All of these are readily available materials.

10.2.3.7 Cost

The total project present worth cost of Alternative 3 is approximately $109.6 million. Table 10.2 provides a detailed breakdown of the escalated project costs. More detailed information is provided in Appendix H.

Direct capital cost. The total escalated cost for the direct capital cost is estimated at $93.9 million. Direct capital cost includes public advisories, permit programs, and source containment.

Indirect capital cost. Total escalated indirect capital cost is estimated to be $1.5 million. Indirect capital cost includes RA integration, RD work plan, RD report, RA work plan, and contingency.

Monitoring and maintenance. No monitoring and maintenance cost are included in Alternative 3.

Contingency. Total escalated cost allowed for contingency is $23.9 million.

10.2.4 Analysis of Alternative 4—Source Removal

Seven areas, all of which are within ORR and controlled by DOE, were identified for remediation based on unacceptable ecological risk. Characterization during design will further delineate and bound these areas. Under Alternative 4:

- shallow or near-shore sediments [i.e., those sediments down to elevation 223 m (733 ft) (msl)], which is 0.6 m (2 ft) below low pool] are removed by excavation at all seven remediation areas,

- deep sediments [i.e., those sediments below elevation 223 m (733 ft) (msl)] are removed by dredging for the seven remediation areas, and

- the remaining sediments are monitored and protected from disturbance by using institutional controls (e.g., public advisories, permit programs) similar to those outlined for Alternative 2.

During removal, the potential for surface water run-on, wind and water erosion, and resuspension of sediments is mitigated using engineering and administrative controls. Once the sediments are removed, the sediments are dewatered, treated as necessary to meet RCRA requirements or disposal WAC, containerized, and then placed in an on-site or off-site disposal facility.

Table 10.2. Cost estimate for Alternative 3 for the Clinch River/Poplar Creek Operable Unit, Oak Ridge, Tennessee

Project cost item		Cost ($thousands)[a]
	Capital cost	
Direct cost		
Public advisories		63
Permit programs		6
Source containment/removal		93,821
	Direct cost total (rounded)	93,890
Indirect cost		
Remedial action integration		827
Remedial design work plan		140
Remedial design report		482
Remedial action work plan		94
	Indirect cost total (rounded)	1,543
	Total capital cost	95,433
	Monitoring and maintenance cost	
Contingency–25 percent		23,858
	Total project escalated cost	119,291
	Total project present worth[b]	109,621

[a]Escalated
[b]Present worth cost based on 30-year present value, 7 percent discount rate.

Note: Costs presented in table are rounded.

10.2.4.1 Overall protection of human health and the environment

Alternative 4 is protective of human health and the environment and meets the RAOs developed for CR/PC OU (see Chapter 8) by removing sediment contaminated above the RGOs. No sediment exposure pathways to ecological receptors will remain after removal. The removal will destroy the existing benthic organisms.

In the short term (i.e., the duration of the remedial action), potential risks to workers increase because of the removal and subsequent processing of large volumes of contaminated sediments. A project health and safety plan would address mitigative measures designed to protect the remediation worker from adverse exposures.

Potential risks also exist in the short term for the community (i.e., K-25 Site personnel and the public, which uses the rivers for recreational purposes). Proper security and safeguards at the remediation area control access to the project site and possible direct contact and inhalation of contaminants. The short time frames and distances for material transport by trucks to on-site disposal areas minimize the transient exposures and possibility of vehicular accidents from loaded trucks. Exposure risks from sediment removal, treatment, and disposal are not expected to exceed the EPA risk targets for noncancer health effects to the community. No residents live near the areas to be remediated, and therefore incremental exposures to residents from inhalation of airborne contaminants would be negligible.

10.2.4.2 Compliance with ARARs

The ARARs for Alternative 4 are the same as for Alternative 3, and they will all be complied with during and after implementation of Alternative 4.

10.2.4.3 Long-term effectiveness and permanence

Magnitude of residual risks. After sediment removal, treatment, and disposition have occurred in Alternative 4, the residual risk for known contamination in both shallow and deep sediments will be below the EPA threshold of 1×10^{-4} or 1.0 HI. Continuing release of contaminants from upstream sources may degrade the remediated areas.

Removed materials will be disposed in approved, permanent repositories and will not be returned to the reservoir system with the exception of dewatering effluent. The effluent will be treated before release, if needed, to meet NPDES release criteria. The residual risk from the effluent will be below that associated with approved release criteria.

Adequacy and reliability of controls. As long as sources of contamination remain on ORR upstream of the remediation areas, there can be no completely reliable method of remediation. Contamination may enter the reservoir system from secondary sources such as surface runoff or groundwater. Heretofore hidden or unknown pockets of risk-significant contamination buried in river sediments may be released to the river and carried downstream during storm events or natural rechanneling. Additionally, the areas of shallow sediment removal may become recontaminated as a result of flooding. In these situations, the long-term protectiveness afforded by the initial cleanup decreases in proportion to the extent and level of recontamination. If the recontamination levels exceed the RGOs, a second cleanup may be required.

For a period of time, Alternative 4 will adequately and reliably meet the RAOs stated in Chapter 8 for known contamination. Assurance of long-term protection, however, will be provided by institutional controls and monitoring. Institutional controls that are maintained would be reliable for preventing unauthorized disturbance of sediments and potential future exposure. Posted warnings would advise the public of the potential health hazards of fish consumption. Periodic monitoring of surface sediment, water, and biota will provide a reliable means to identify changes in the reservoir system, such as new sources or sediment migration.

10.2.4.4 Reduction of toxicity, mobility, or volume through treatment

Although the primary focus of Alternative 4 is on removal rather than treatment, treatment is evaluated as required to facilitate meeting RCRA requirements, disposal WAC, or release requirements for excavated sediments and soils.

Under Alternative 4, excavated and dredged sediments would be dewatered by decanting water from the sediment placed on the dewatering pad and then filtering the effluent with appropriate filter media. Decanting and filtration are forms of physical treatment that merely separate the liquid and solid media, but do not fix the contaminants in either of the host media. This collected aqueous waste stream would be managed either by releasing it to the river, if contaminant concentrations are below NPDES limits, or by treating it to meet NPDES release limits and then properly disposing of the residuals. The latter option reduces the mobility of the contaminant. Sampling and analysis of the water would reveal which management option would be appropriate.

Sediment waste streams will be disposed of at a permanent on-site or off-site repository. Based on analyses of the sediment core samples, it is expected that the waste streams will be radioactively contaminated above background levels, but will not be classified as RCRA-hazardous or mixed waste. However, if RCRA-hazardous or mixed waste streams are found, they will be treated (e.g., through solidification or thermal desorption/destruction) to meet land disposal restrictions, thereby reducing contaminant mobility, and in some cases, contaminant toxicity.

10.2.4.5 Short-term effectiveness

Protection of the community during remedial action. Community in this context refers to nonproject K-25 Site personnel and recreational users of the rivers addressed in this study. All the remediation sites are located within ORR and, as such, are remote from any residential or commercial areas. However, the areas are accessible by recreational users for fishing, hiking, or similar activities.

Under Alternative 4, excavation, treatment, and disposal are planned for the remediation areas, and exposures during these activities would be possible outside of the immediate project-related areas. Fugitive dust would transport contaminants outside the remediation area, where it could be inhaled by members of the community. The short-term inhalation risks associated with sediment excavation activities have not been quantified. However, standard mitigating measures would be implemented to minimize uncontrolled and significant releases. In addition, during much if not all of the material handling, the sediment will be muddy or caked from moisture in the sediment and will not easily be airborne. Therefore, risks to the community exposed to dust would not be expected to exceed EPA risk targets. Once the sediments are packaged or covered for transportation, airborne risks should be negligible.

Migration of the contaminants to areas outside the remediation area could also occur through sediment resuspension and downstream dispersion during removal activities. Such resuspension and transport downstream could result in exceedances to surface water quality criteria and the destruction of benthic invertebrates. Silt curtains installed around the perimeter of the work site are proposed to mitigate this transport pathway.

Remediation involves the transportation of contaminated sediment to disposal sites. (Staging and treatment areas are expected to be located near the excavation or dredging site.) Because the trucks would be covered immediately and decontaminated before leaving the remediation area, exposures to hazardous materials would not be expected to occur during transport. In addition, any spillage from the truck should be minimal if the truck is packaged or covered properly. In the event spillage occurs, it could be easily cleaned up and reloaded on the truck. The primary risk from material transport is from vehicular accidents rather than from contaminant exposure. A waste transportation plan including routes, spill prevention, and cleanup will be prepared before remedial implementation.

Protection of workers during remedial action. A health and safety plan for the project would be developed to ensure that risks to workers do not exceed EPA target levels.

Environmental impacts. Under Alternative 4, an estimated in situ volume of 173,172 m^3 (226,500 yd^3) of sediment would be excavated, treated, and disposed of for remediation. This is a significantly larger volume of processed sediment than in Alternative 3. The remediation areas would be restored to roughly original grade, but no borrow soil would be added for site restoration.

Disturbance of the sediments would have potential short-term impacts on the surface water. Potential increases in concentrations of contaminants and total suspended solids are expected to be minimal and within regulatory guidelines, because best management practices and appropriate engineering controls would be used to prevent adverse impacts to water quality. Contaminants released to the rivers by treated dewatering effluent would be below the NPDES limits.

Excavation and handling of untreated sediment could be an air pathway to potential receptors. Removal operations such as loading, dumping, grading, and excavation would cause increased concentrations of contaminated dust, if not properly controlled. Storage piles of dewatered sediment and large areas of bare sediment could also serve as sources of fugitive emissions. Control methods, such as wetting and limiting the number of exposed storage piles, would significantly lower emissions from these activities. Although not quantified, it is expected that off-site risk from controlled fugitive dust emissions from excavation and sediment handling activities would be within acceptable limits. Emissions from heavy equipment operation (i.e., fuel combustion) would be negligible and would have an insignificant effect on local or regional air quality.

Disruption of habitat should be temporary in the remediation areas. The benthic organisms, as well as aquatic plants and other vegetation, would be destroyed in the excavated areas, treatment areas, along the shoreline where heavy equipment is used, and in floodplain or shoreline areas where access roads are provided. However, the disruption of localized habitat for the plants and organisms should be temporary, and the plants and organism population may begin to thrive again after a period of several years. The remediation activities are not anticipated to have any effects on a regional level.

The effects of source removal on the creek or river banks would be minimal but long lasting. The removal inevitably alters the existing shoreline by the depth of material removed, but because a meter or less of sediment is expected to be removed, the material remaining should have a slope that will be close to the current natural slope.

Wetland areas could be encountered in the remediation areas. If so, the portion of the wetlands that overlaps the construction area would be destroyed under Alternative 4. The remaining portion of the wetlands outside the construction area could be indirectly impacted because of the amount of work that would occur adjacent to the wetlands. A wetland analysis will be required if this alternative is chosen as the preferred alternative. Constructing a wetland elsewhere to replace the loss of habitat at the remedial site would potentially be required.

Before construction activities begin, the remediation areas will be searched for the tall larkspur (*Delphinium exaltatum*), a state-listed endangered species and a federal candidate for listing as threatened or endangered. If found, consultation with the state would help determine an appropriate protective measure.

Cemeteries and known archaeological sites are near the Poplar Creek remediation areas. All construction activities will allow for at least a 15-m (50-ft) buffer around the cemeteries and, if archaeological resources are discovered during any construction activities, appropriate measures will be taken to comply with the Archaeological Resources Protection Act.

Duration of remedial activities. The short-term remedial action objectives would be achieved upon completion of these activities. The actual time to complete this alternative, after approval by the regulatory agencies, would include approximately 12 to 18 months for additional characterization, remedial design, and the associated approval of the remedial action work plan before beginning the remedial action. Therefore, after final regulatory approval of the ROD, it would take approximately 4–5 years to complete the action, not including long-term environmental monitoring.

10.2.4.6 Implementability

Technical feasibility. The technical feasibility for Alternative 4 has been demonstrated at other locations around the country. Removal and restoration use standard construction techniques and good engineering practices to contain turbidity and reduce runoff and dispersal. Providing access roads will be a minor obstacle. Because the sediment contains radioactive contaminants, the removal techniques are integrated with occupational health physics programs and other safety regimen that are also well established. Additional characterization after removal and restoration will confirm the degree to which the remediation areas have been cleaned up.

Ex situ sediment treatment for dewatering uses standard processes such as decanting and filtration. If treatment of the separated water is necessary, ORR has several water treatment plants available for use.

Disposal of dewatered sediments is available at on-site (on ORR) and off-site disposal facilities. These facilities have ample capacity for the range of volumes currently predicted. These facilities are required under the licenses or permits to adequately contain the waste and provide comprehensive environmental monitoring.

Administrative feasibility. Alternative 4 is administratively feasible to implement; the fact that all the remediation areas and some potential disposal facilities are within ORR and under DOE control significantly enhances this feasibility. Nevertheless, the sediment disturbance activities will need to be coordinated with, at a minimum, EPA, USACE, TVA, and TDEC. The removal will need to be performed during the period of year when the reservoir water level is lowered to winter pool. Removal at any other time would be substantially more difficult and costly.

Excavation and dredging activities would unavoidably disrupt and inconvenience recreational activities in the remediation areas. Dredging would block river traffic because of the cables extending to the shorelines. The fact that removal will occur during winter pool when recreational use is down will minimize these impacts. Short-term impacts on aesthetic quality could be expected as a result of removing vegetation and a sediment layer, although replanting and eventual natural restoration would occur. However, DOE would coordinate with other federal and state agencies to minimize short-term disturbances from removal activities.

Waste transportation impacts would be minimized by cooperation with local and state authorities to determine the most appropriate haul routes and trip times to reduce traffic congestion, accident potential, and road deterioration. This will be especially critical if the sediment is disposed off site.

The removal of contaminated sediment under Alternative 4 reduces estimated human health and ecological risks below the RGOs. This alternative would not, however, allow the unrestricted selling, buying, and development of these properties in accordance with normal market pressures, because the properties are on ORR. Because DOE is expected to maintain ownership and control of DOE-owned property for the foreseeable future, no impacts to the institutional environment would be likely to result from this alternative on ORR.

Availability of services and materials. All of the services and materials required to implement Alternative 4 are available. Advance planning is critical, however, for scheduling the services and obtaining the materials, particularly with regard to timing (winter pool) constraints and obtaining any needed permits for disturbing sediments and transporting wastes.

10.2.4.7 Cost

The total project present worth cost of Alternative 4 is approximately $123.5 million. Table 10.3 provides a detailed breakdown of the escalated project costs. More detailed information is provided in Appendix B.

Direct capital cost. The total escalated cost for the direct capital cost is estimated at $109.6 million. Direct capital cost includes deed restrictions, public advisories, permit programs, and source containment.

Indirect capital cost. Total escalated indirect capital cost is estimated to be $2.0 million. Indirect capital cost includes RA integration, RD work plan, RD report, RA work plan, and contingency.

Monitoring and maintenance. No monitoring and maintenance cost are included in Alternative 4.

Contingency. Total escalated cost allowed for contingency is $27.9 million.

10.3 COMPARATIVE ANALYSIS

A summary of the comparative analysis of alternatives for CR/PC OU is presented in Table 10.4.

Table 10.3 Cost estimate for Alternative 4 for the Clinch River/Poplar Creek Operable Unit, Oak Ridge, Tennessee

Project cost item		**Cost ($thousands)[a]**
	Capital cost	
Direct cost		
Public advisories		63
Permit programs		6
Removal		109,498
	Direct cost (rounded)	109,567
Indirect cost		
Remedial action integration		1,137
Remedial design work plan		169
Remedial design report		575
Remedial action work plan		111
	Indirect cost total (rounded)	1,992
	Total capital cost	111,559
Contingency–25 percent		27,890
	Total project escalated cost	139,449
	Total project present worth[b]	123,527

[a]Escalated

[b]Present worth cost based on 30-year,present value, 7 percent discount rate.

Note: Costs presented in table are rounded.

Table 10.4. Detailed analysis summary and assessment for alternatives for the Clinch River/Poplar Creek Operable Unit, Oak Ridge, Tennessee

Alternative 1: No Action	Alternative 2: Institutional Controls and Advisories	Alternative 3: Containment, Removal, and Disposal	Alternative 4: Removal and Disposal
Overall protection of human health and the environment			
Not protective of human health or the environment because risks from exposure would continue	Benthic organisms would receive negligible risk reduction. However, the alternative would be protective of human health because the pathways of human exposure are controlled or monitored	Protective. Pathways of exposure would be significantly reduced. Higher risk sediments would be contained or removed. Institutional controls and monitoring would address sediment disturbance, fish consumption, and recontamination	Protective. Pathways of exposure would be significantly reduced. Higher risk sediments would be removed. Institutional controls and monitoring would address sediment disturbance, fish consumption, and recontamination
Compliance with applicable and relevant or appropriate requirements			
This alternative does not comply because it is not protective of human health and the environment	Would comply with action-specific TBCs through use of institutional controls. Surface water will be addressed at a later time.	Would comply with location-specific ARARs, although complicated or extensive efforts may be required to avoid adverse effects to wetlands, threatened and endangered species, floodplains, and archaeological resources. Would comply with chemical- and action-specific ARARs for containment and sediment removal. Surface water will be addressed at a later time.	Would comply with location-specific ARARs, although complicated or extensive efforts may be required to avoid adverse effects to wetlands, threatened and endangered species, floodplains, and archaeological resources. Would comply with chemical- and action-specific ARARs for removal. Surface water will be addressed at a later time.

Table 10.4. (continued)

Alternative 1: No Action	Alternative 2: Institutional Controls and Advisories	Alternative 3: Containment, Removal, and Disposal	Alternative 4: Removal and Disposal
		Long-term effectiveness	
No risk reduction occurs for either human or ecological receptors Future risks may increase if contaminated sediment is disturbed or redistributed without notice, or if existing controls are discontinued Natural sedimentation will eventually cover most contaminated sediments and make them biologically unaccessible	Negligible risk reduction occurs for ecological receptors Human health protection is provided by sediment-disturbing permit requirements, posted warnings, and periodic monitoring. Long-term continuance of these institutional measures would be required to provide long-term protection Monitoring would provide a reliable means to identify changes in the reservoir system Natural sedimentation will eventually cover the contaminated sediments and make them biologically unaccessible	Containment and removal of all sediment above RGOs would reduce unacceptable risks to human health or ecological receptors. Contaminated deep sediments would be properly disposed. Posted warnings would advise people of the potential health hazards of fish consumption The shallow-sediment cap would require long-term maintenance. Institutional controls would be required for long-term protection from residual contamination. Monitoring of sediment, water, and biota would provide a reliable means to identify changes to the reservoir system, such as sediment migration or recontamination of remediated areas by upstream sources. Capping and removal would initially have impacts on biota and habitat; impacts would range from localized and temporary, to more widespread and permanent. It is uncertain whether the benthic community will ever recover in the capped aras.	Removal of all sediment contaminated above RGOs would eliminate unacceptable risks to human health and benthic organisms. Contaminated sediments would be properly disposed. Posted warnings would advise people of the potential health hazards of fish consumption. Institutional controls would be required for long-term protection. Monitoring of sediment, water, and biota would provide a reliable means to identify changes to the reservoir system, such as sediment migration or recontamination of remediated areas by upstream sources. Removal would initially have negative impacts on biota and habitat; impacts would be localized and temporary. The benthic organisms may eventually recover.

Table 10.4. (continued)

Alternative 1: No Action	Alternative 2: Institutional Controls and Advisories	Alternative 3: Containment, Removal, and Disposal	Alternative 4: Removal and Disposal
		Short-term effectiveness	
Current risks continue without mitigation as no action is implemented for the OU	Risks to workers, the community, and the environment are minimal during implementation. (However, ecological risk remains at current levels.)	During implementation, risks from fugitive dust, downstream dispersion of resuspended sediments, and transportation accidents are minimized using administrative and engineering controls	During implementation, risks from fugitive dust, downstream dispersion of resuspended sediments, and transportation accidents are minimized using administrative and engineering controls
	No disturbance of the sediments occurs. No environmentally sensitive resources would be affected by this alternative	Capping and removal would initially have negative impacts on biota and habitat; the existing benthic community would be destroyed; impacts could range from localized and temporary, to more widespread and permanent	Removal would initially have negative impacts on biota and habitat; the existing benthic community would be destroyed; impacts would be localized and temporary
		Reduction in toxicity, mobility, or volume through treatment	
No action to reduce toxicity, mobility, or volume	No reduction in toxicity, mobility, or volume would result from implementation of this alternative	Deep sediments removed during remediation will be treated as required to facilitate meeting RCRA requirements, disposal waste acceptance criteria, or release requirements. In some cases, these treatments may reduce contaminant mobility or toxicity	Shallow and deep sediments removed during remediation will be treated as required to facilitate meeting RCRA requirements, disposal waste acceptance criteria, or release requirements. In some cases, these treatments may reduce contaminant mobility or toxicity

Table 10.4. (continued)

Alternative 1: No Action	Alternative 2: Institutional Controls and Advisories	Alternative 3: Containment, Removal, and Disposal	Alternative 4: Removal and Disposal
		Implementability	
No implementation actions required	Institutional actions could be readily implemented but would require oversight and coordination of multiple agencies or groups, as well as recurring activity over a long time period	Capping, dredging, slurry treatment, waste transportation, and disposal are well known and could be readily implemented using standard techniques and equipment. Sediment and habitat disturbance activities will need to be coordinated with multiple agencies or groups. Construction would need to occur when the reservoir water level is at winter pool	Dredging, excavating, slurry treatment, waste transportation, and disposal are well known and could be readily implemented using standard techniques and equipment. However, the volume of sediment to be removed and processed in this alternative is much greater than in Alternative 3. Sediment and habitat disturbance activities will need to be coordinated with multiple agencies or groups. Construction would need to occur when the reservoir water level is at winter pool
		Present worth cost	
$0	$3.6 million	$109.6 million	$123.5 million

RCRA = Resource Conservation and Recovery Act of 1976
RGO = remedial goal option
TBC = to be considered

10.3.1 Overall Protection of Human Health and the Environment

Alternative 1, no action, would not protect human health or the environment because no action would be taken. Unregulated sediment-disturbing activities of deep sediments would also cause concern as most of the contaminants released during the peak release years are within these sediments. Exposure to deep sediments either by direct ingestion, ingestion of crops grown on contaminated sediment, or consumption of beef or beef products raised on these crops is of concern. Environmental exposure to disturbed deep sediments are also a concern as most of the deep sediment contamination is typically unavailable to all but a few benthic organisms. Disturbance would probably increase availability of contaminants to a broad range of biota. Loss of existing public advisories could increase health risks to humans because ingestion of fish above advised levels may cause certain health problems.

Alternative 2 would protect humans from exposure to the deep sediments by regulating dredging activities, thus maintaining the existing cover. Alternative 2 would be more protective of the environment than Alternative 1 because of permit programs limiting the types of sediment-disturbing activities that may occur. Monitoring of site conditions would enable faster responses to contaminant movement should it occur and minimize risks to the environment or humans.

Alternative 3 would be more protective to human health and the environment than Alternatives 1 and 2 by containment of shallow sediment contaminants with a geomembrane cap, removal of deep sediments, and institutional controls and advisories. Physical separation of receptor from contaminants provides an increased level of protection by ensuring no pathway of concern is complete. Initially, capping and removal would destroy the benthos and their habitat on the capped or dredged area. Benthic habitat similar to the existing habitat may never return to the capped area. The dredged area may return to a similar habitat over a period of years.

Alternative 4 would be most protective to human health and the environment by completely removing all sediments above the risk thresholds.

During remediation, Alternatives 2, 3, and 4 would protect the community and workers through the use of engineered and institutional controls.

10.3.2 Compliance with ARARs

The no action alternative is not protective of human health and the environment. There are no ARARs governing institutional controls and advisories; however the NCP [40 CFR 300.430(e)(3)(ii)] and DOE Order 5400.5(IV)(6)(c) recommend certain controls that may be used in Alternative 2. All ARARs for Alternatives 3 and 4 would be met without waivers. TBCs for Alternatives 3 and 4 are the same as for Alternative 2. Consultations will be held with TVA, USACE, and TDEC to ensure substantive requirements are met for the TVA Act, the USACE Nationwide Permit Program, and the TDEC Aquatic Resource Alteration Permit Program. If Alternative 3 or 4 is chosen as the preferred alternative, a wetland and/or floodplain analysis would be required before initiating actions for containment or removal. Potential mitigation for the tall larkspur (*Delphinium exaltatum*), wetland mitigation (if required), and a Phase I archaeological resource survey will also be needed for Alternatives 3 and 4. ARARs for surface water are met by meeting the limit on mercury or arsenic in fish flesh.

10.3.3 Long-Term Effectiveness and Permanence

Alternative 1 (no action) would have no effect in the long term. The risk assessment indicates that the current site conditions with no waste management may not be protective of human health or the environment. Sediment-disturbing activities could occur, resuspending contaminated sediment and increasing risk levels without forewarning. No remedial activities would be conducted on any areas of Clinch River or Poplar Creek under this alternative.

Alternatives 2, 3, and 4 would be effective in the long term at minimizing risks to potential human receptors. Institutional controls provided in Alternative 2 would be protective to human receptors under current land uses, whereas Alternative 3 would provide on-site containment, removal, and institutional controls to reduce the residual risks to potential future human receptors and current ecological receptors. The cap in Alternative 3 could degrade with time, especially without proper inspections and maintenance. Proper maintenance would keep the cap in good repair for many years. Alternative 4 would provide removal of all sediments posing a human health or ecological risk and would be effective in the long term. Alternative 3 or 4 would be impacted by the continuing releases from upstream sources on ORR. It may be more cost-effective to accomplish further source cleanup before attempting to remove or contain the contaminated sediments in the CR/PC OU. The benthic organisms and habitat destroyed by capping may never recover. The areas of removal may eventually become fertile benthic habitat after a number of years.

10.3.4 Reduction of Toxicity, Mobility, or Volume Through Treatment

Alternatives 1 and 2 will not reduce toxicity, mobility, or volume because no treatment options are used. Alternatives 3 and 4 will reduce the mobility of the contaminants through removal or containment, but not by treatment, and may reduce toxicity and volume depending on which, if any, treatment technologies are needed to achieve proper disposal of the sediments.

10.3.5 Short-Term Effectiveness

Alternative 1 would not involve any action; therefore, there would be no increase in short-term risks and no short-term environmental effects. Alternative 2 would provide similar protection to the existing conditions.

Alternatives 3 and 4 approximately equally protect the local community during implementation. Through the use of engineering and institutional controls, access to the work site would be controlled. Air or dust emissions from the installation of the geomembrane cap or removals is not expected to have an effect off DOE property. Erosion and turbidity would be controlled, but some releases and spread of contaminated sediment would be expected.

The risks to workers are expected to be similar in implementing Alternatives 3 and 4. By planning activities in accordance with as low as reasonably achievable principles, industry and Occupational Safety and Health Administration codes and requirements, and DOE Orders, worker risks from contaminant exposure would be controlled to acceptable levels during construction of a cap or transport of sediment.

The short-term environmental effects would vary for the alternatives. Alternative 2 would have little or no short-term impacts. Disruption of habitat would occur along the inland areas used to haul construction equipment and materials into the site, but would be temporary. Permanent impact on habitat would occur on capped and excavated and dredged areas. All benthic organisms in the capped or dredged areas will be destroyed. The benthic community may or may not recover and inhabit the remediated areas. Capped areas (Alternative 3) would be the least likely to recover.

The duration of the activities varies significantly between alternatives. Alternative 1 would have no implementation time because there are no activities. Alternative 2 would take approximately 6 months to implement, depending on the area to be covered. Alternatives 3 and 4 could be completed within 4–5 years if the summer pool season is used to gather all the materials and set the stage for accelerated construction during winter pool.

10.3.6 Implementability

There would be no insurmountable technical implementability issues for any of the alternatives. Technical experience and materials for caps and excavation are available locally. Alternatives 3 and 4 would be more difficult to implement because of health and safety considerations and the need to control migration of sediments and contaminants. However, all of the technologies and experience are available. No difficulty in sampling and analyzing for contaminants in sediments, water, or biota would occur because all of them have been conducted many times in the past.

Administrative implementability would arise from meetings and information exchanges among TVA, USACE, TDEC, and DOE concerning the substantive requirements of permit programs, monitoring results (and reports), and record keeping.

10.3.7 Cost

Cost estimates are included in this FS for each remedial alternative. The estimates are based on feasibility level scoping and are intended to aid in making project evaluations and comparisons among alternatives. The estimates have an expected accuracy of +50 percent to -30 percent for the scope of action described in Chapter 3 for each alternative.

The estimates have been divided into capital cost and M&M cost. All estimates have been escalated using DOE-approved escalation rates and a schedule for the various activities based on similar project experience. Escalation rates used are 3.4 percent for FY 1996, 3.0 percent for FY 1997, 3.1 percent for FY 1998, 3.2 percent for FY 1999, 3.3 percent for FY 2000, 3.4 percent for FY 2001, 3.4 percent for 2002, and no escalation added for FY 2002 and after.

Contingency has been included in each estimate and is based on the degree of difficulty of the RA, the technology status, and the uncertainty level of the action scope.

Capital costs are defined as those expenditures required to initiate and install an alternative. These are short-term costs and exclude costs required to maintain the action throughout the project lifetime. Capital costs consist of direct and indirect costs. Direct costs include construction costs (material, labor, and equipment to install an action), service equipment, process and new process buildings, utilities, and waste disposal costs. Indirect costs include Titles I and II engineering, Title III inspection, project integration, project administration and management, and project contingencies.

M&M costs are long-term costs associated with ongoing remediation at a site. These costs occur after construction and installation are completed. The costs include labor, materials, utilities, and services required to monitor, operate, and maintain the facilities for a period of 30 years.

The estimated present worth of each remedial alternative was determined on a discount rate of 7 percent and a base M&M period of 30 years according to EPA guidance.

Detailed cost estimates, proposed project schedules, and the major assumptions used to develop the cost estimate for each remedial alternative are presented in Appendix H.

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