DOE/OR/01-1393V1&D1

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

Volume 1. Main Text

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DOE/OR/01-1393/V1&D1
ORNL/ER-315/V1&D1

Energy Systems Environmental Restoration Program
ORNL Environmental Restoration Program

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

Volume 1. Main Text

Date Issued—September 1995

Prepared by
Environmental Sciences Division
Oak Ridge National Laboratory
and
Jacobs Engineering Group, Inc.

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

Environmental Restoration and Waste Management Programs
OAK RIDGE NATIONAL LABORATORY
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

ARAR	applicable or relevant and appropriate requirement
AWQC	ambient water quality criteria
BEIAS	Biomedical and Environmental Information Analysis Section
BERA	baseline ecological risk assessment
BMAP	Biological Monitoring and Abatement Program
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	*Code of Federal Regulations*
CNF	Central Neutralization Facility
COC	contaminants of concern
COE	U.S. Army Corps of Engineers
COPC	contaminants of potential concern
CR/PC	Clinch River/Poplar Creek
CR-ERP	Clinch River Environmental Restoration Program
CRM	Clinch River mile
CRRI	Clinch River Remedial Investigation
DO	dissolved oxygen
DOE	U.S. Department of Energy
ECR	excess cancer risk
EFPC	East Fork Poplar Creek
EPA	U.S. Environmental Protection Agency
EPT	ephemeroptera, plecotera, tricoptera
ER	Environmental Restoration
EROD	ethoxyresorufin-o-deethylase
FDA	Food and Drug Administration
FFA	Federal Facility Agreement
FS	Feasibility Study
HAI	health assessment index
HDPE	high-density polyethylene
HEAST	Health Effects Assessment Summary Tables
HI	hazard index
HPRR	Health Physics Research Reactor
HQ	hazard quotient
IRIS	integrated Risk Information System
IWMF	Interim Waste Management Facility
LCL	lower confidence limits
LHS	Latin Hypercube Sampling
LOAEL	lowest observed adverse effect level
LSI	liver-somatic index
MBM	McCoy Branch Embayment mile
MBM	McCoy Branch Embayment mile
MCL	maximum contaminant level
MFO	mixed function oxidase
NCP	National Contingency Plan
NOAEL	no observed adverse effect level
NTS	Nevada Test Site
O&M	operation and maintenance

ORGDP	Oak Ridge Gaseous Diffusion Plant
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
ORREM	Oak Ridge Reservation Environmental Monitoring
OU	operable unit
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyls
PCM	Poplar Creek mile
PLE	product limit estimate
PNL	Pacific Northwest Laboratory
PPE	personal protective equipment
QA	quality assurance
QC	quality control
RAO	remedial action objective
RCF	risk conversion factor
RCRA	Resource Conservation and Recovery Act
RfC	reference concentration
RfD	reference dose
RGO	remedial goal option
RI/FS	remedial investigation/feasibility study
RME	reasonable maximum exposure
ROD	record of decision
SI	sensitivity index
SWSA	solid waste storage area
TBC	to be considered
TCLP	Toxicity Characteristic Leaching Procedure
TDEC	Tennessee Department of Environment and Conservation
TKN	total Kjeldahl nitrogen
TOC	total organic carbon
TRE	trivalent rare earths
TRM	Tennessee River mile
TVA	Tennessee Valley Authority
TWRA	Tennessee Wildlife Resources Agency
UCL	upper confidence limit
UV	ultraviolet
VOC	volatile organic contaminants
VSI	visceral-somatic index
WAC	waste acceptance criteria
WBM	Walker Branch Embayment mile
WBR	Watts Bar Reservoir
WBRIPG	Watts Bar Reservoir Interagency Permitting Group
WOC	White Oak Creek

PREFACE

This Remedial Investigation/Feasibility Study Report on the Clinch River/Poplar Creek Operable Unit (DOE/OR/01-1393V1&D1) was prepared in accordance with requirements under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) for reporting the results of a site characterization for public review. This work was performed under Work Breakdown Structure 1.4.12.3.1.02.41 (Activity Data Sheet 9302 "Clinch River/Poplar Creek Operable Unit"). This document provides the Environmental Restoration Program with information about the results of investigations performed in Clinch River/Poplar Creek. It includes information on risk assessments that have evaluated long-term impacts to human health and the environment. Information provided in the document forms the basis for the record of decision in the Clinch River/Poplar Creek Operable Unit.

EXECUTIVE SUMMARY

This document is the combined Remedial Investigation/Feasibility Study Report for the Clinch River/Poplar Creek (CR/PC) Operable Unit (OU). The CR/PC OU is located in Anderson and Roane Counties, Tennessee and consists of the Clinch River and several of its embayments in Melton Hill and Watts Bar Reservoirs. These waters have received hazardous substances released over a period of 50 years from the U.S. Department of Energy's Oak Ridge Reservation (ORR), a National Priority List site established under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). In accordance with CERCLA, a remedial investigation of this OU has been conducted to determine the current nature and extent of any contamination and to assess the resulting risk to human health and the environment. The feasibility study evaluates remedial action alternatives to identify any that are feasible for implementation and that would effectively reduce risk in the OU.

Historical studies had indicated that current problems would likely include ^{137}Cs in sediment of the Clinch River, mercury in sediment and fish of Poplar Creek, and PCBs and pesticides in fish from throughout the OU. Peak releases of mercury and ^{137}Cs are known to have occurred over 35 years ago, and current releases are low. Past releases of PCBs from the ORR are poorly quantified, and any current releases are difficult to quantify because levels are so low.

The current site characterization focused on evaluating contaminants in surface water, sediment, and biota. Contaminants in surface water were all found to be below Ambient Water Quality Criteria throughout the OU. Very few analytes were detected at levels that exceeded concentrations at reference sites. However, through the use of sensitive analytical techniques, a significant increase in mercury levels could be detected in Poplar Creek downstream of East Fork Poplar Creek (EFPC).

Several metals were shown to be elevated in the sediment of McCoy Branch as a result of historical ash disposal to upstream surface water. As expected, ^{137}Cs and mercury were elevated in Clinch River sediments downstream of source streams. Because of their concurrent release histories, peak concentrations of both are roughly coincident, generally being found buried under cleaner sediment in areas of stable sediment deposition. Concentrations of ^{137}Cs and ^{60}Co in near-shore sediments were shown to be low throughout the OU. In Poplar Creek, increased levels of several metals in sediments, including mercury, are found downstream of the mouth of EFPC. Additional metals, uranium, and ^{99}Tc are elevated in sediments downstream of Mitchell Branch, and several more metals are elevated at the coal ash disposal area in lower Poplar Creek. Organic contaminants in sediment consisted primarily of a few relatively ubiquitous environmental pollutants, all of which were found infrequently and at low concentrations. Sediment transport modelling in the Clinch River indicates that, under normal flow conditions, existing ^{137}Cs inventories would decline due to decay. A local storm in the White Oak Creek watershed would result in a temporary increase in contaminated sediment, but normal flows would soon transport these recent deposits downstream. A major regional storm would result in the immediate decline in ^{137}Cs as a result of scour of existing sediment and eventual deposition downstream.

PCBs and the pesticide chlordane were elevated above reference values (Norris Reservoir) in several species of fish throughout the CR/PC OU. Mean levels in all species were below the federal Food and Drug Administration (FDA) action levels (2.0 mg/kg) at all sites. Significant year-to-year and spatial variation in average concentrations were taken to be indicative of multiple sources of these contaminants; moreover, similar levels are known in fish from other East Tennessee reservoirs situated upstream of the ORR. Mercury levels in fish from Poplar Creek were shown to be elevated downstream of EFPC. Mean values were below the FDA action level of 1.0 mg/kg. Uptake of mercury and PCBs at several trophic levels (benthic invertebrates, insectivorous fish, piscivorous fish and wildlife) was documented.

The greatest risk to human health from contaminants in the CR/PC is from the consumption of fish containing PCBs and pesticides. This is the only pathway resulting in a cancer risk greater than 10^{-4}, with two species (catfish and striped/hybrid bass) having average concentrations that result in a greater than 10^{-3} cancer risk. Fish from Poplar Creek contain sufficient mercury to pose a toxic risk to children via the consumption pathway. The state of Tennessee has posted both Poplar Creek and the Clinch River, advising persons to avoid or to limit the consumption of fish taken from these waters. Several radionuclides in sediment would pose an unacceptable risk if dredged and placed on shore so as to allow unrestricted access. Cesium-137 and ^{60}Co each pose an excess cancer risk greater than 10^{-5} via external exposure. If dredge spoil were used for agricultural purposes, modelled concentrations of several contaminants in produce (meat, milk, and vegetables) were predicted to pose an unacceptable risk. Several noncarcinogenic contaminants, primarily metals, posed an unacceptable hazard via direct ingestion of surface water. No unacceptable risks were identified for the swimming or irrigation scenarios.

Significant ecological risks were identified in Poplar Creek, but not the Clinch River. The fish and benthic invertebrate communities in Poplar Creek were both found to be at risk of experiencing a 20% reduction in species richness and abundance. Poplar Creek water was toxic to fish embryos and periodic increases in dissolved metal concentrations reached toxic levels. Fish from Poplar Creek were also in the poorest health as indicated by a suite of biological parameters. Poplar Creek sediment contained several contaminants at concentrations that have been reported toxic by other studies. No significant effects were found for piscivorous or insectivorous wildlife inhabiting Poplar Creek. Cottontail rabbits feeding on vegetation growing on dredge spoil were predicted to be at significant risk from mercury.

A range of remedial options were evaluated and screened in the feasibility study, with three being retained. The no-action alternative must be retained, under CERCLA, for purposes of comparison; however, this alternative is not protective of human health and the environment. Alternative 2, the use of institutional controls and advisories, would be protective of human health but would not address ecological risk. Alternative 3, source containment, would reduce both human health and ecological risks by isolating existing contaminants in sediment thereby reducing the potential for future exposure.

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), 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—the Oak Ridge National Laboratory (ORNL), the Y-12 Plant, and the 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 have been identified that require environmental evaluation. 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 1994).

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 1994). 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 and 8). 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 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 action 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 in early 1996.

DOE will document the selection of the final remedy in a formal Record of Decison (ROD), to be signed by EPA and TDEC. This ROD is scheduled for completion in early 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.

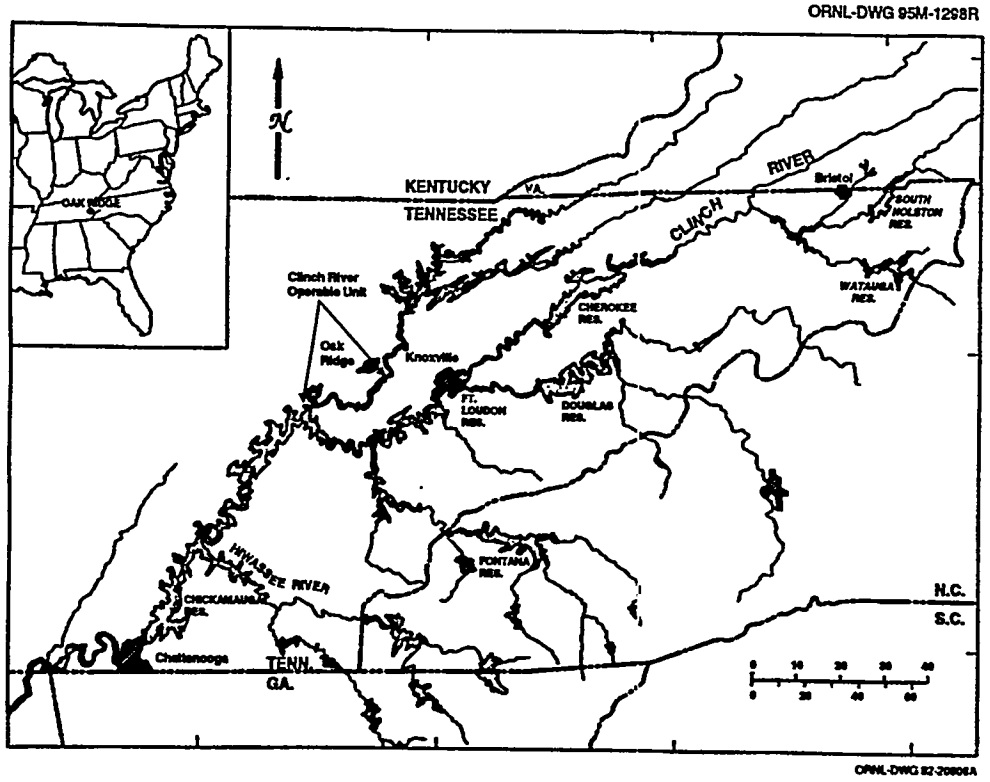

Fig. 2.1. Location of the Clinch River/Poplar Creek Operable Unit in eastern Tennessee.

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, while 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.

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 approximately 20 miles to the east. The nearest community downstream of the ORR is Kingston (pop. 4,552), situated at the confluence of the Clinch and Tennessee rivers approximately 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 above 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).

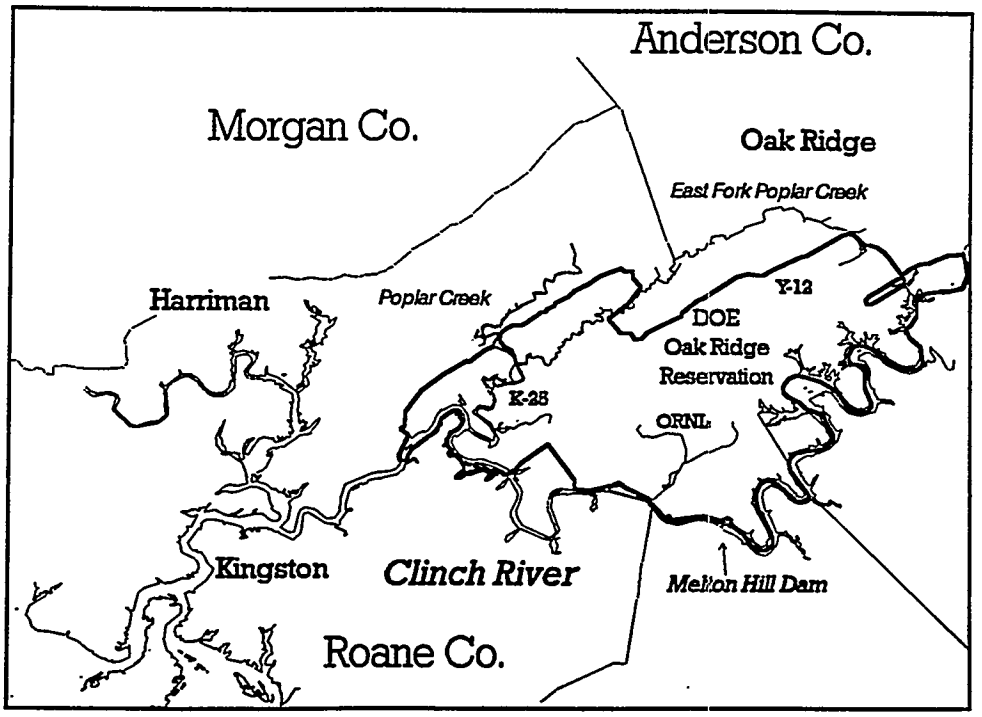

Fig. 2.2. The Clinch River/Poplar Creek Operable Unit, indicating its relation to DOE's Oak Ridge Reservation.

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 Scarboro, 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 or downstream of the ORR.

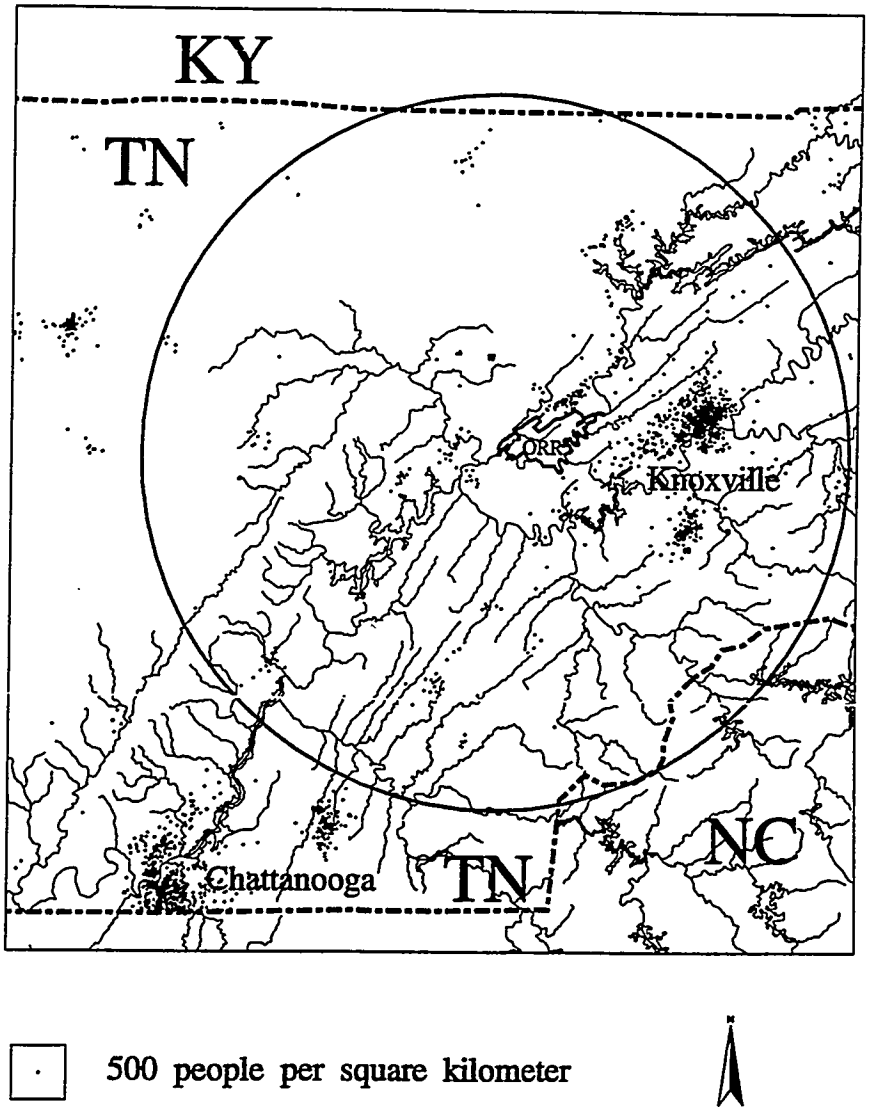

Fig. 2.3. Population density and distribution within 50 miles of the Oak Ridge Reservation.

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.

In addition to providing domestic water supply and recreational opportunities, the Clinch River provides commercial navigation as far as CRM 66 at the city of Clinton. Other uses of the Clinch River include industrial water supply, wildlife and aquatic habitat, and likely some limited use for irrigation and watering of livestock.

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, with a mean temperature of 38°F. The hottest month is July, with a mean temperature of 77°F (Energy Systems 1993).

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, with wind speeds less than 7.4 mph occurring 75% of the time (Energy Systems 1993).

Precipitation in the area averages between 50 and 55 in. per 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 from 32 to 35 in., or 66% of rainfall, leaving an average of 20 to 23 in. available for runoff and infiltration. Evapotranspiration is greatest during the growing season, often exceeding the rate of precipitation, resulting in soil moisture deficits. In winter, precipitation exceeds evapotranspiration, resulting in increased water available for runoff and groundwater recharge (Energy Systems 1993).

Rainfall in 1994 (when data for this RI were being collected) was approximately 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 1993).

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 (1993) 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 1993).

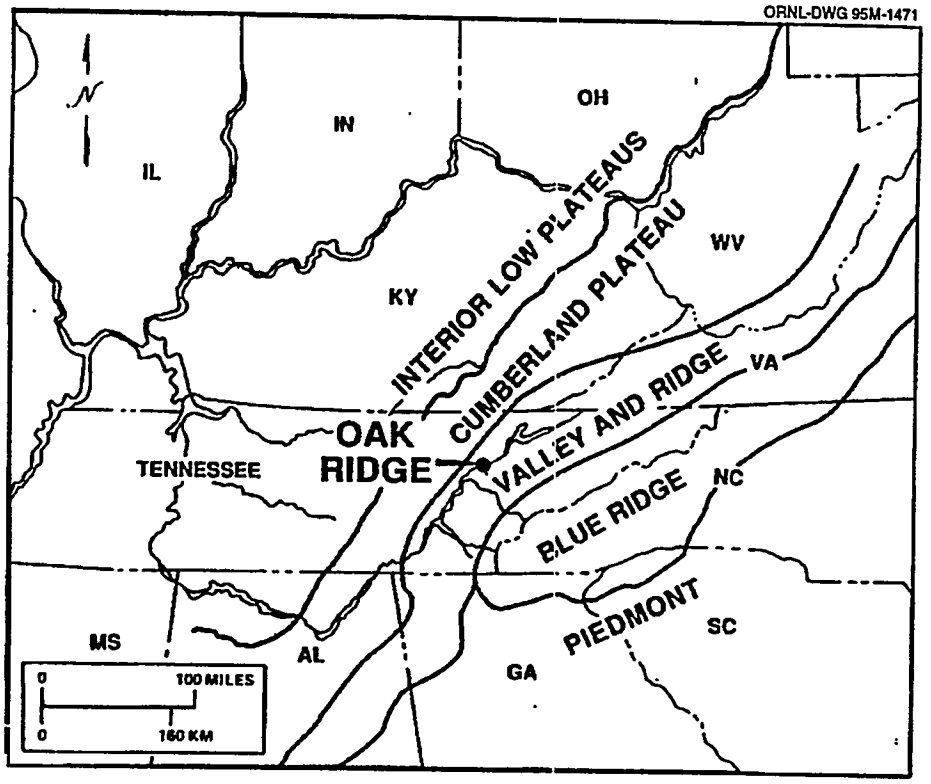

Fig. 2.4. Regional physiographic map depicting the location of the Oak Ridge Reservation and the Clinch River/Poplar Creek Operable Unit in the Valley and Ridge Province (Energy Systems 1993).

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, while 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 1993). 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 via 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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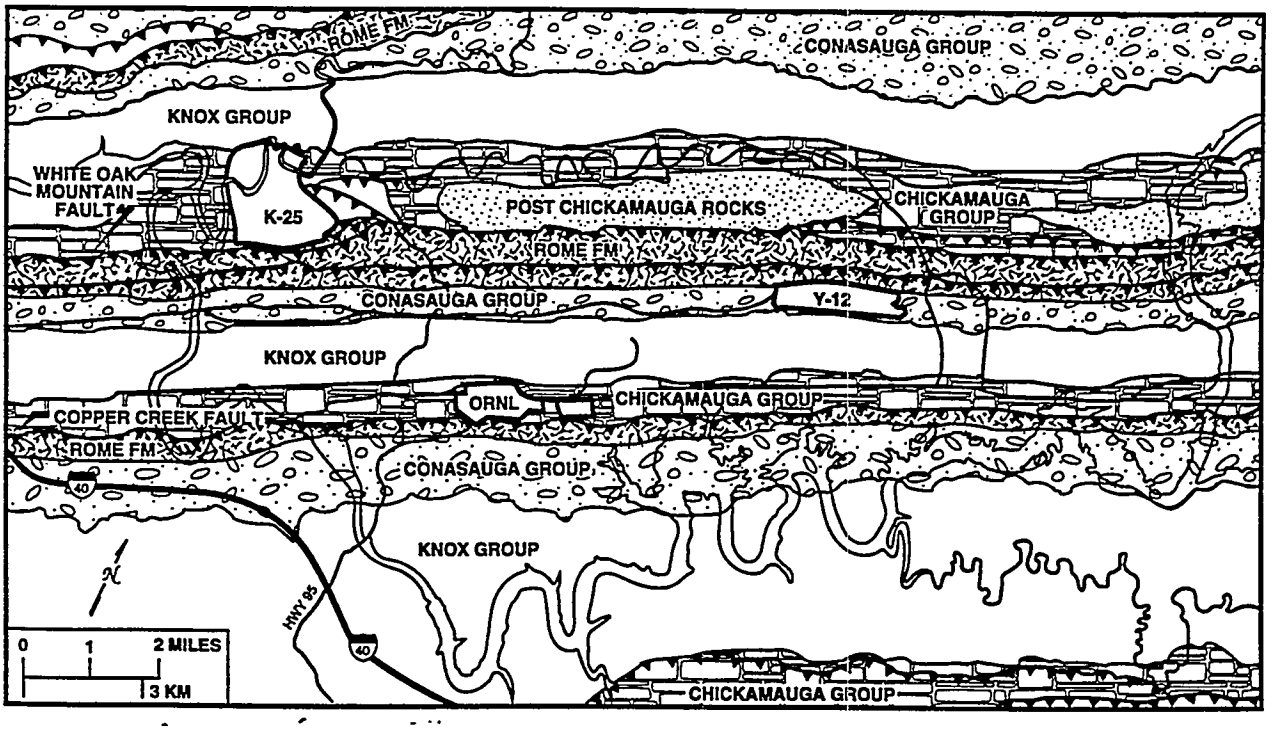


Fig. 2.5. Geologic map of the Oak Ridge Reservation (Energy Systems 1993).

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, while the shallower, loamier soils of Pine Ridge and Haw Ridge have formed from the Rome Formation. In the valleys underlain by the Chickamauga Limestone, such as Bethel Valley and much of EFPC Valley, the Talbott and Colbert soils, characterized by plastic clay subsoils, predominate. 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 is being addressed separately from the CR/PC OU (DOE 1994).

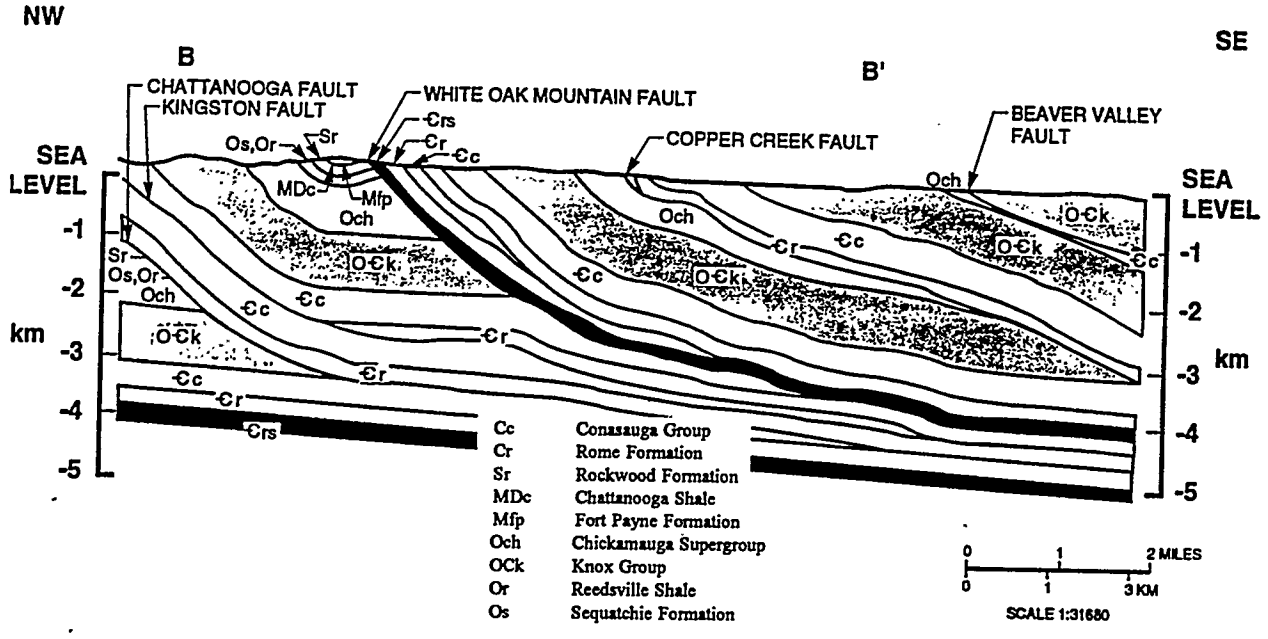


Fig. 2.6. Generalized geologic cross-section of the Oak Ridge Reservation (Energy Systems 1993).

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 1993). The Knox aquifer consists of the Knox Group and the underlying Maynardville Limestone of the Conasauga Group (Energy Systems 1993), 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 1993), and the yield of some wells penetrating the larger solution conduits within this unit yield in excess of 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 1993). 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 1993).

Within both hydrologic units, three major zones of groundwater occurrence can be identified (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 1993). 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 200 m), the quality of the water changes from freshwater to brine, which is sometimes used to delineate the functional basement of this zone (Energy Systems 1993).

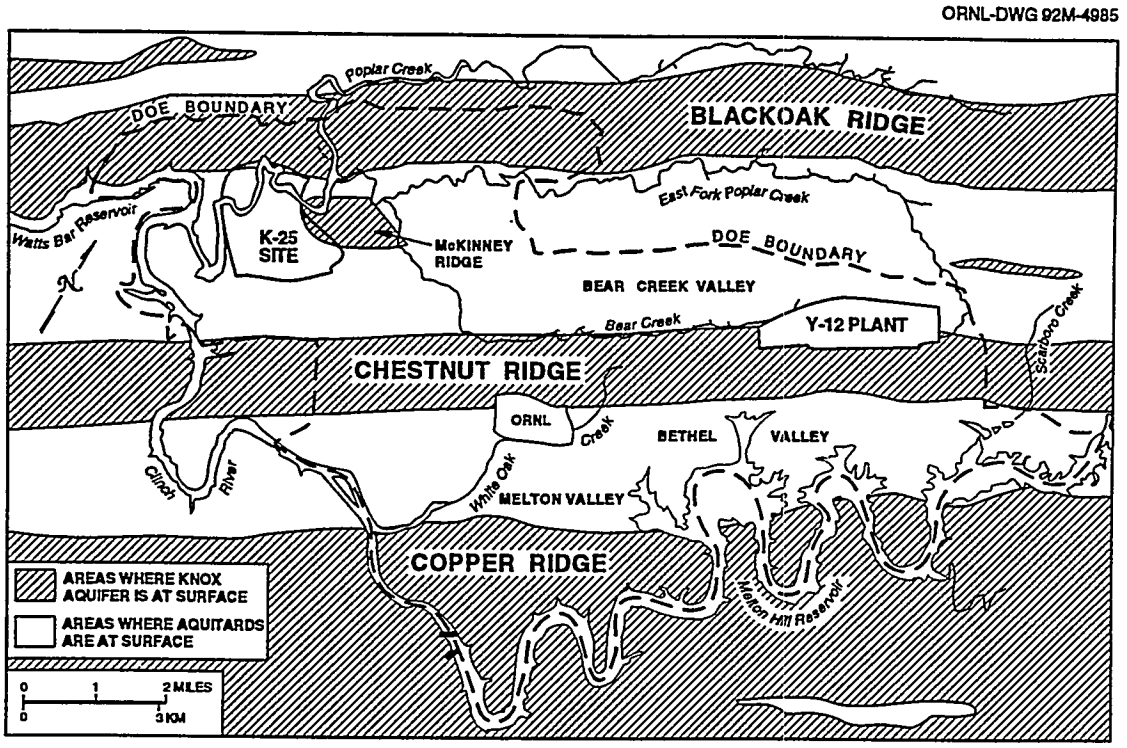


Fig. 2.7. Location of aquitards and the Knox Aquifer on the Oak Ridge Reservation (Energy Systems 1993).

On the ORR, as well as the surrounding region, the majority (95%) of the active subsurface flow is via the shallow stormflow zone to nearby surface streams. The vadose zone transmits water vertically to the saturated zone, necessary for recharge of 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 is to nearby surface streams. In the Knox aquifer, a few solution conduits may be as much as 2 miles long (Energy Systems 1993).

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 mi^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

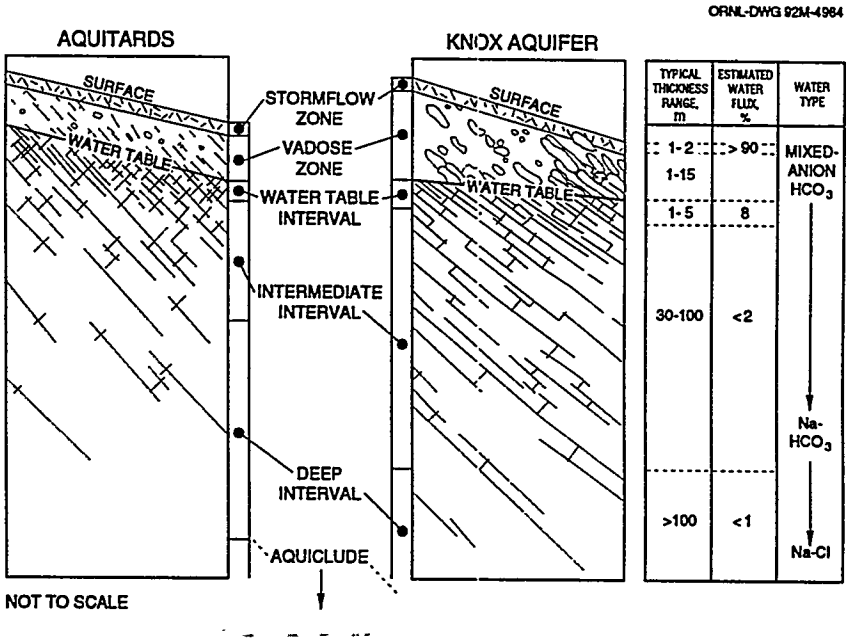


Fig. 2.8. Schematic representation of vertical relationships characteristic of flow zones on the ORR: estimated thickness, water flux, and water type (Energy Systems 1993).

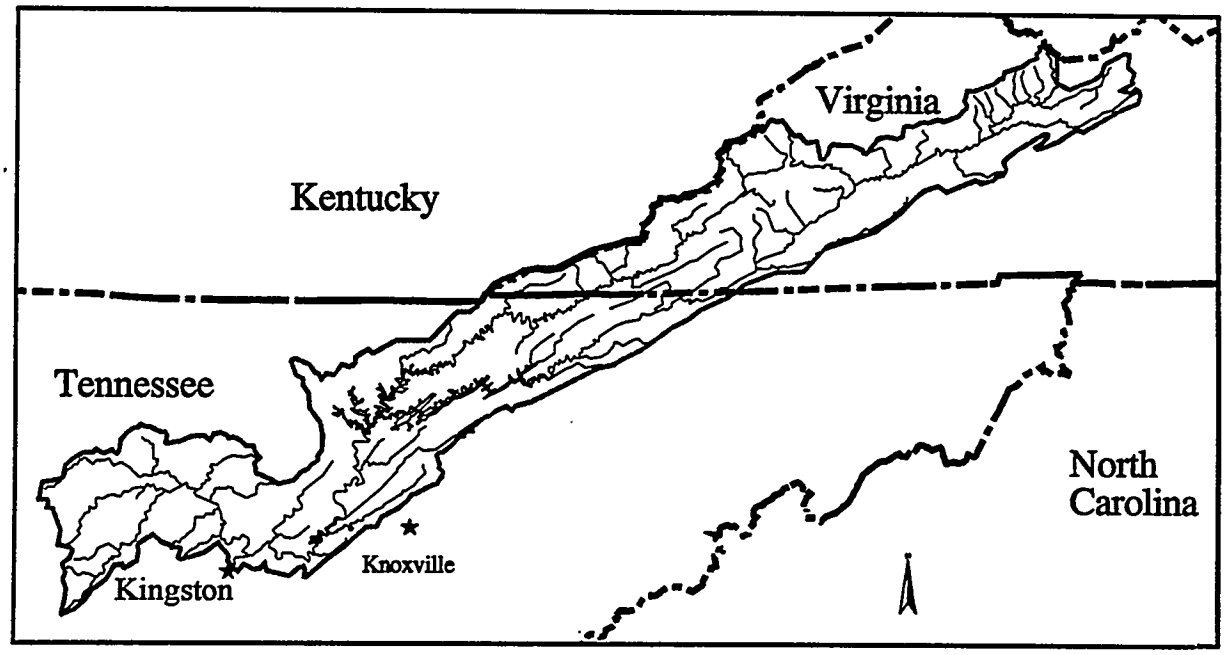


Fig. 2.9. The Clinch River drainage basin—4413 square miles in Virginia and Tennessee.

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.

The Clinch River arm of WBR extends approximately 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 approximately 39,000 acre-feet of water at full pool (elevation 741 ft above msl). Average annual discharge at the mouth is approximately 6800 cfs, and the average hydraulic retention time is approximately 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 approximately 1020 cfs on average, or about 20% of the flow in the Clinch River at Kingston. Most of its 865 mi^2 watershed is located in the Cumberland Plateau. Poplar Creek has a drainage area of 135.9 mi^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 mi^2 of the ORR, including ORNL, and has an average annual flow of 13.8 cfs.

The Clinch River arm of WBR is primarily riverine through most of its length. More lake-like (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. Due to 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. Since 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, forms the eastern and northern boundaries of the ORR. 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)

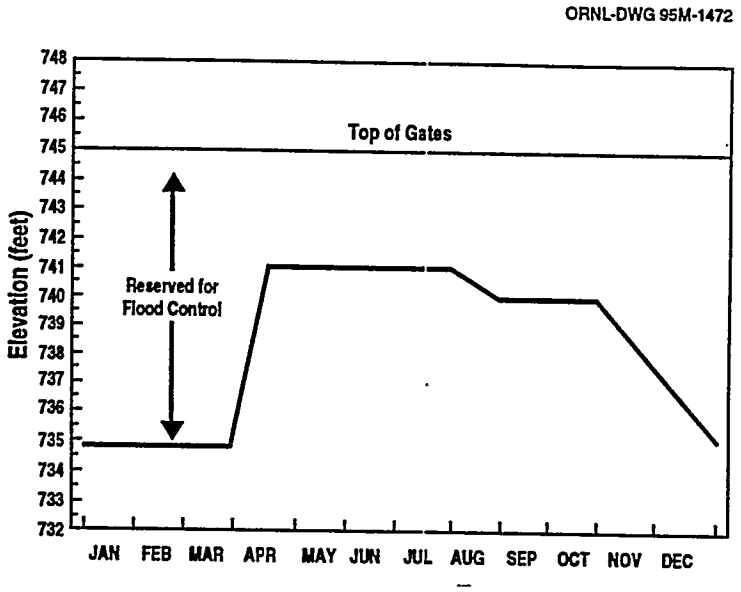
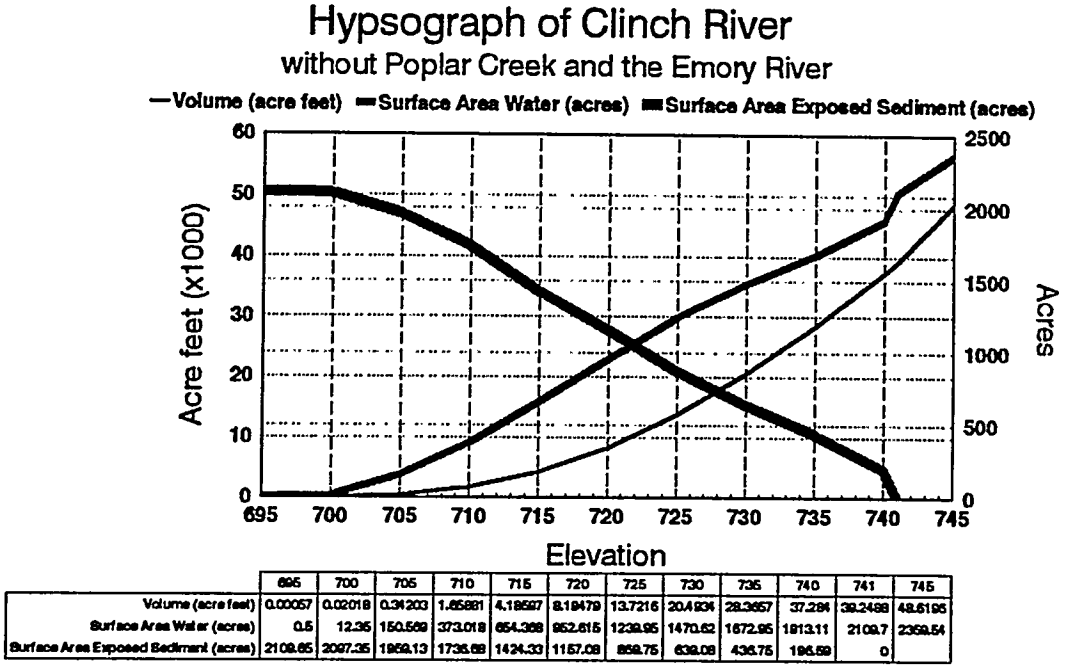


Fig. 2.10. Seasonal water level elevations in Watts Bar Reservoir as maintained by the Tennessee Valley Authority.



	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	1813.11	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.

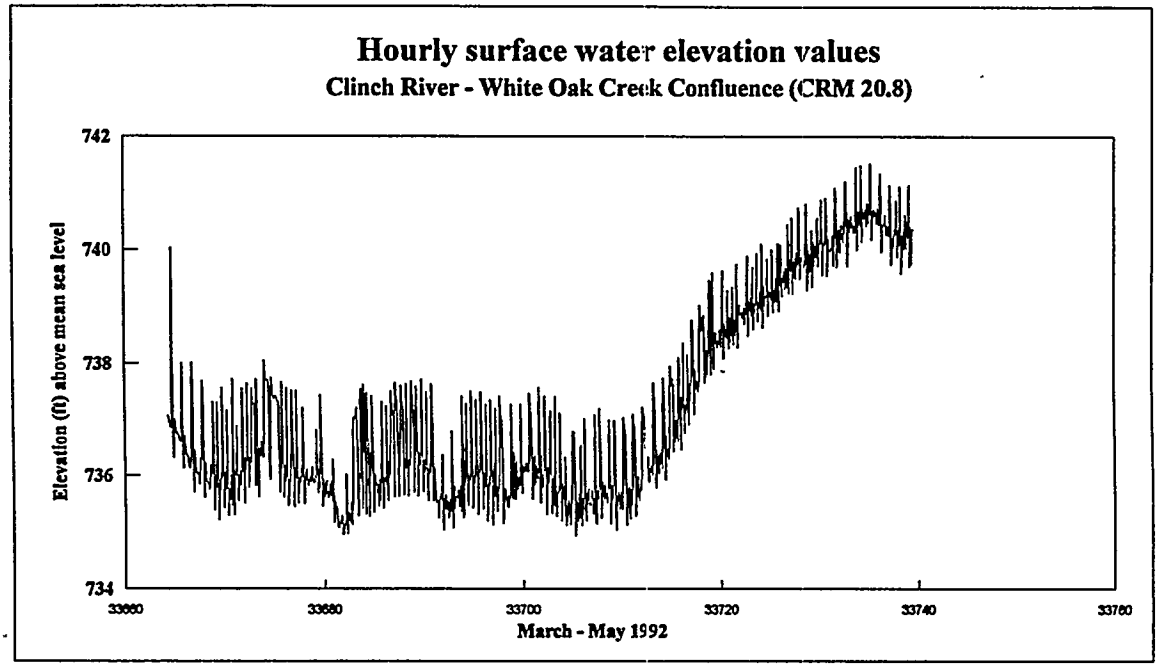


Fig. 2.12. Variation in water level elevations typical of the Clinch River immediately below Melton Hill Dam.

relative to its length (Fehring and Meinert 1993). Full pool in Melton Hill is 795 ft above msl, with intermittent water level fluctuations depending on the system-wide reservoir management needs of the TVA. Unlike Watts Bar, there is no seasonal drawdown of Melton Hill.

Though it is narrow and relatively shallow, Melton Hill Reservoir does exhibit 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 to 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 Site were formerly discharged into Kerr Hollow Quarry, which flows 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 to McCoy Branch, which flows into the Clinch at CRM 37.3. The average annual discharge of each stream is less than 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).

Table 2.1. Minimum, maximum, and mean (in parentheses) values for selected water quality parameters, Clinch River and Poplar Creek[1]

Water Quality Parameter	Clinch River		Poplar Creek	
	Historical Values[2]	CRRI Values	Historical Values[3]	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)

[1] 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.
[2] Melton Hill Dam Tailrace, USGS Data.
[3] Poplar Creek above EFPC confluence.

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, while 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 due to coal mining activities on the Cumberland Plateau (Energy Systems 1993). Averages and ranges for these values observed during the RI are presented in Table 2.1.

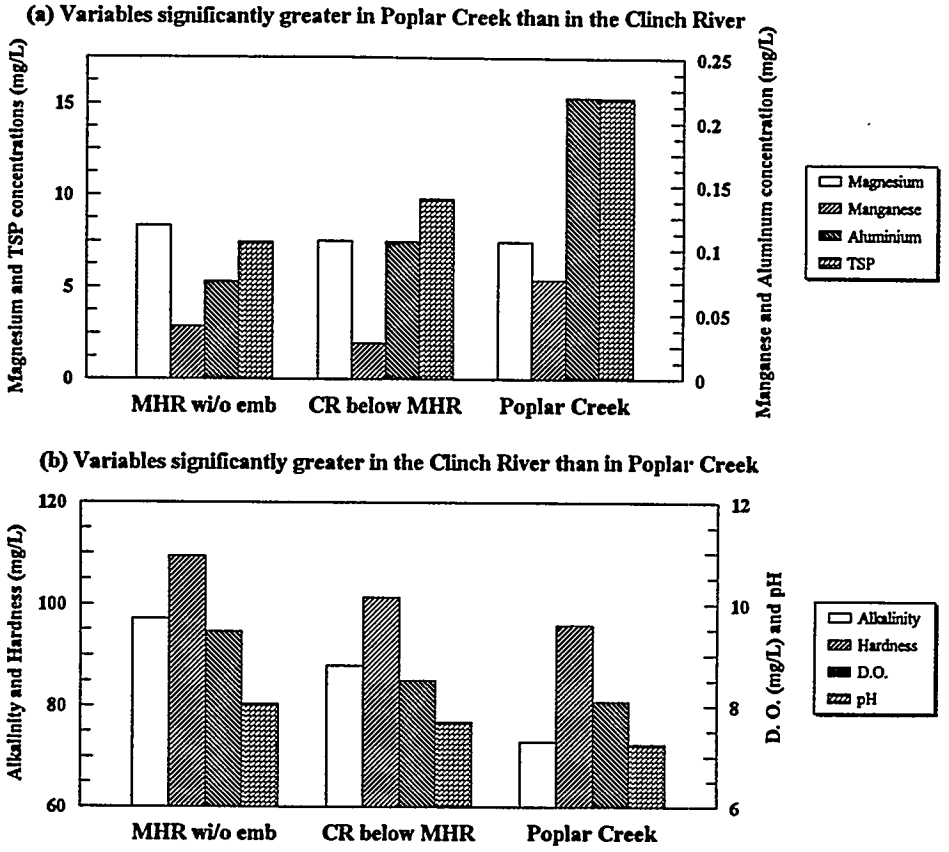


Fig. 2.13. Water quality differences between Clinch River and Poplar Creek.

Several water quality parameters [e.g., temperature, dissolved oxygen (DO), 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.

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 (via 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, there was relatively uniform deposition of sediment 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 mi^2 to 1501 mi^2. Melton Hill Reservoir retains approximately 60% of the sediment entering from upstream and further reduces the local sediment basin to 1070 mi^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, while 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 periods of 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 contibutor 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 weekly a 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 pre-impoundment 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 where little long-term deposition occurs, functioning 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, with more sediment accumulating in the straight portions of the Clinch River than in sharp bends. 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 pre-date 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 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.

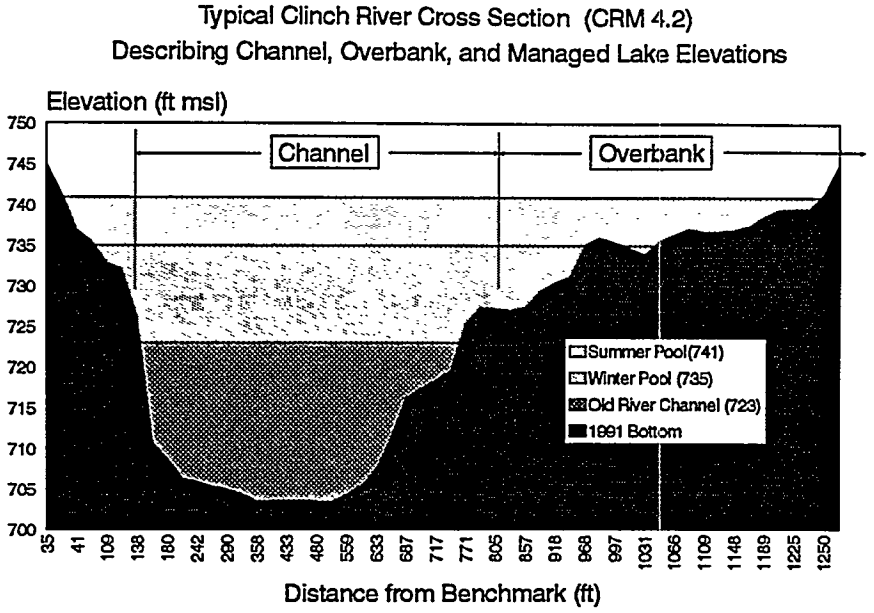


Fig. 2.14. Typical Clinch River cross section (CRM 4.2) depicting the river channel, overbank areas, and managed surface water elevations.

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).

Table 2.2. Density ranges for various descriptive sediment type classes (Hamilton 1972)

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

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 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. 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%

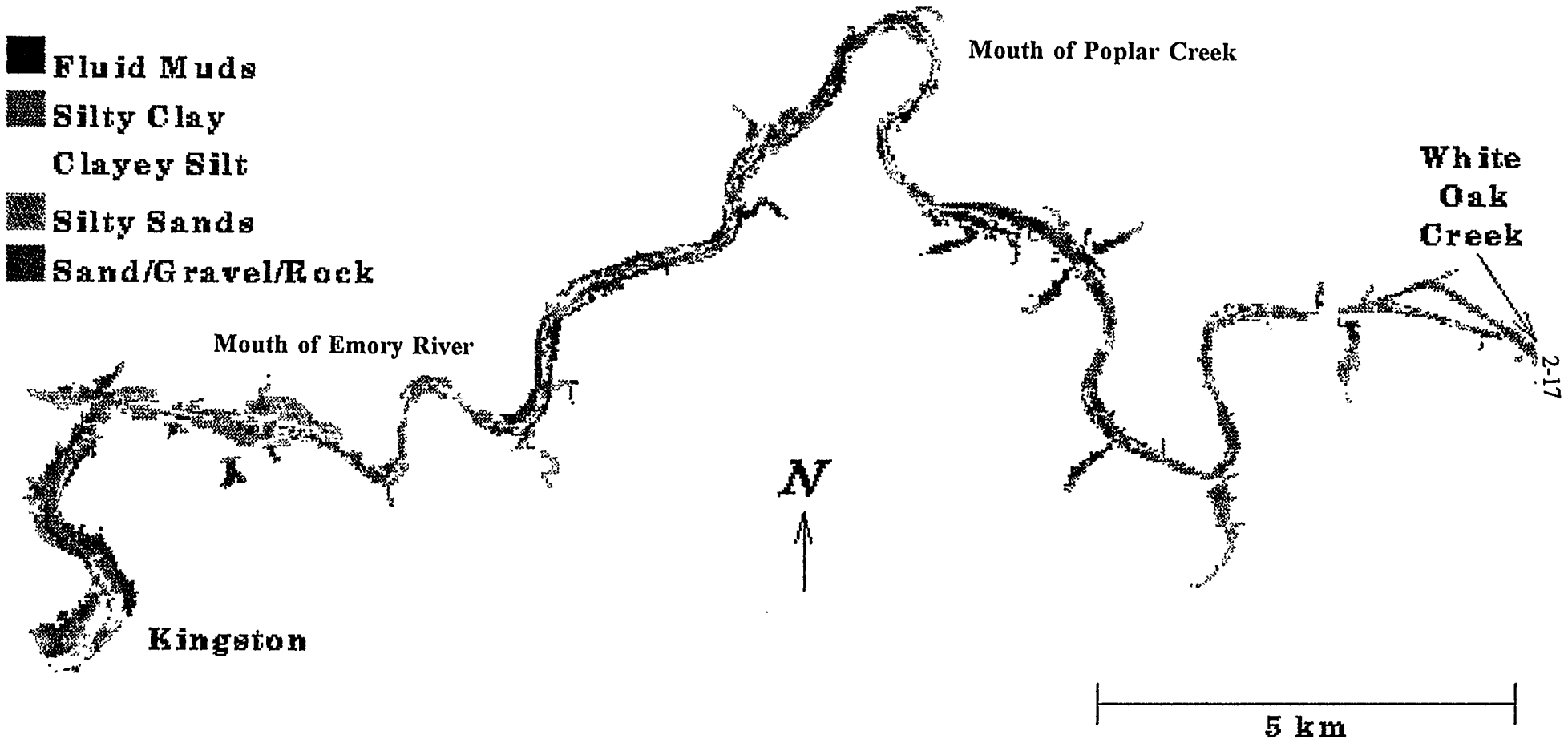


Fig. 2.15. Density of surface sediment in the Clinch River downstream of the Oak Ridge Reservation.

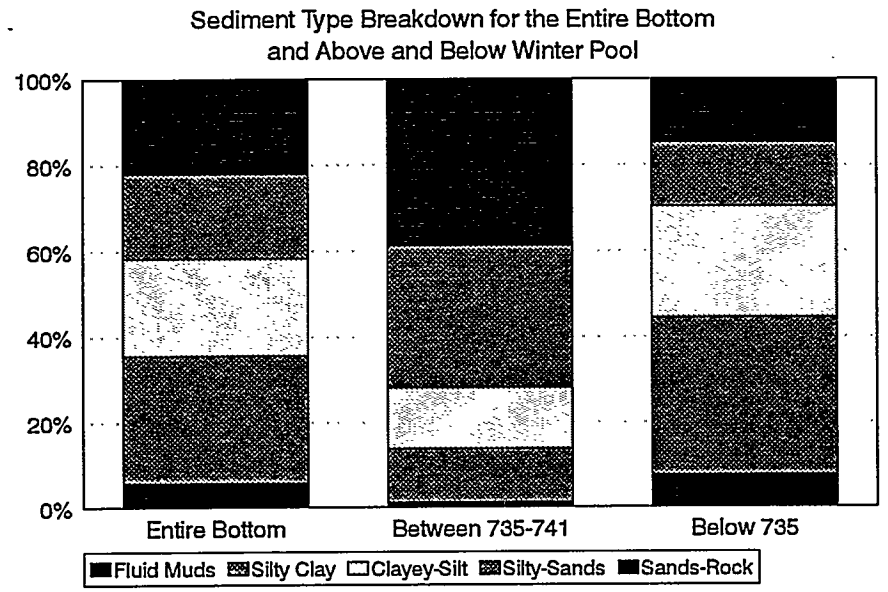


Fig. 2.16. Relative proportions of sediment types in the Clinch River downstream of the Oak Ridge Reservation.

of the sediment falls within the densest two categories, which is not surprising given that the near shore is exposed to wave action for part of the year. This wave action effectively washes fine sediment into deeper water.

Micro-topography also has an influence on deposition pattern. The geologic formations in the area are often exposed and cross the river bottom at various angles. The tilted nature and the varying erosional characteristics of these formations make for a "jagged" bottom where the harder formations stick up and the softer formations are eroded away. This 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 occurred that affect 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 approximately 44,000 yd^3 of material. In the same year, a discharge channel was dredged in the Clinch River, and approximately 124,587 yd^3 of material were 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 was dredged during construction of the dam. An estimated 454,600 yd^3 of material were removed, approximately half of which was sediment and half 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

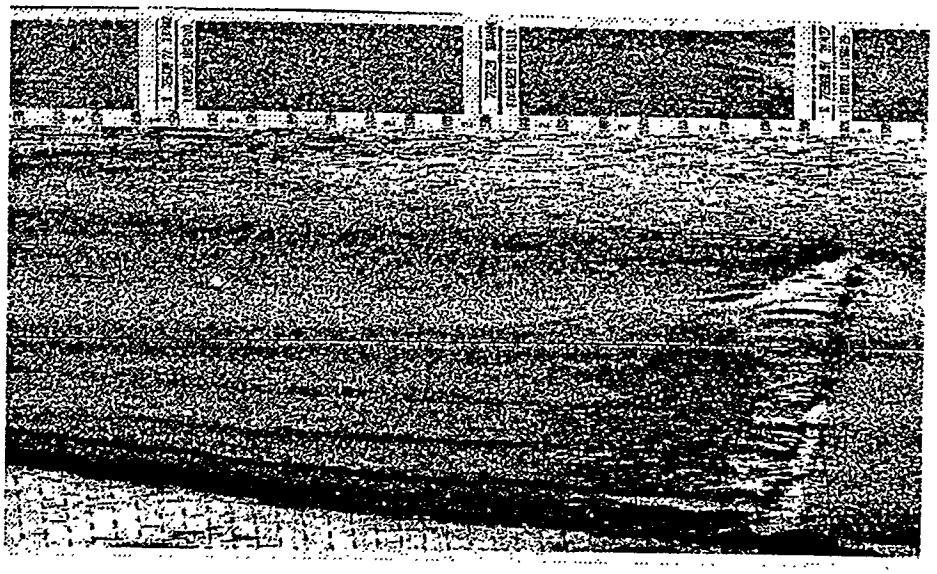


Fig. 2.17. Sonographic image of a portion of the Clinch River bottom. Darker shades represent denser sediments. The structure at the right side of the image is the weir at CRM 3.9 that deflects cooler Clinch River water to TVA's Kingston Steam Plant.

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 principle 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 riverine 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 through direct 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).

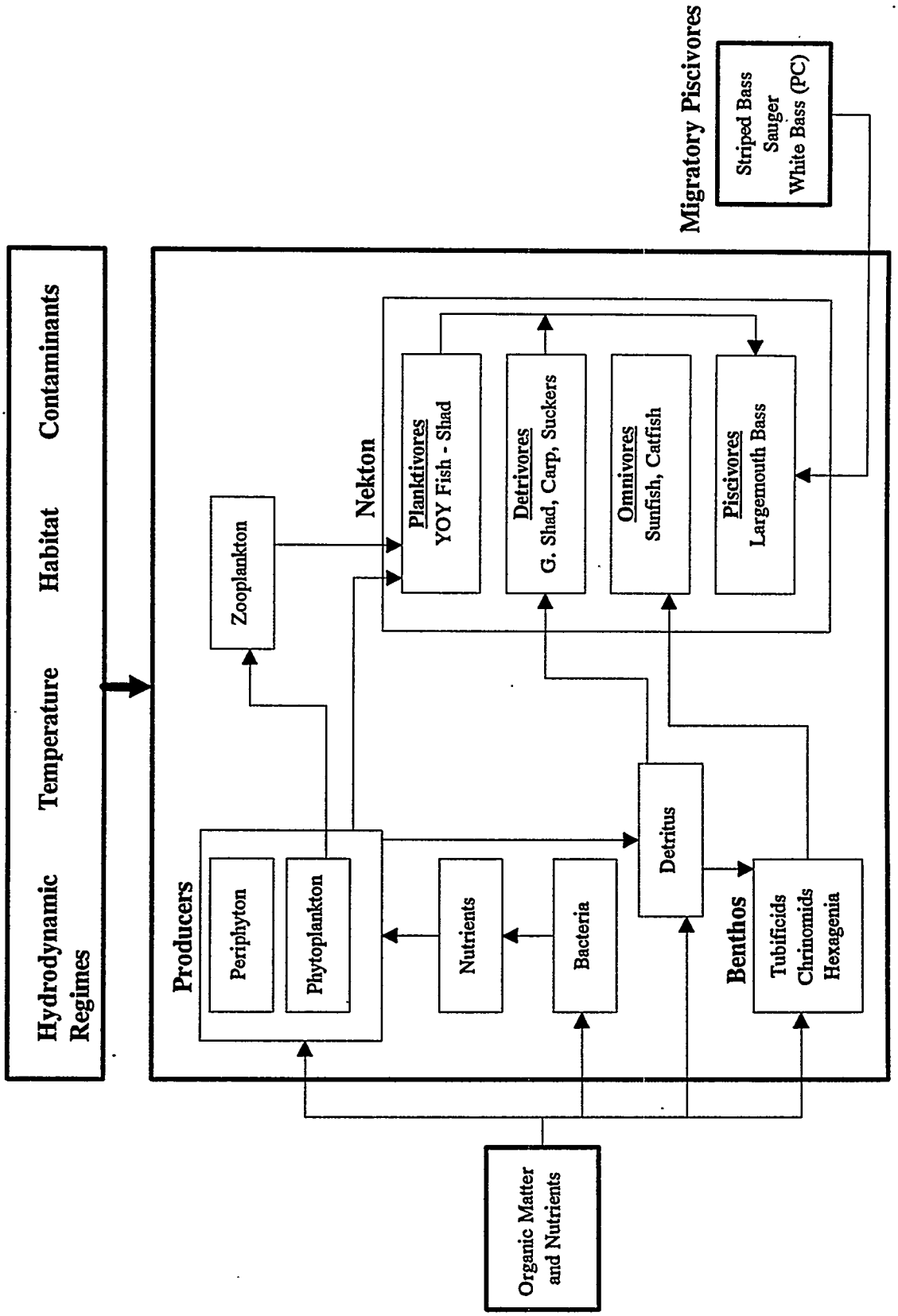


Fig. 2.18. Simplified aquatic ecosystem diagram for the Clinch River and lower Poplar Creek illustrating the principal energy flow pathways and environmental factors that regulate these pathways.

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 compared to 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 1990). 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). Benthic communities in both systems suffer from a lack of species diversity; 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., while taxa in the lower section of the Clinch River (miles 0–9) are composed primarily of *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, with *Tubificidae, Chironomidae,* and *Hexagenia* dominating and with very few EPT taxa being 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 is dependent upon a host of environmental factors, the most important being 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 Poplar Creek systems is characterized by relatively low species diversity with representatives of all four main functional feeding groups 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 consist primarily of largemouth bass. In both systems, striped bass and sauger are seasonal migratory species, with white bass using 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) comprising 50–60% of the abundance, omnivores (carp, suckers) comprising 10–20%, insectivores (sunfish) comprising 10–15%, and piscivores (black bass, striped bass, and striped bass hybrids) comprising 5–10% of the abundance. The most abundant species in the Clinch River during the fall of 1993 and 1994 were found to be (listed in order of importance) gizzard shad, bluegill minnows, *Cyprinidae,* and largemouth bass. 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 between Poplar Creek and the Clinch River are due to the seasonal shifts in the distributions of various species as well as the nature and range of habitat (quantity and quality) found in these two areas. Fish communities surveys suggest that there may be a relationship between species diversity and richness with 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, while 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 in particular lower Poplar Creek, 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 relative 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 approximately 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 and Ashwood, in preparation). 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

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.

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 approximately 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 J1.

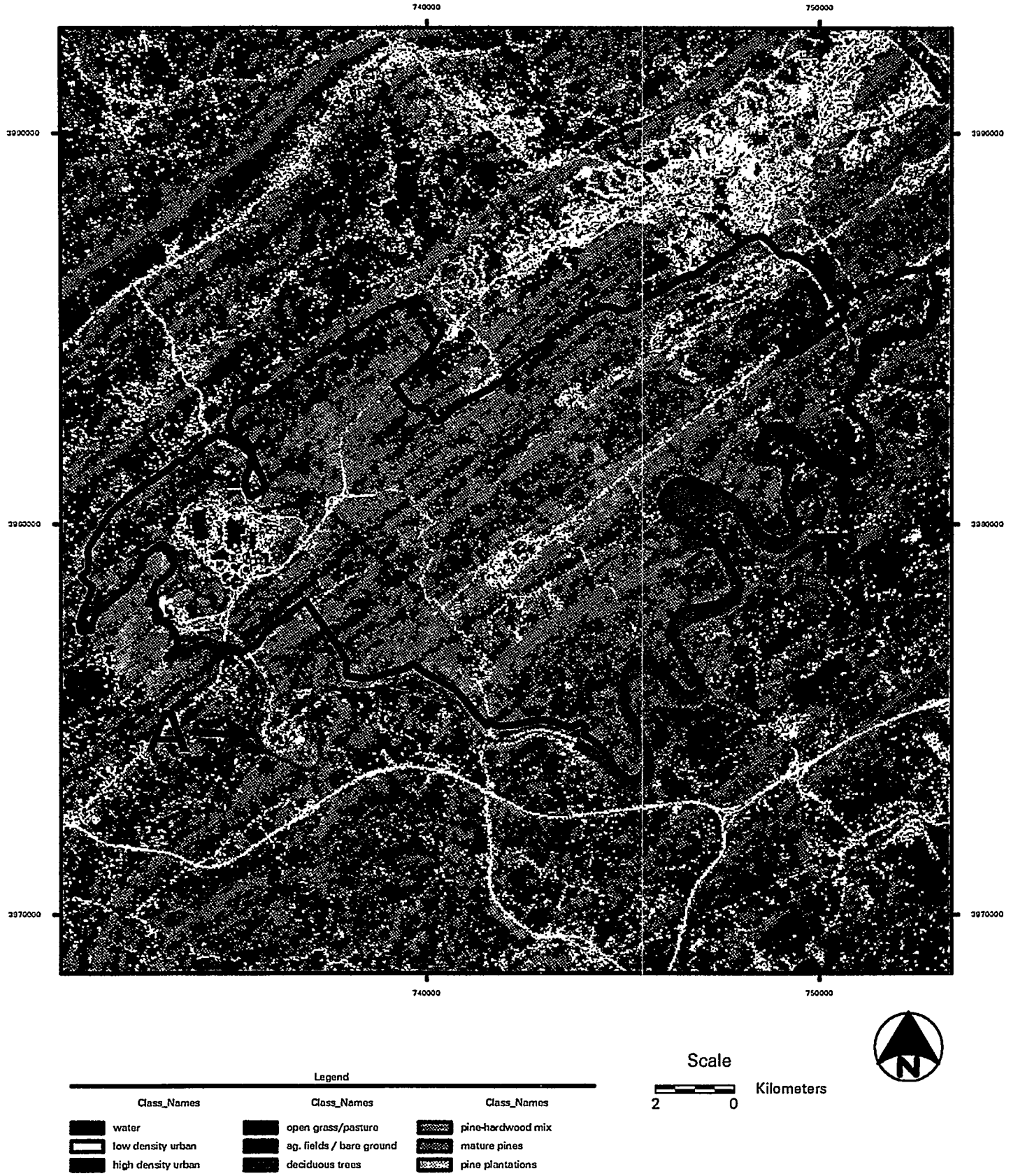


Fig. 2.19. Land use on and immediately surrounding the Oak Ridge Reservation. Site A was being developed at the time for the TVA's Clinch River Breeder Reactor and is not urban land as the legend suggests. Site B is the Freels Bend agricultural station.

Table 2.4. Land use/land cover categories, percentage and area (hectares) of coverage within each Clinch River and Poplar Creek reach (extending 100 meters 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

Rare and endangered plant surveys have been performed for most of the ORR. Table J2 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 (HPRR) 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 J3). 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 riverine system. Table J3 also includes more than 40 species that are listed by either the Tennessee Wildlife Resources Agency (TWRA)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. However, several of the characteristics of the OU are evident from this description which affect the range of remedial options available.

The volume of surface water in the OU is very large. Surface water is also very transient, with short average retention times. Therefore, any remedial action (other than posting of waters) would most likely be effected at the contaminant source, at one or more of the on-site OUs.

The greatest quantity of sediment-associated contaminants will 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.

There are aquatic and terrestrial biological receptors in the CR/PC OU, 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. Remediation of these biota would be difficult and alone would not address the ultimate contaminant source.

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 desription 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 G and evaluated in detail by Holladay et al. (1995).

3.1 SITE DESCRIPTION

To facilitate the conduct of the RI, the CR/PC was divided into nine study 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 in an effort to distinguish between areas where (1) natural environmental conditions are known to differ or (2) effects are anticipated from plausible contaminant sources.

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 the upstream boundary of the ORR almost 20 miles to CRM 52.

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 the upstream boundary of the ORR at CRM 43.7. The only known source of contaminants to reach 1 is from the Y-12 Plant via 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:

Subreach 7.1—Upper McCoy Branch is the portion of the embayment upstream of the 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 7.2—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 4 subreaches:

Subreach 2.1—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 2.2—The mouth of WOC (CRM 20.8) to the mouth of Pawpaw Creek (CRM 19.0). Contaminants from ORNL would be the least dilute in this subreach of the Clinch River.

Subreach 2.3—The mouth of Pawpaw Creek (CRM 19.0) to the mouth of Grassy Creek (CRM 14.5). This subreach is still riverine, but there is a zone of sediment accumulation at the downstream end of Jones Island. The K-25 potable water intake is at the lower boundary of this subreach.

Subreach 2.4—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 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 and K-25 facilities, 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 the creek upstream 5.5 miles to the mouth of EFPC. There are four subreaches, established primarily on the basis of potential contaminant sources:

Subreach 3.1—The mouth of EFPC (PCM 5.5) to the mouth of Mitchell Branch (PCM 4.6). Any effluents or runoff containing contaminants from the Y-12 Plant or the City of Oak Ridge enter Poplar Creek via EFPC.

Subreach 3.2—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 effluents to Mitchell Branch.

Subreach 3.3—The CNF outfall (PCM 3.4) to the Poplar Creek ash disposal area (PCM 1.0). Historically several K-25 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 3.4—The ash disposal area (PCM 1.0) to the mouth of Poplar Creek. 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:

Subreach 4.1—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 upstream and immediately downstream of Brashear Island.

Subreach 4.2—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.

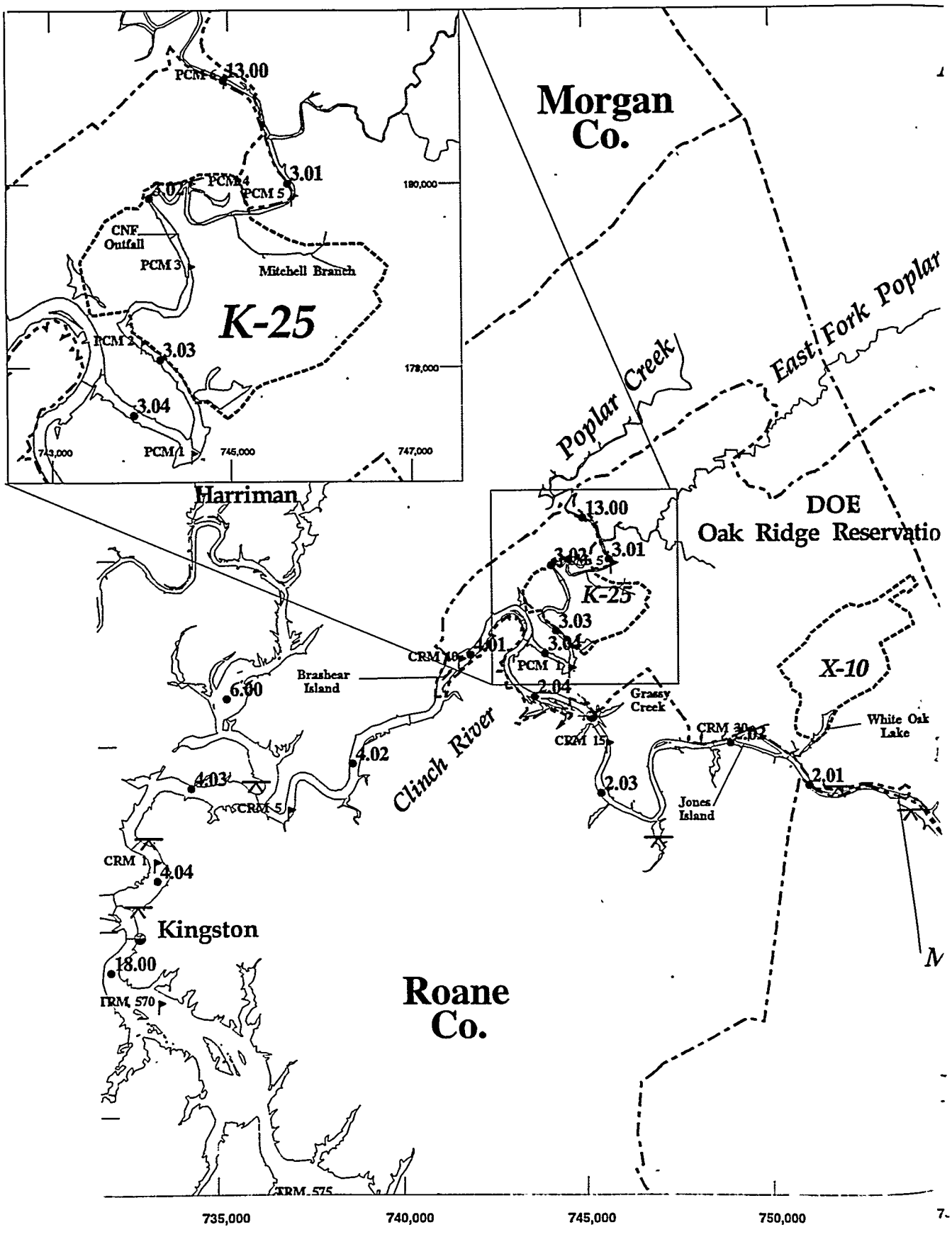
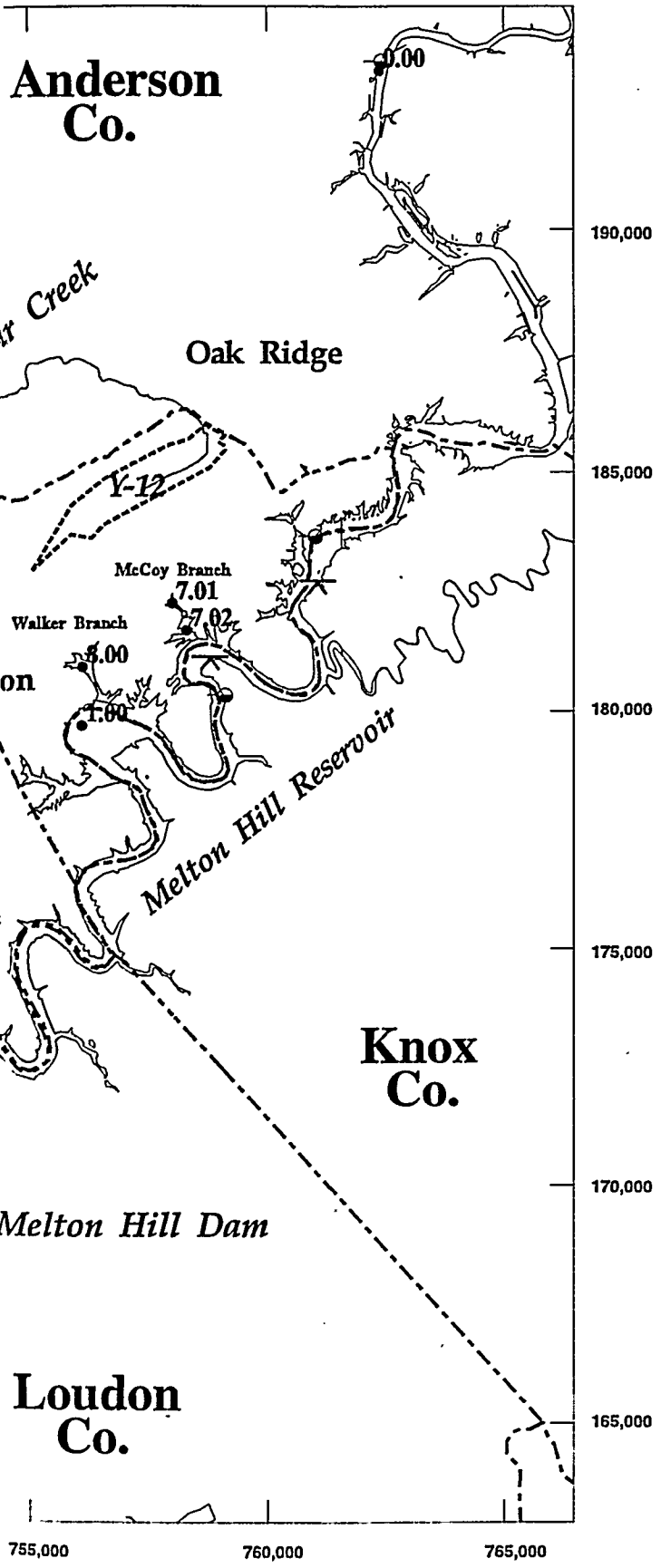
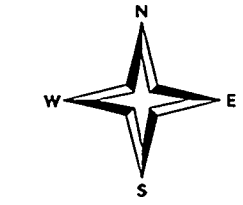


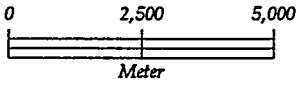
Fig. 3.1. Reaches and subreaches studied as part of the Clinch River/Poplar Creek Remedial Investigatio



Clinch River Environmental Restoration Program

Study Sites





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 Clinch River Environmental Restoration Program

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Subreach 4.3—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 effluents from TVA's Kingston Steam Plant. Urban areas within the Emory River watershed include Harriman and Crossville.

Subreach 4.4—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. This area was investigated previously as a separate CERCLA OU (DOE 1995). It serves as a positive reference reach for ORR contaminants.

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. Very 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 the Los Alamos 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 thorium, lithium, beryllium, and lead. In addition, the industrial-scale machining operations at Y-12 involved the use of

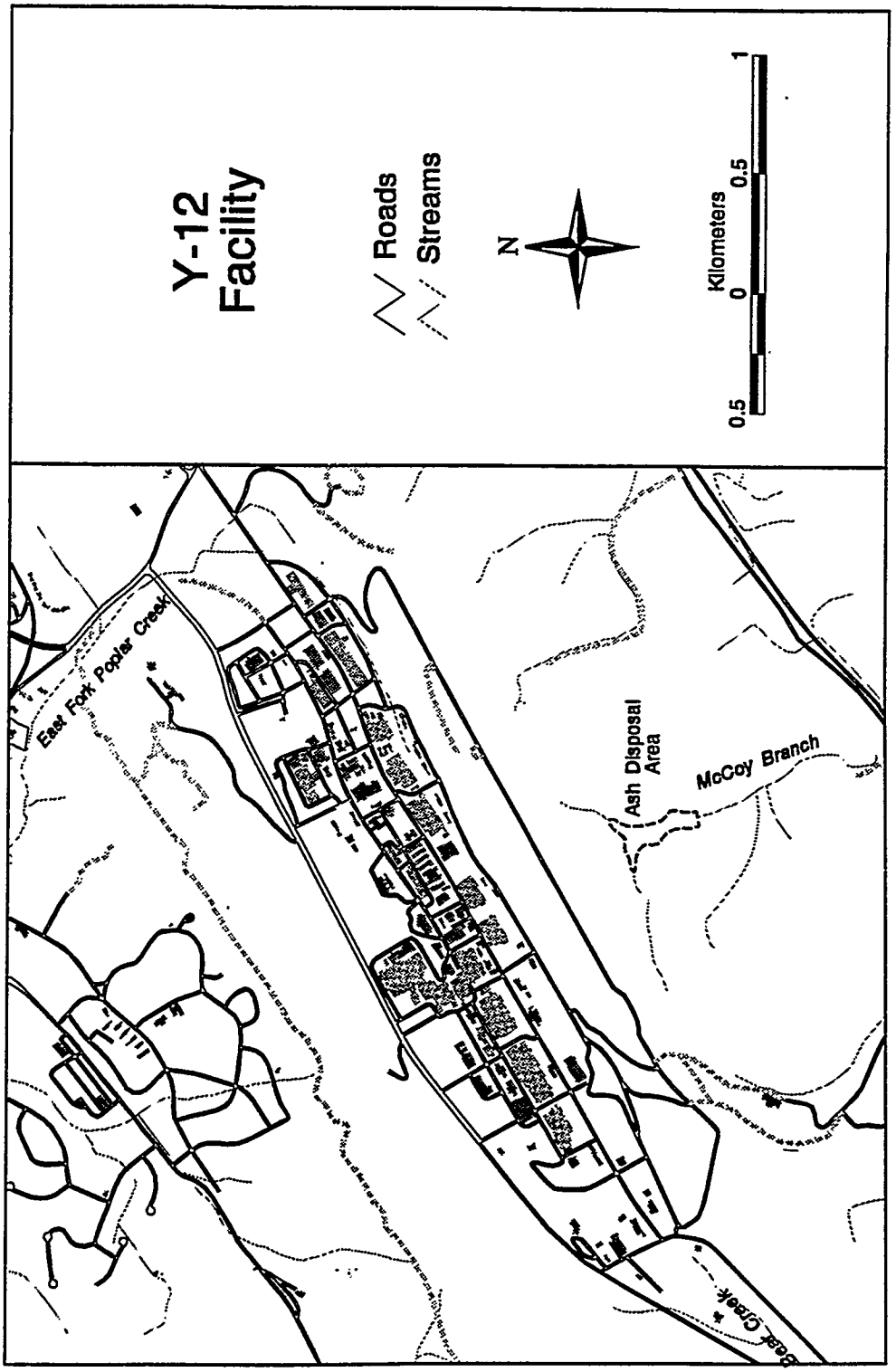


Fig. 3.2. The Y-12 Plant.

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 also 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, which was a column exchange process that used an aqueous lithium hydroxide solution and a solution of lithium in mercury. Millions of pounds of mercury were required for the process, with large quantities accidentally released to process buildings, soils, and surface water (Wilcox 1983). By the time the Colex process was closed in 1963, an estimated 239,000 lbs of mercury were lost to EFPC, and another 428,000 lbs were lost to the ground in the plant area (Wilcox 1983). Peak releases of mercury occurred in 1957 and 1958, when approximately 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 Y-12. Mercury releases declined dramatically after 1958, but the contaminated storm sewer system, buildings, and soils at Y-12 continue to contribute several grams per day of mercury 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.

In the early 1950s Y-12 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, technetium, and plutonium) in the resulting waste stream, which was then transferred 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 a 10-million-gal capacity 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 to EFPC. Groundwater in the area beneath the site of the former S-3 ponds is contaminated with nitrates, volatile organic contaminants (VOCs), 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 to 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 via the gaseous diffusion process (DOE 1994a). This process produces highly enriched ^{235}U from ^{238}U by the

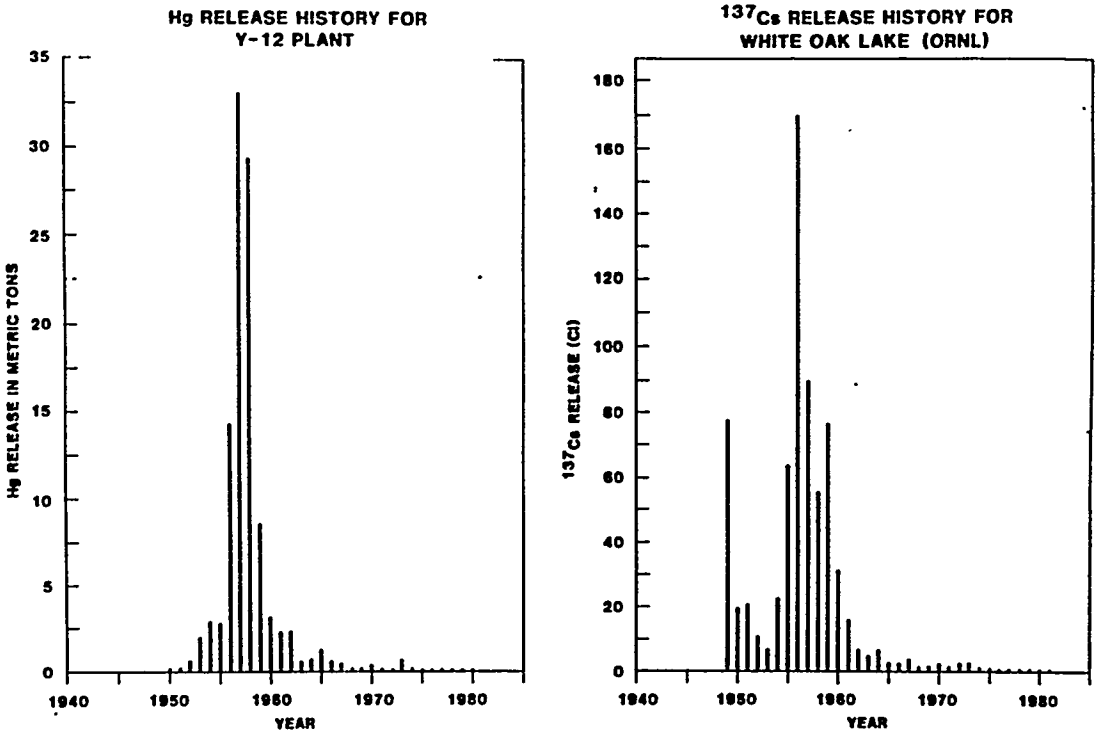


Fig. 3.3. Mercury release history from the Y-12 Plant and ^{137}Cs release history from White Oak Lake (ORNL).

diffusion of gaseous uranium hexafluoride across a long series of porous barriers. At each barrier in the diffusion cascade, the product becomes slightly enriched in ^{235}U as a result of its greater diffusion rate across the barrier as compared to the slightly heavier ^{238}U. Prior to 1964, a long series of diffusion cascades were 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 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 clorinated 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 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

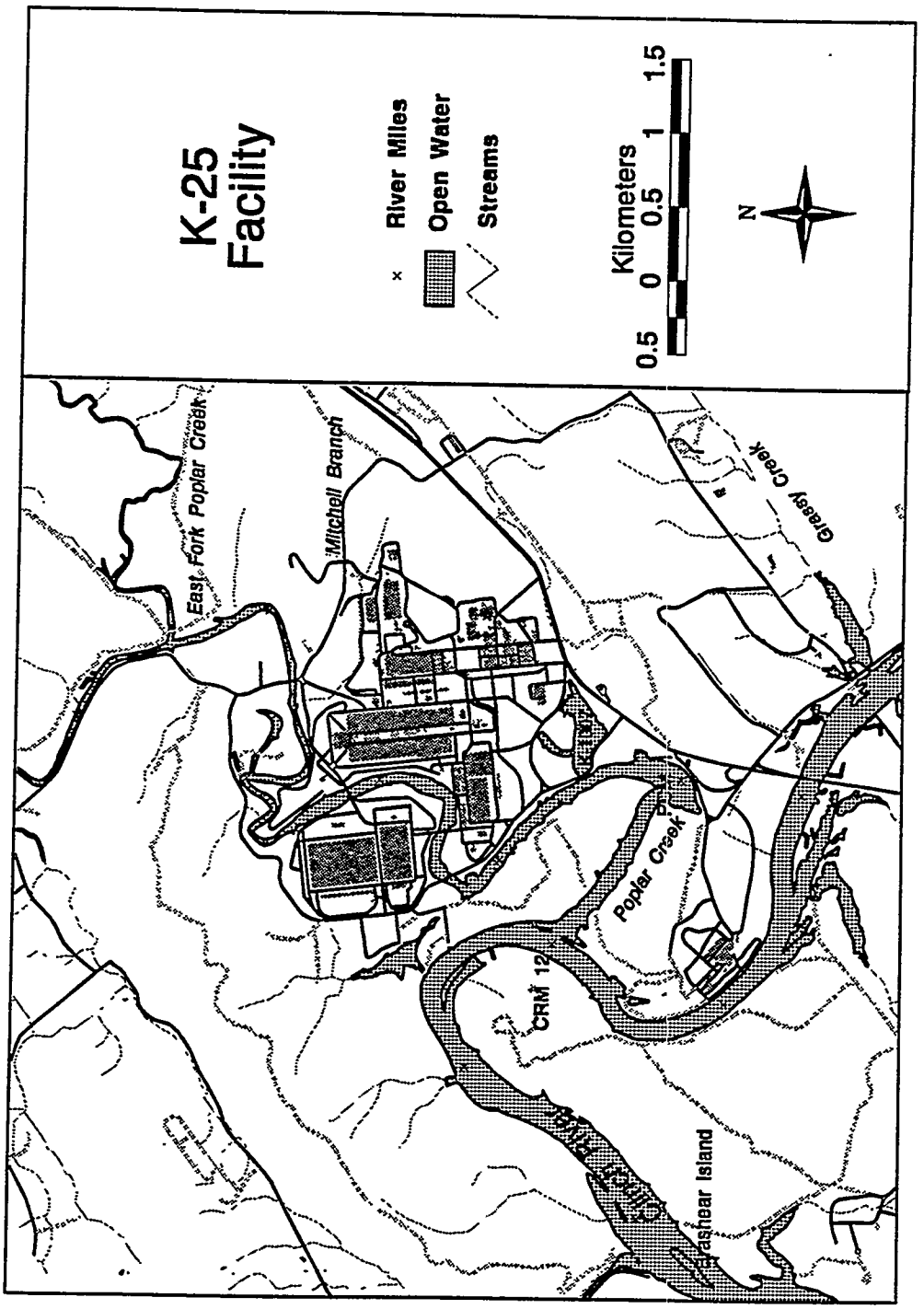


Fig. 3.4. The K-25 site.

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 (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 also 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 remaining contamination of the underlying soil with metals and radionuclides was found and 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 subsequently 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 effluents 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 via 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 1965–1966 to create the holding pond proper. In addition to the cooling system wastes described above, the pond was used to dispose of compressed

various fluoridated and chlorinated hydrocarbons (Goddard et al. 1991). The cylinders were reportedly breached prior to placement in the pond (Bruce et al. 1993). The K-901-A 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. Releases of PCBs from the switchyards (Goddard et al. 1991) have likely allowed migration of these chemicals 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 prior to 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 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, which could have contributed to contamination of on-site and off-site surface waters via 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 approximately 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 subsequently 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 Gunnite tanks

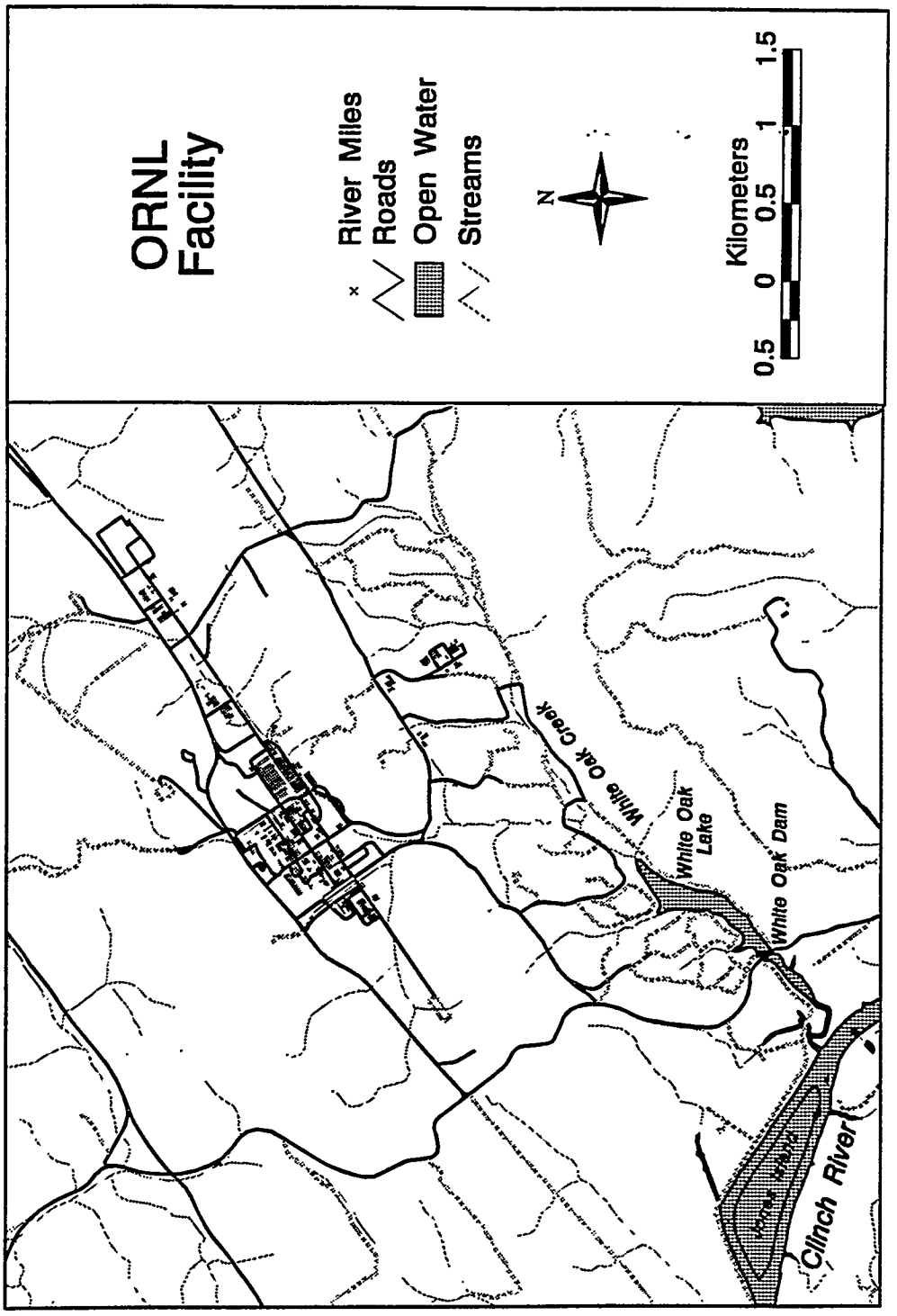


Fig. 3.5. The Oak Ridge National Laboratory.

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 Gunnite 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, with large quantities of diluting water, to WOC. 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 holding pond was built to provide an additional settling basin for liquids pumped from the Gunnite tanks and to allow for the additional decay of short-lived radionuclides prior to 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 Gunnite tanks. 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 waste water treatment plant was built to recover fission products from these (and other) liquid wastes prior to 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 down slope 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 disposal of liquid wastes using the hydrofracture technology was used (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 approximately 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 was a 1-acre site in use from 1943 to 1944, SWSA 2 was a 4-acre site in use from 1944 to 1946, and SWSA 3 was a 6-acre site used from 1946 to 1951. SWSAs 4 through 6 were located in Melton Valley, where the soil types were 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 was a 23-acre site used from 1951 to 1959, and SWSA 5 was a 50-

acre site in use 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 by being either buried on site or discharged as liquid waste to pits and trenches, since 1944, 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 was disposed of in pits and trenches; almost 60% of the ^{137}Cs and 78% of the ^{90}Sr was 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/yr 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 would 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 by 1975 annual releases were less than 2 Ci, where they have remained since (Table 3.1, Fig. 3.5). Total ^{137}Cs released to the Clinch River from ORNL is approximately 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 less than 4Ci/yr 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, with ^{3}H and transuranics being of somewhat less significance. Another 10% of the releases originates from operating facilities at ORNL, such as reactors, laboratories, and processing plants. The remaining 10% originates 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

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 regarding the amount of these various contaminants are generally impossible because of a lack of quantitative monitoring information. 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 prior to 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.

Table 3.1. Estimated[a] discharges (Ci) of radionuclides from White Oak Creek to the Clinch River, 1949–1992 (DOE 1988, Blaylock et al. 1993, Energy Systems 1992 and 1993, Vol II).

Year	^{137}Cs	^{106}Ru	^{90}Sr	TRE[b]	^{144}Ce	^{95}Zr	^{131}I	^{60}Co	^{3}H	TRU[c]
1949	77	110	150	77	18	180	77	[d]		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

[a] All digits carried through to avoid rounding errors. Only first two are significant.
[b] Total Rare Earth Elements, exclusive of cerium.

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 sediments. The greater solubility of the other radionuclides mentioned results in their being rapidly diluted and flushed downstream. Other possible contaminants released from the ORR include various metals (including As, Pb, Cr, Be, and Ni), various organic compounds (including PCBs, chlorinated and nonchlorinated hydrocarbons, and various laboratory chemicals), and various radionuclides (including transuranic radionuclides and fission products).

3.3 WATER CHARACTERIZATION

This section characterizes the nature and extent of contaminants in surface waters of the CR/PC OU. Data from historical studies are summarized (Sect. 3.3.1), and recent data from the ORR environmental monitoring program is then used to provide a general characterization of current conditions (Sect. 3.3.2). The CRRI Phase 2 investigation involved 6 tasks; data from these tasks is presented in Sects. 3.3.3–3.3.7. An originally planned task, the characterization of contaminant distribution downstream of point sources (DOE 1993), was discontinued after some initial data were collected. These data are not discussed in this section but are summarized in Tables A1–A3. Conclusions regarding the characterization of contaminants in surface water are presented in Sect. 3.3.8.

3.3.1 Historical Studies

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.

Clinch River Study. The comprehensive Clinch River Study was a multi-agency, interdisciplinary, 5-year (1960–1964) 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, the 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 to (1) determine the fate of radioactive materials that were currently being discharged to the river, (2) develop an understanding of the mechanisms of dispersion of radionuclides released to the river, (3) evaluate the limitations of the river for receiving radioactive effluents safely, and (4) suggest long-term monitoring procedures (Struxness et al. 1967).

The radionuclides of interest in the study (determined *a priori* based on 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 river (Struxness et al, 1967). The study area included the Tennessee River 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, approximately 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 6,409 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).

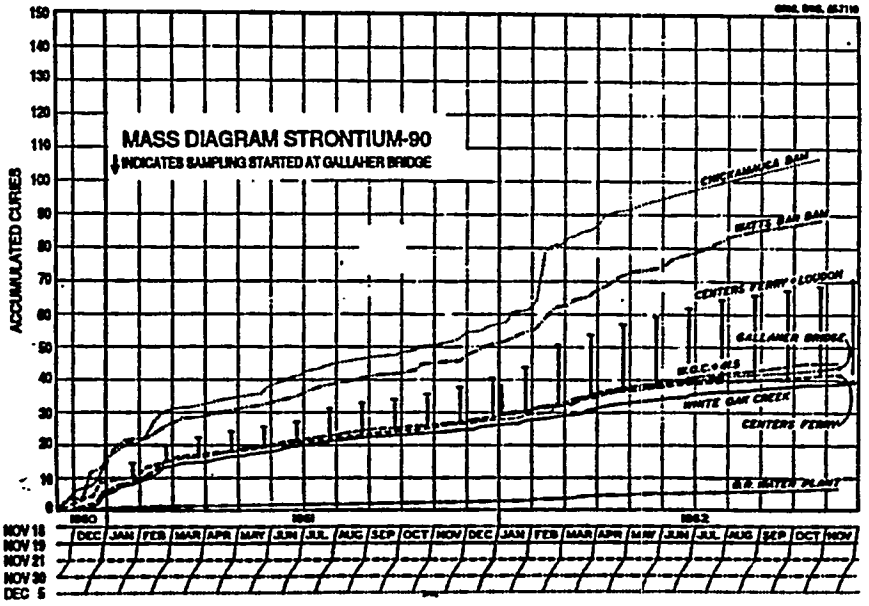


Fig. 3.6. Mass balance diagram for ^{90}Sr in surface water downstream of ORNL (Churchill et al. 1965). Note that accumulated Curies is consistently greatest at downstream stations, indicating no net loss of contaminant mass.

Oak Ridge Reservation Environmental Surveillance Reports. Monitoring data has been collected for a number of years for surface waters on and around the ORR. Table 3.2 summarizes the mean concentrations of selected metal and radionuclide analytes in surface water at CRM 9.9 for the period 1987–1992 (Energy Systems 1988–1993). The CRM 9.9 (Brashear Island, Fig. 3.1) station is downstream of all known ORR contaminant sources. Mean values are consistently below ambient water quality criteria (AWQC) and are expected to be representative of the Clinch River at this location. Organic analytes have rarely been detected in Clinch River surface water at this location.

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 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). The majority of potential contaminants-of-concern were detected in reaches 2 and 3. Inorganic mercury and lead were detected in reach 3. Elevated levels of cadmium and lead were observed in reach 4. Bis-(2-ethylhexyl)-phthalate was the only organic contaminant observed in any of the samples collected for Phase 1 site characterization. These Phase 1data have been combined with the Phase 2 data for purposes of site characterization and risk assessment.

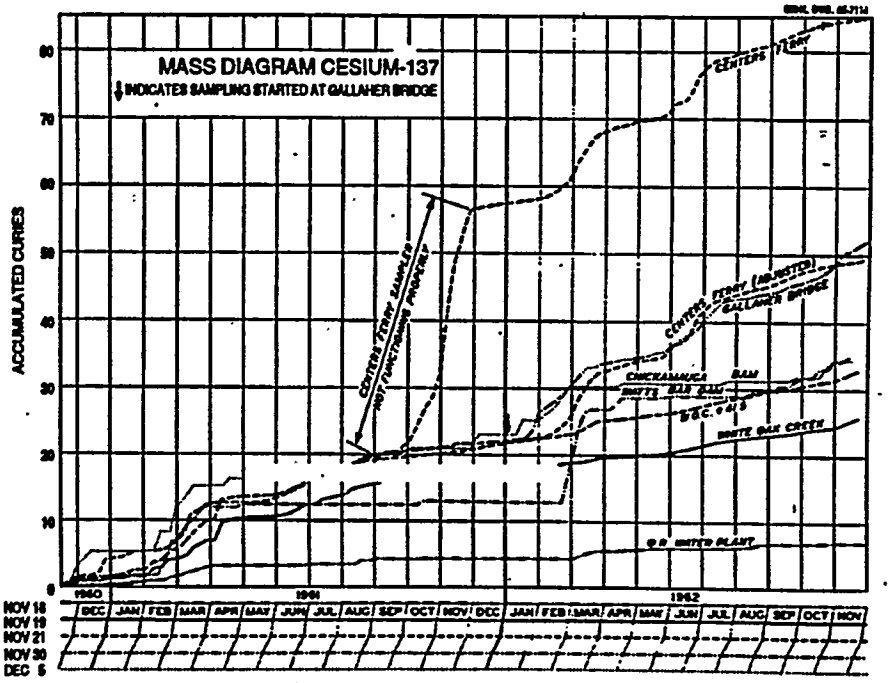


Fig. 3.7. Mass balance diagram for ^{137}Cs in surface water (Churchill 1965). Note that total accumulated Curies is less at Watts Bar and Chickamauga dams than at Gallaher Bridge, indicating a net loss of contaminant mass (to sediments) in Watts Bar Reservoir.

Arsenic Biogeochemistry in the Clinch River and Watts Bar Reservoir. The purpose of this study was to evaluate the distribution of arsenic species in the off-site surface water environment, with an emphasis on identifying potential sources of arsenic to the water column (Ford et al. 1995). This study involved the collection of surface water, sediment, and sediment pore water from the Clinch River and WBR for analysis of several arsenic species. Sensitive and precise processing and analytical techniques were used to measure arsenic species (arsenate, arsenite, and organic arsenic) at levels well below the ecological (190 μg/L for arsenate and 8.1 μg/L for arsenite) and human health (4.7 μg/L for arsenite) risk screening criteria; absolute detection limits using these techniques are approximately 0.0015 to 0.003 μg/L.

Inorganic arsenic levels were elevated (up to 1.5 μg/l) in surface waters (0 to 5 m) in the Kingston area. Values dropped with depth in the water column and were lowest near the sediment-water interface, suggesting an arsenic source directly to surface water in the Kingston area, downstream of the ORR.

Table 3.2. Mean, maximum , and minmum concentrations of selected analytes in surface water at Clinch River Mile 9.9 (Brashear Island) for the period 1987–1992[1]

Analyte	AWQC[2]		Year					
	HH[2]	Aquatic Life[3]	1987	1988	1989	1990	1991	1992
Arsenic	50	190[4]	ND[5] <5	ND <5	ND <5	ND <5	ND <5	4 2/5[6]
Cadmium	5	1[7]	<2 <2/3	ND <2	ND 2	ND <2	ND <2	1.2 0.5/2
Chromium	100	100	ND <10	ND <10	ND <10	18 <10/ 440	ND <10	
Lead	NA[8]	3[7]	<7 <4/25	<4 <4/11	ND <4	ND <4	ND <4	3.3 2/4
Manganese	50	NA	20 <10/74	67 20/200	79 69/ 100	49 25/94	74.4 30.6/ 150	69.3 30/140
Mercury	2	0.012			ND <0.2	ND <0.2	ND <0.2	
Nickel	100	160	ND <50	<52 <50/74	ND <50	12 <10/21	ND <10	23.3 10/50
^{137}Cs	NA[9]	NA[10]	ND <100	<3.8 -100/ <100	ND 0	0.75 0/1.97	0.037 -21.6/ 18.3	2.23 -15.3/ 17

[1] All units are μg/L except ^{137}Cs, which is pCi/L. Blank cells indicate no analyses reported. Data from Energy Systems (1988–1993).
[2] AWQC = ambient water quality criteria. HH = human health.
[3] Criteria continuous concentration
[4] Arsenic III
[5] ND = analyte not detected.
[6] Minimum/maximum values for the year.
[7] Calculated for hardness = 100 mg/L (approximate hardness of Clinch River water); values are for dissolved form.
[8] No AWQC for water supply available.
[9] AWQC is 4 mrem/yr for all radionuclides combined.
[10] DOE Order 5400.5 specifies 1 rad/day

Sediments in Poplar Creek deeper than 15 cm were grossly contaminated with arsenic. Sediment pore water values averaged 2850 $\mu g/L$. However, surface sediment arsenic concentrations did not differ from those at Kingston or in Melton Hill Reservoir. The Poplar Creek site received fly ash directly from disposal operations at the K-25 site on the ORR until 1963. Evidence suggests that deposition and accumulation of cleaner sediment, in combination with stream-like flow rates and oxic conditions throughout the water column, may be preventing significant release of this arsenic to surface waters of Poplar Creek.

Arsenic concentrations in Rogers Quarry outfalls (McCoy Branch mile 1.0) dropped greatly during this study as a result of actions taken to curtail and eventually eliminate fly ash disposal in Rogers Quarry. However, arsenic concentrations in the Rogers Quarry discharge were in excess of human health and ecological risk screening criteria for all sample collection periods. Arsenic concentrations in surface waters (0- to 2-m deep) decreased with distance downstream in McCoy Branch.

Seasonal increases in dissolved inorganic arsenic were observed both in the sediment pore waters and surface waters throughout the McCoy Branch Embayment. In summer, maximum values in upper McCoy Branch Embayment exceeded human health risk screening criteria. The maximum summer values in lower McCoy Branch Embayment were elevated, but did not exceed human health risk (drinking water) criteria. Changes in surface water arsenic values in McCoy Branch Embayment may be attributed largely to seasonal changes in sediment pore water chemistry, rather than to changes in Rogers Quarry outflows.

TVA Recreational Areas and Water Intakes Study. The purpose of this study was to provide screening-level data to evaluate the potential threat from contaminants to persons whose drinking water supply is Melton Hill or WBR, or to persons using the recreational facilities on these waters (TVA 1991). The analysis of water samples was limited to ^{3}H. A single grab sample was collected from each of 18 recreational areas and 11 water intake locations in Watts Bar, Melton Hill and Norris reservoirs in May and June 1991. Tritium levels were very low, with only six samples above detection limits (200 to 230 pCi/L) and activities that 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 3 beach areas in WBR and 2 water intakes in the Clinch River (TVA's Kingston steam plant and the DOE intake at Grassy Creek).

Summary of Historical Data. The historical data, considered together with the hydraulic conditions in the Clinch River (Chap. 2), tentatively point to the following site conceptual model for contaminants in water. Organic contaminants are infrequently detected and are present at very low concentrations when detected. Inorganic analytes are present at levels generally below AWQC. Radionuclide concentrations are generally very low and do not exceed water quality criteria. Sources of contaminants to off-site waters are generally via point-source discharges, tributary flow, or non-point source runoff. However, McCoy Branch surface water receives soluble arsenic that is released seasonally from contaminated sediment; maximum values exceed certain human health criteria at some locations. Despite gross arsenic contamination in Poplar Creek sediment, migration of arsenic to surface water appears limited by sediment and hydraulic conditions.

3.3.2 Oak Ridge Reservation Environmental Monitoring Data

The Oak Ridge Reservation Environmental Monitoring (ORREM) Program is designed to collect and analyze surface water data to assess the impact of contaminants on the ORR on human health and the environment (DOE 1994a). Surface grab samples are collected bimonthly from several sites (Fig. 3.8). Samples are analyzed for general water quality parameters, total metals, and volatile organics.

Samples are also screened for radioactivity and analyzed for specific radionuclides when appropriate. Data from 14 ORREM sites collected during 1993/1994 (n = 9 to 10 at each site) are used here to verify the conceptual model above.

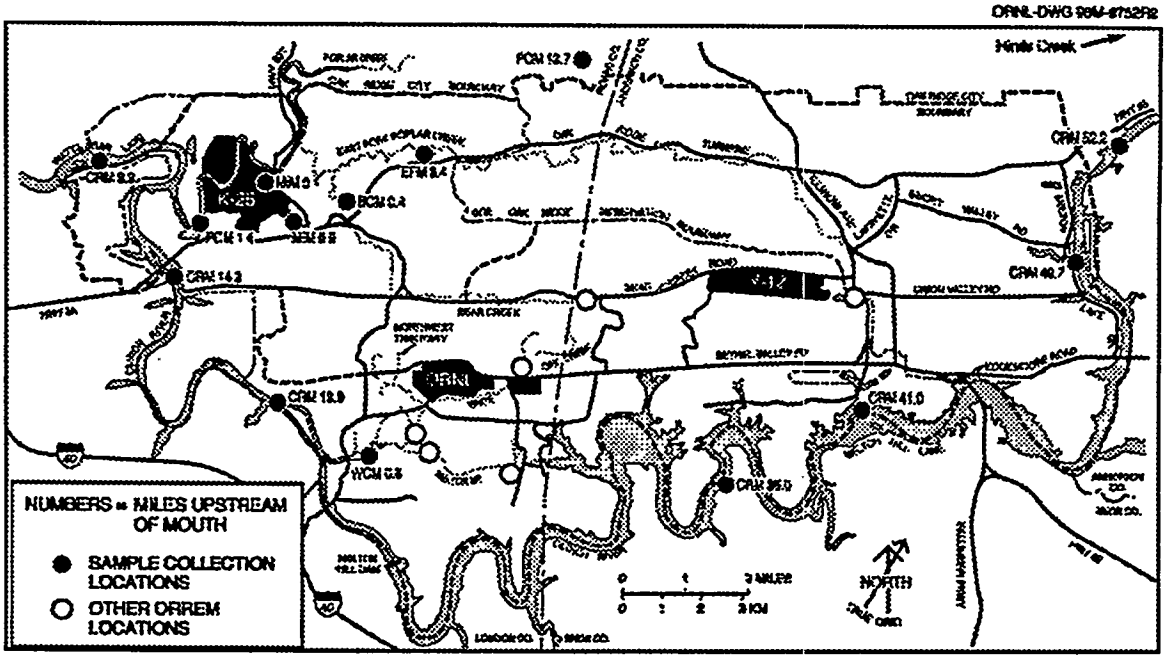


Fig. 3.8. Selected surface water sampling sites from the Oak Ridge Reservation Environmental Monitoring Program.

Many analytes were detected infrequently (i.e., in less than half of the samples at every location), including all organic compounds, cyanide, Sb, As, Be, Cd, Co, Cu, Pb, Se, Ag, and Tl (Table A4). Analytes detected in more than half the samples from any site are listed in Table A5 along with summary descriptive statistics. In general, mean concentrations of these analytes did not differ significantly ($p < 0.05$) among sites in the Clinch River or Poplar Creek. [Gross beta values below WOC (CRM 19.9) were significantly higher than at one reference site (CRM 52.2), and gross alpha values at PCM 1.4 were significantly higher than at one reference site (CRM 41.0).] The mean concentrations of each analyte in Table A5 are below AWQC for all Clinch River and Poplar Creek stations.

These data are combined with data collected during the CRRI and used to calculate representative concentrations for surface water exposure pathways in the baseline risk assessment (Chaps. 5 and 6).

3.3.3 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), to maximize contaminant flux from both groundwater and runoff but allowing high dilution; (2) low baseflow/high runoff (summer rain) to minimize contaminant flux from groundwater while maximizing contaminant flux via surface runoff; and (3) low baseflow/low runoff (summer dry), minimizing contaminant flux from both sources

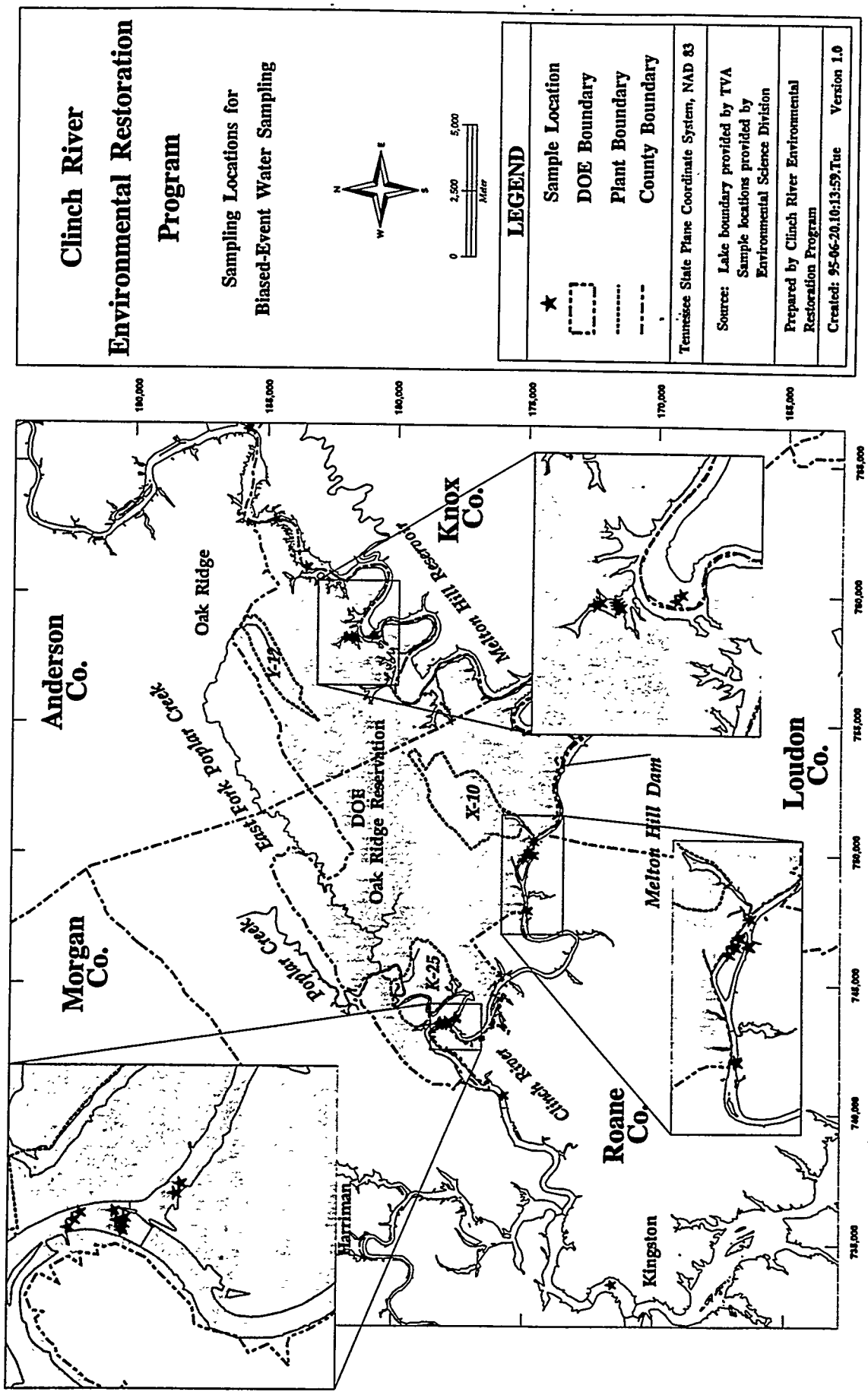


Fig. 3.9. Sampling locations for characterizing contaminant concentrations associated with extreme hydrologic events.

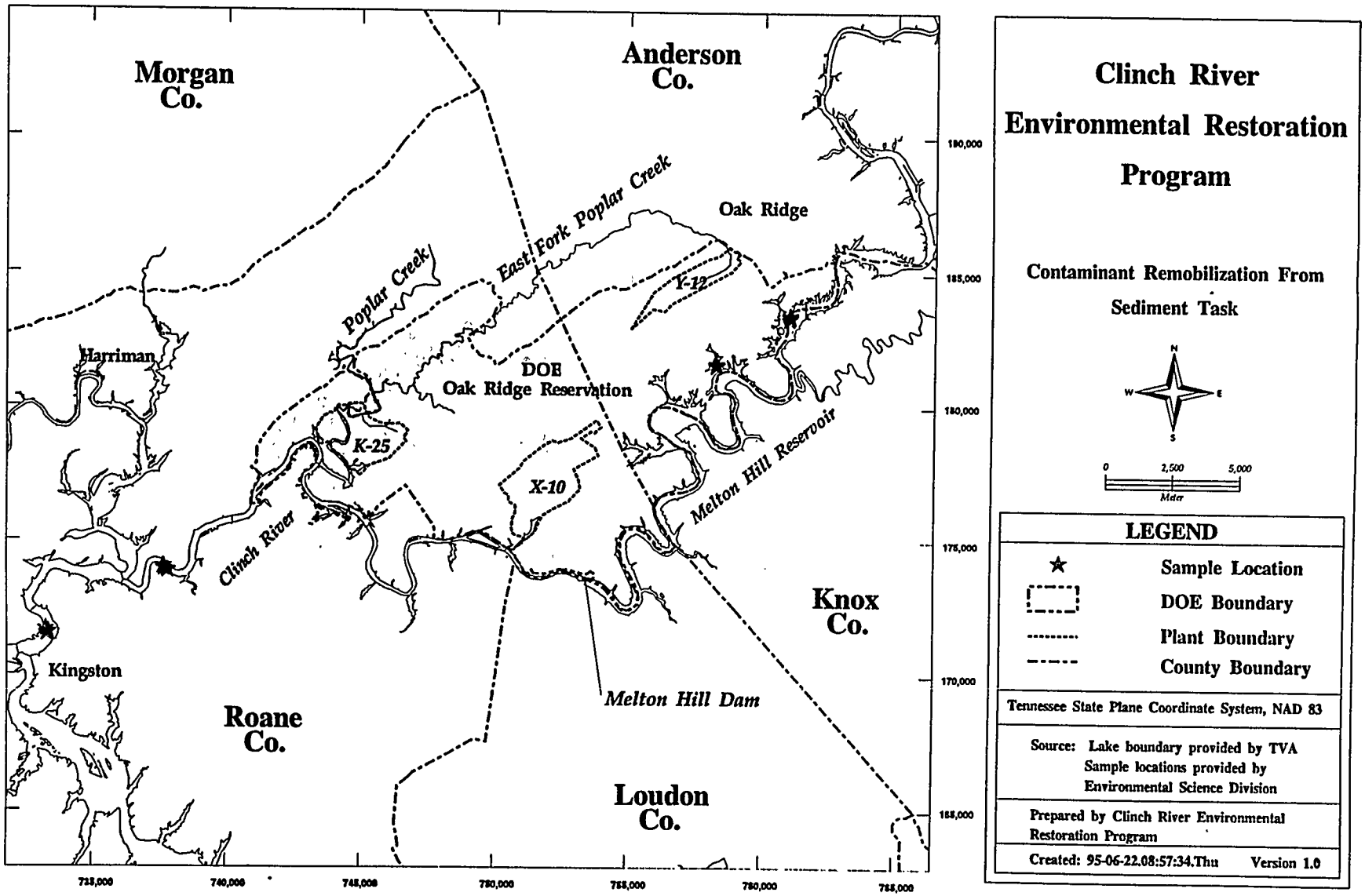


Fig. 3.10. Sampling locations for evaluation of contaminant remobilization for sediments.

but also minimizing dilution (DOE 1994b). Initially a fourth condition, high baseflow/low surface runoff was to be sampled, but these hydrologic conditions were not observed during the study period.

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 "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, 1994 (Figs. A1 and A2). Ideally, flow in the Clinch River would be minimal during sampling to minimize dilution, but due to 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 5 locations (Fig. 3.9) 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 sufficiently far downstream to ensure uniform mixing of contaminants. Four reference sites were also sampled (Fig. 3.9). Each of the 19 sites were sampled once per flow event. Information regarding analytical classes, sampling locations, and frequency of collection for these data is listed in Table A6. Water quality parameters for all sites reflected seasonally expected norms (Sect. 2.5.3).

Results for each source stream are presented in Tables A7–A10. 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 were higher than at all study sites except the McCoy Branch source location. 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 expected decrease in concentrations expected from dilution of Poplar Creek water, but the data are too scant to draw any conclusions.

3.3.4 Contaminant Remobilization from Sediment Study

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. 3.10), three times per season for two seasons and analyzed for metals, radionuclides, and organic contaminants (Table A6). Results are presented in Tables A11–A17. 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.

3.3.5 Characterization of Additional Sources of Contaminants to the CR/PC OU

(Data for this task were received too late for inclusion in this report. Text and data will be included in the D2 version. Sample locations will be identified in Fig. 3.11. Data will be summarized as Table A18 in that report.)

3.3.6 Characterization of Contaminants in Poplar Creek Surface Water

The goal of this task was to characterize the average ambient conditions in Poplar Creek with respect to contaminant concentrations. Samples were collected from eight sites in Poplar Creek (Fig. 3.12) four times in each of three seasons (winter, spring, and summer). 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 A6). Results are presented in Tables A19–A26.

Mean concentrations of all analytes were below AWQC at all locations over the study period as a whole. Concentrations of several inorganic analytes (Tables A19–A21) varied seasonally and were often highest in summer. Spatially, concentrations were similar across all sites (including reference sites) and were within the range historically reported (Gaydos et al. 1983) for a reference site on Poplar Creek. An exception to this was nitrate/nitrite, concentrations of which were significantly elevated below the mouth of Mitchell Branch (Fig. 3.13). In addition, total inorganic mercury and total methyl mercury concentrations were significantly elevated downstream of EFPC (Fig. 3.14), as expected. Although it was expected that mercury flux would be largely associated with periods of increased suspended sediment load, correlation of mercury concentrations and total suspended particle values were weak ($R^2<0.2$). Contrary to expectations, the highest mercury concentrations were observed near the mouth of Poplar Creek rather than immediately below EFPC. One explanation may be resuspension of sediment between EFPC and the mouth of Poplar Creek. Few organics were detected (Tables A22–A23) and reference concentrations were similar to those at the study sites. Measures of gross-alpha and gross-beta emission in water samples did not differ significantly among sites over the study period as a whole. In winter, however, both measures were significantly elevated below the mouth of Mitchell Branch compared to reference sites (Fig. 3.15). No individually measured radionuclides demonstrated spatially significant patterns.

3.3.7 Aqueous Toxicity Tests

Ambient water from the CR/PC OU was also evaluated for toxicity to aquatic organisms. 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 via microbial assay. Study sites (Fig. 3.16) 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 in Poplar Creek was coordinated with samples collected for the characterization task in that reach (Sect. 3.3.6). Samples for contaminant analysis were also collected in conjunction with toxicity testing at sites in Melton Hill Reservoir and in the the Clinch River below Melton Hill Dam (for three of the toxicity sampling dates). In the fathead minnow and *Ceriodaphnia* tests, data from the Clinch River were analyzed separately from data from Poplar Creek. These data are summarized in Tables A27–A29. Standard test protocols for fathead minnow and *Ceriodaphnia* tests are described by Kszos et al. (1994). Medaka test procedures can be found in ER/CR-SOP 2162.

Fig. 3.11. Sampling locations for characterization of additional contaminant sources to the CR/PC OU (TVA 1995). (TO BE PROVIDED IN THE D2 VERSION.)

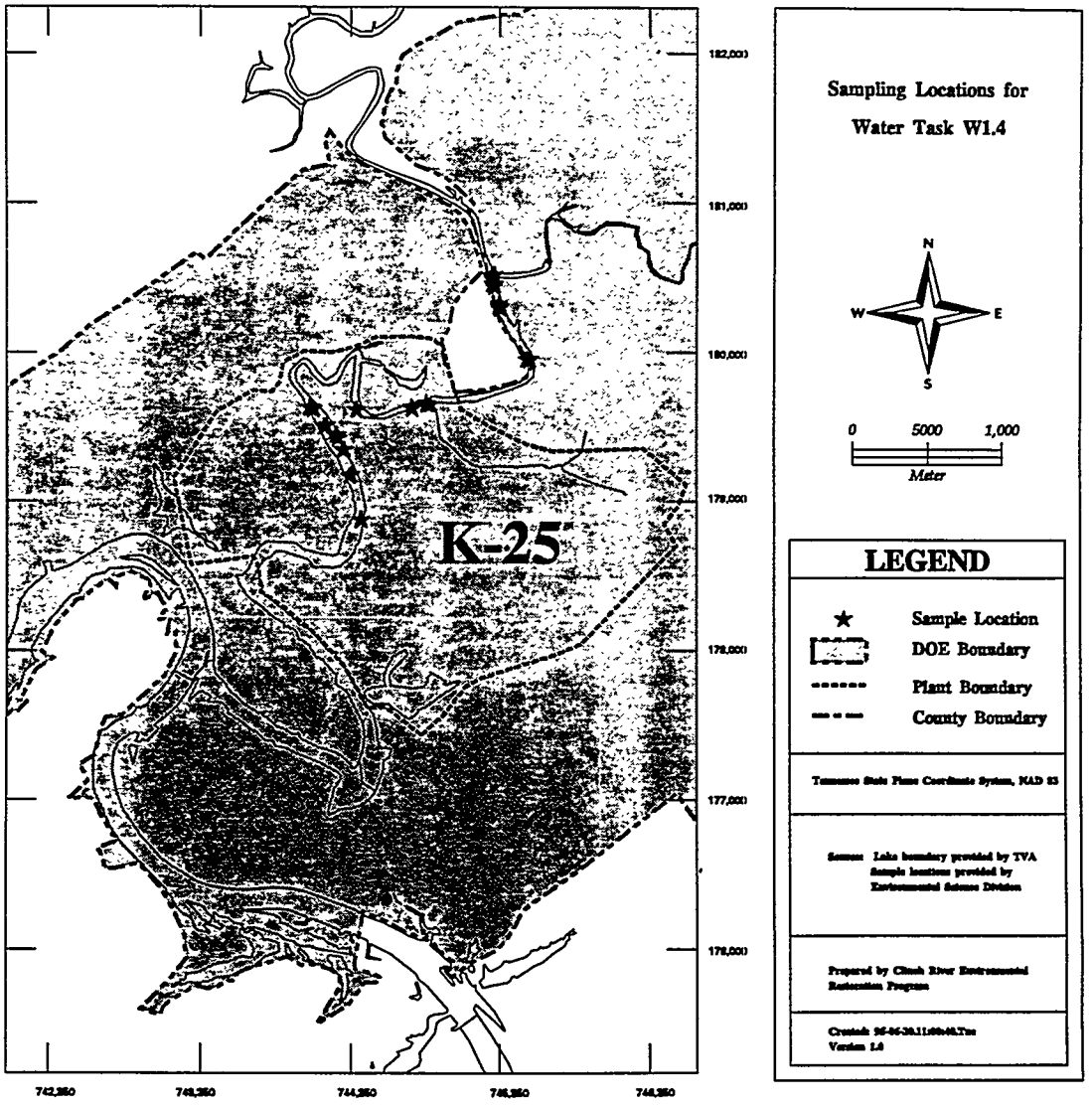


Fig. 3.12. Sampling locations for characterization of average ambient contaminant concentrations in Poplar Creek.

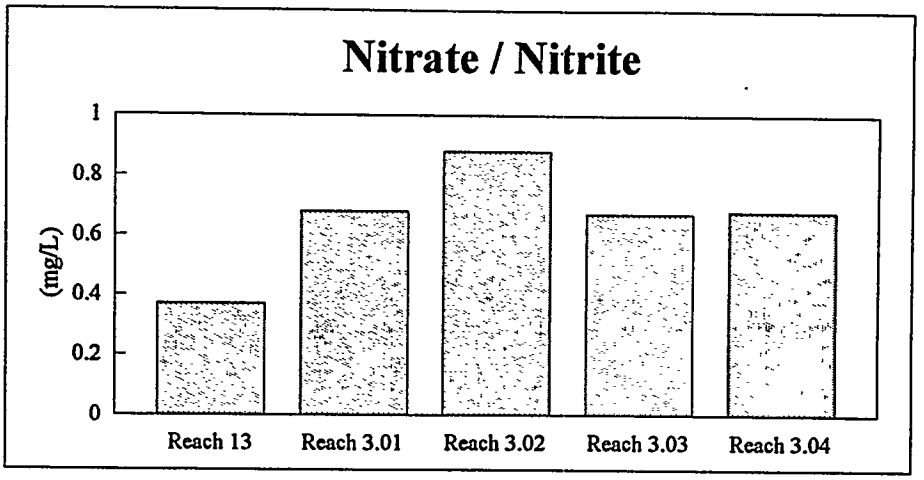


Fig. 3.13. Mean dissolved nitrate/nitrite concentrations in Poplar Creek by reach.

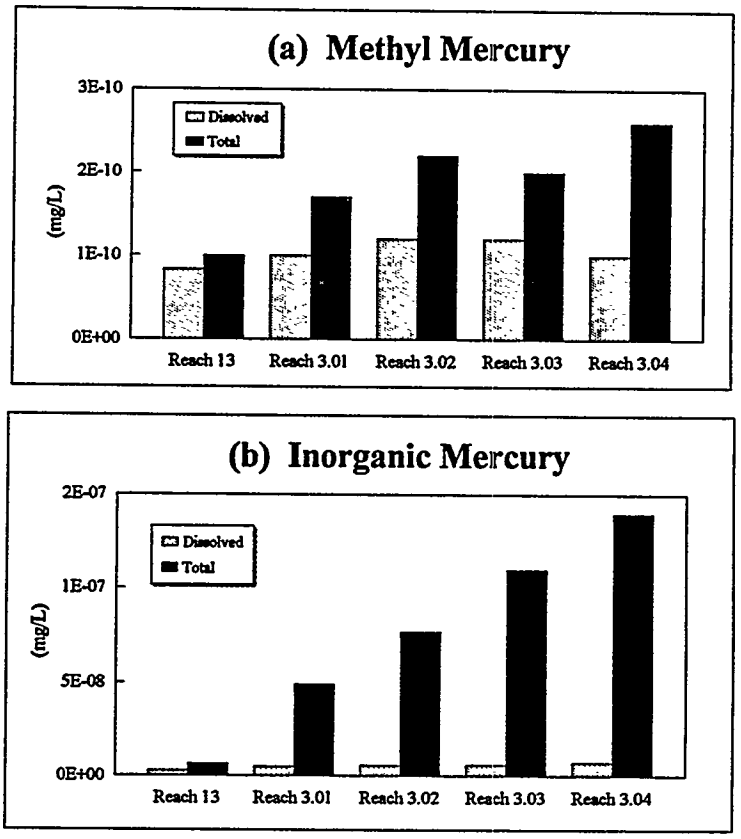


Fig. 3.14. Mean mercury concentrations by reach in Poplar Creek (a) methyl mercury and (b) inorganic mercury. The Tennessee water quality criteria for designated uses for inorganic mercury is 0.0024 mg/L maximum concentration and 0.000012 mg/L continuous concentration.

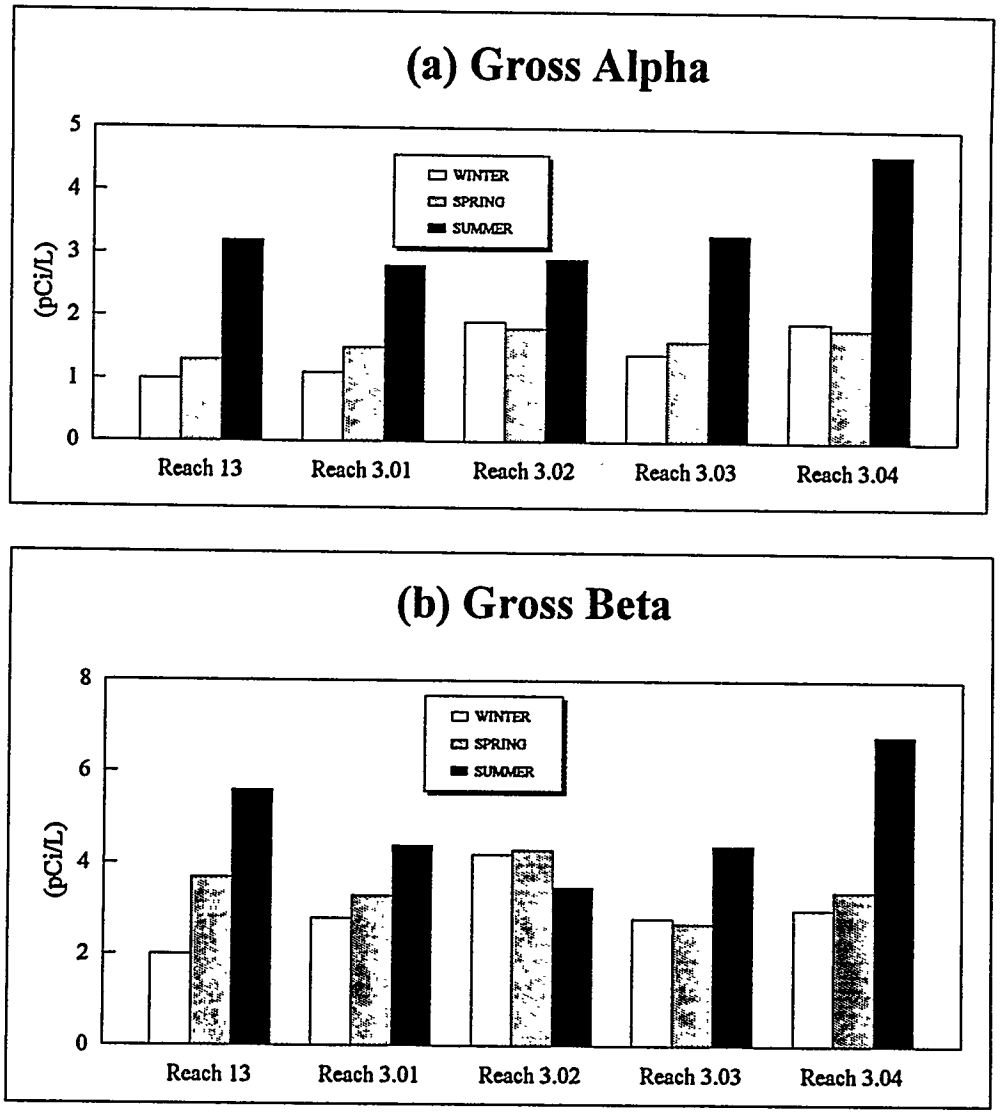


Fig. 3.15. Mean radioactivity concentrations by season within each reach for Poplar Creek (a) gross alpha and (b) gross beta.

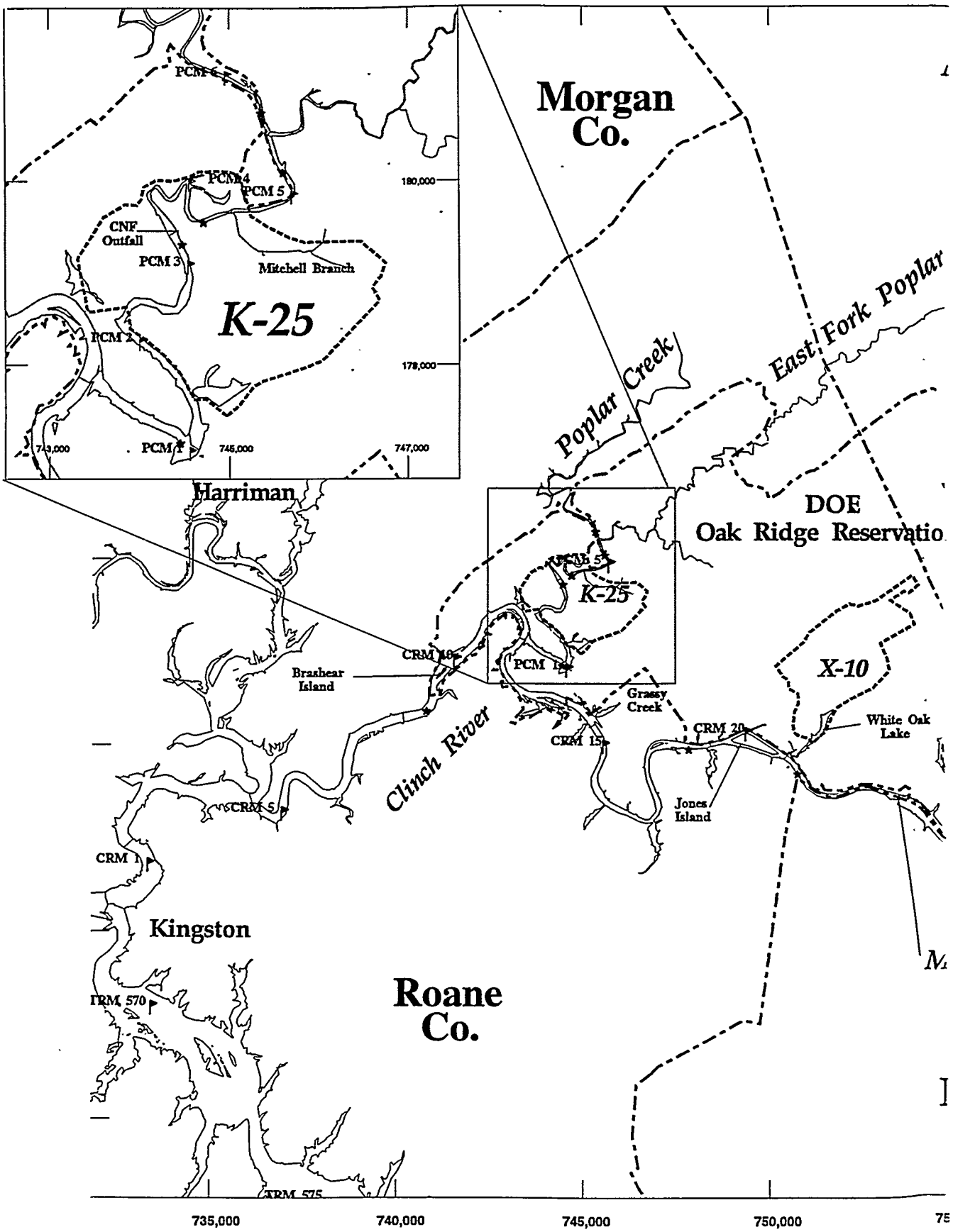
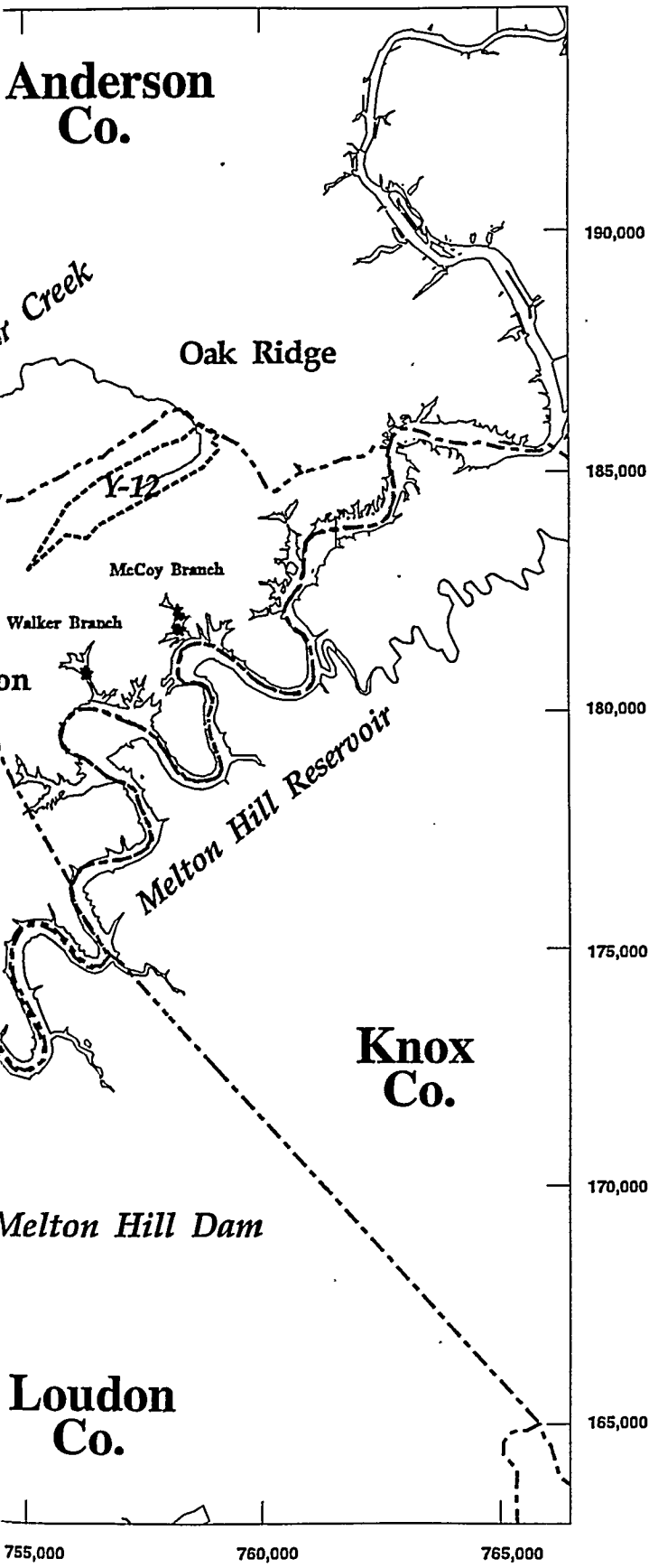
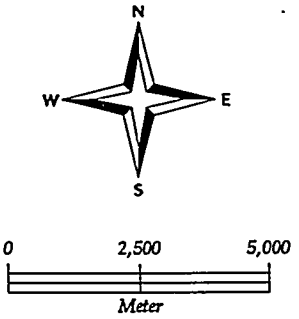


Fig. 3.16. Sample locations for aqueous toxicity testing.



Clinch River Environmental Restoration Program

Sampling Locations for Water Toxicity Task



LEGEND

*	Sample 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 Clinch River Environmental Restoration Program

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3.3.7.1 Fathead minnow test results

An analysis of variance (Table A28) revealed no significant differences in fathead minnow survival or growth among the Poplar Creek and Clinch River ambient test sites during each of the six tests conducted, with one exception. In the April 1994 test, minnow survival in subreach 3.01 (PCM 5.1) was significantly lower than survival in subreach 3.04 (PCM 1.0) (Fig. 3.17). There were no significant spatial or temporal effects between the McCoy Branch and Walker Branch embayments. Fathead minnow survival data are summarized in Table A30.

Large among-replicate variation in fathead minnow survival was encountered in nearly all of the tests using water samples from the Poplar Creek and Clinch River ambient sites. Generally, minnow survival was high (90–100%) during the first few days of the test and then declined thereafter. This phenomenon has been observed in other studies conducted using ambient water samples collected from both upstream reference sites and sites downstream of facility discharges on the ORR. The large among-replicate variation in minnow mortality is suspected to be caused by indigenous facultative pathogenic bacteria or aquatic fungi. In other studies conducted by the Toxicology Laboratory, treatment of water samples with ultraviolet (UV) light improved minnow survival and eliminated the large among-replicate variation. Though there is strong circumstantial evidence for pathogens, the possibility also exists that organic compounds could be destroyed or altered by exposure to UV light (Kszos et al. 1992).

Minnow survival in the Poplar Creek sites and at CRM 9.0 was lowest during the test conducted in April 1994 (Fig. 3.17). Mean survival in the Poplar Creek sites ranged from 42.5% to 77.5% and at CRM 9.0 was 45%; in comparison, control survival was 97.5%. Water samples from the ambient sites were collected during a rain event, and water levels in Poplar Creek were estimated to be 3–4 ft above normal flow (Sect. 2.5.3). The increase in turbidity, particularly exposure to colloidal clay particles, may have clogged gill filaments and interfered with respiration (Bennett 1970), which may have contributed to the higher mortality during this test.

Fathead minnow growth in the ambient water samples was generally greater than minnow growth in laboratory control water (Table A31); however, growth in all test sites and the control was lower in the April 1994 test (Fig. 3.18). It appears that the organisms used in this particular test were more sensitive relative to other test periods. The younger age of the larvae (< 8-h old) used for this test may be a contributing factor.

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. 3.19); however, the two Poplar Creek reference sites also differed from each other by 20% during several tests. No 20% reductions in growth or survival were noted for the Clinch River stations.

3.3.7.2 *Ceriodaphnia dubia* test results

No spatial or temporal patterns in *Ceriodaphnia* survival were observed; survival was uniformly high (>80%) in all tests (Table A32). 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 (WBM 0.4, reach 8) was significantly greater than reproduction in upper McCoy Branch Embayment (MBM 0.4, reach 7.02) during both test periods.

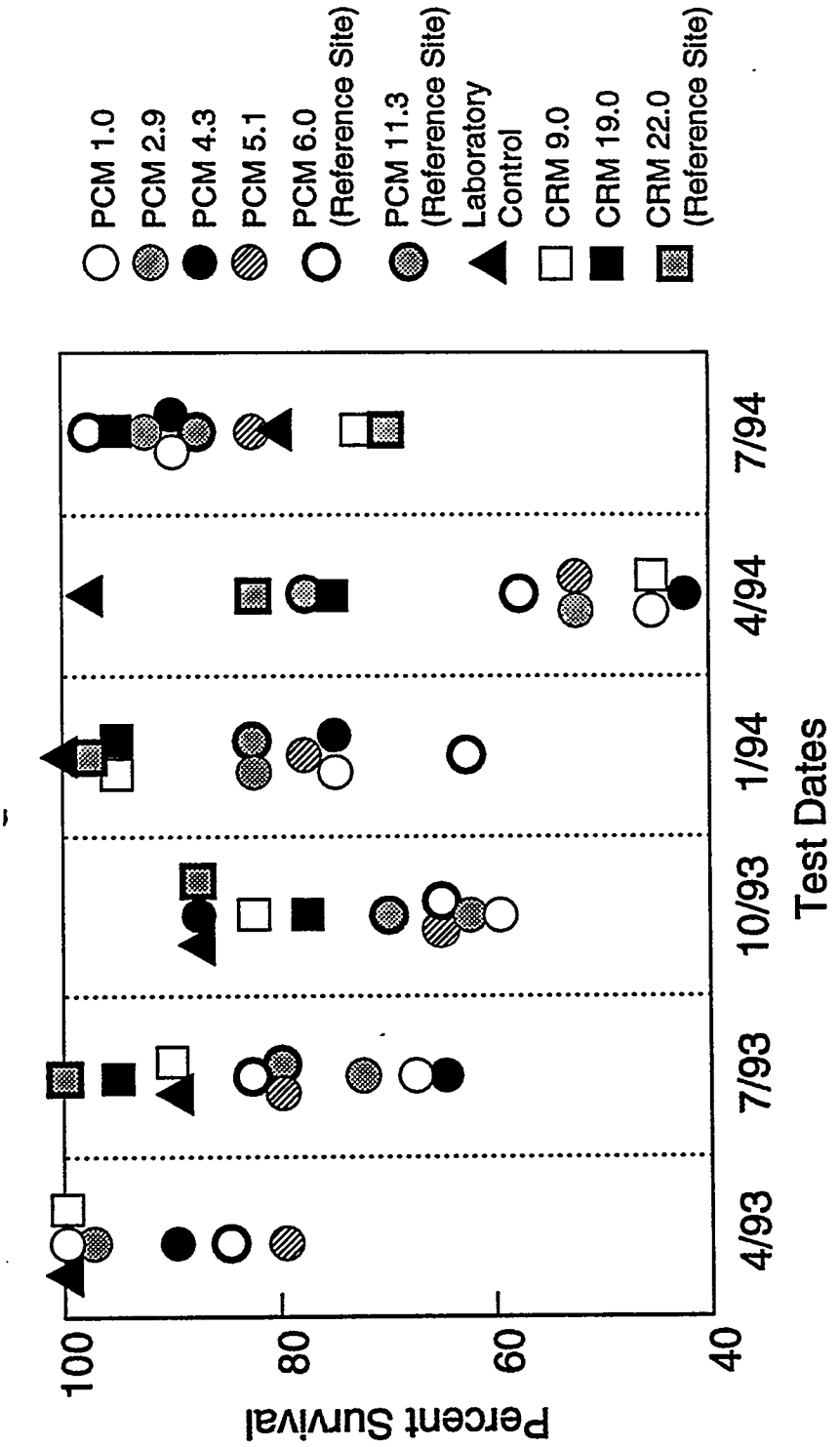


Fig. 3.17. Fathead minnow survival in water samples from Poplar Creek and Clinch River sites by test date.

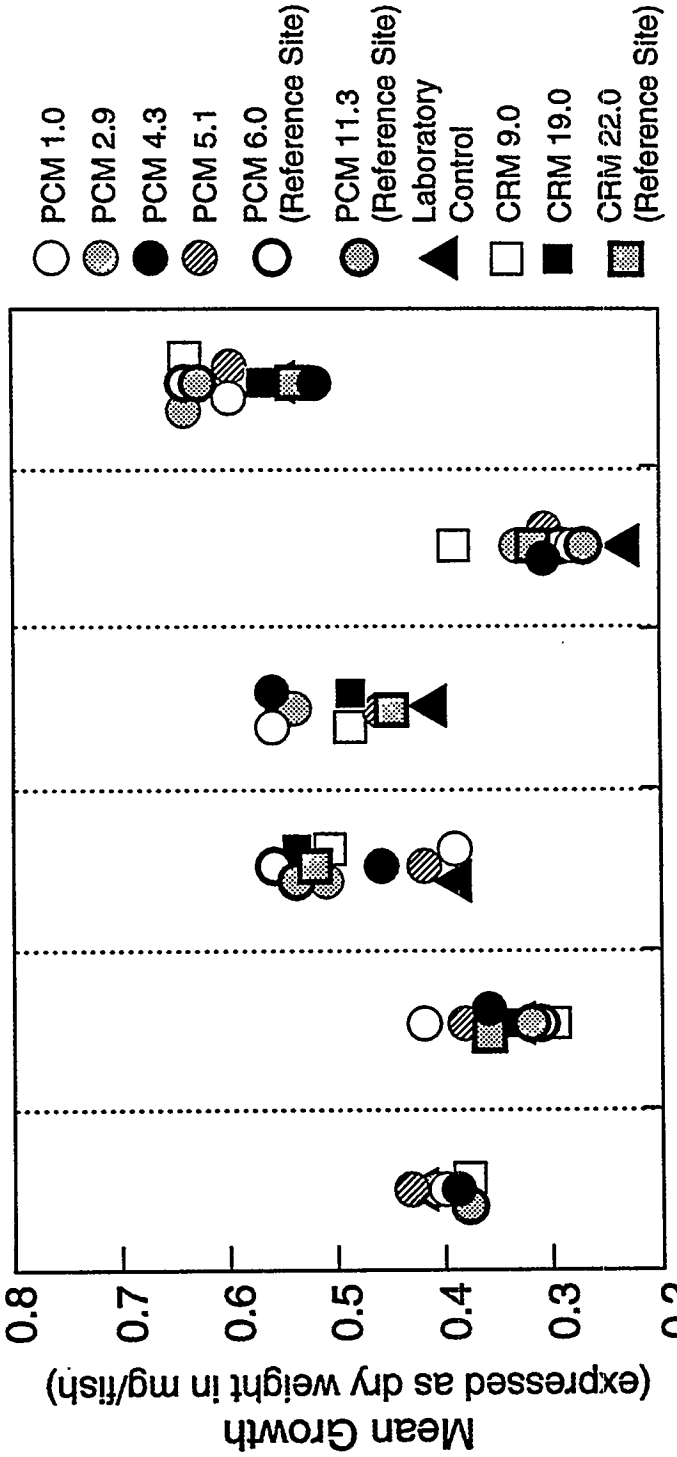
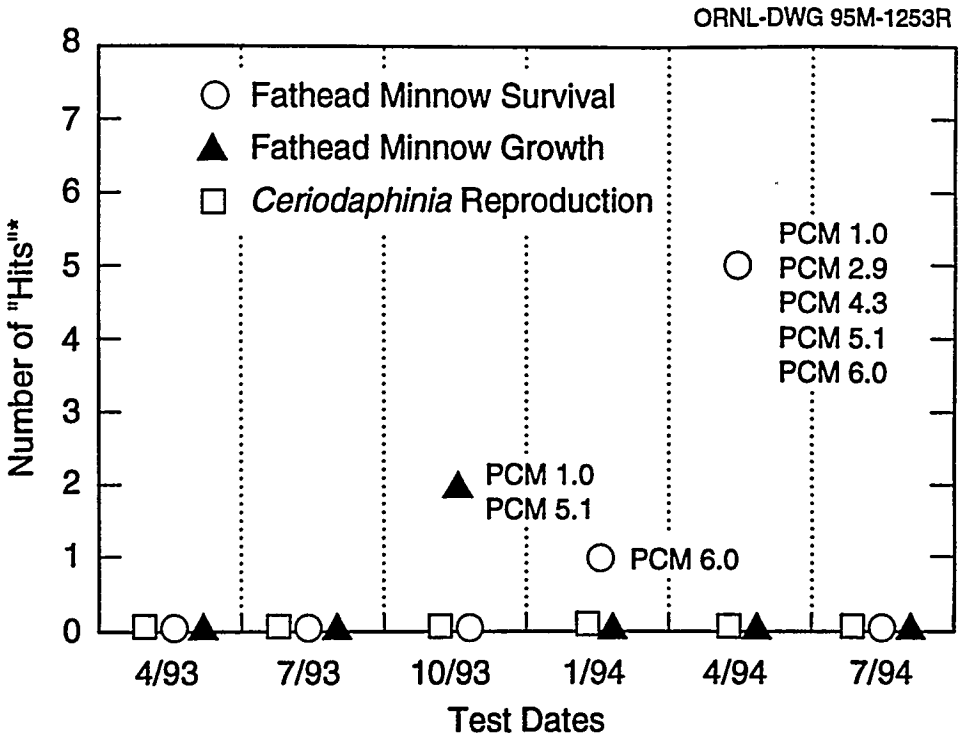
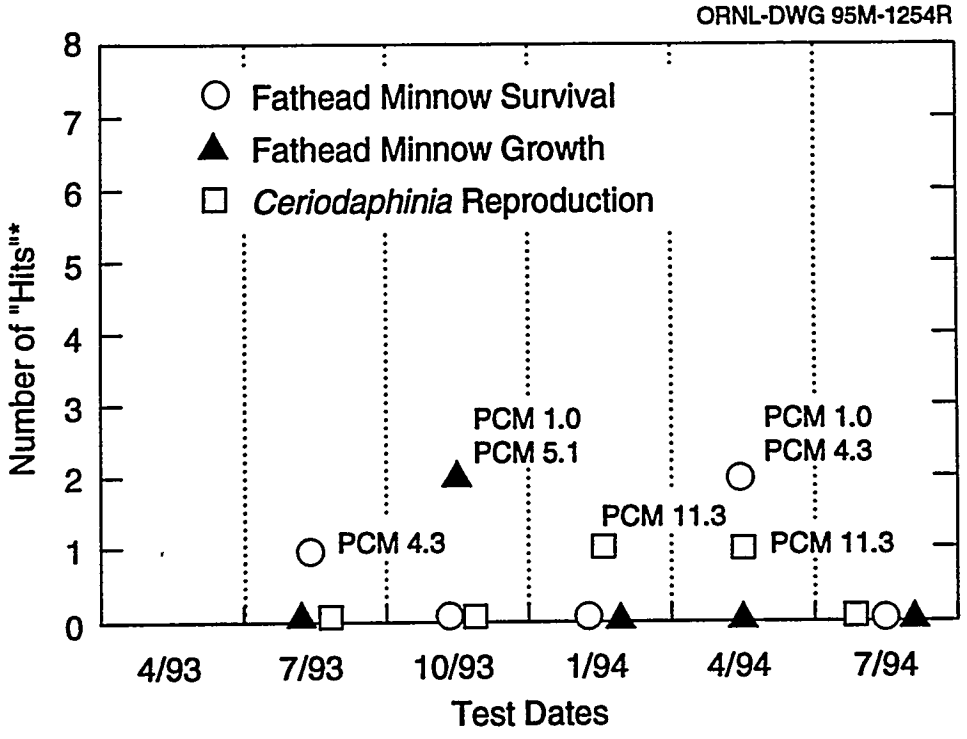


Fig. 3.18. Fathead minnow growth in water samples from Poplar Creek and Clinch River test sites by test date.



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

Fig. 3.19(a). Comparisons made to reference site PCM 11.3 of fathead minnow survival and growth and *Ceriodaphnia* reproduction; number of hits by test date.



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

Fig. 3.19(b). Comparisons made to reference site PCM 6.0 of fathead minnow survival and growth and *Ceriodaphnia* reproduction; number of hits by test date.

Though there were significant temporal differences, 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 Table A32.

The only (statistically nonsignificant) reductions in fecundity of 20% or more between sites were reductions in one Poplar Creek reference site (PCM 11.3) relative to the other Poplar Creek reference (PCM 6.0) (Fig. 3.18).

3.3.7.3 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 . Subsequent to these tests, two additional medaka embryo-larval tests were conducted on many of the same sites for the K-25 Site 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 incidences 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 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 (α=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 pre-hatching interval (10–12 days on the average), the medaka embryo appears most sensitive to adverse water conditions in the period prior to hatch. Further discussion will thus focus 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 A34) and in both of the K-25 Site BMAP tests (Table A35). 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 A34 and A35). Nonetheless, the general pattern throughout the tested portion of Poplar Creek was for a decrease in mean survival in comparison to the reference situation for all downstream sites adjacent to the ORR (Fig. 3.20), with greatest impacts observed in subreach 3.02 (PCM 4.3) and subreach 3.04 (PCM 1.0).

The incidences 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 A34 and A35).

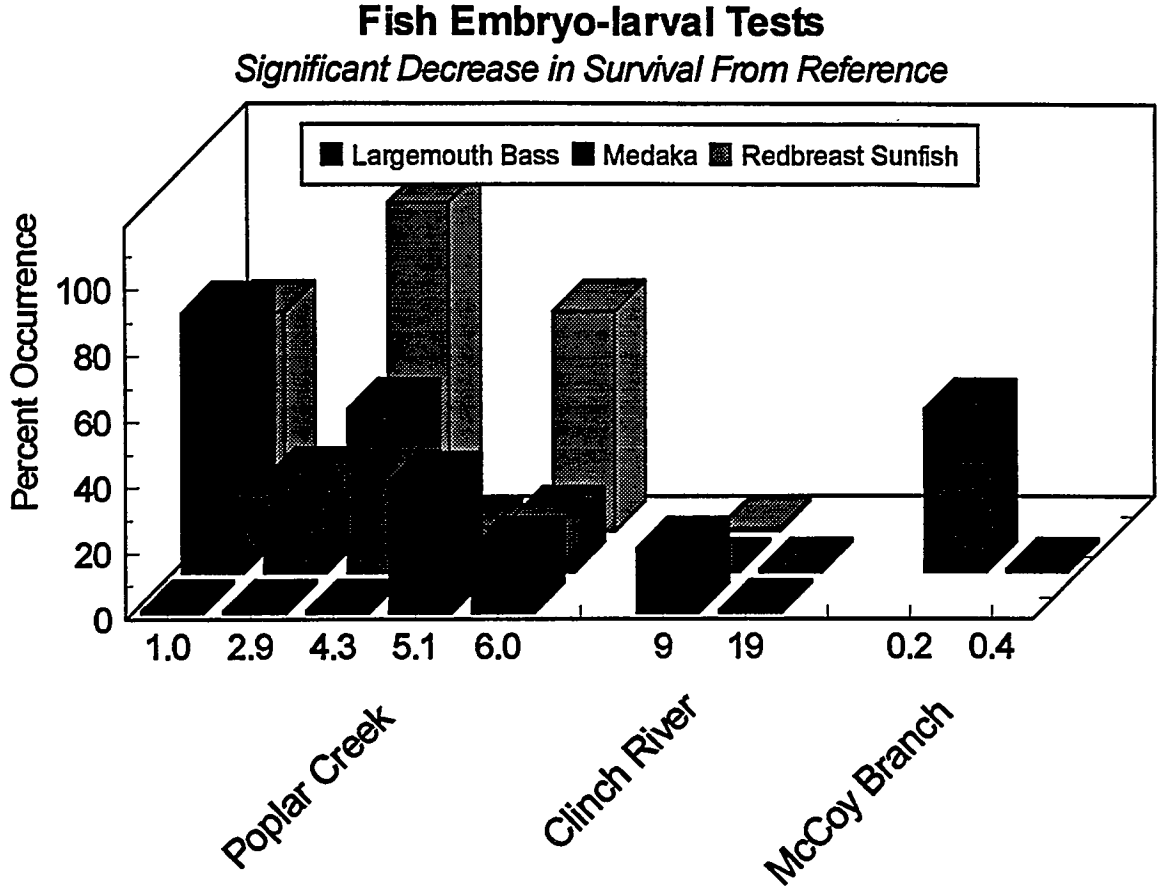


Fig. 3.20. Mean frequency of occurrence of significant decreases in fish embryo-larval survival in water from the various study sites as compared with the appropriate reference. For medaka, the values are based on a series of five to eight different tests of water from Poplar Creek and Clinch River sites 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).

Certain abnormalities were seen more frequently at the two downstream sites (subreach 2.02 at CRM 19 and subreach 4.01 at CRM 9) as compared with the upstream reference site (subreach 2.01 at CRM 22) (Fig. 3.21), 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.

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 A34 and A35). For example, overall incidences of developmental abnormalities significantly higher than in controls were noted in six of seven medaka embryo-larval tests conducted on water samples from subreach 3.02 (PCM 4.3). The incidences of many specific abnormalities were also greatest in embryos exposed to water from sites in lower Poplar Creek adjacent to the ORR (Fig. 3.21). 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 appears due in many instances to problems with the proper maintenance of chorionic integrity, leading to premature and largely unsuccessful attempts to hatch.

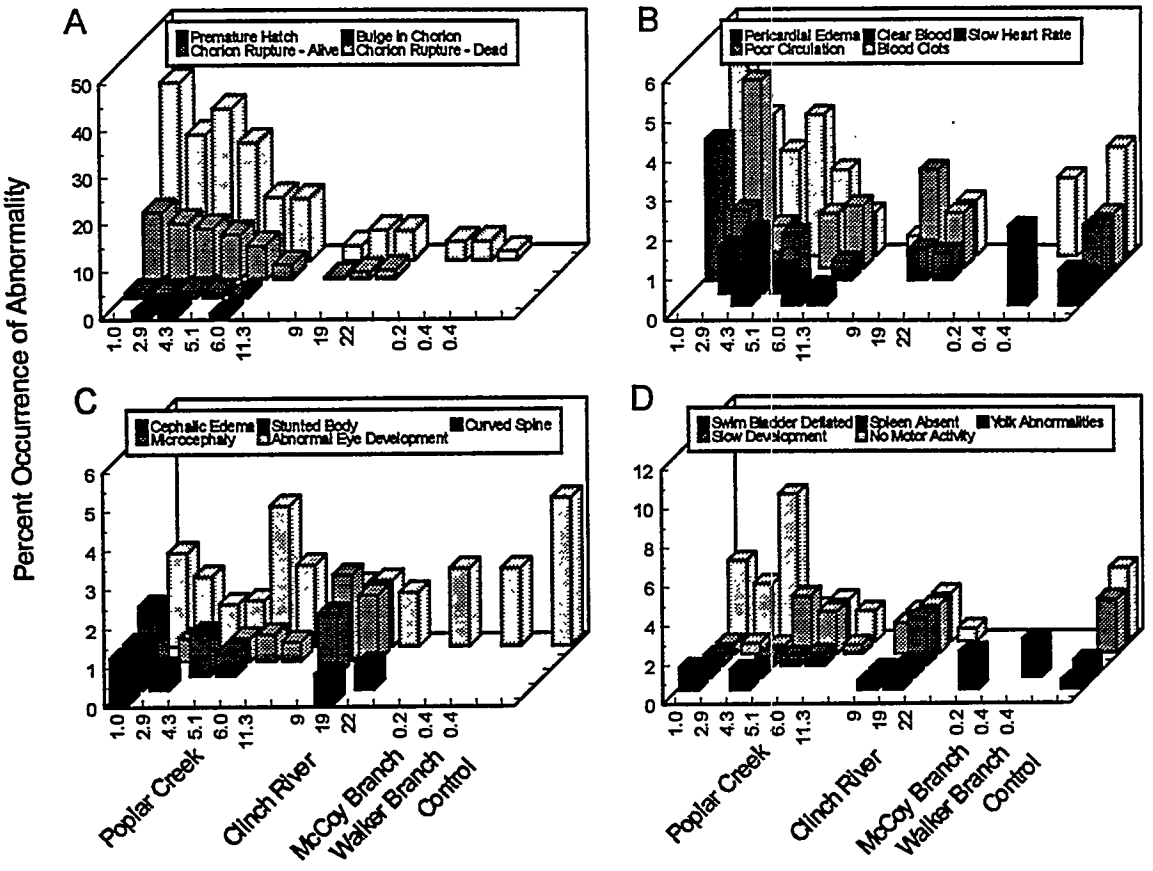


Fig. 3.21. Percent occurrence of developmental abnormalities during medaka embryo-larval tests. Abnormalities are grouped according to the following major classifications: (a) chorion-associated abnormalities; (b) circulation-associated abnormalities; (c) skeletal-muscular and head region abnormalities; and (d) other abnormalities.

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 A35). Although survival was significantly less at MBM 0.2 as compared with the Walker Branch reference site (WBM 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 A35 and Fig. 3.21).

Redbreast Sunfish Toxicity Tests. A set of fish embryo-larval tests was initiated in July 1994 for the purpose of comparing the responses of medaka and redbreast sunfish embryos and fry to water from select sites in Poplar Creek and the Clinch River. Medaka tests were conducted as outlined in Table A27; 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 as to the suitability of the millipore-filtered distilled water control for use with redbreast sunfish. Instead, all comparisons within tests were made to the appropriate reference site.

In this comparison test, survival was again very high (80 and 85%, respectively), and the incidences of developmental abnormalities relatively low (10 and 20%, respectively) 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 A31). Survival was only slightly less (70%), and the incidences 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 incidences 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 as the upstream reference site on Poplar Creek (Table A37; Fig. 3.20). 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 of 1994 on largemouth bass embryos and fry exposed during early development to water from several sites in the Clinch River and Poplar Creek and one site in the Tennessee River. Because of time-constraints caused by difficulties in collecting female largemouth bass that possessed mature, but not over-ripe, 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 A37 and summarized in Fig. 3.20. Survival was uniformly high in these tests; however, small but statistically significant decreases in survival as compared with controls were observed in two of five groups of embryos tested against water from CRM 9.0. There were also minor, but again statistically significant, decreases 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 A38).

Conclusions and Summary. The results of the medaka embryo-larval tests clearly demonstrate an intermittent toxicity to fish embryos by water from lower Poplar Creek. This toxicity does not appear to extend into the Clinch River. Redbreast sunfish embryos appear to respond similarly to 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.

3.3.7.4 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, using the SOS-Chromotest kit (Table A27). 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.

3.3.8 Summary of Water Characterization

Historical data indicated that the quality of water in the CR/PC OU was generally good and that contaminant concentrations in the Clinch River in the past generally met applicable water criteria.

Monitoring data from ambient sites throughout the OU were used to augment data collected during the remedial investigation. For the Clinch River, the present characterization attempted to answer specific questions regarding what were presumed to be worst-case conditions for contaminant flux to the river and the potential for mobilization of contaminants from sediment. Average ambient conditions in Poplar Creek were characterized over three seasons. Finally, the toxicity of surface water from a number of sites in the OU was evaluated using several test organisms and several measures of toxicity.

Ambient concentrations in the Clinch River, as evidenced by the ORREM data, revealed no contaminants whose mean concentrations exceeded AWQC or that were elevated over reference values.

No evidence of large-scale migration of contaminants from known source streams during extreme hydrologic events, leading to increased ambient concentrations in the Clinch River, was found. Maximum concentrations of contaminants in source streams were generally within the range of historically reported mean values.

No evidence was found of significant contaminant remobilization from sediment to surface water, although the study was limited in scope and design.

Both the ORREM data and the site characterization data indicated that mean concentrations of all analytes in Poplar Creek were below AWQC, and, with few exceptions, mean concentrations did not differ significantly from upstream reference concentrations. The site characterization data did reveal statistically significant elevated levels of total mercury and total methylmercury downstream of EFPC. Mean concentrations were highest near the mouth of Poplar Creek rather than immediately below EFPC. In addition, concentrations of dissolved nitrate/nitrite were significantly elevated downstream of Mitchell Branch. Gross measures of both alpha-emitting radionuclides and beta-emitting radionuclides were also significantly elevated downstream of Mitchell Branch in winter. As expected, concentrations of organic compounds were low throughout the system, with many compounds undetected anywhere (Table A39).

No site-related patterns of toxicity were observed in the majority of toxicity tests. The exception was the Japanese medaka embryo-larval series of tests, which clearly demonstrates intermittent toxicity to fish embryos of water from study sites in Poplar Creek. These results are corroborated by similar results using redbreast sunfish, a species native to the Clinch River/Poplar Creek area. Toxic effects apparently do not extend into the Clinch River downstream of Poplar Creek. While medaka fry were not sensitive to water from Poplar Creek, the susceptibility of embryos is thought to be due to disruption of chorionic integrity, resulting in premature hatching and embryo death.

3.4 SEDIMENT CHARACTERIZATION

3.4.1 Introduction

Many contaminants of concern in the Clinch River system have an affinity for, and rapidly become associated with, particulates. Uptake or 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 in off-site areas over time. Therefore, knowledge of the depositional patterns and contaminant levels within deposition zones is critical in the characterization of sediments in Poplar Creek and the Clinch River. This section summarizes current understanding of the sediment contaminant distribution in Poplar Creek and the Clinch River by presenting a review of historical studies and a synthesis of the data collected during the CRRI.

3.4.2 Historical Studies

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

3.4.2.1 Annual surveys of gamma activity in Clinch and Tennessee rivers bottom sediments

From 1951 to 1966, various researchers (Garner and Kochitzky 1956, Cottrell 1960, ORNL 1961–1967) measured total gamma radiation at the surface of the sediment using 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 taken annually at points along the 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. 3.22). The distribution of activity along a transect was generally proportional to the depth of the water (e.g., Fig. 3.24); most of the activity was found in the main channel.

In general, gamma count rates in the Clinch River gradually increased downstream from the mouth of WOC. The highest count rates found throughout the entire Clinch and Tennessee rivers system typically occurred between CRM 11.0 and 8.0, although annual variations exist (Fig. 3.23). 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 2.6.

An increase in activity in 1952 (Fig. 3.23) was attributed to the release of an unspecified short-lived radionuclide, possibly barium, which had largely decayed by 1953 (Garner and Kochitzky 1956). The large increase in activity during the period 1956–1960 was due primarily to the draining of White Oak Lake with 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 the 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.

3.4.2.2 Clinch River Study

The purpose and scope of the Clinch River Study are outlined in Sect. 3.3.1 The following briefly summarizes salient findings with respect to contaminated sediment.

In 1962, 113 cores were collected from 14 locations in the Clinch River and used to estimate the radionuclide inventory of the sediment. A total inventory of 200 Ci was estimated for the river as of July 1, 1962 (Table 3.3). The bulk of this inventory was ^{137}Cs (150 Ci, or 77% of total). The remaining inventory was composed of an estimated 18 Ci (8.7%) of ^{60}Co, 16 Ci (7.7%) ^{106}Ru, 10 Ci (5.1%) TRE elements, and 2.9 Ci (1.5%) ^{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 approximately 1.5% of the total activity released from ORNL to that time, including approximately 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).

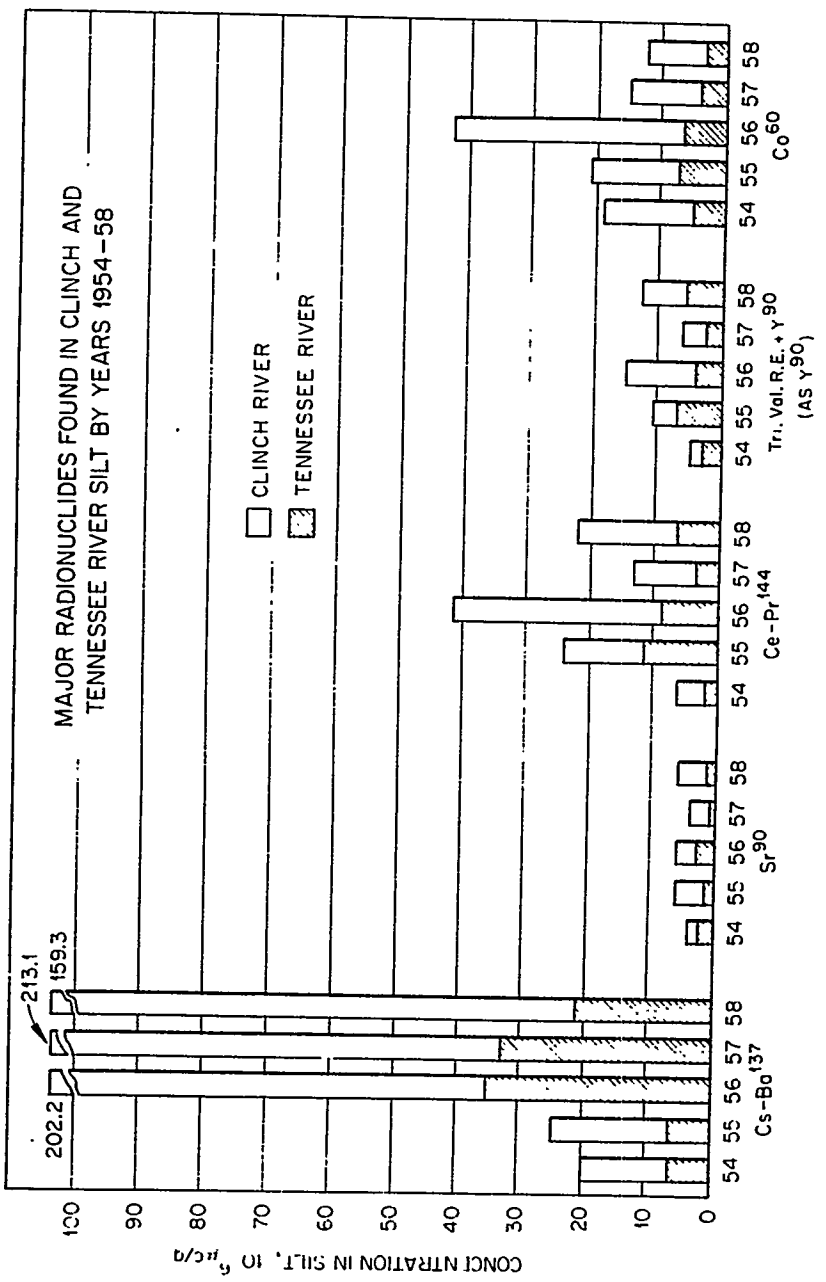


Fig. 3.22. Major radionuclides found in Clinch and Tennessee rivers silt by years, 1954 through 1958 (Cottrell 1960).

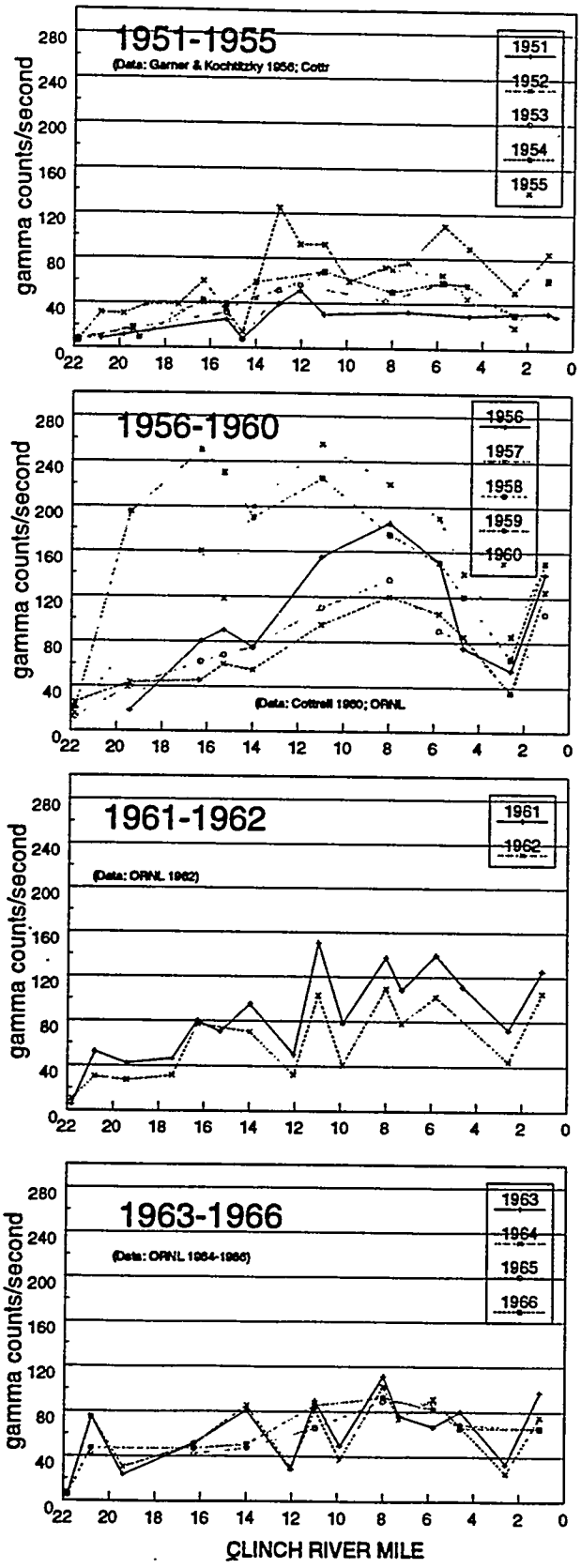


Fig. 3.23. Results of gamma surveys in the Clinch River, 1951 through 1966.

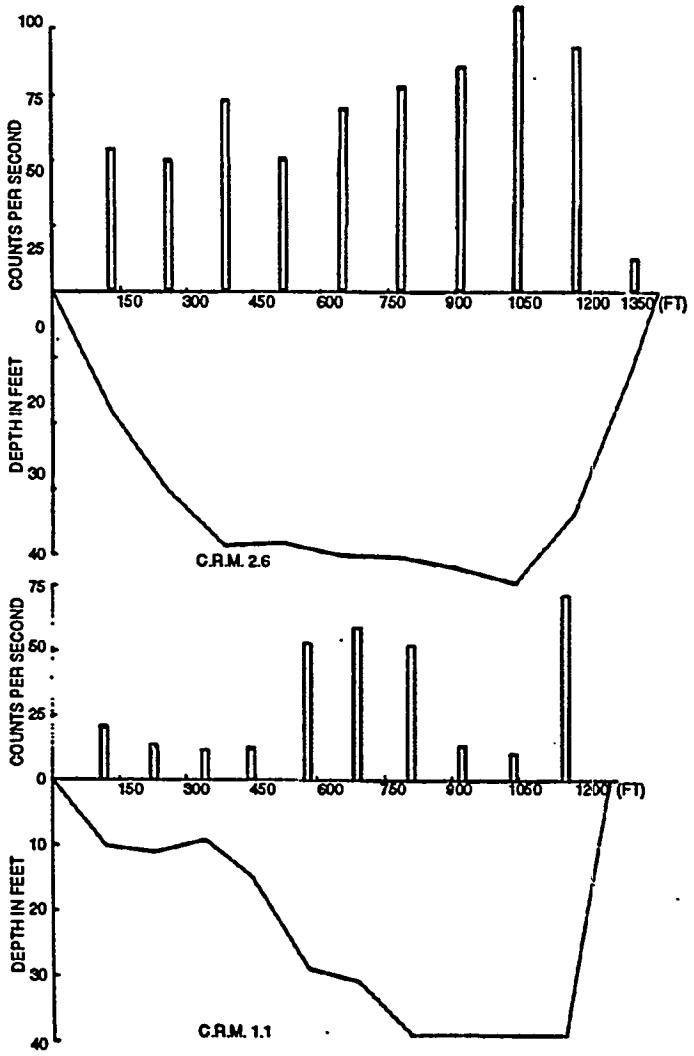


Fig. 3.24. Gamma activity vs. depth profile for selected locations in the Clinch River (Cottrell 1960).

Table 3.3. Estimated radionuclide inventory of the Clinch River below the Oak Ridge Reservation, as of July 1, 1962 (Carrigan 1969)

Location of Subreach		Total identified activity (Ci)		Volume of Radioactive Sediment (acre-feet)	
From CRM	To CRM	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

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. 3.25) and each exhibited the same general pattern (e.g., Fig. 3.26) identified in the gamma surveys (Sect. 3.4.1). Although areas of greater and lesser deposition are evident, there was a general tendency for the thickness of radioactive sediment to increase linearly from CRM 21 to the mouth of the Emory River (Fig. 3.27) (Carrigan 1969). Laterally, radiation levels in sloughs of the Clinch River were not as high as those in sediments of the main channel, and radionuclide levels in areas exposed during winter drawdown were less 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 3.28). The ^{137}Cs profile in a core was found to correlate with the release history from ORNL (Fig. 3.29).

Pickering (1969) found that, in general, radioactive bottom sediment in the Clinch River could be classed as clayey silt, composed of approximately 35% mica and other clays and 65% quartz. The potential for desorption of radionuclides was investigated using contaminated sediments from WOC

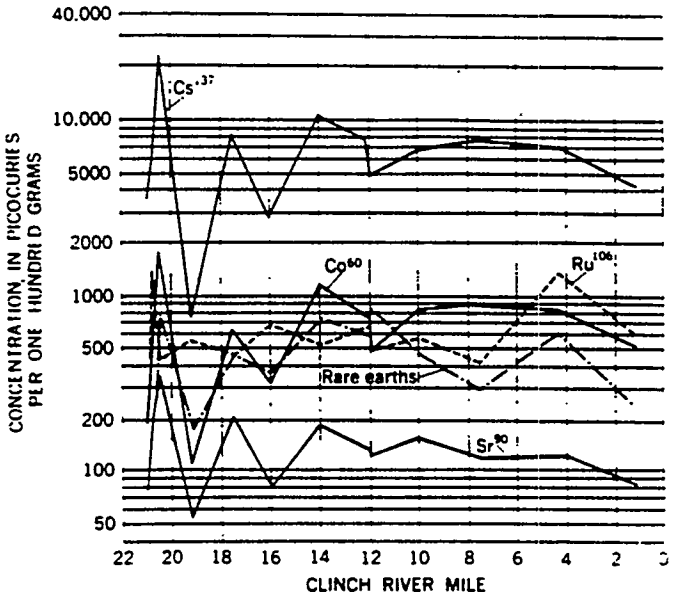


Fig. 3.25. Longitudinal distribution of radionuclides in the Clinch River downstream of the Oak Ridge Reservation (from Carrigan 1969).

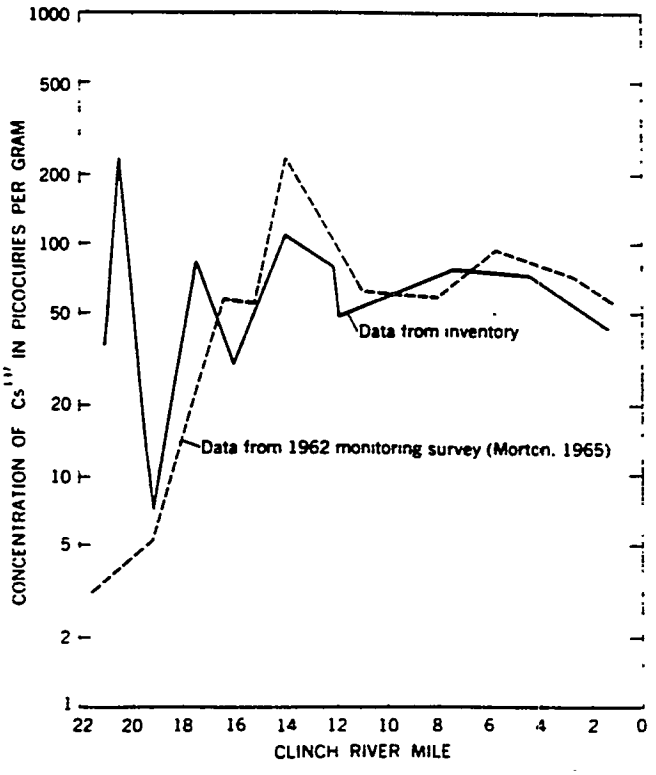


Fig. 3.26. Comparison of the longitudinal distribution of ^{137}Cs with gross gamma activity in the Clinch River downstream of the Oak Rdige Reservation (from Carrigan 1969).

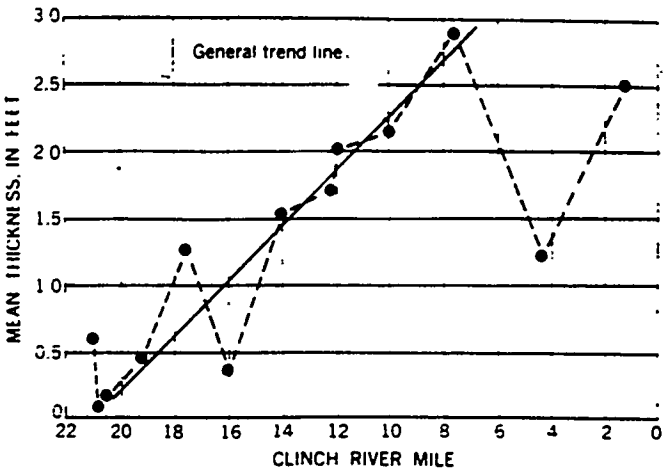


Fig. 3.27. The thickness of radioactive sediment in the Clinch River was found to increase between the mouth of White Oak Creek and CRM 8, after which it declined sharply (from Carrigan 1969).

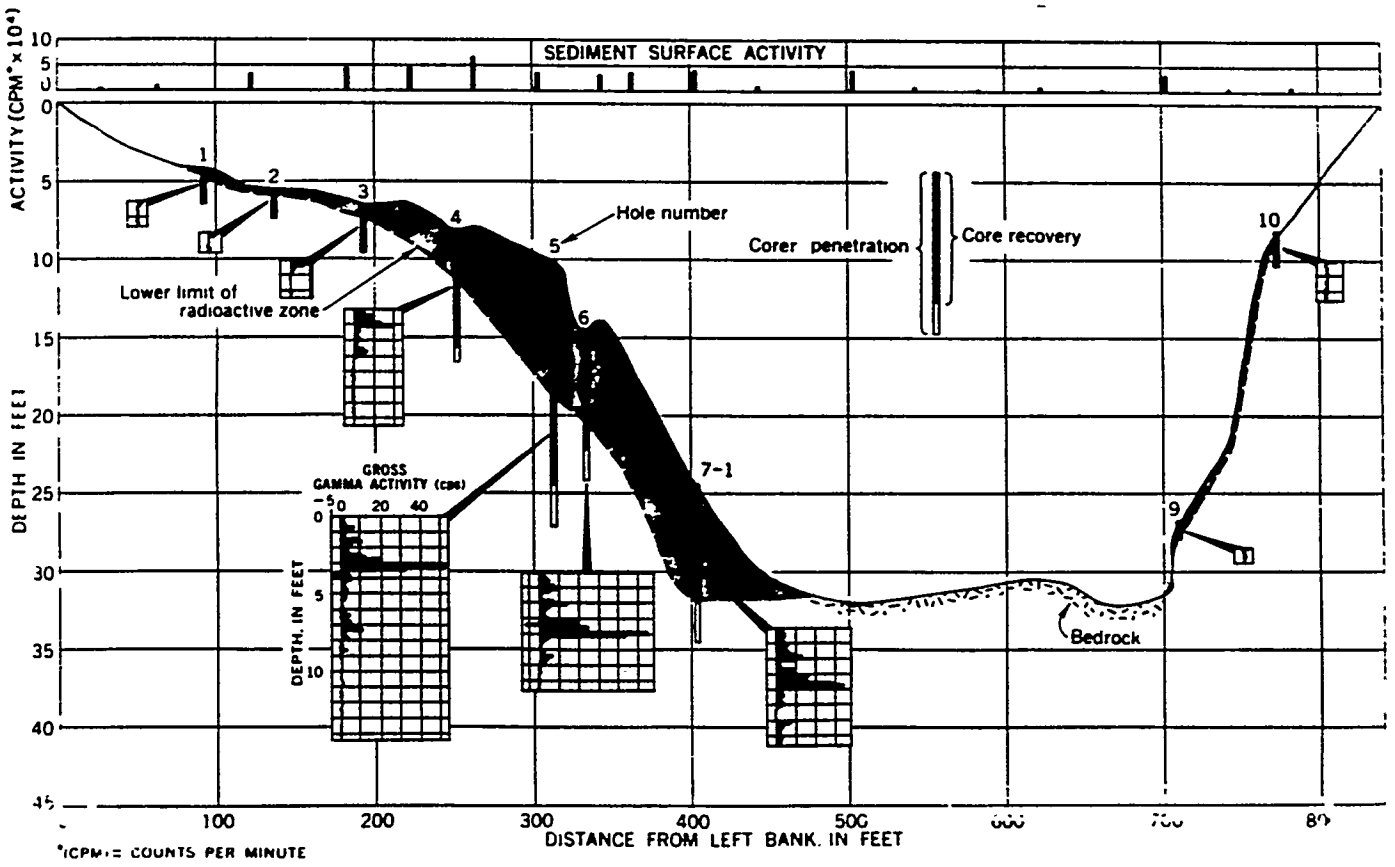


Fig. 3.28. Cross section of the Clinch River at Mile 7.5, showing lateral distribution of sediment and vertical distribution of gross gamma activity (from Carrigan and Pickering 1967).

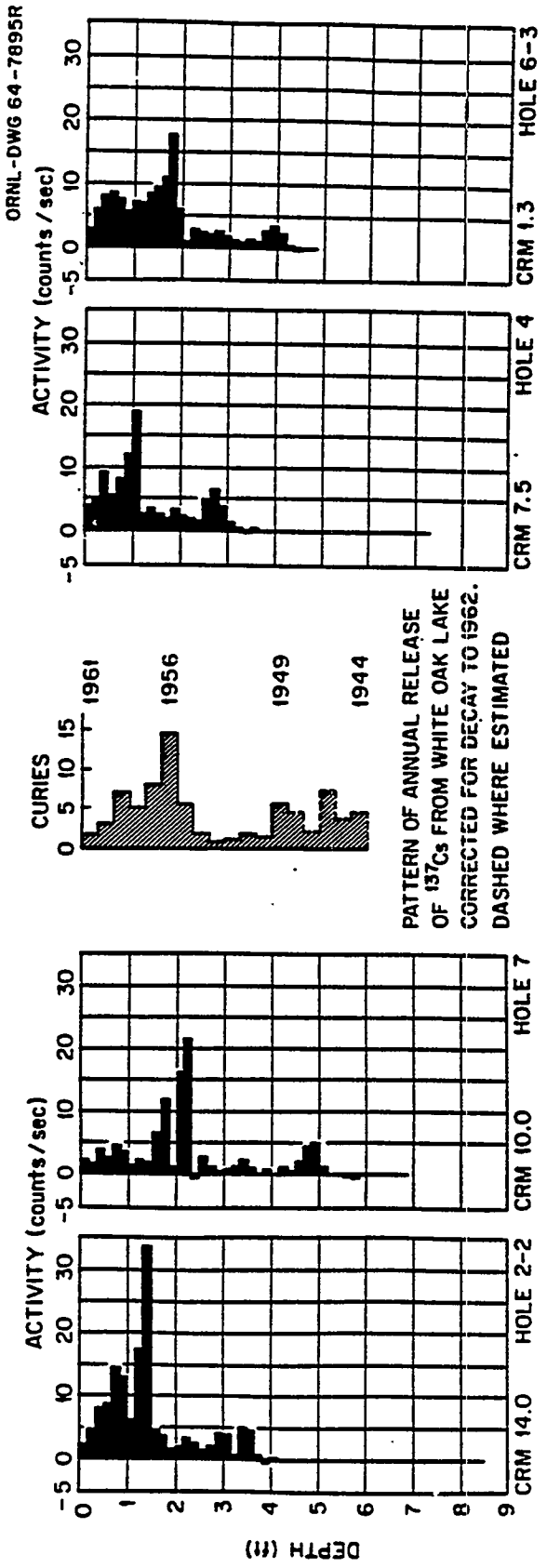


Fig. 3.29. Comparison of patterns of variation with depth of gross gamma radioactivity in four bottom sediment cores to variations in annual releases of ^{137}Cs to Clinch River (Carrigan and Pickering 1967).

(Morton et al. 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, while strongly alkaline solutions were more effective in desorbing ^{106}Ru. Regardless of treatment, less than 6% of the ^{137}Cs was desorbed, indicating a strong affinity for particles. This affinity is greatest for certain clay minerals.

3.4.2.3 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 of the altered flow regime, resulting from operations at Melton Hill Dam, 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 purposely 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 now 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.-diameter 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 analysis for ^{90}Sr and transuranic radionuclides were made of subsamples taken from the top, middle, and bottom of about half of the cores (Oakes et al. 1982).

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 3.4.

Table 3.4. Maximum concentrations of radionuclides reported in the 1977 Clinch River Sediment Survey (data from Oakes et al. 1982)

Radionuclide	Maximum Concentration	Location	Depth in Core
^{137}Cs	606 pCi/g	CRM 14.25	28 cm
^{60}Co	12.2 pCi/g	CRM 14.25	28 cm
^{90}Sr	11.7 pCi/g	CRM 14.25	31 cm
$^{238,239}Pu$	6.5 pCi/g	CRM 14.0	23 cm

3.4.2.4 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 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. 3.30. 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 in the CRM 1.0 core (7.8 mg/kg) 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).

3.4.2.5 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 1986).

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 (CRM 24.0) and Norris reservoirs (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 B1. Values for selected radionuclides are presented in Table B2.

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 based on sediment particle size fraction: (1) the fine sands and smaller fraction (containing particles $< 500\mu$ m) and (2) the silt and clay fraction only (containing particles $<62\mu$m). 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. 3.31.

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.

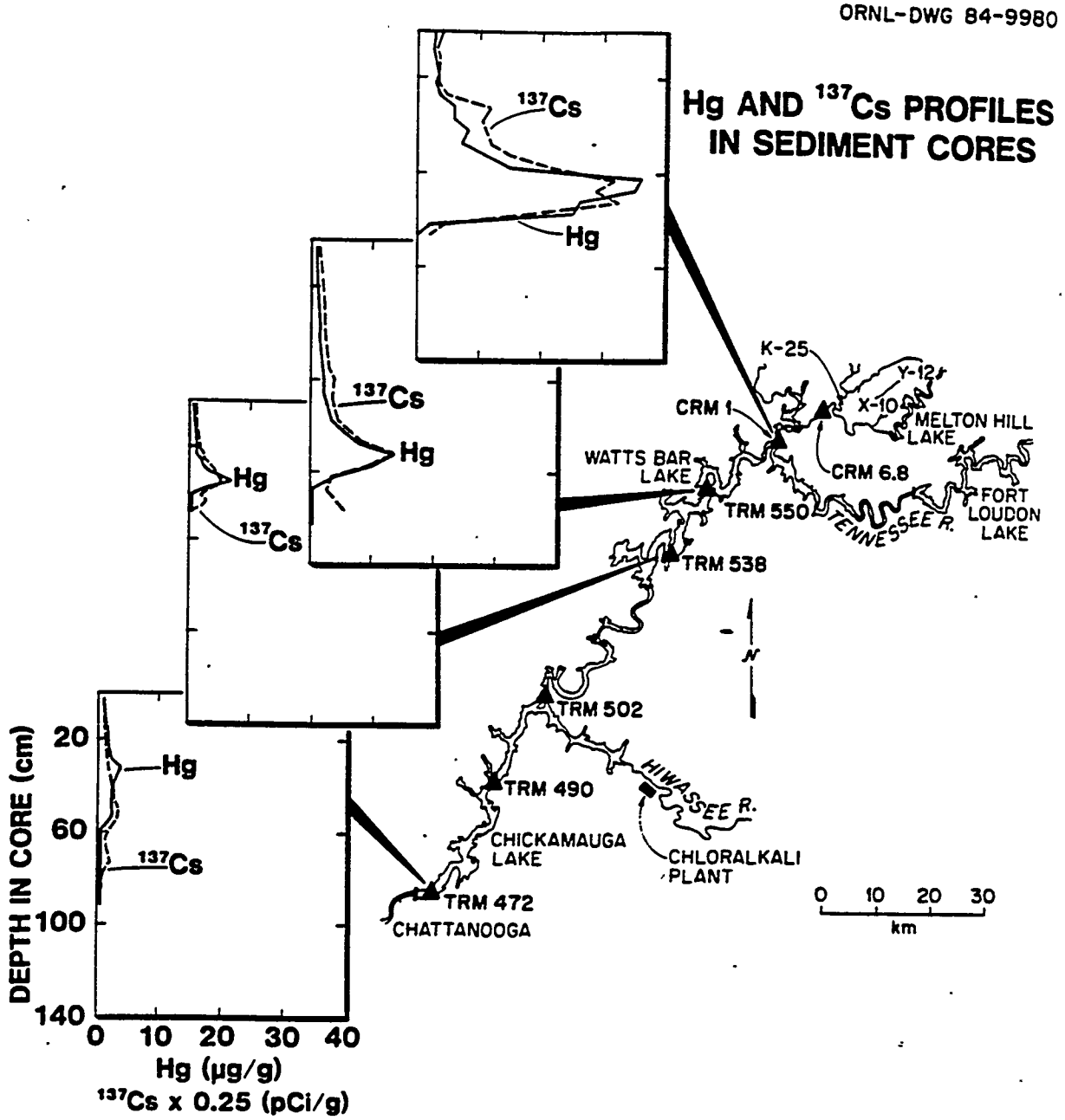


Fig. 3.30. The vertical distributions of mercury and ^{137}Cs in selected sediment cores obtained from Watts Bar and Chickamauga reservoirs (Turner et al. 1984).

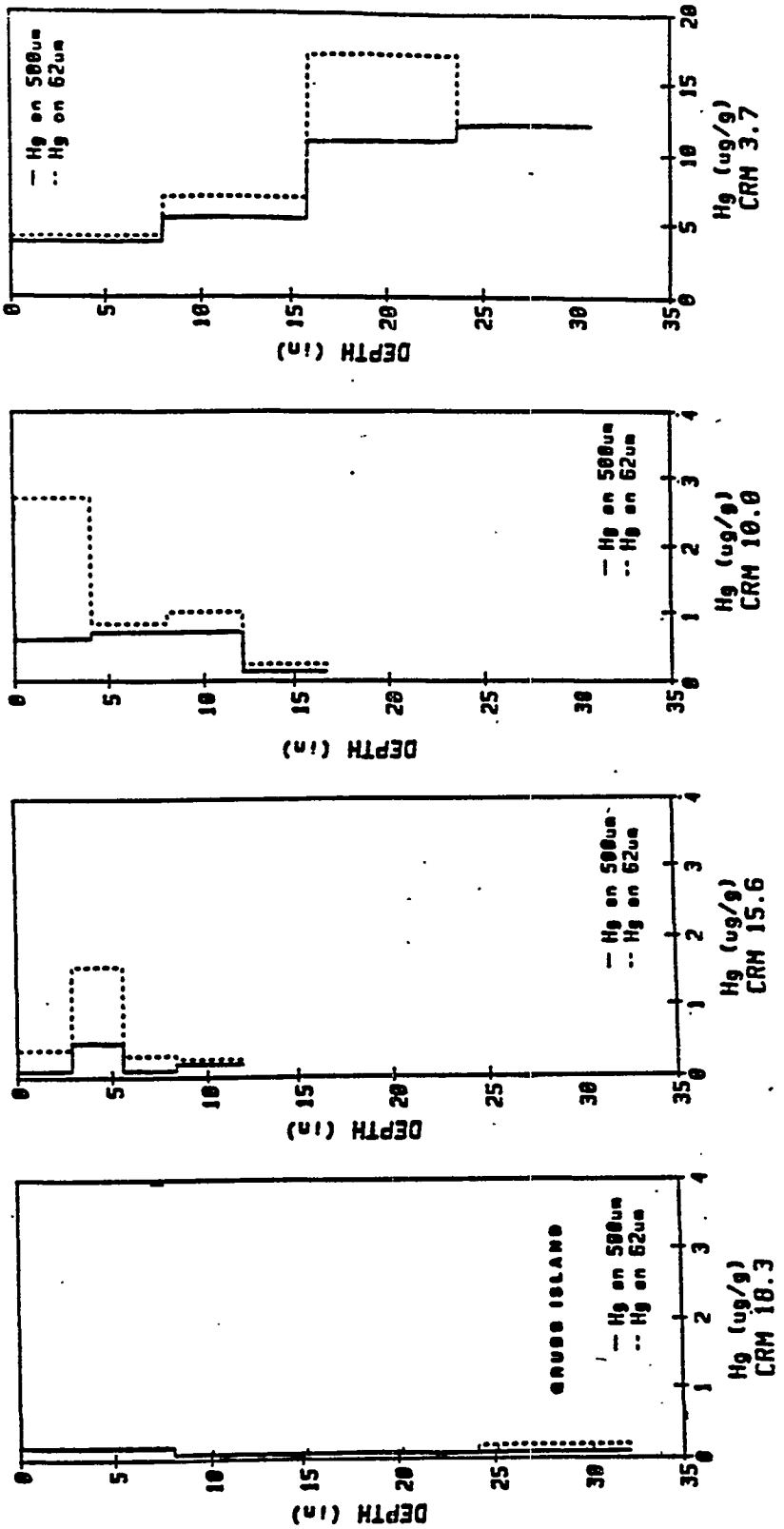


Fig. 3.31. Mercury concentrations vs. depth and particle size for Clinch River core sampling (TVA 1985a).

3.4.2.6 ORGDP sediment survey

Ashwood et al. (1986) collected approximately 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 anlayzed 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 from which pollutants originating at K-25 had historically entered or were currently entering the surface water. Samples were collected in January and February 1985. The study focused on contaminants identified by Hoffman et al. (1984), based on data from the TVA Instream Contaminants Study (TVA 1986), 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, 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 5.5 pCi/g; K-25 mean = 30 pCi/g) relative to K-25 mean values (i.e., 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).

3.4.2.7 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 which were composited for analysis. Volatile organic analyses 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 1991). Results of the sediment analyses are presented in Table B3.

No radionuclides other than ^{137}Cs were detected at any location during the study. Mercury levels downstream of Poplar Creek were elevated above maximum reference concentrations (Table B3). 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.

3.4.2.8 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. 3.34). Cores were sectioned vertically into 6-cm increments (upper 18 cm of core only) and 4-cm increments (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 2 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. 1992). 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 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.

The data from CRRI Phase 1 are combined with the CRRI Phase 2 data to aid in characterizing the nature and extent of contamination in the CR/PC OU.

3.4.2.9 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. 3.32 and 3.33. In general, mean near-shore surface sediment concentrations are greatest in reach 4, 7 to 22 miles downstream of the mouth of WOC. 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 due to 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 due to reverse flows in the system. Reference concentrations were at background, except for the expected increase in ^{60}Co downstream of Braden Branch.

Data from the near-shore surface sediment characterization task are used in Sect. 3.4.3 of this report to assist in the characterization of contaminants in surface sediment and in the ecological (Chap. 6) and human health (Chap. 5) baseline risk assessments included in this report.

3.4.2.10 Summary of historical studies

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.

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

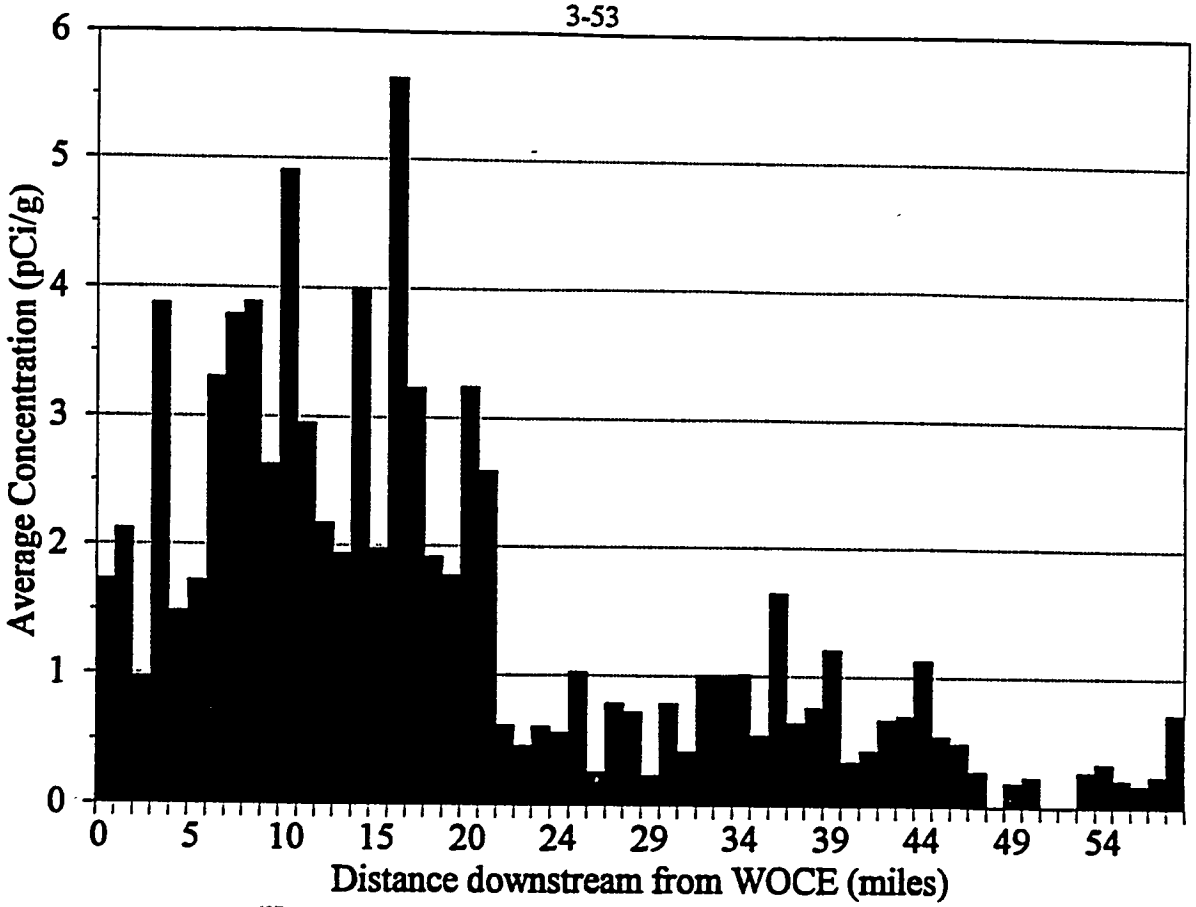


Fig. 3.32. Mean ^{137}Cs concentrations relative to distance downstream from White Oak Creek Embayment.

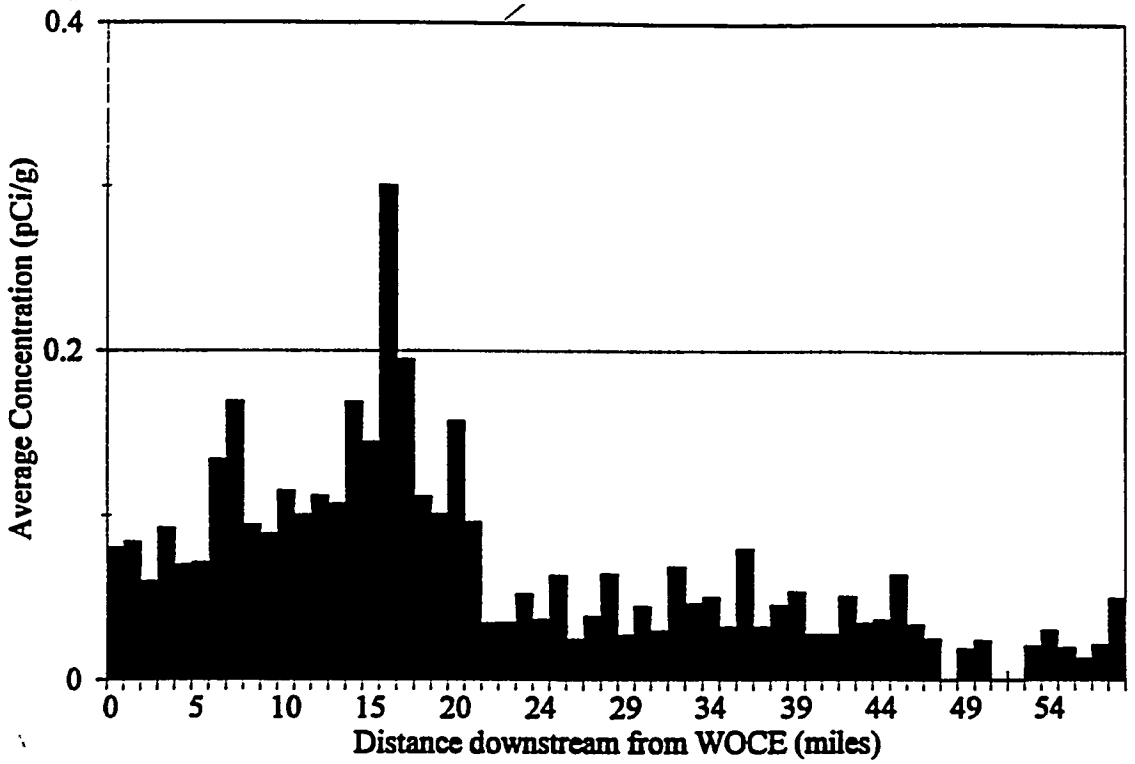


Fig. 3.33. Mean ^{60}Co concentrations relative to distance downstream from White Oak Creek Embayment.

characterized, but concentrations would be expected to be highest in areas of sediment deposition, as with the contaminants above. The areas of greatest potential for contaminant accumulation 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. Although limited in scope, historical data on organic pollutants in sediment indicate low levels of a relatively few compounds, most of which are ubiquitous in sediments near or downstream of urban or industrial areas (phthalate esters and PAHs).

3.4.3 Clinch River Remedial Investigation—Phase 2 Characterization of Sediments

Phase 2 of the CRRI included several sediment characterization tasks: spatial characterization of contaminantion in the Clinch River and Poplar Creek, characterization of sediment toxicity to benthic organisms, and sediment and contaminant transport modeling. Characterization focused on areas of sediment deposition, primarily downstream of Melton Hill Dam. These areas were indicated by the release histories and historical data as being the areas of greatest potential concern.

3.4.3.1 Characterization of contaminants in sediment and pore water

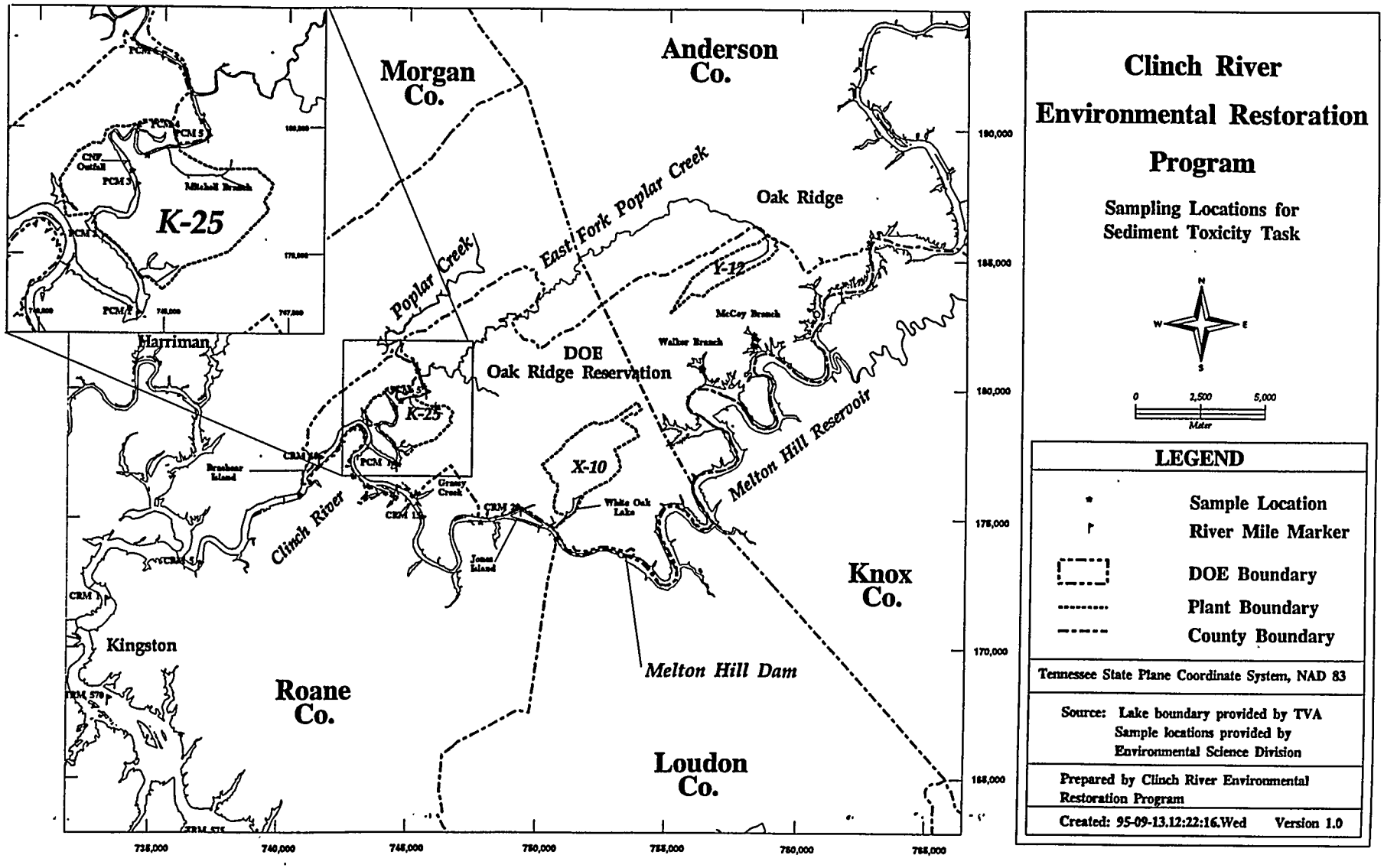
Data were collected regarding contaminant concentrations in surface sediment, in sediment cores, and in sediment pore water. These data are used in Chaps. 5 and 6 to characterize risk to human and ecological receptors. The following discussion attempts to identify meaningful spatial patterns of contamination in these media and interpret these patterns both in relation to known or suspected contaminant sources and to the dynamics of the river system.

Sample locations are illustrated in Fig. 3.34. Sample collection methods are described in Table B4. Sample sizes for each reach and subreach are tabulated in Table B5. Data for surface sediment, core samples, and pore water are summarized in Tables B6–B11 (inorganics), B12–B15 (organics), and B16–B21 (radionuclides).

Data regarding 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. 3.35.

To identify trends in contamination, mean contaminant concentrations in a study reach or subreach are compared to those from the appropriate reference reach and upstream study reaches. Discussion focuses on contaminants that are elevated above reference values in at least two of the three sediment media sampled. For this analyses, "elevated" means the concentrations that are statistically elevated (at the 0.05 level) as measured by one-way ANOVA and subsequent pairwise comparisons. In addition, the concentrations that are elevated by a factor of two or more as compared to 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


Clinch River
Environmental Restoration
Program
Sampling Locations for
Sediment Toxicity Task
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W
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LEGEND
Sample 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 Clinch River Environmental
Restoration Program
Created: 95-09-13.12:22:16.Wed Version 1.0
Morgan Co.
Anderson Co.
Oak Ridge
DOE
Oak Ridge Reservation
K-25
K-25
K-10
K-12
Poplar Creek
East Fork Poplar Creek
Clinch River
Melton Hill Reservoir
Melton Hill Dam
Harriman
Kingston
Roane Co.
Knox Co.
Loudon Co.
Bradner Island
Grassy Creek
Jones Island
White Oak Lake
McCoy Branch
Walker Branch
Mitchell Branch
CMP Outfall
PCM 1
PCM 2
PCM 3
PCM 4
PCM 5
CRM 1
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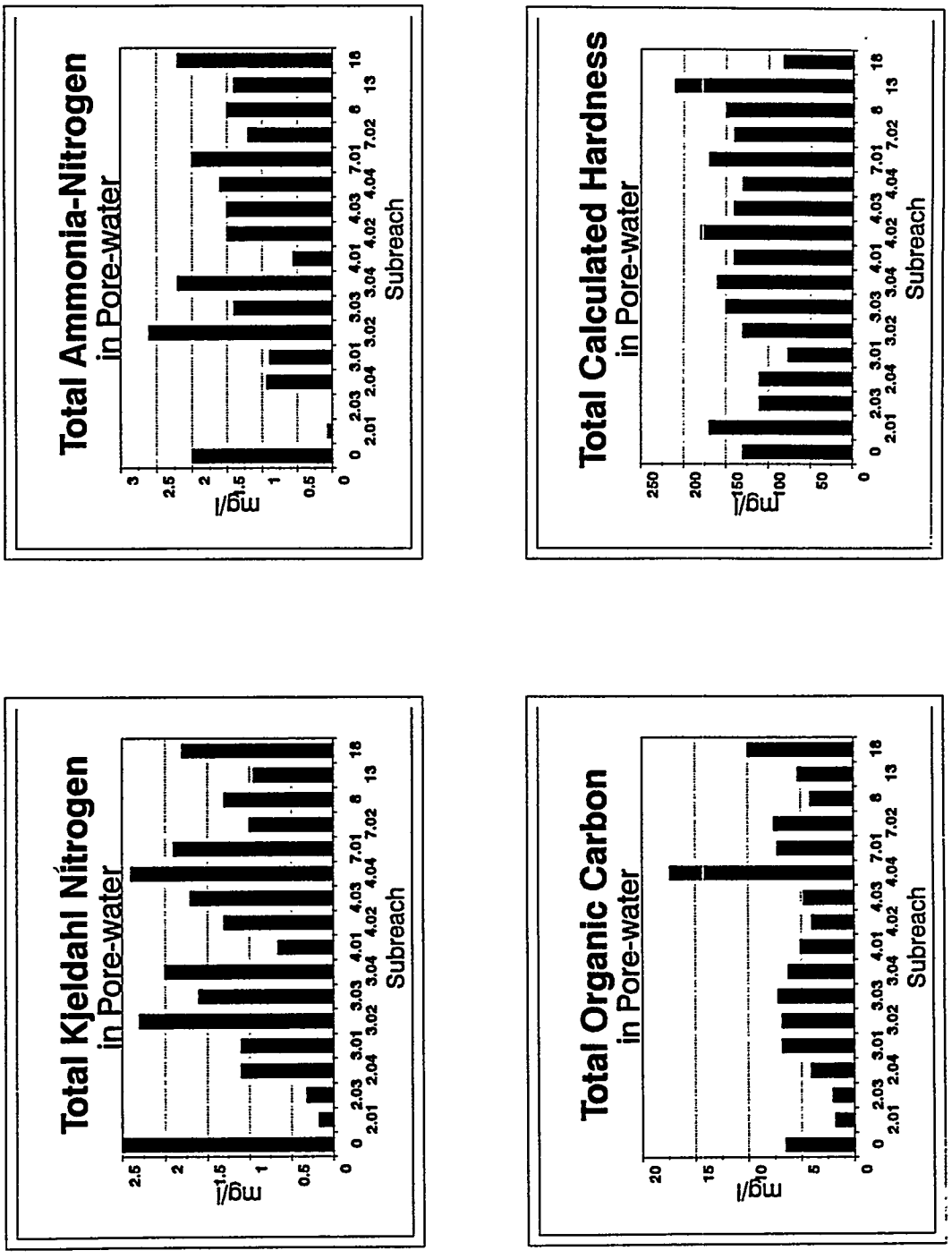
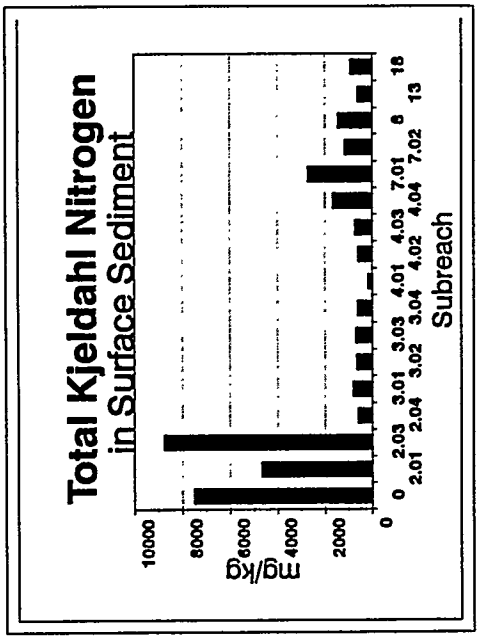
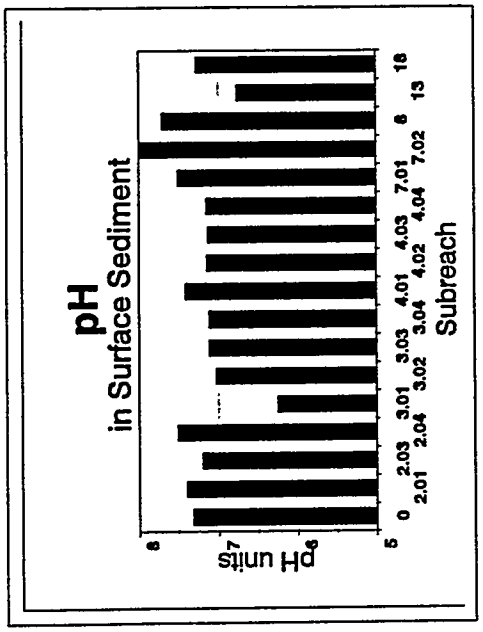


Fig. 3.35. Mean concentrations of selected noncontaminant parameters relevant to ecological health of benthic communities.





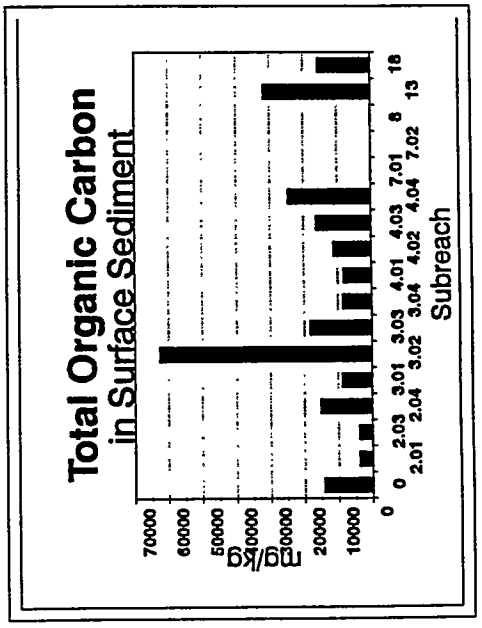


Fig. 3.35 (continued)

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. 3.36–3.38 (inorganics), 3.39–3.40 (organics) and 3.41–3.43 (radionuclides).

Poplar Creek.

Subreach 3.01—Below EFPC. Interpretation of data in this subreach is limited by sample size (n=2), but mean concentrations of mercury, cadmium, chromium, and copper are elevated above reference values (reach 13) in at least two sediment media downstream of EFPC (Figs. 3.36–3.38). 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 above. However, none of these analytes exhibits its peak core or surface sediment concentrations in subreach 3.01, probably due to 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. 3.38).

Subreach 3.02—Below Mitchell Branch. Mean concentrations of mercury, cadmium, copper, and chromium remain elevated over reference values and are generally increased over values in subreach 3.01, probably partly a result of increased sediment deposition (Figs. 3.36–3.38). However, concentrations of chromium and copper in surface sediment peak here, are significantly elevated over subreach 3.01, and likely indicate a significant source of these analytes from Mitchell Branch. Analytes that are significantly elevated in two or more sediment media for the first time are nickel, silver, several isotopes of uranium, ^{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 above, except benzo(b)fluoranthene and silver, exhibits peak core and surface sediment values in this subreach; silver exhibits mean surface sediment concentrations here. The sediment of Mitchell Branch has previously been identified as containing elevated concentrations of Ni (Fig. 3.44) Cr, Cu, Ag, Zn, U isotopes (Fig. 3.45), and ^{99}Tc (Ashwood et al. 1986).

Subreach 3.03—Below CNF. Mean concentrations of each analyte above 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. 3.37). Concentrations of nickel, copper, and benzo-(b)fluoranthene remain elevated in core samples (Fig. 3.37) but drop significantly in surface sediments (Fig. 3.38). Levels of Arochlor-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, PCB-1254, U, and ^{99}Tc remain elevated in both surface and core sediment samples relative 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. 3.36–3.37) and are significantly elevated in both media for the first time. Increased core values are attributed to the historical disposal of fly ash in Poplar Creek (Sect. 3.1). Surface sediment concentrations of B and V, while elevated relative to subreach 3.03, are also relatively high in the reference reach. Cesium-137 and ^{60}Co are also elevated in core and surface sediment in subreach 3.04, most likely the result of backflow from the Clinch River or the discharge of cooling water drawn from the Clinch River and released to Poplar Creek from the K-1770 steam plant.

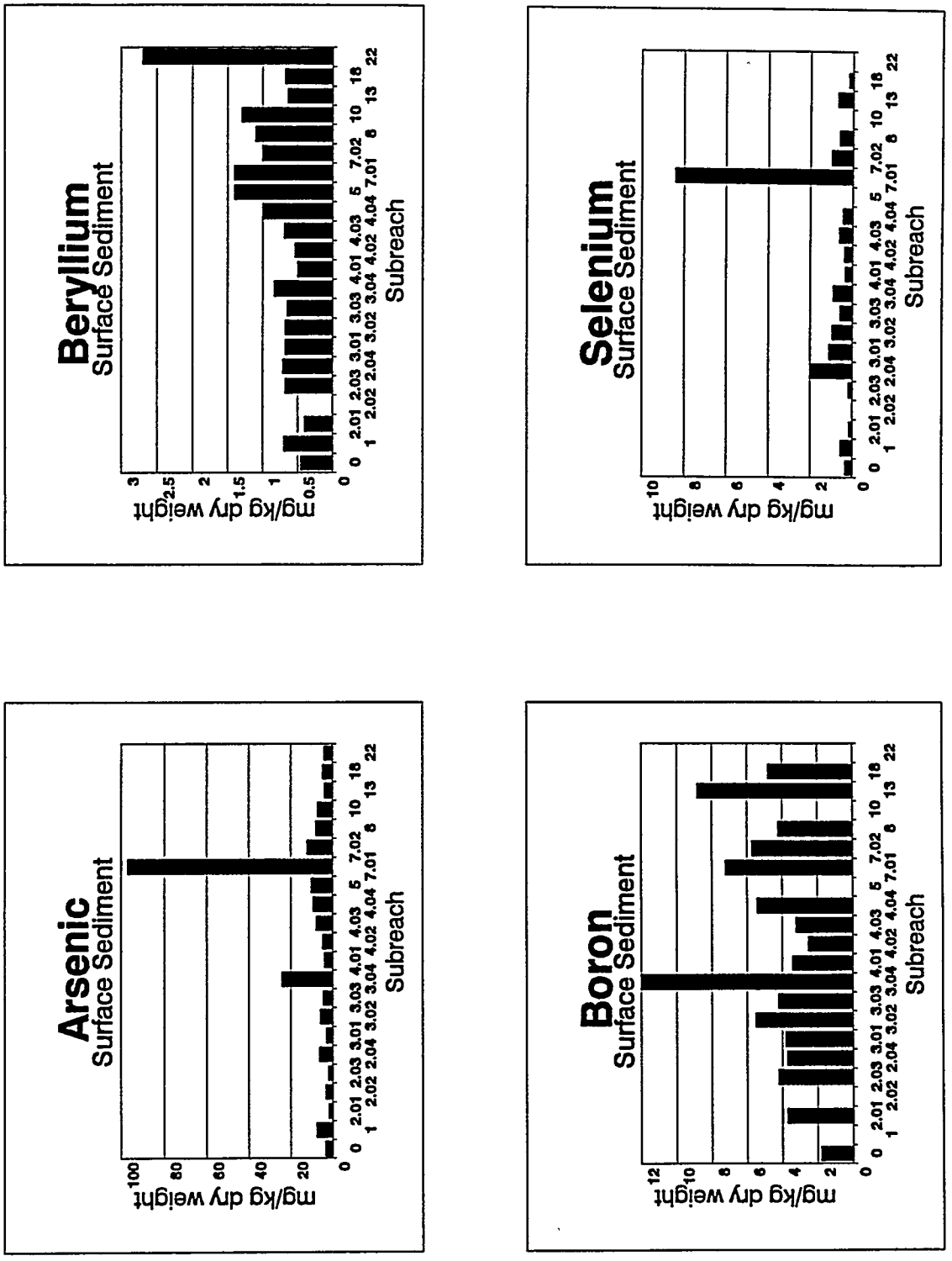
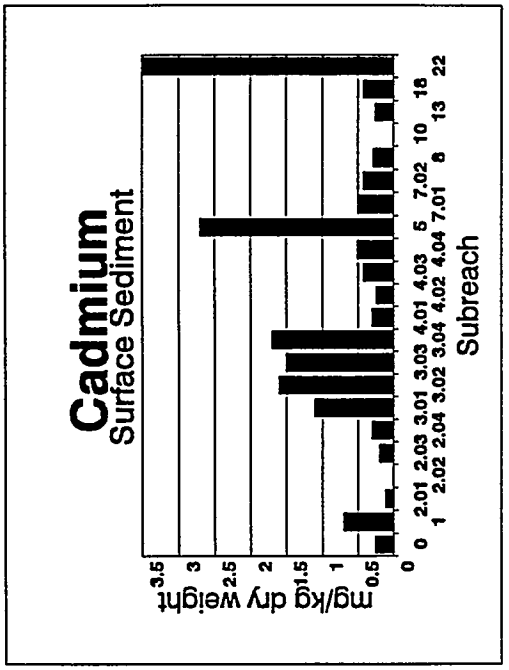
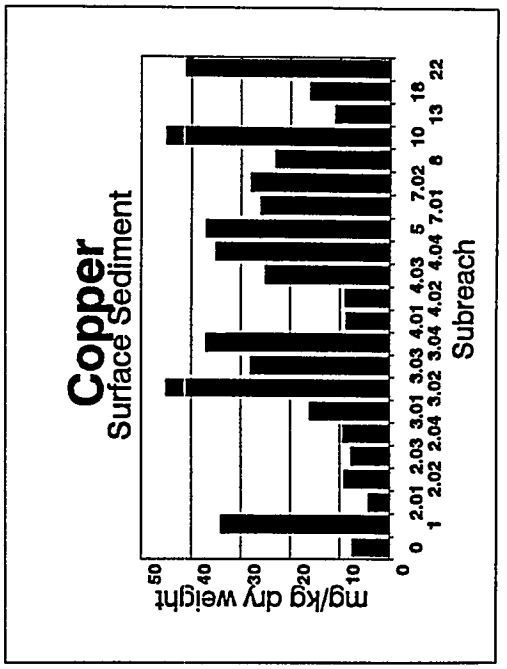
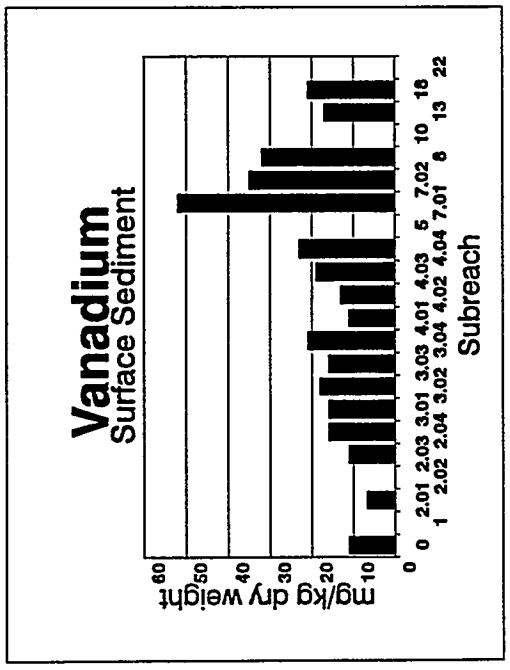
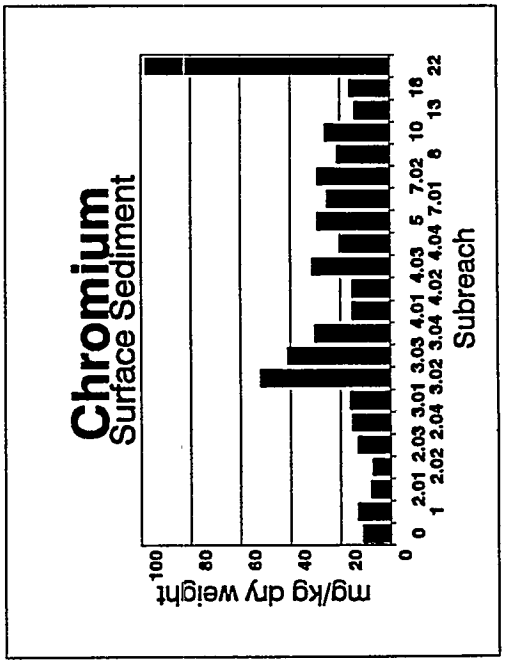


Fig. 3.36. Surface sediment mean concentrations for subreaches of inorganic contaminants.









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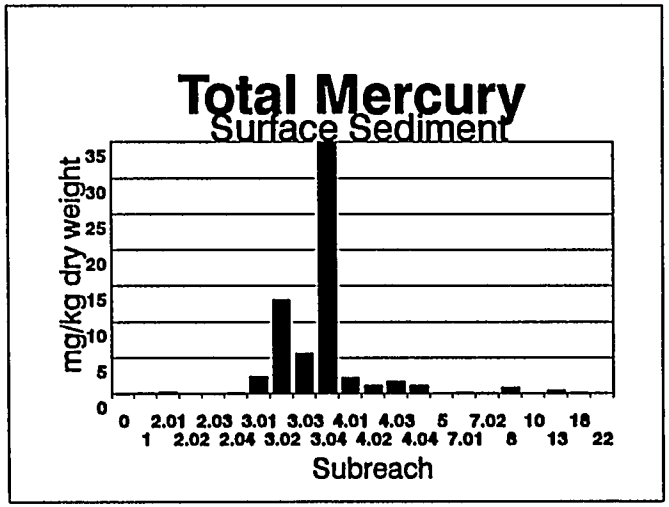

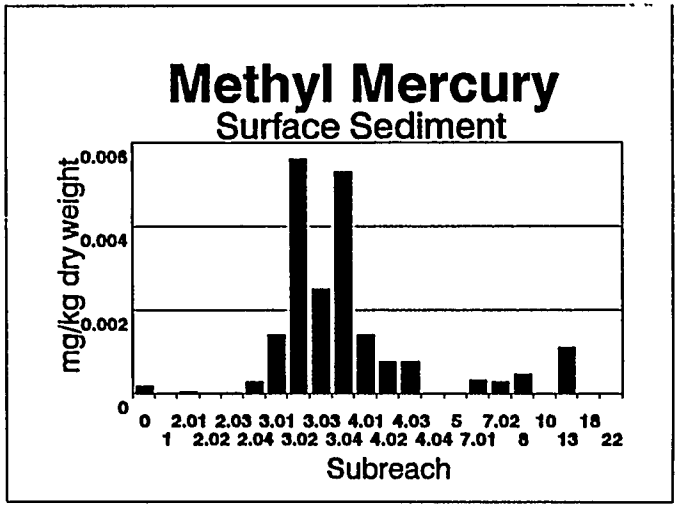

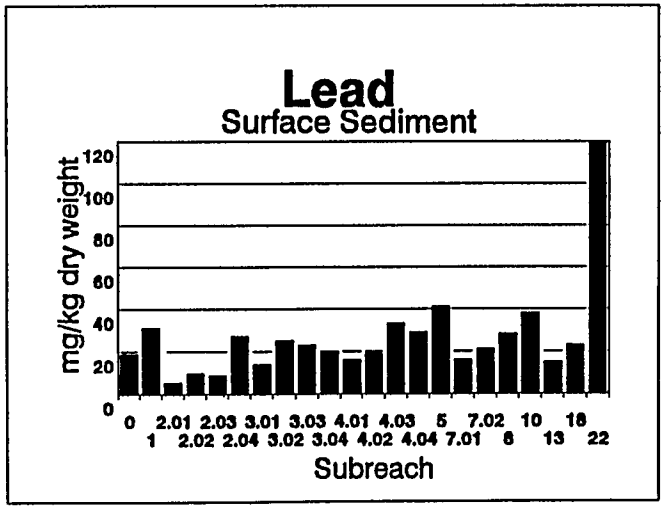

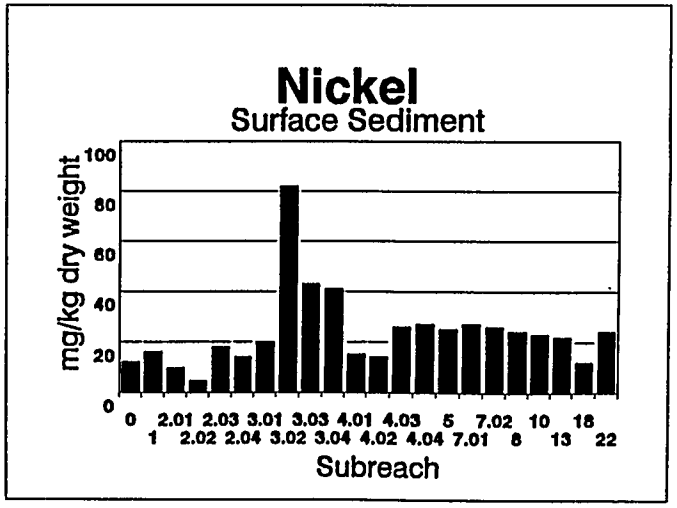

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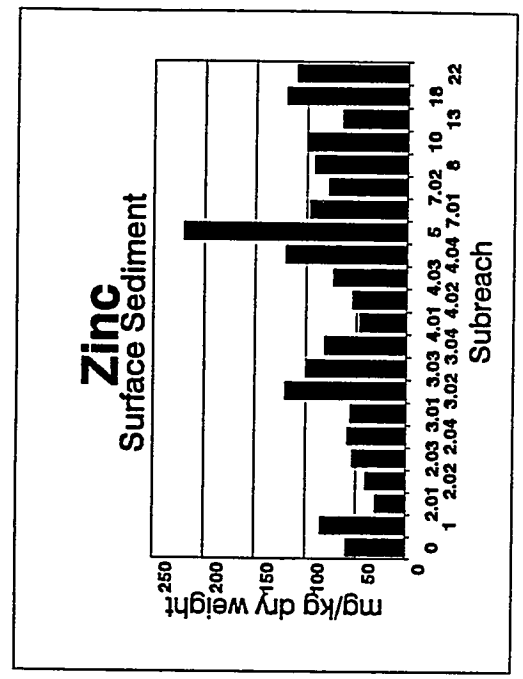

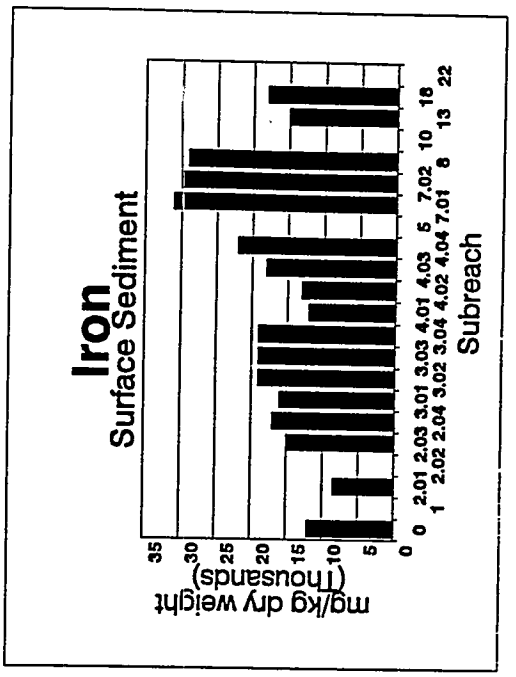

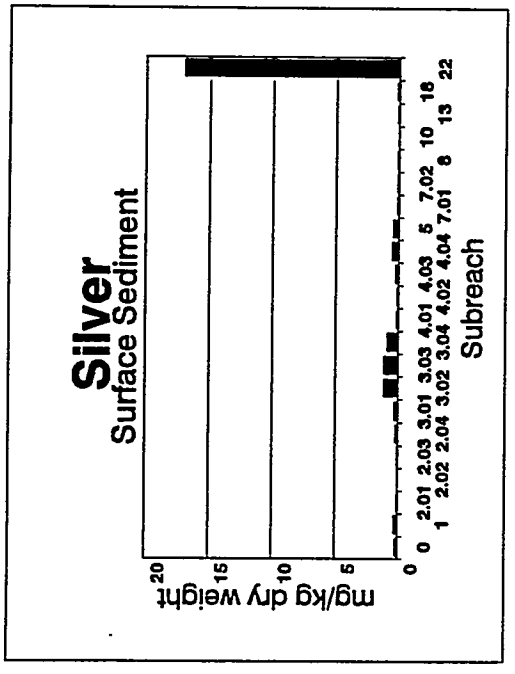

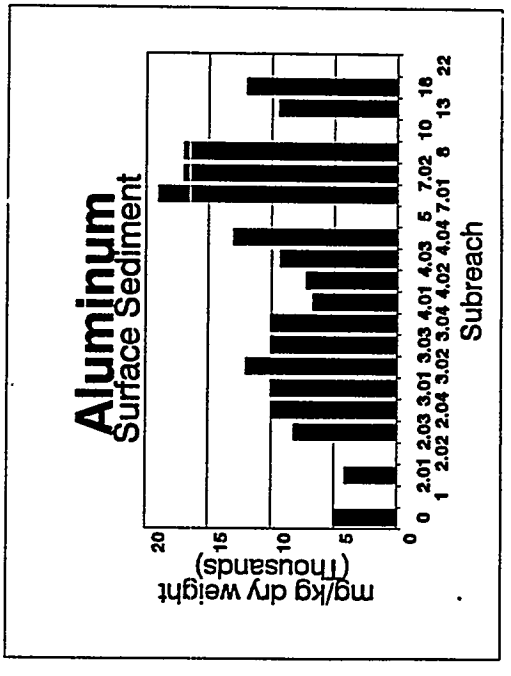

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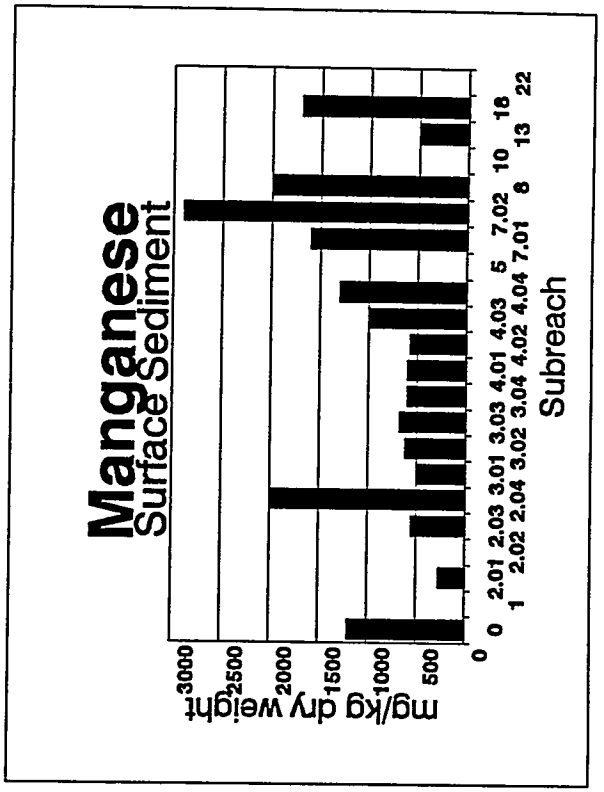

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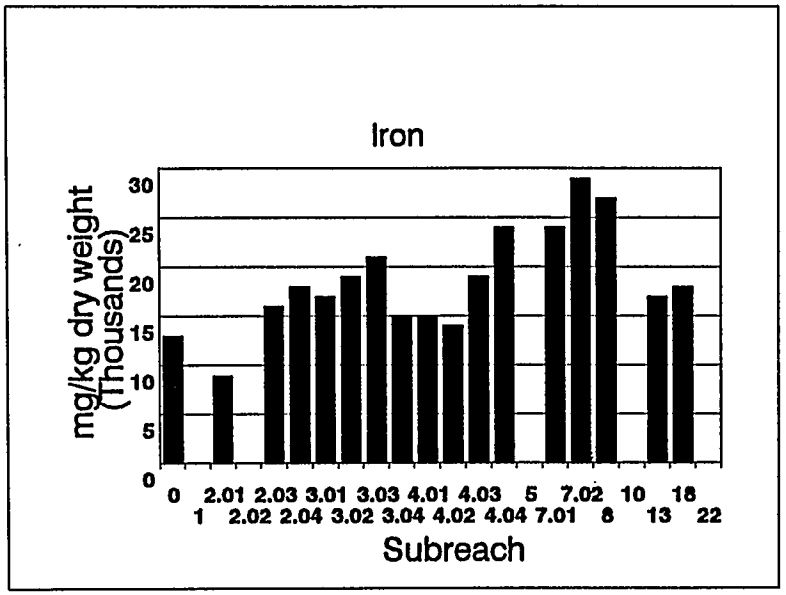
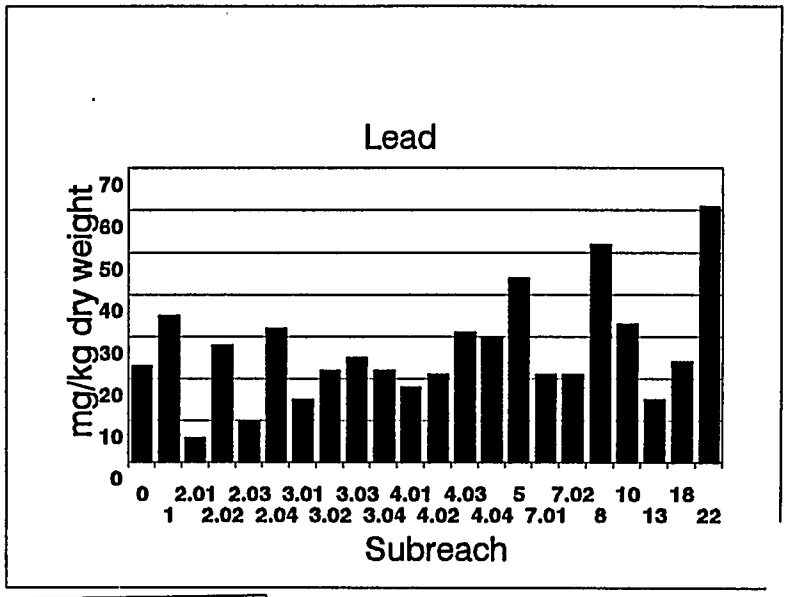




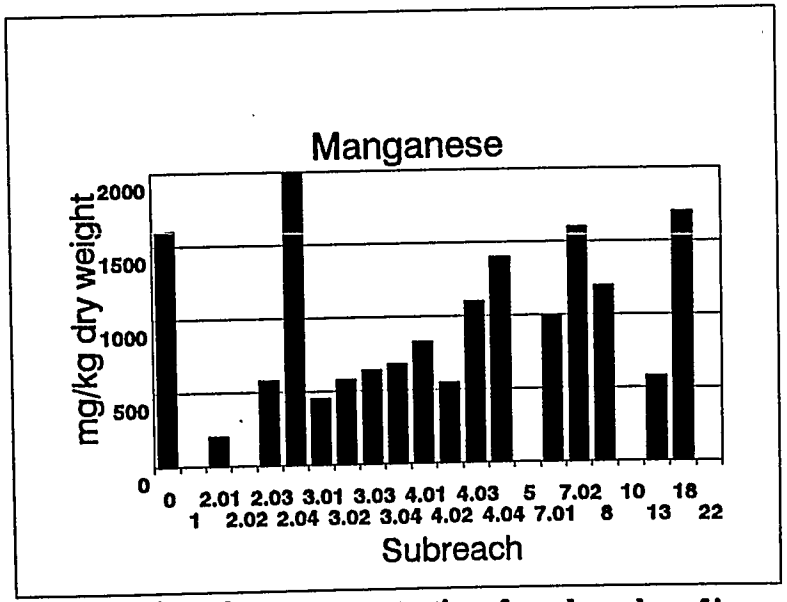
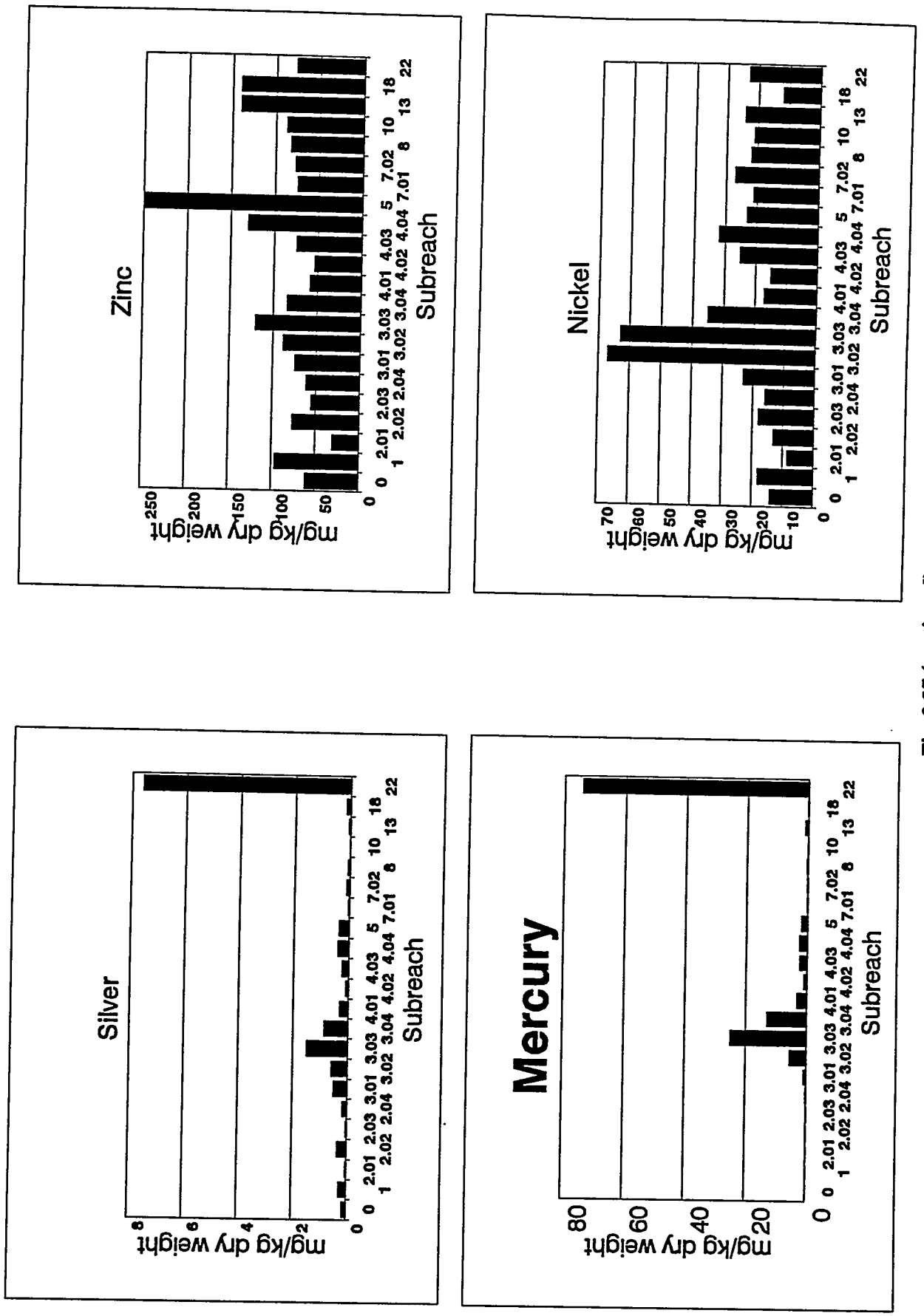


Fig. 3.37. Core-weighted mean concentrations for subreaches of inorganic contaminants.



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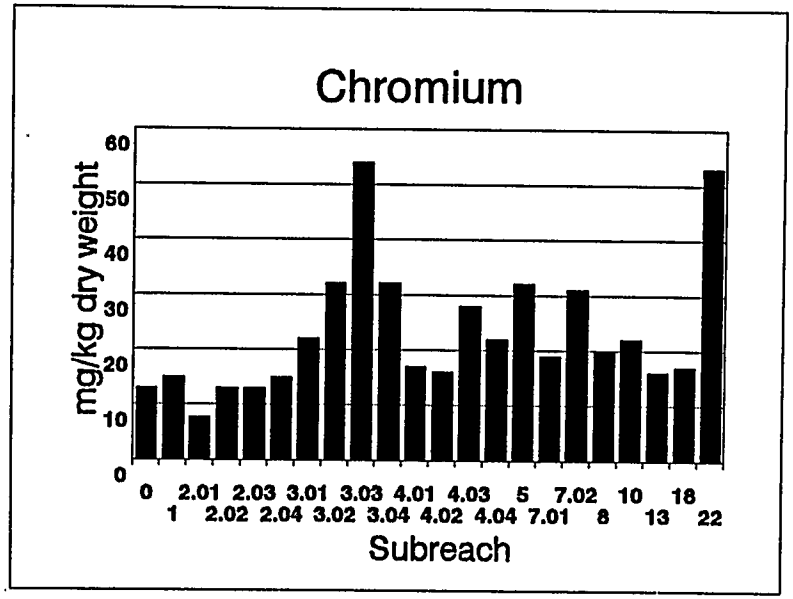

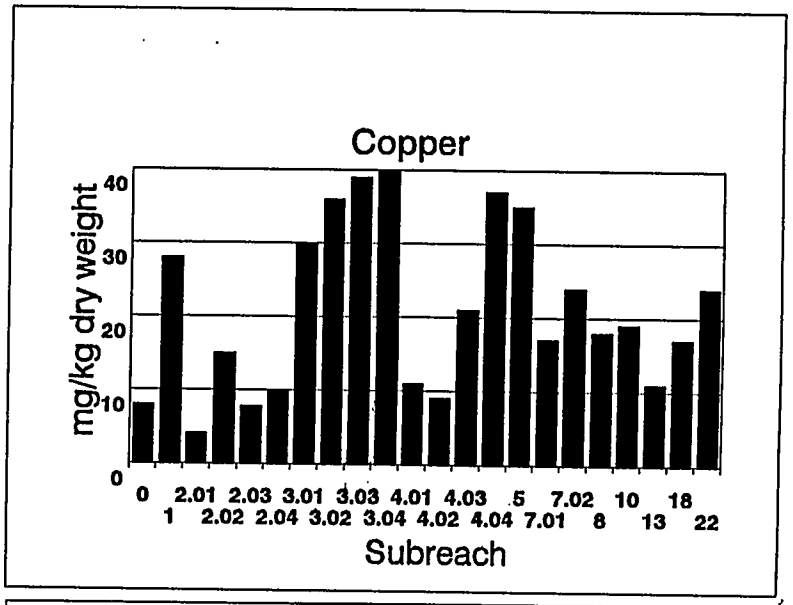

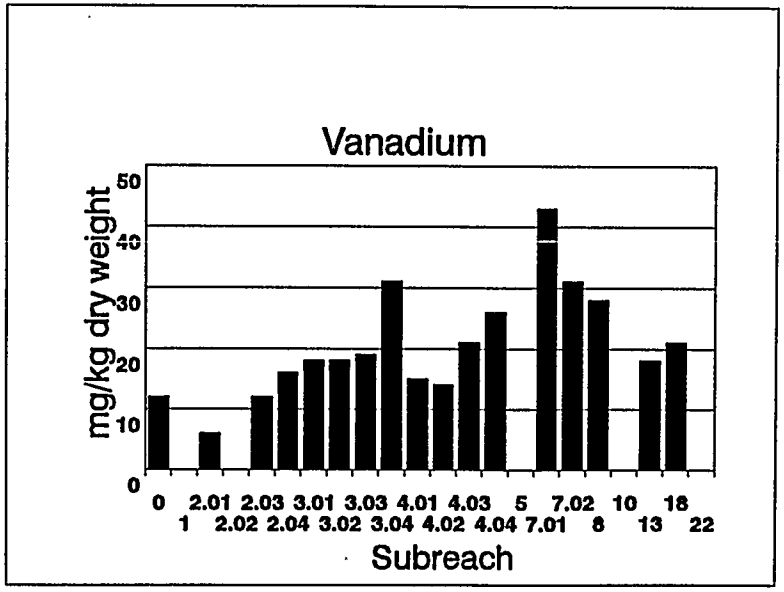

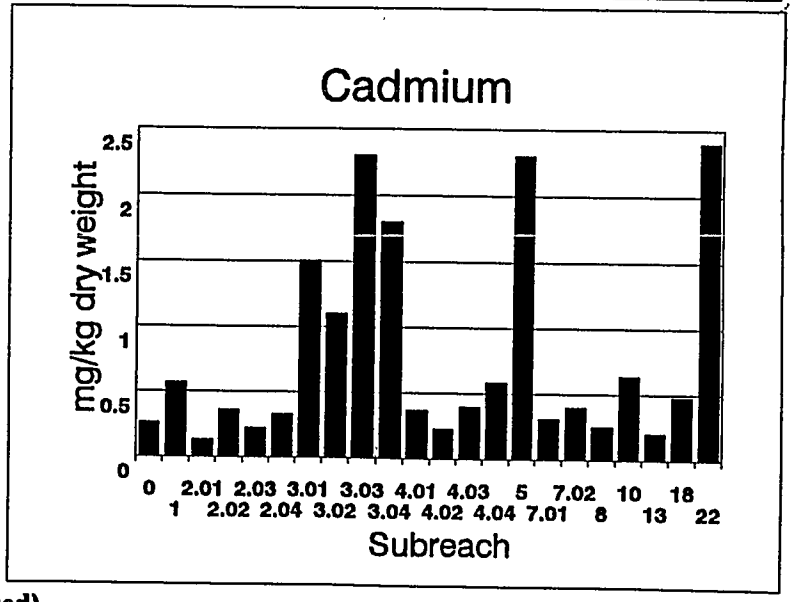

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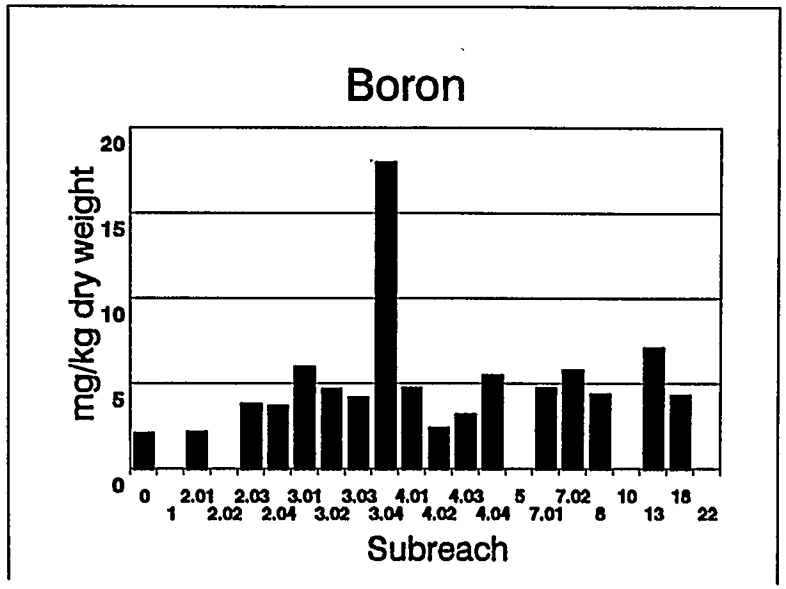

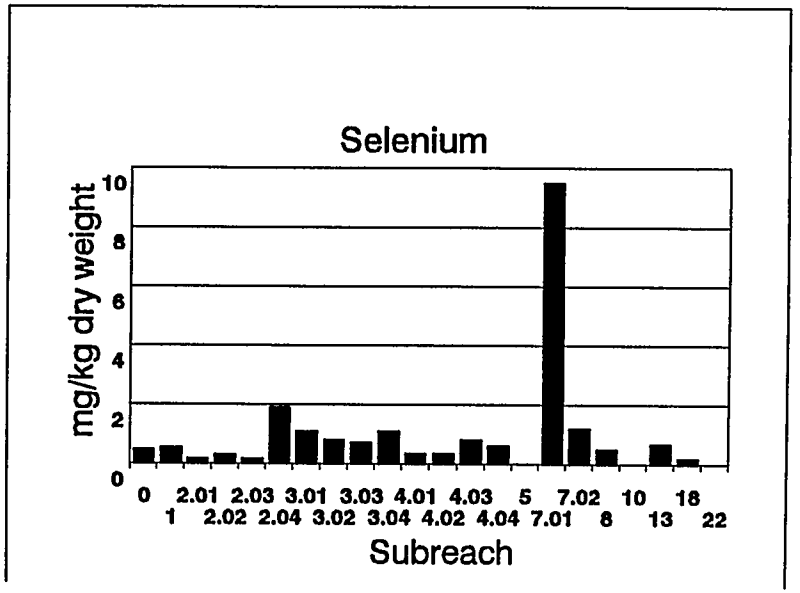

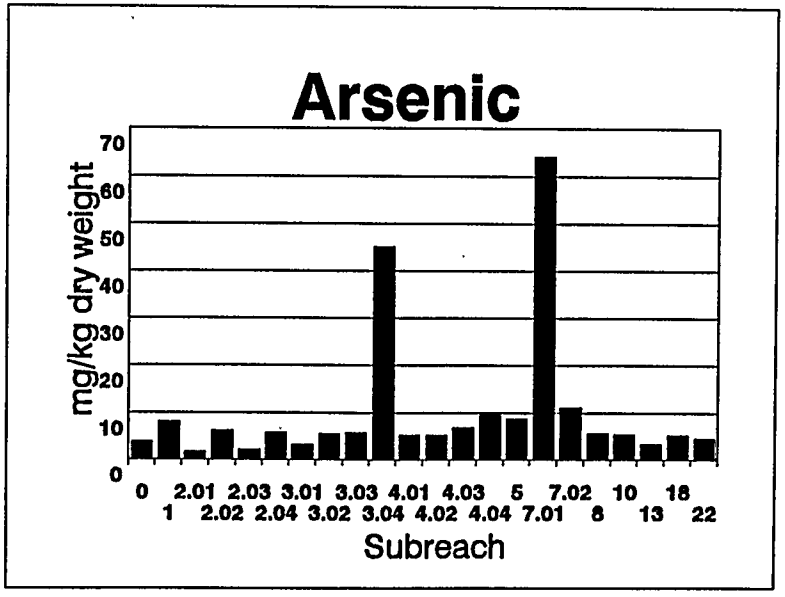

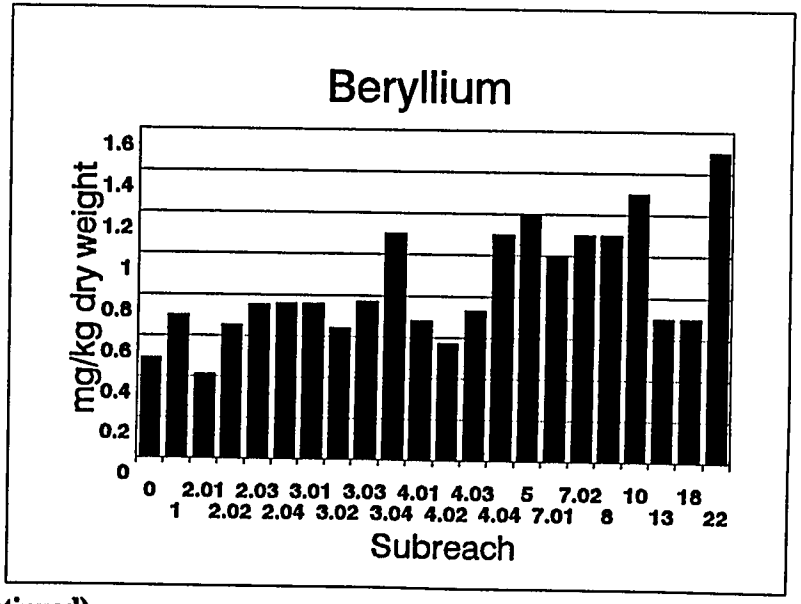

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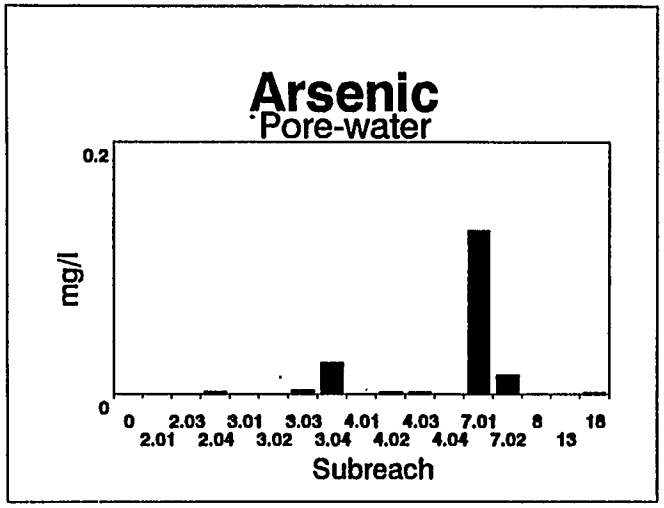
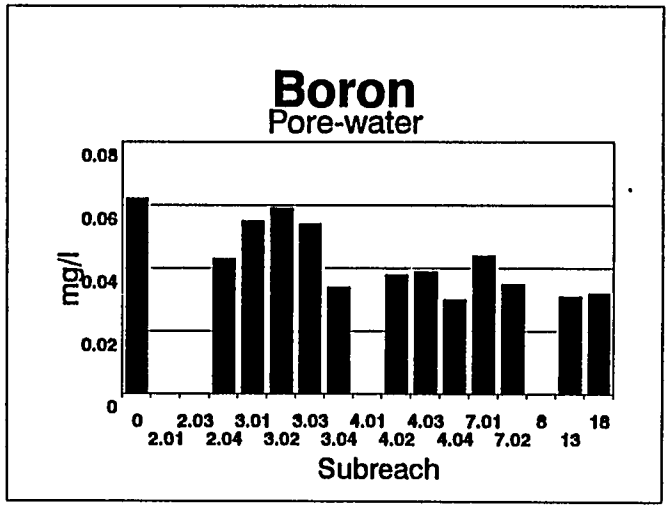
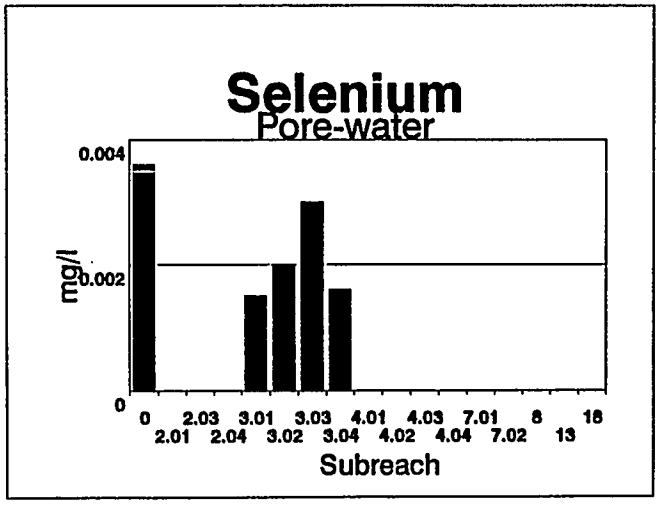






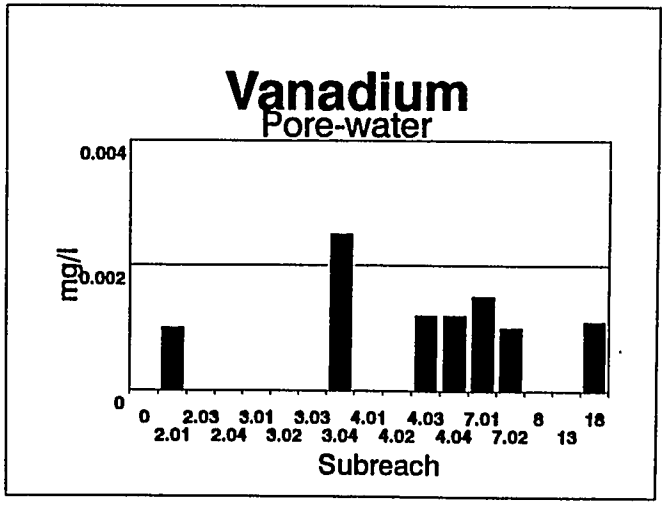
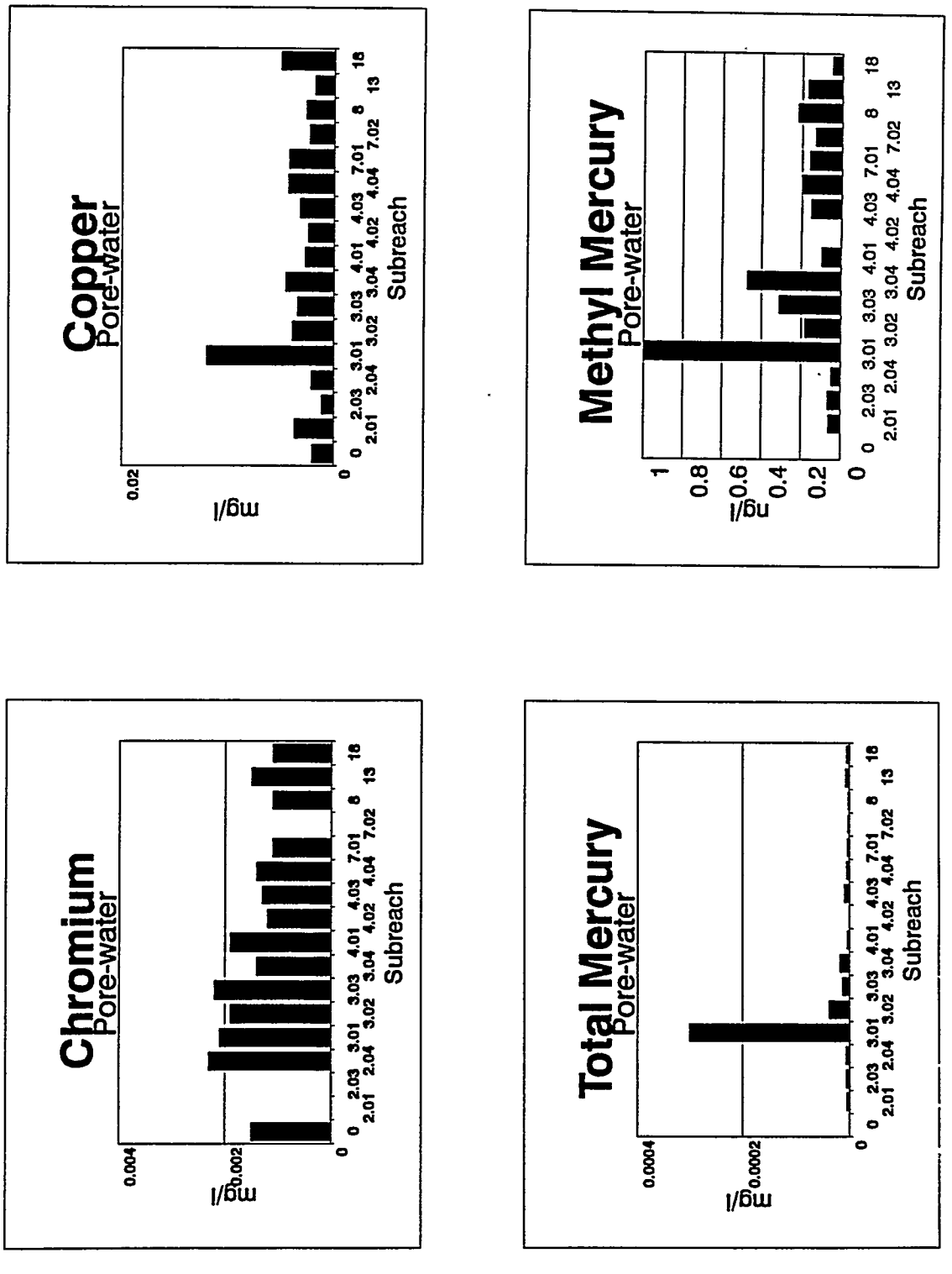


Fig. 3.38. Pore water mean concentrations for subreaches of inorganic contaminants.



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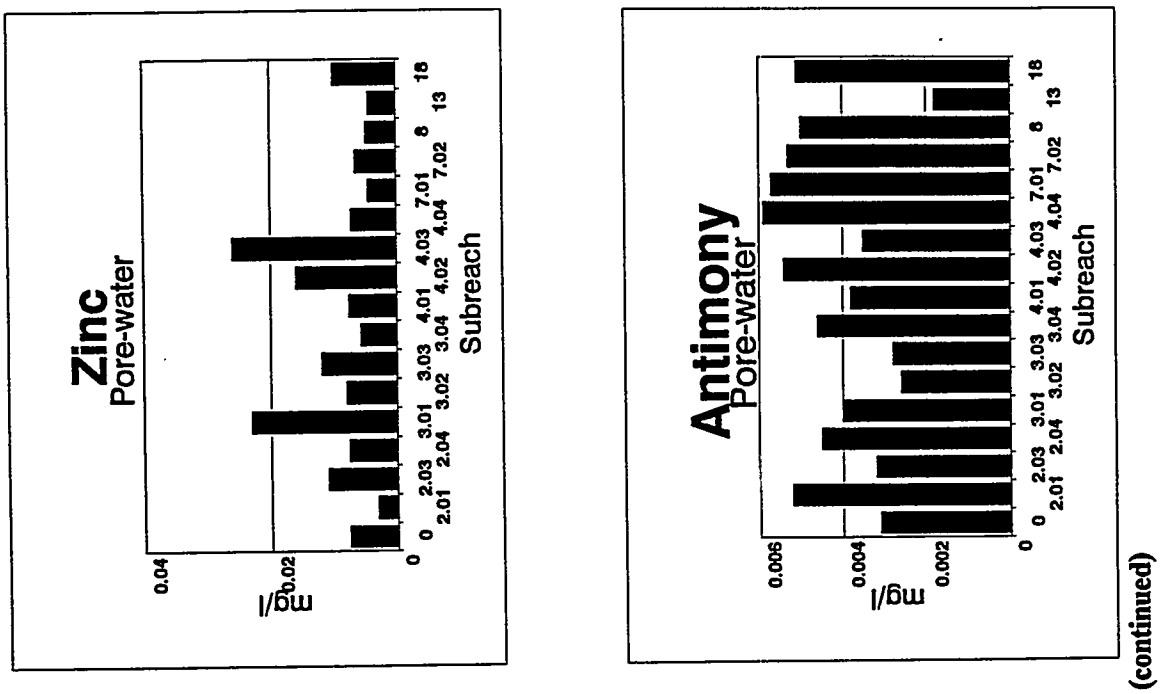

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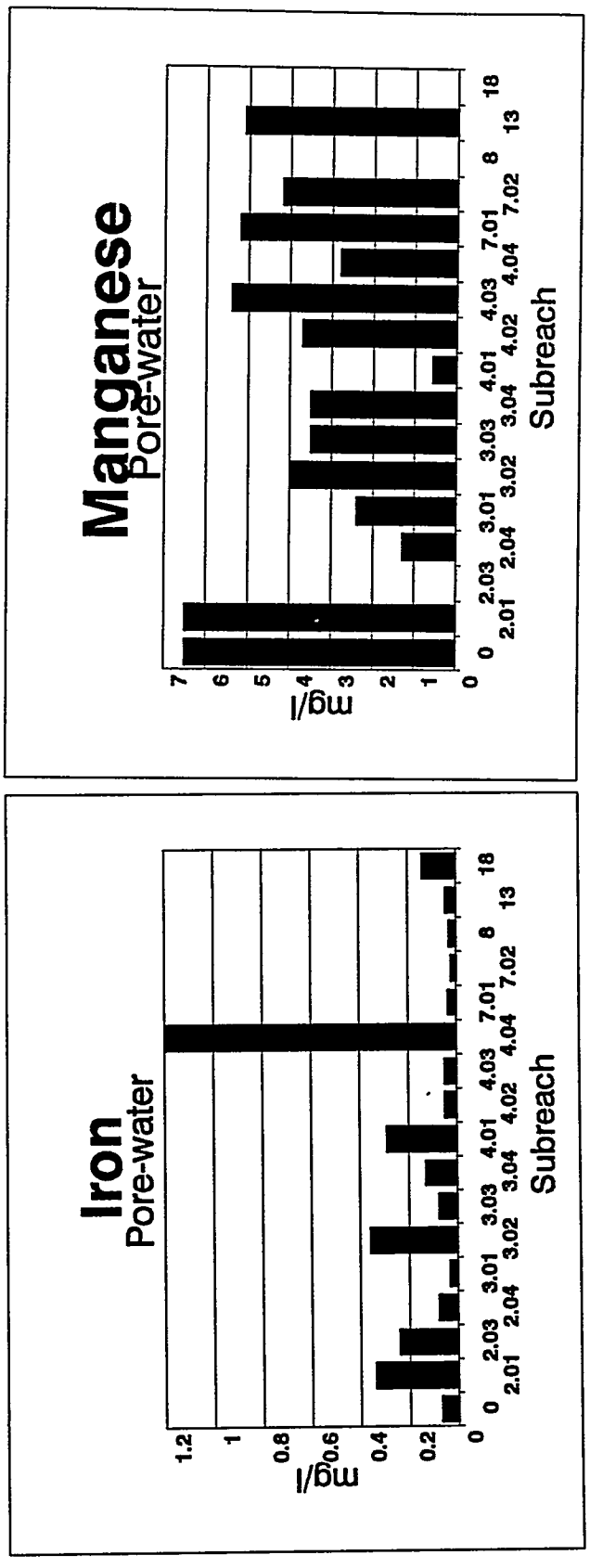

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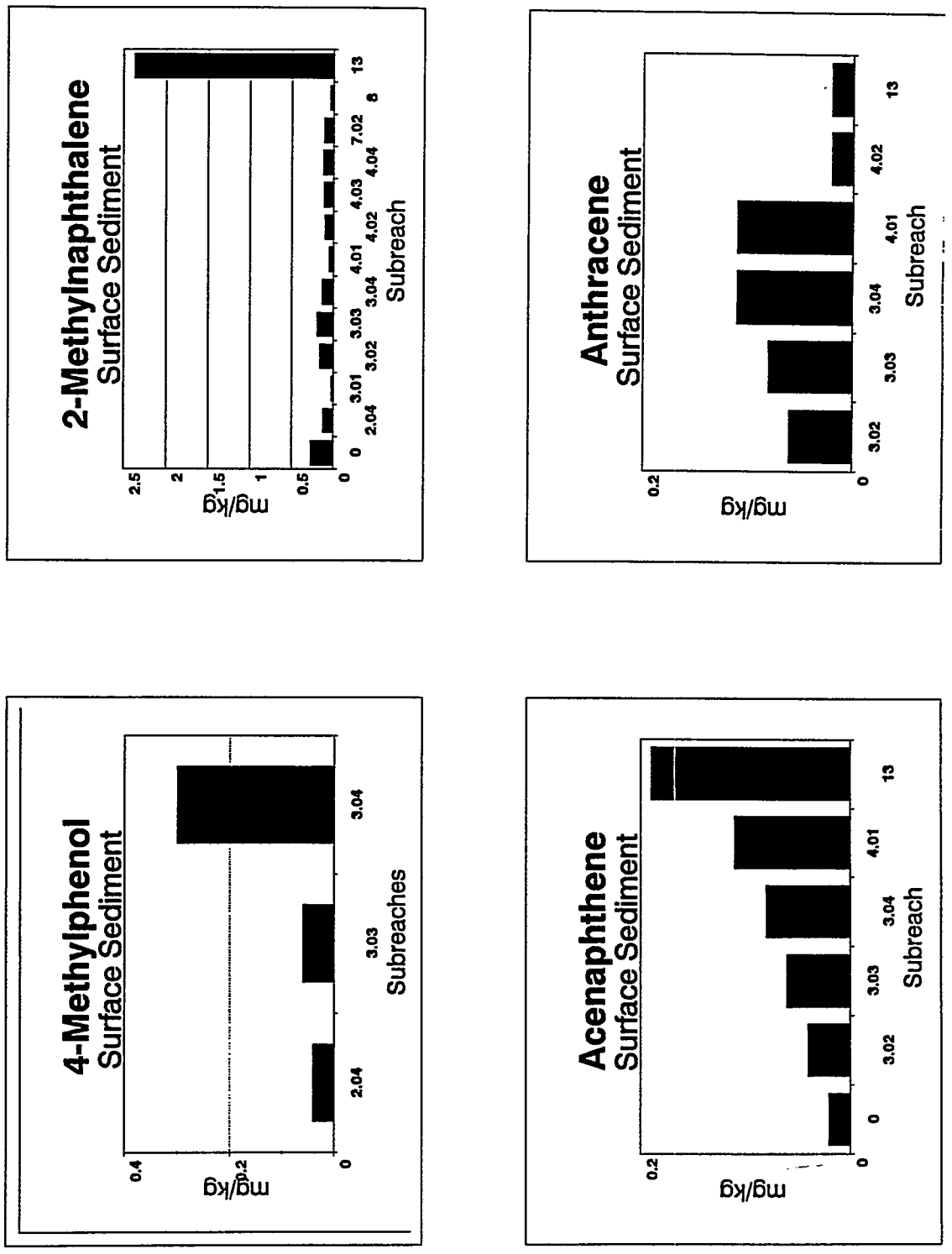
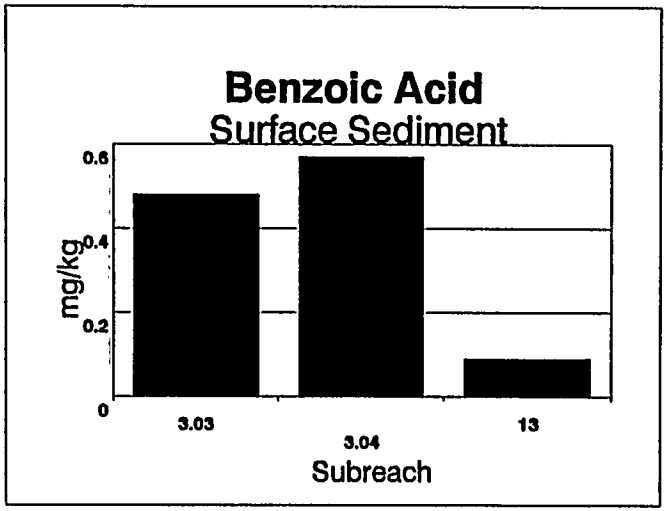
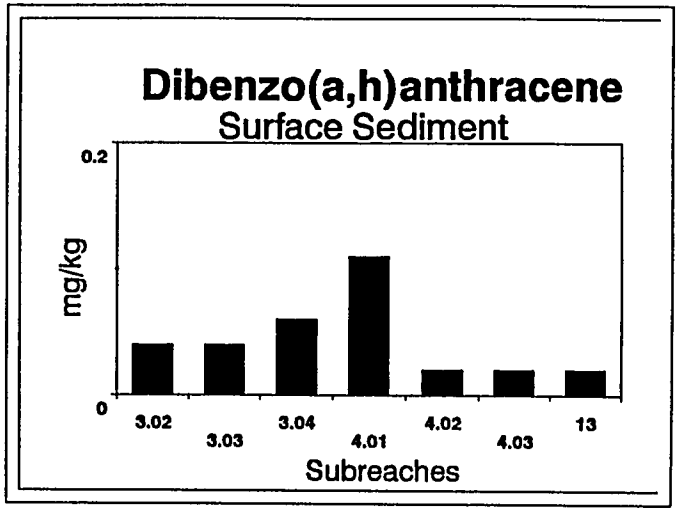
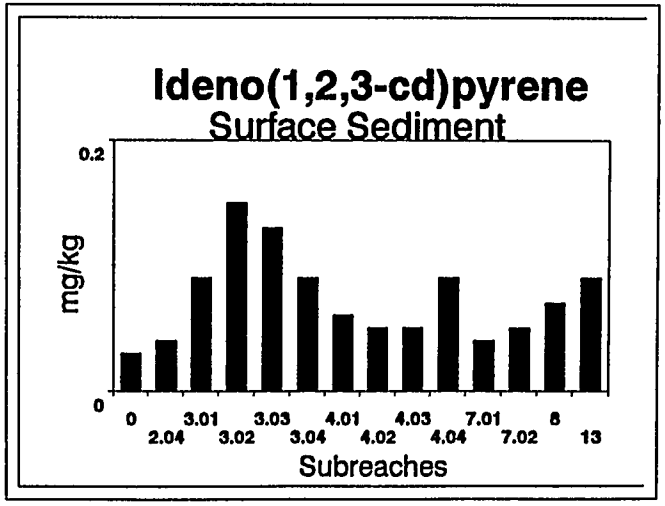
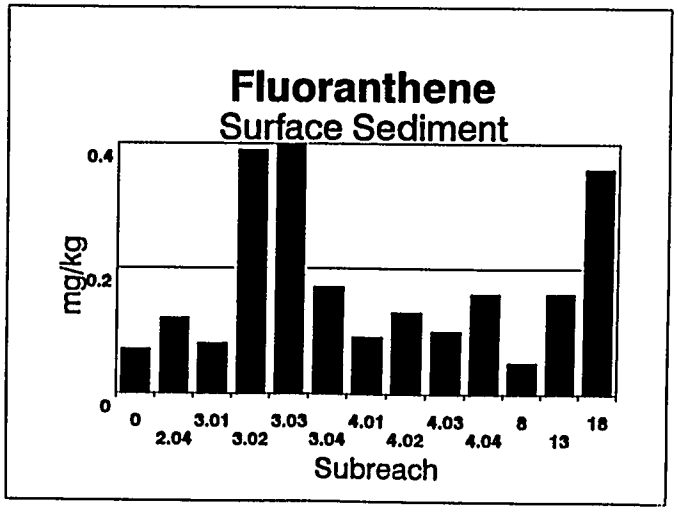


Fig. 3.39. Surface sediment mean concentrations for subreaches for organic contaminants.









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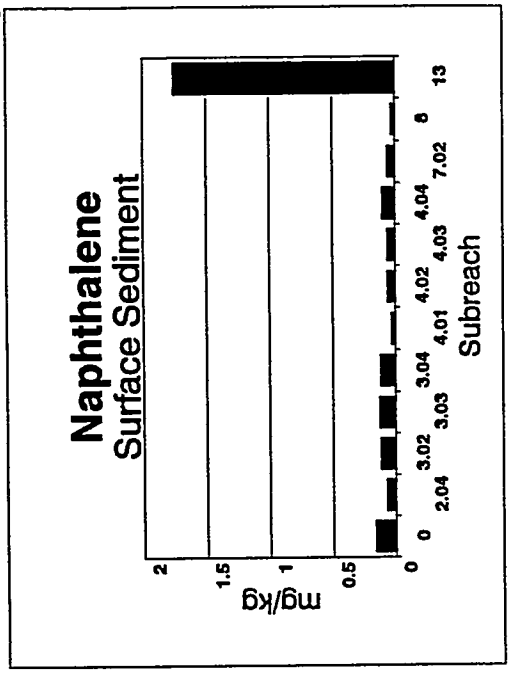

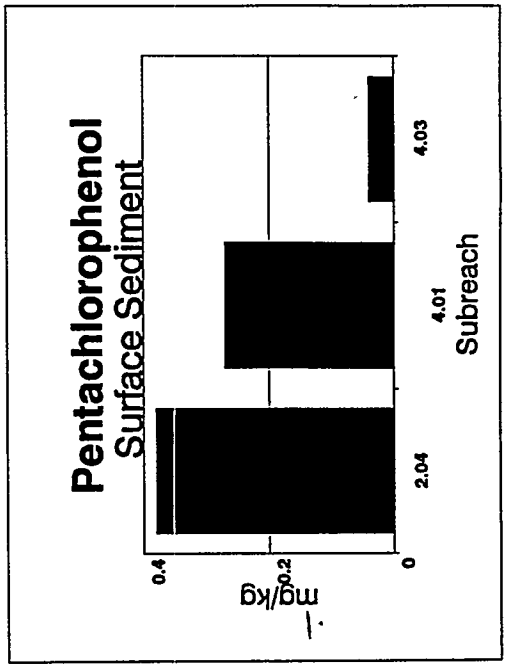

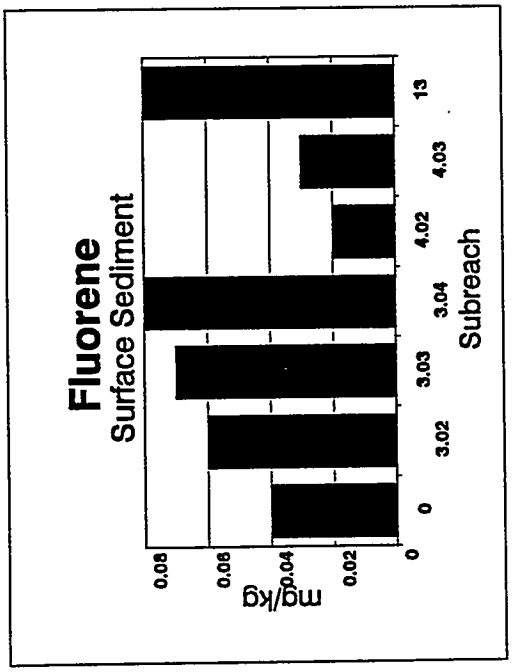

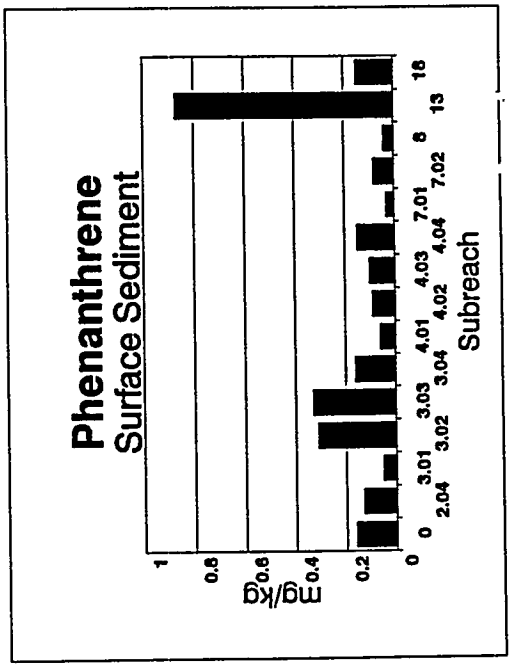

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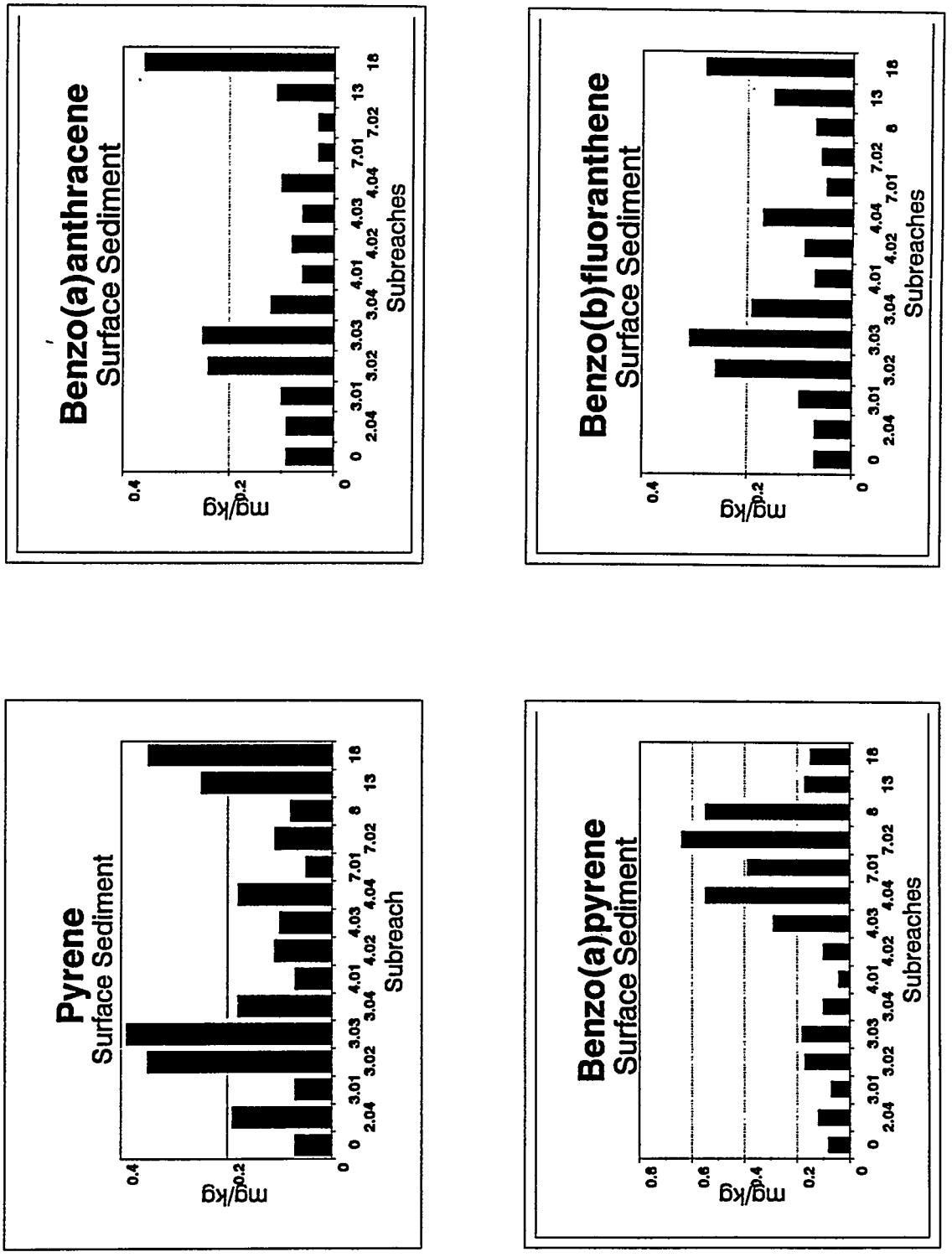

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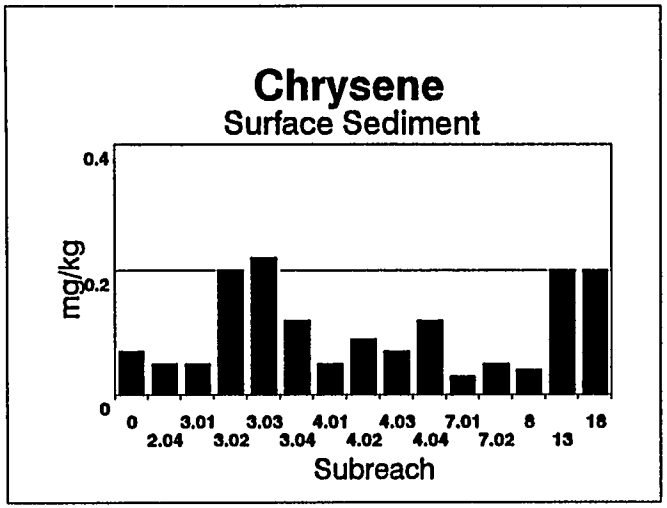

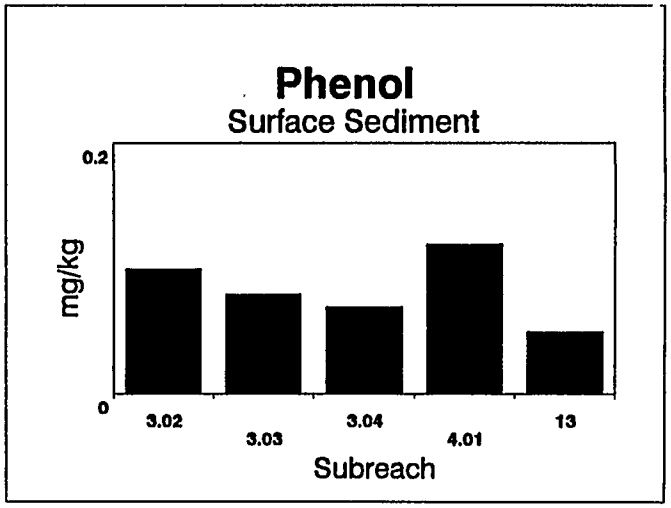

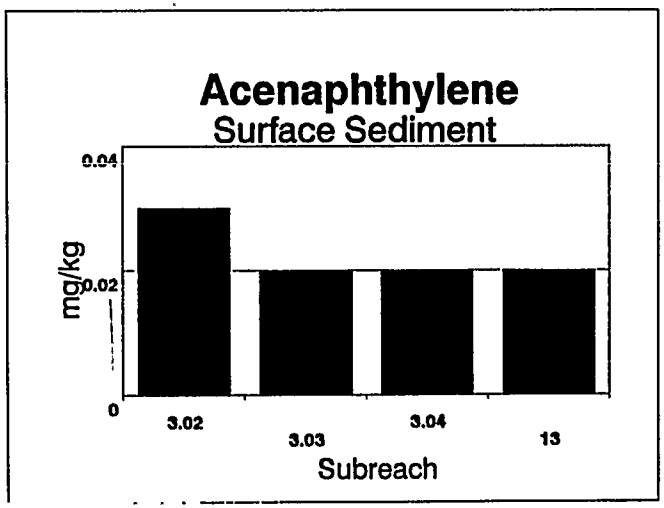

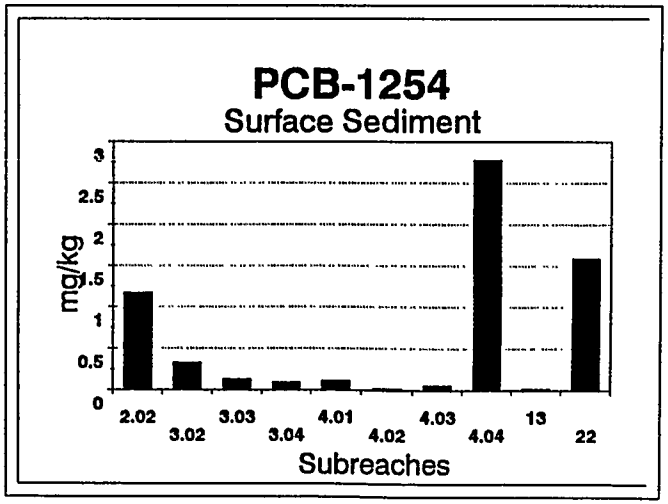

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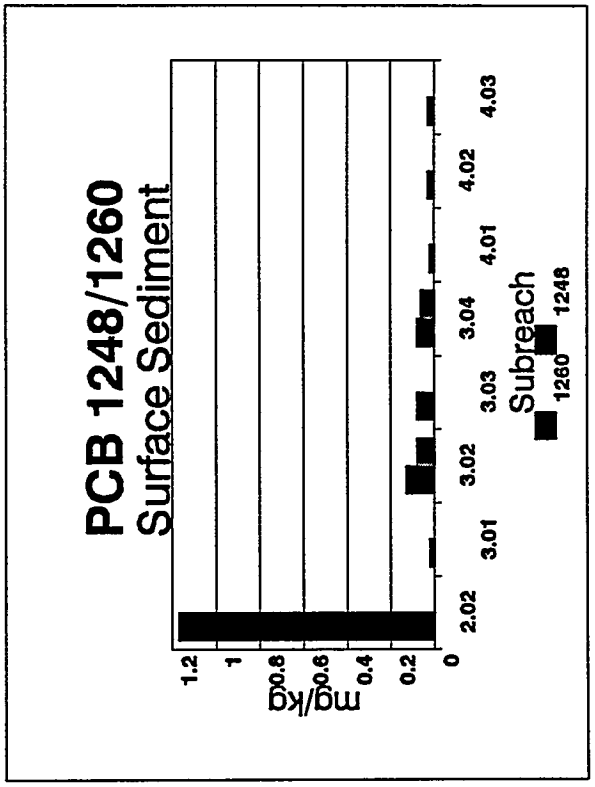

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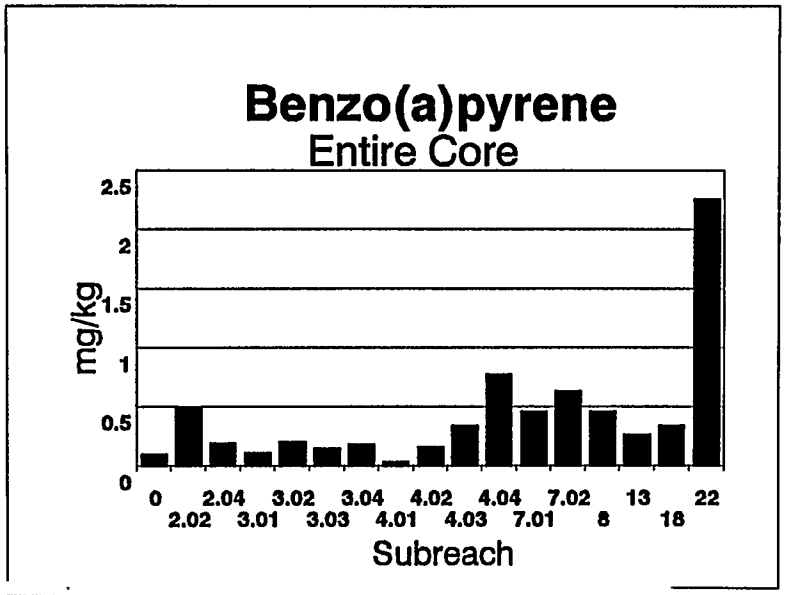
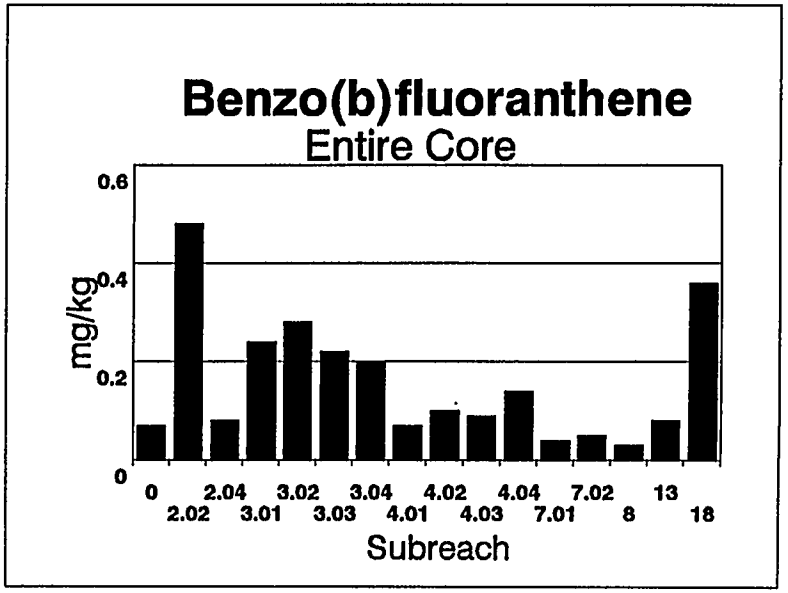
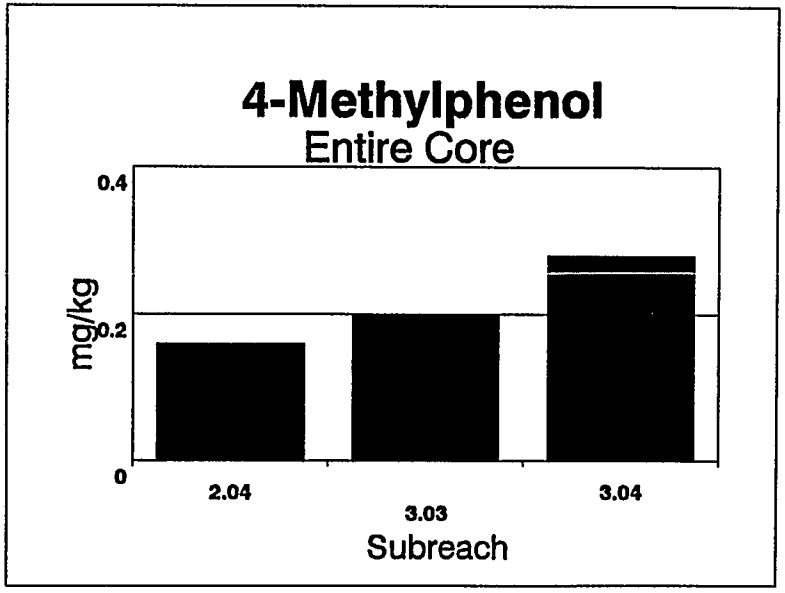






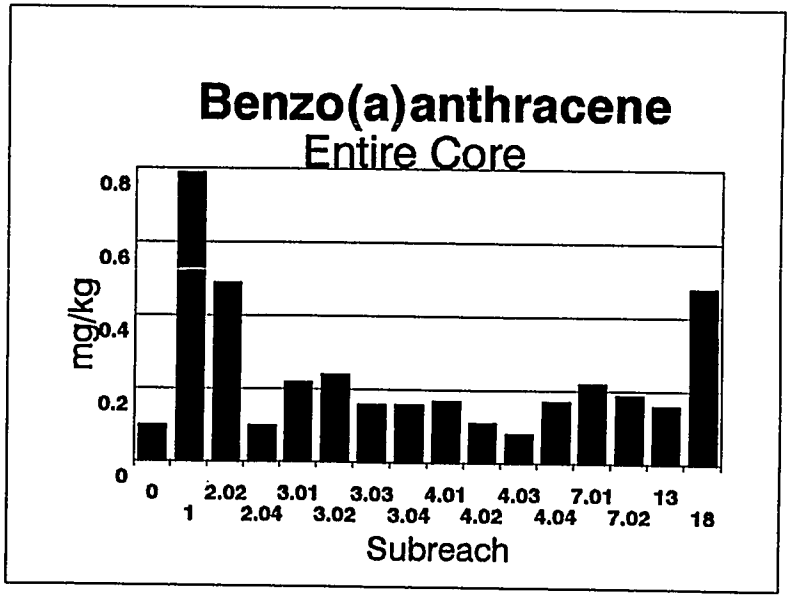
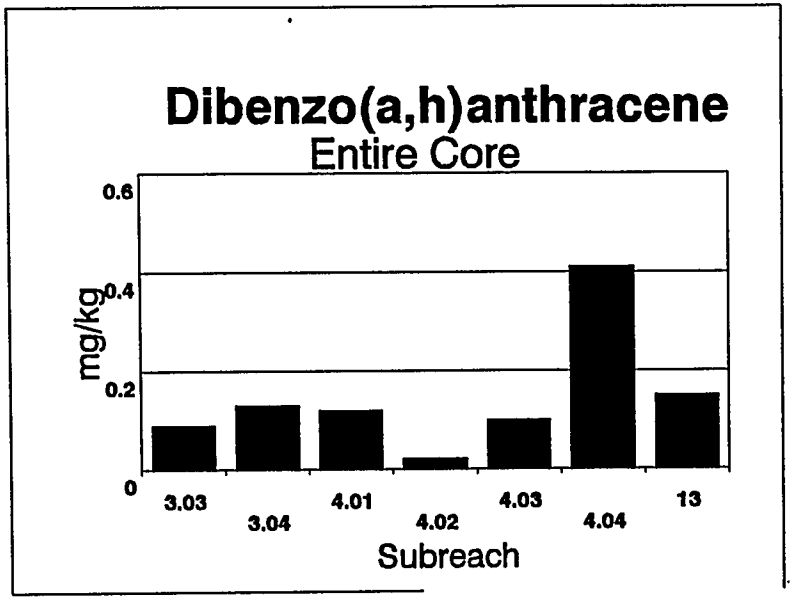
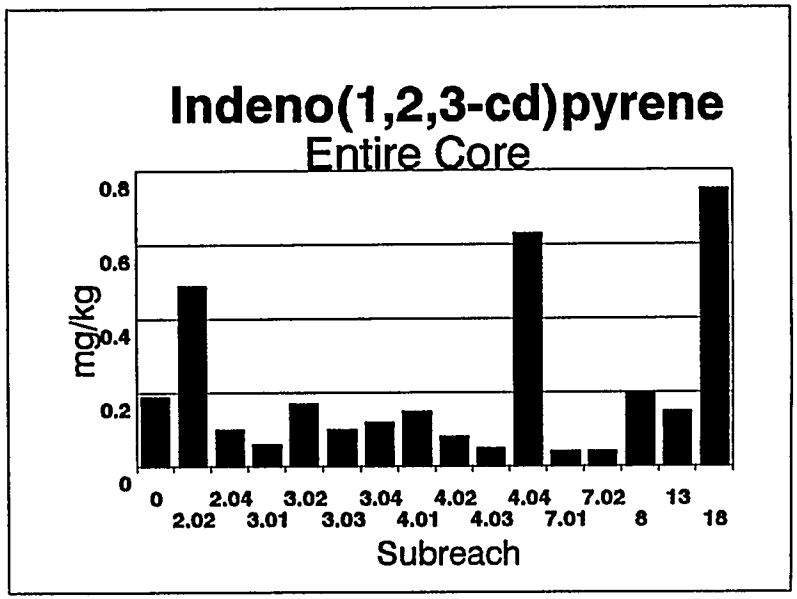
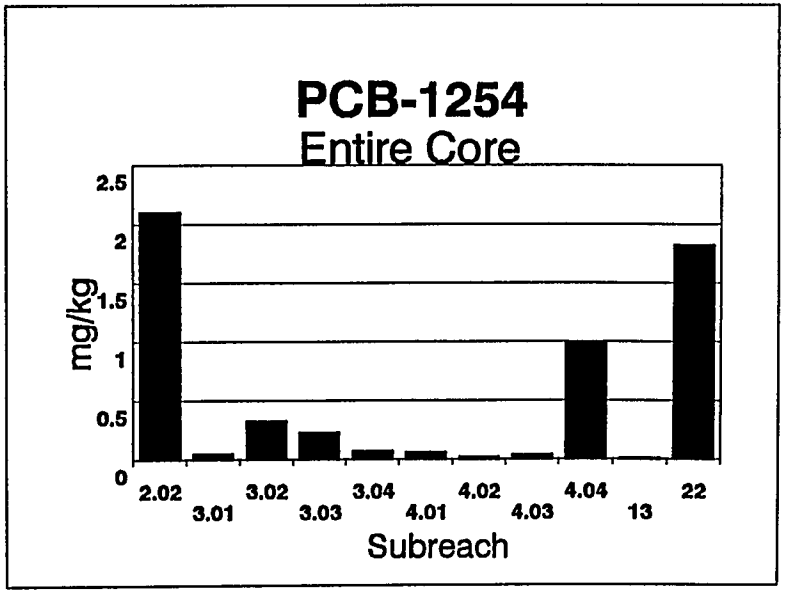


Fig. 3.40. Core-weighted mean concentrations for subreaches of organic contaminants.







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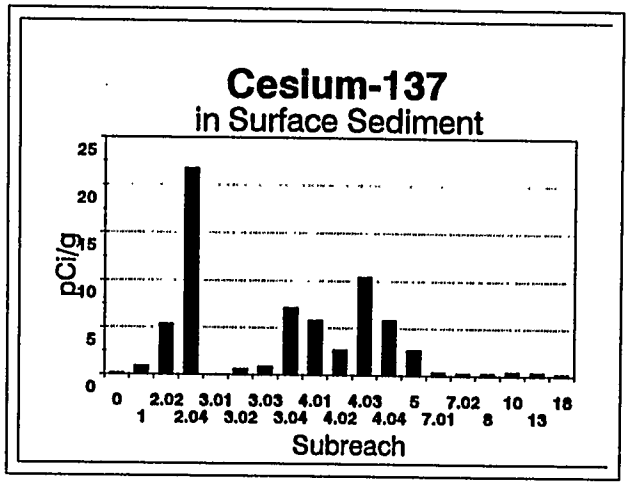
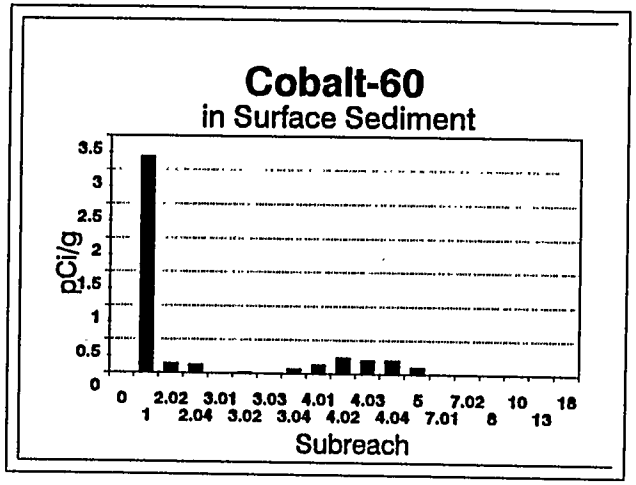
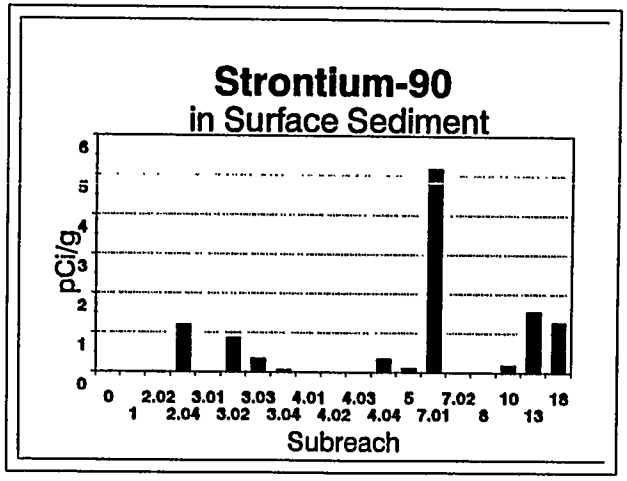






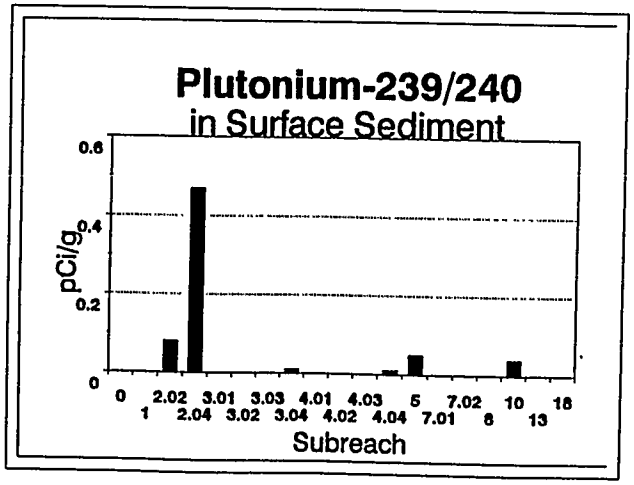
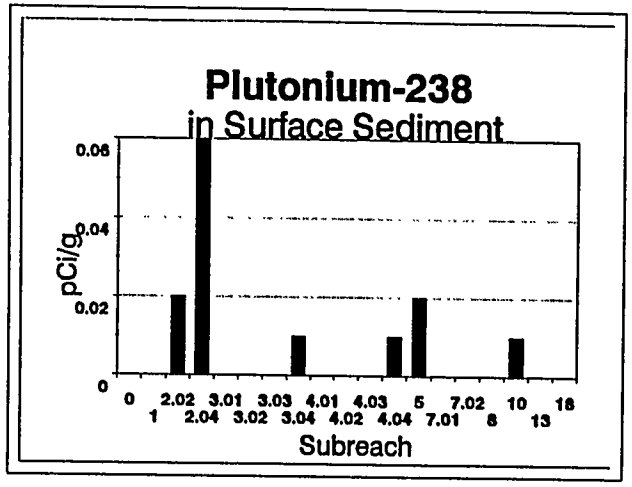
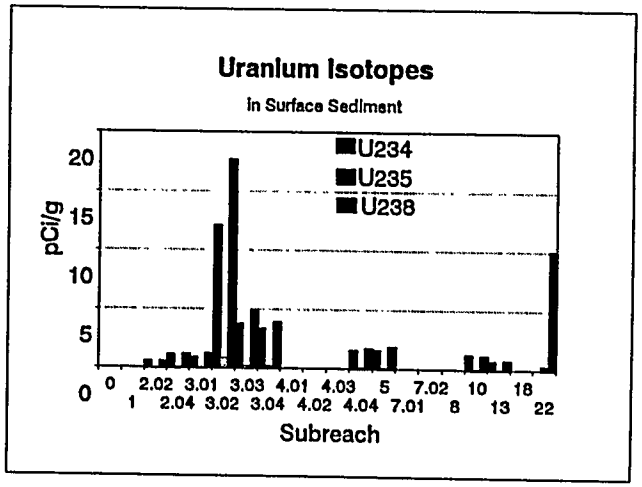
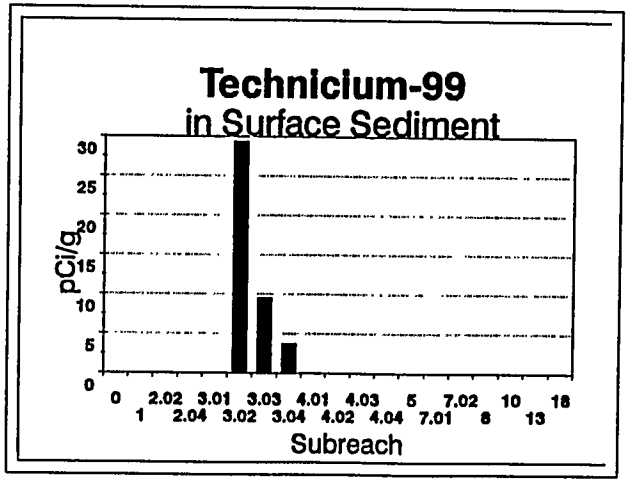


Fig. 3.41. Surface sediment mean concentrations for subreaches of radiological contaminants.







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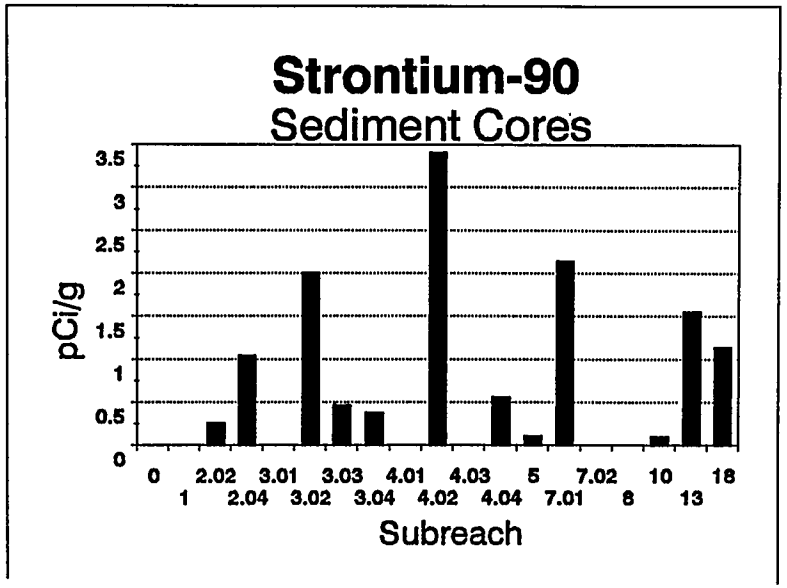
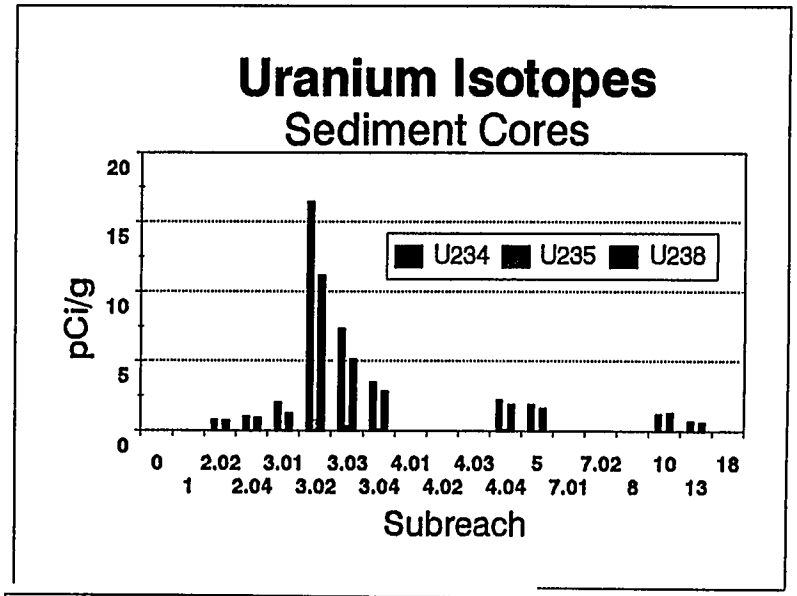
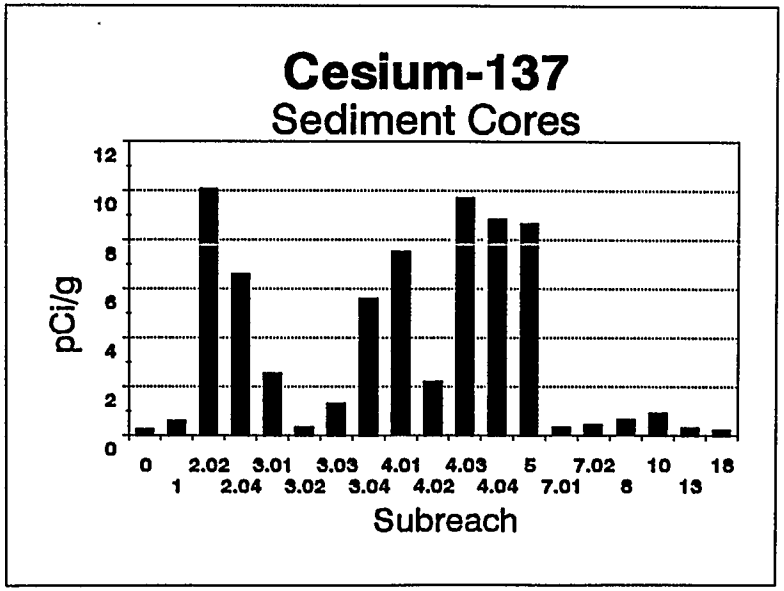






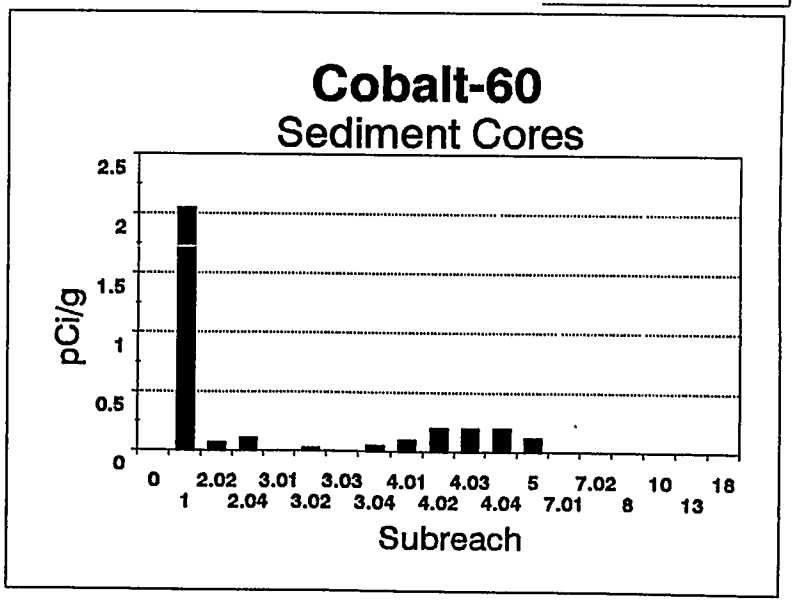
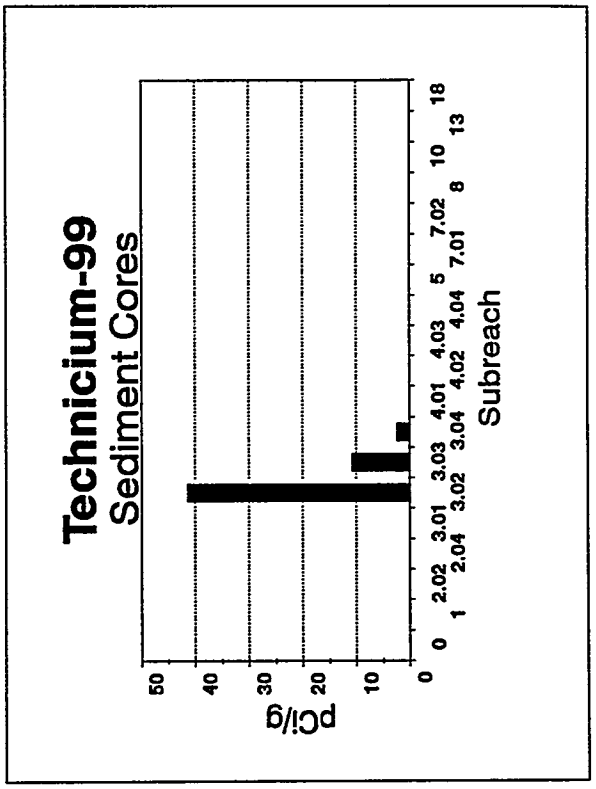


Fig. 3.42. Core-weighted mean concentrations for subreaches for radiological contaminants.



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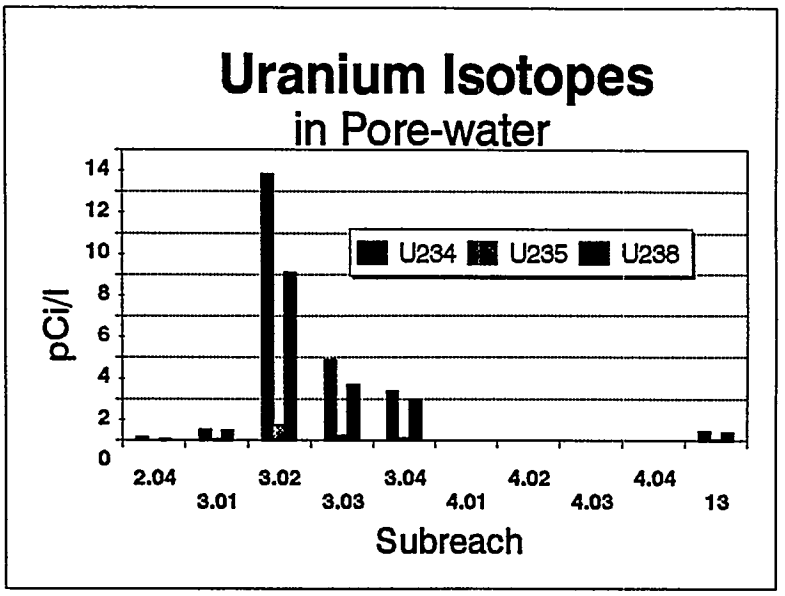


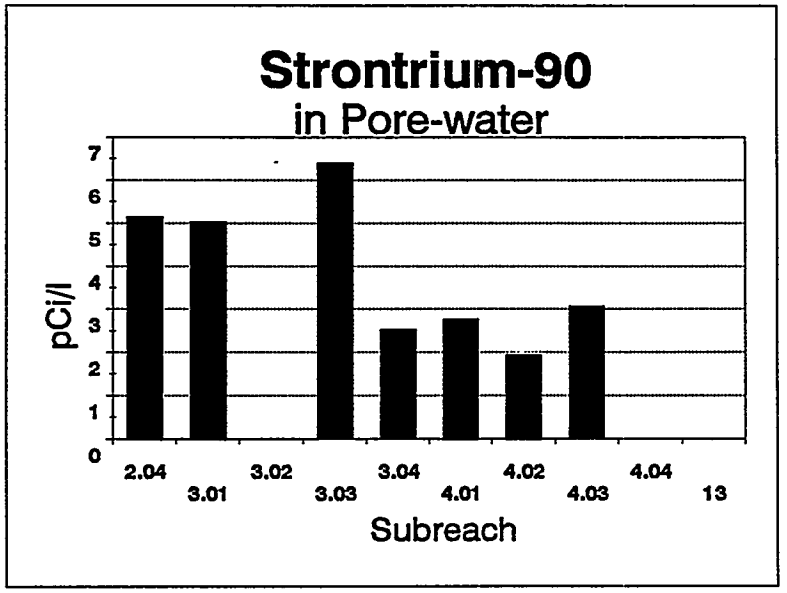


Fig. 3.43. Pore water mean concentrations for subreaches for radiological contaminants.

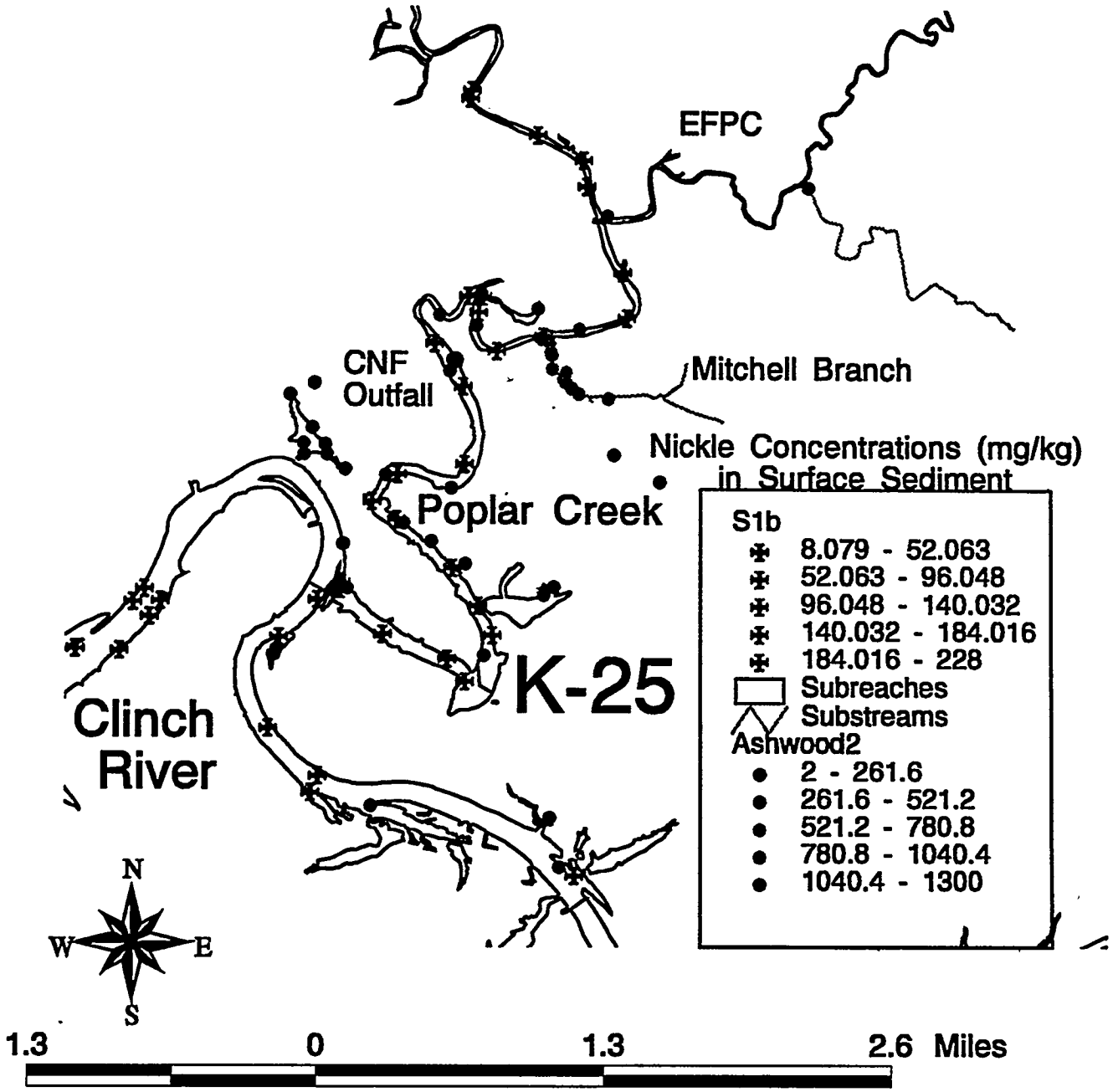


Fig. 3.44. Nickel concentrations in surface sediment in Mitchell Branch and vicinity (data from Ashwood et al. 1986 and CRRI Phase 2).

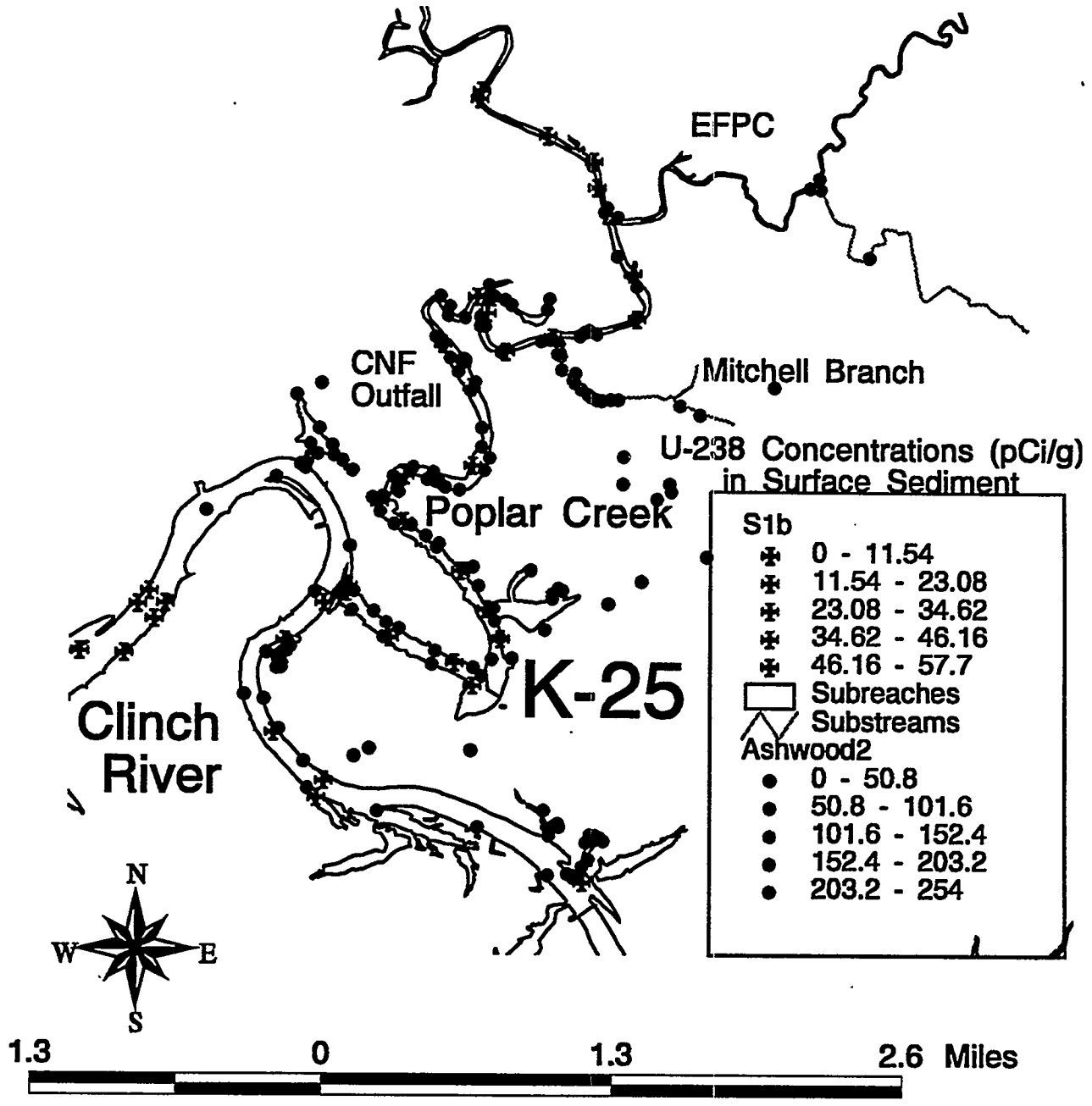


Fig. 3.45. Uranium concentrations in surface sediment in Mitchell Branch and vicinity (data from Ashwood et al. 1986).

Clinch River

This discussion considers spatial trends in contaminant distribution from upper Melton Hill Reservoir (reach 0) 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. Organic contaminants were detected only infrequently, consisted mostly of relatively ubiquitous PAHs and phenols, were generally present at low concentrations (<1 mg/kg dry sediment), and were often present in one or more reference reaches at similar concentrations (Tables B12–B15). No clear pattern of organic contaminant distribution in the Clinch River is is evident (Figs. 3.39 and 3.40). Therefore, the discussion below 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. 3.41–3.43). This increase is attributed to known releases from the American Nuclear Corporation site on Braden Branch (see Sect. 3.1) and not the ORR. Cobalt-60 remains elevated in both surface and deep sediment at levels above reference values throughout the Clinch River, but at concentrations much lower than those observed in reach 1. Copper is 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, it is difficult to detect trends in reach 2 with confidence. 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. 3.41 and 3.42). 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 appears elevated in both surface and deep sediment in subreach 2.02, below WOC (Figs. 3.39 and 3.40). Although the data are derived from a single core sample, the spatial pattern is consistent with known PCB contamination in WOC sediment (subreach 22), 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. 3.36 and 3.37). Subreach 2.04 is adjacent to the the K-1770 steamplant 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. 3.36 and 3.37), but at levels much reduced from those within Poplar Creek itself. Mean mercury concentrations are relatively constant throughout reach 4. As noted above, ^{60}Co remains elevated throughout reach 4, but mean concentrations are low (<1 pCi/g). Cesium-137 remains elevated throughout reach 4 as well, where core values are generally highest (Fig. 3.42); surface concentrations appear slightly higher in reach 2 (Fig. 3.41).

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, as this pattern is evident for contaminants whose sole or primary source is the ORR (e.g., ^{137}Cs and 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. 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. 3.36 and 3.37). Aroclor-1254 was also elevated below the Emory River, but while 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 subreach 7.01, 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. 3.36–3.38). Selenium was elevated in surface and deep sediment in subreaches 7.01 and 7.02. Vanadium appeared elevated in deep sediment and pore water and to a lesser degree in surface sediment. Each of these metals is associated with coal ash, and its presence is presumed to result from the historical disposal of fly ash from the Y-12 Plant into McCoy Branch and Rogers Quarry.

3.4.4 Sediment And Pore-water Toxicity

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).

3.4.4.1 Introduction

The sensitivity of an organism is related to the route of exposure and the biochemical response to contaminants (EPA 1994). For sediment, three general exposures can be defined: exposure to particles (whole sediment), to pore water, and to overlying water (elutriate). A suite of organisms were chosen to assess the toxicity of each exposure route. Organisms that were chosen to evaluate exposure to whole sediment were *Hyalella azteca* (survival endpoint) and *Anodonta imbecillis* (survival endpoint); the solid phase Microtox® test was also used to evaluate toxicity of whole sediment. Two tests were chosen to evaluate exposure to pore water: the *Ceriodaphnia dubia* acute test (survival endpoint) and the liquid-phase Microtox® test. *C. dubia* is a crustacean that is commonly used to evaluate water quality (EPA 1993). Elutriate exposure was evaluated using the crustacean *Daphnia magna* (survival and reproduction endpoints). Direct exposure to *D. magna* may occur from dissolved or colloidal contaminants.

The methods used for evaluating sediment toxicity with these organisms are described in Table B22. Care was taken to avoid excessive alteration and manipulation of sediment samples during sediment storage and preparation and pore water extraction. Several common water quality parameters were analyzed in the water used in the test (alkalinity, hardness, pH, DO, conductivity, and ammonia). Except for ammonia, potentially toxic contaminants were not analyzed. All measured water quality parameters were within organism tolerances and, therefore, do not affect the sediment toxicity results.

The study period (July 1993 to September 1994) involved 9 sampling events: event 1—July 1993; event 2—August 1993; event 3—November 1993; event 4—February 1994; event 5—May 1994; event 6—June 1994; event 7—July 1994; event 8—August 1994; and event 9—September 1994. A total of 78 samples from 15 sites in 6 reaches were collected for toxicity testing (Fig. 3.46); not all sites were sampled during each sampling event. A total of 304 tests were conducted, using 5 test species and 3 endpoints.

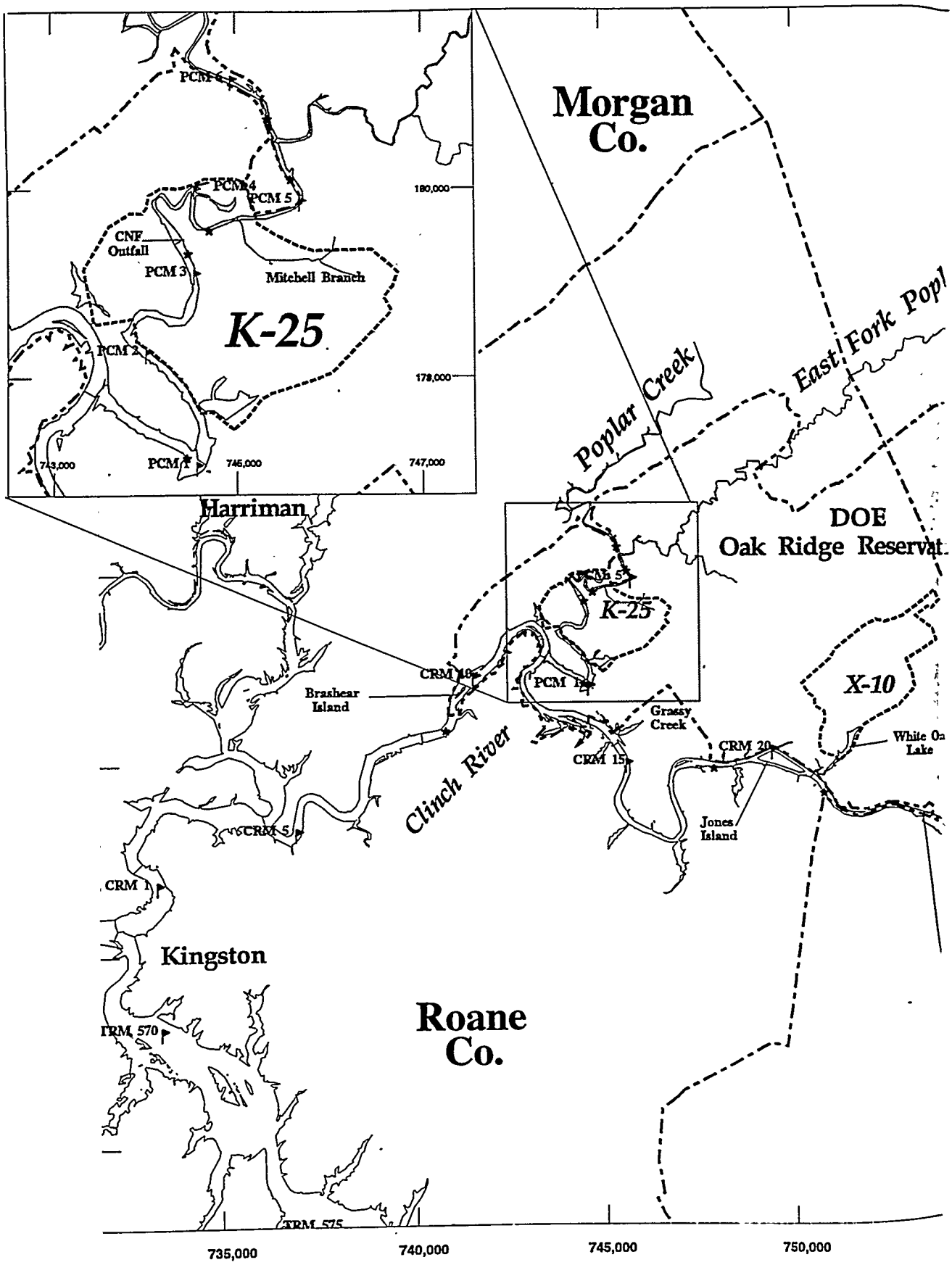
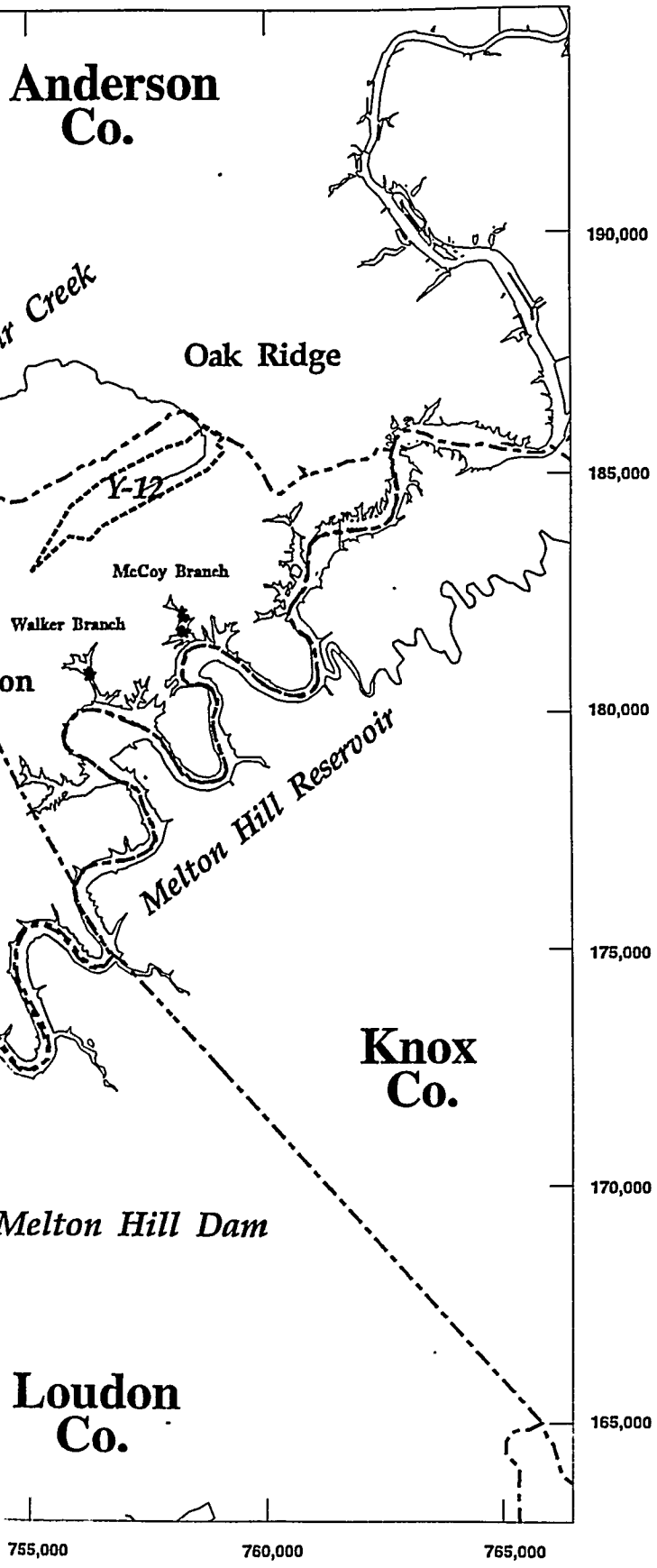


Fig. 3.46. Sampling locations for sediment and pore water toxicity samples.



Clinch River Environmental Restoration Program

Sampling Locations for Sediment Toxicity Task

N
W E
S

0 2,500 5,000
Meter

LEGEND

- Sample 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 Clinch River Environmental Restoration Program

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3.4.4.2 Sediment Toxicity—Results and Discussion

The interpretation of sediment toxicity test results, presented below, is made within the context of two important test issues. First, the sites of interest were not sampled in replicate, introducing the issue of pseudoreplication into the study and precluding an analysis of spatial variability of sediment toxicity at the sites of interest. Also, an appropriate definition of "significance," due (in part) to the unavailability of a true control sediment and to the very large scale of ecological systems, is an unresolved issue with respect to sediment toxicity evaluations.

Either of two criteria were used to identify a significant toxic response: (1) if the study site had a statistically significant ($p<.05$) toxic response relative to the reference site or (2) if the study site had a toxic response that was 20% greater than the reference site.

Whole Sediment. A total of 62 individual tests were conducted with *H. azteca*, resulting in 46 site-to-reference comparisons. Only one test (PCM 4.3 in event 1) showed a statistically significant reduction in survival compared to the reference site. Eleven of the 46 comparisons (23.9%) showed a ≥ 20% reduction in survival relative to the reference sites. These include four sites during event 1 (PCM 1.0, 2.9, and 4.3 and CRM 19.0), site CRM 9.0 in event 3; sites PCM 5.1, CRM 9.0, and CRM 19.0 in event 4; site PCM 2.9 in event 5; site CRM 19.0 in event 7; and CRM 9.0 in event 9. No significant reductions in survival were observed in any McCoy Branch/Walker Branch tests.

The generally high survival rate (85% of all *H. azteca* tests resulted in ≥ 70% survival) and the low percentage of statistically significant site-to-reference comparisons (1 of 42) indicates that either the sediment samples were not contaminated, or the *H. azteca* toxicity test was not sensitive to any contaminants present. Many studies have shown this organism to be sensitive to many particle-associated contaminants, supporting the former conclusion.

A total of 24 individual tests were conducted with the freshwater mussel *A. imbecillis*, resulting in 18 site-to-reference comparisons. Only four of the 24 (22.2%) showed a ≥ 20% reduction in survival relative to reference sites: PCM 4.3 in events 2 and 9 and site CRM 19.0 in events 1 and 9. The only statistically significant reduction in survival was at CRM 19.0 in event 9.

Seventy-five percent of all *A. imbecillis* tests resulted in ≥ 75% survival. However, this test has many serious limitations and uncertainties, including a high sensitivity to technician proficiency and a very small database of whole sediment toxicity test studies with which to validate the method as sensitive and useful. The results presented here are suspect for at least two reasons: (1) organism survival in split sample tests conducted by TVA's Aquatic Research Laboratory was consistently higher than our results (survival was often 100% at TVA), and (2) organism recovery in our tests were often below 100%.

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.

The excellent *C. dubia* survival (96% of all tests resulted in ≥ 90% survival) is strong evidence for the absence of toxicity in the pore water samples tested, as the sensitivity of this test organism to aqueous toxicants is well established. However, the major issue with this test is not organism response, but rather the pore water sample characteristics and stability. 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.

Elutriate. A total of 54 individual tests were conducted with *D. magna,* resulting in 40 site-to-reference survival comparisons. Five of the 40 (12.5%) showed a ≥ 20% reduction in survival relative to reference sites: PCM 1.0 in events 1 and 7; CRM 9.0 in event 4; and CRM 19.0 in events 1 and 5. None of these comparisons showed statistically lower survival.

Forty site-to-reference reproduction comparisons were obtained from the same 54 *D. magna* tests. Seven of the 40 (17.5%) showed a ≥ 20% reduction in reproduction: PCM 1.0 in events 4 and 5; PCM 4.3 in event 4; PCM 5.1 in event 4; CRM 9.0 in events 4 and 5; and CRM 19.0 in event 4. Note that all significant reductions in reproduction occurred in either event 4 or 5. The reduction in reproduction was statistically significant in two of the seven tests above (CRM 9.0 and 19.0, both in event 4).

Toxicity was not suggested by either the *D. magna* survival or toxicity data. Overall survival was high, with about 85% of the tests resulting in ≥ 80% survival. Water-only reference toxicant tests and other water-only studies have demonstrated *D. magna* to be relatively sensitive to water-borne contaminants, implying that contaminants are either not present in Clinch River/Poplar Creek sediment or are not migrating into the overlying water column.

3.4.4.3 Conclusions

The results above reveal very little toxicity from sediment at the sites tested. The total percentage of tests that met either of the the significance criteria was very low, suggesting a minimal impact to test species in the Poplar Creek, Clinch River, and McCoy/Walker Branch systems.

There were no striking spatial trends in the sediment toxicity data. Of the overall low number of positive responses at test sites, the higher percentage of them were found at PCM 4.3 and CRM 9.0 and CRM 19.0. Temporal trends are difficult to determine given the minimal response detected. The greatest probability of detecting a hit appears to occur in the winter/spring, but without other data (chemistry, sediment transport dynamics, etc.), this subtle trend cannot be confirmed. No historical sediment toxicity data are available for this study area.

The most sensitive tests were the *H. azteca* and the freshwater mussel (*A. imbecillis*) whole sediment tests, where 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).

3.4.5 Sediment Transport Modeling in the Clinch River

3.4.5.1 One dimensional modeling of sediment and ^{137}Cs transport

Introduction. The fate of historically released ^{137}Cs (1949–1991) was simulated using three separate one-dimensional water and sediment transport models: HEC-6-R (USACE 1993), CHARIMA (Holly et al 1990), and TODAM (Onishi et al 1980). The models were implemented independently by ORNL, by TVA in cooperation with the Iowa Institute of Hydraulic Research, and by the 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 (TVA) models simulate hydrodynamic conditions, sediment deposition, erosion and transport, and contaminant fate. The TODAM model simulates sediment and contaminant fate processes, but receives hydrodynamic information from CHARIMA's hydrodynamic module. The three models were configured similarly for the total Clinch

River/WBR system, but differed in numerical solution techniques and in algorithms for calculating sediment transport and contaminant fate.

Calibration. All three models were calibrated using TVA data on historical sedimentation rates and validated using existing data on ^{137}Cs distributions (Cottrell 1960; Morton 1965; Struxness et al. 1967; Oakes et al. 1982; Olsen et al. 1991). 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 1951–1956 and 1961–1991.

The "known" ^{137}Cs inventory in the Clinch River in the summer of 1977 was extrapolated via 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. 3.47). 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 different predictions above are due in part to differences between the models in the assumed particle size distribution of inflowing sediments. The TODAM modelling group assumed a greater quantity of relatively coarse sediment entering the Clinch River as compared to the other modelling groups. TODAM's higher predicted inventories may be partly explained by the fact that coarser sediments would deposit more quickly and closer to the source than finer sediments. The HEC-6-R assumptions were modified, after the initial calibration above, to mimic the particle-size distributions of inflowing sediment used in the TODAM and CHARIMA models. When HEC-6-R mimicked TODAM's coarse sediments, the predicted ^{137}Cs inventories increased (from 57 to 72 Ci). Similarly, when HEC-6-R mimicked CHARIMA's finer incoming sediments, the predicted ^{137}Cs inventories decreased (from 57 to 55 Ci) (Fig. 3.47). In effect, the TODAM model failed to simulate the observed erosion and downstream transport of contaminated sediment that had been deposited in the Clinch River during the periods of peak ^{137}Cs flux from WOC in the late 1950s.

Each of the models above predicts deposition over relatively large reaches of the Clinch River, and, therefore, none predicts small-scale patterns of ^{137}Cs distribution very well. The strong spatial heterogeneity in erosion and deposition in Clinch River sediment is at a much finer scale than the modeled reaches. For example, only about 25% of the over 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 was represented by only 5 silt ranges. This resulted in some necessary additional bathymetry assumptions to obtain successful hydrodynamic simulations, which might have affected the pattern of sediment deposition adversely. Nonetheless, CHARIMA and HEC-6-R predicted total ^{137}Cs inventory very well.

Simulations. Several future scenarios were simulated by each model: (1) predictions assuming no major storm events or releases of ^{137}Cs, (2) predictions following an assumed 100-year-return-period storm localized in the WOC watershed, and (3) predictions following an assumed system-wide 100-year-return-period storm. All simulations evaluated both short-term (2 week, 1 and 6 month) and long-term (1 and 10 year) responses. The increased ^{137}Cs flux from WOC was based on preliminary predictions by the HSPF watershed model (Bickwell et al. 1993) applied to the WOC Watershed. The scenarios begin on April 30, 1991, each modelling group beginning with the ^{137}Cs inventories predicted for that date by their respective models. Results are summarized in Table 3.5.

Curies Present in the Clinch in 1977

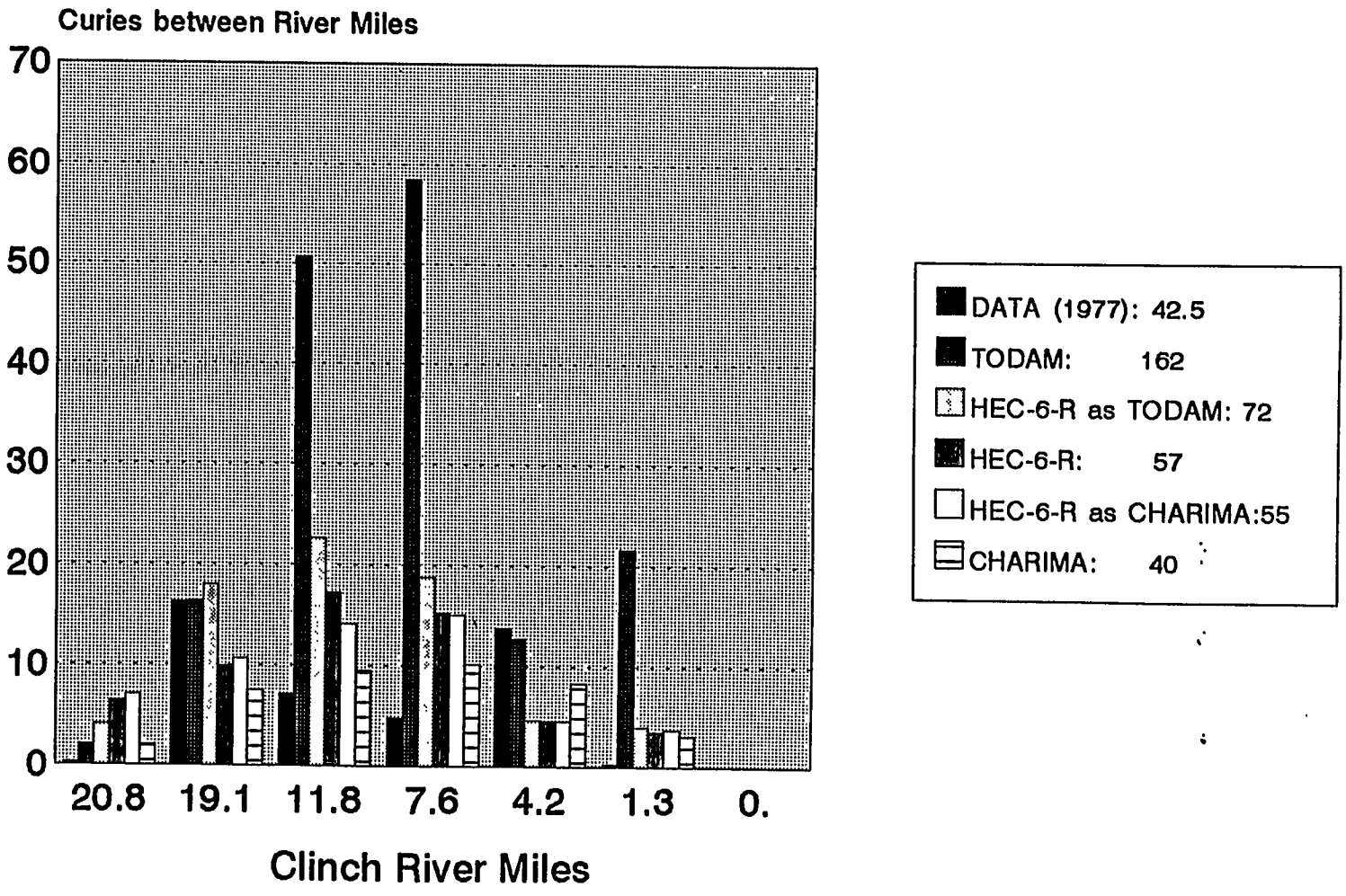


Fig. 3.47. Results of model simulations for curies within river reaches over different time periods.

HEC-6-R. 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, with 6.7 Ci accumulating in the Clinch within two weeks after onset of the storm. However, within 6 months the level in the Clinch decreases to a level lower than before the storm (i.e., the curies deposited in the Clinch 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 in the Clinch occurs 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 via 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 via radioactive decay.

Table 3.5. Clinch River storm responses with regard to ^{137}Cs in bed sediments

	onset	2 weeks	1 month	0.5 year	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

CHARIMA. 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. Again, some erosion occurs in the short term. CHARIMA predicts less of an increase in contaminant levels than HEC-6-R as a result of the local storm (i.e., only 3 Ci) 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 onset of the simulations) was lower than that of HEC-6-R, and its storm scenario response was, therefore, similarly muted.

TODAM. 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 what is 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, as compared to HEC-6-R and CHARIMA, TODAM assumes bed sediments are less erodable, which is explained by the assumption of generally coarser bed particle size distributions used in the TODAM model. The local storm scenario results differ in that a larger increase of ^{137}Cs occurs in the Clinch (+19 Ci) compared to HEC-6-R (+6.7 Ci) and CHARIMA (+2.5 Ci). 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, while both HEC-6-R and CHARIMA showed an immediate decrease due to erosion.

Conclusions. 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 following a system-wide, 100-year storm, despite increased contaminant flux from WOC, contaminant levels in the Clinch River will decrease in the short-term due to 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 a greater proportion of coarse sediment being transported, generally predicting 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.

3.4.5.2 Simulation of sediment transport in the Clinch River using a two-dimensional, finite-element model

Introduction. 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 based on data previously surveyed by TVA, ORNL, and U.S. Army Corps of Engineers. Grids are formed by connecting the nodes into triangular or quadrilateral elements. Grid generation was performed using 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).

Module calibration. The RMA2 hydrodynamic module was calibrated using 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

$$Y = 17.4583 + 0.92286X$$

where Y is the observed stage and X is the simulated stage. The regression R^2 was 0.9892, implying good correspondence between simulated and observed values. The simulation results also showed that reverse flow occurred several times in November 1993 around the Kingston Steam Plant.

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 approximately 90% of ^{137}Cs released was adsorbed to suspended sediments (Churchill et al. 1965). Simulated sediment transport was deemed realistic based on predicted deposition locations of clay, silt, and sand being similar to qualitative patterns of these sediment classes observed in the Clinch River (Levine et al. 1994).

Model scenarios. Six simulations using the coupled hydrodynamic and sediment transport modules were performed. The six simulations involve 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 sediment introduced that deposited in the WOC Embayment (WOC dam to coffer dam), upper Clinch River region (between WOC Embayment and Poplar Creek), and lower Clinch River region (between Poplar Creek and 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).

Results and conclusions. Predicted percentages deposited or exported out of the system for the six simulations after 30 days are presented in Table 3.6.

The following conclusions can be drawn from the simulation results:

(a) As expected, under the same flow conditions, clay was transported further downstream than silt.
(b) A higher percentage of sediment was retained in WOC Embayment under baseline conditions compared to the local and system-wide storm scenarios; however, over 70% of the clay was exported out of the system under baseline conditions.
(c) No significant deposition occurred in the lower Clinch River in any of the simulations.

(d) Significant deposition only occurred in the upper Clinch River region under the local storm scenario.
(e) Simulations using clay and the system-wide storm using silt resulted in significant export (>27%) of sediments out of the Clinch River and into lower WBR.

Table 3.6. 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 the White Oak Creek under baseline, local storm, and system-wide storm conditions

	Clay (%)			Silt (%)		
	Baseline	Local storm	System storm	Baseline	Local storm	System storm
WOCE	29.36	4.27	4.63	85.38	39.77	46.06
Upper CR	0.00	68.17	0.53	0.00	48.50	25.97
Lower CR	0.00	0.00	0.00	0.00	0.00	0.00
Out of CR	70.38	27.57	94.65	14.62	11.73	27.92

3.4.6 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, and 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 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 into McCoy Branch has resulted in elevated concentrations of several metals (As, Se, and V) in sediment of the 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 contamination, have average ^{137}Cs concentrations that are lower than sediment accumulation zones in and along the river channel. Mercury is also elevated in the Clinch River downstream of Poplar Creek, where peak concentrations are found coincident with those of ^{137}Cs due to 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 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 as compared to upper subreaches of Poplar Creek, possibly a result of backflow from the Clinch River.

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 as compared to the appropriate reference tests, and no meaningful pattern of toxicity could be discerned.

One-dimensional sediment transport modelling 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 via 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 subsequent scouring and transport downstream of these recently deposited sediments would occur within 6 months. A regional 100-year storm would result in the immediate scour and transport downstream of existing contamination, such that levels decline immediately as a direct result of the storm.

3.5 CHARACTERIZATION OF BIOTA

The characterization of biota consists of five interrelated tasks. The fish contaminant study (Sect. 3.5.1) characterizes the nature and extent of organic and heavy metal contaminants in key fish species of the CR/PC system. An analysis of contaminants in the aquatic foodweb is presented in Sect. 3.5.2. Section 3.5.3 investigates contaminant dynamics and biological effects of other important biota of the CR/PC system such as mink and the great blue heron. The biological indicators component (Sect. 3.5.4) and the reproductive effects study (Sect. 3.5.5) are designed to evaluate the effects of contaminants and other environmental stressors on fish health and fish reproductive function, respectively. Information from these five tasks are used not only to characterize the nature and extent of contamination in biota and to evaluate the effects of this contamination on important ecological species, but they also provide information that is used in the human health and ecological risk assessments.

3.5.1 Contaminants in Sportfish

3.5.1.1 Historical studies of contaminants in fish

Clinch River Study. Fish were collected from the Clinch River downstream of ORNL during the period 1960–1963 and analyzed for several radionuclides for the purpose of estimating potential risk to humans via 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.

Results are presented in Table C1. Bottom feeders were analyzed in two ways: flesh only and as whole fish. Results are presented for each bottom feeder species. Sight feeders were analyzed flesh only; because of limited sample size, no interspecific or year-to-year results are 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 in effect at that time (Cowser and Snyder 1966). The authors noted that approximately 45% of the ^{90}Sr and 20% of the ^{137}Cs measured in fish tissues were attributable to atmospheric fallout from nuclear tests.

TVA Instream Contaminant Study. In preparing for this study, TVA (1983) conducted an extensive review of existing data. These data led TVA to expect levels of mercury, chromium, nickel, and PCBs in Clinch River and Poplar Creek fish that were above background in at least one species (Tables C2 and

C3). In both the Clinch River and in Poplar Creek, mean mercury concentrations, while elevated, were below the Food and Drug Administration (FDA) action level of 1 mg/kg.

During the study, TVA (1985) collected fish (primarily bass and bluegill) from several sites in the Clinch River, one site in Poplar Creek, from several streams on the ORR, and from reference areas. TVA concluded that the 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). Considering all species combined, mean mercury concentrations in Poplar Creek and the Clinch River were elevated above background but were below the FDA action level of 1.0 mg/kg, confirming the historical data.

Organic contaminants other than PCBs were detected only 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 relative to catfish from Melton Hill Reservoir. Mean concentrations (<1.0 mg/kg) in the Clinch River were below the FDA action level of 2.0 mg/kg (Table C4).

Of several radiological parameters that were measured, only gross beta and ^{137}Cs appeared elevated in Clinch River fish as compared to historical values for fish from the Tennessee River (Table C5).

CRRI Phase 1. Three fish species (bluegill, channel catfish, and largemouth bass) were collected from locations in Watts Bar, Melton Hill, and Norris reservoirs and analyzed for metals, organic compounds, and radionuclides (Cook et al. 1992). Compared to the Norris 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.

TVA Fish Tissue Studies. The TVA monitors contaminant concentrations in fish tissue throughout the Tennessee Valley. Elevated levels of PCBs have been found in fish from Fort Loudon, Watts Bar,and Melton Hill reservoirs since the late 1980s. As a result, the TDEC (1992) 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 data collection efforts have focused on PCBs in catfish, results for which are summarized in Figs. 3.48–3.49.

3.5.1.2 Introduction

Both water-soluble and particle-reactive contaminants can accumulate in WBR biota. The extent to which this has occurred in WBR fish has been under investigation since the mid 1980s (TVA 1985; Dycus and Hickman 1988, 1989, 1990; Meinert 1991; Cook et al. 1992; Meinert and Fehring 1992; Fehring and Meinert 1993). The upper Tennessee River arm of the reservoir was the first area of the reservoir to be investigated, but eventually, investigators realized that a problem extended throughout the reservoir. Phase 1 of the CRRI identified seven contaminants of concern to human health in fish — As, Be, Hg, Se, PCBs (Aroclors 1254 and 1260), and chlordane (Cook et al. 1992). Radionuclides occurred at relatively low levels in fish tissue and none were listed as a contaminant of concern via the fish consumption pathway. The ecological risk assessment did not identify any additional contaminants of concern.

The Phase 1 sampling and analysis left many questions unanswered, however. Some important sport species (i.e., largemouth bass, *Micropterus salmoides*, and striped bass, *Morone saxatilis*) were

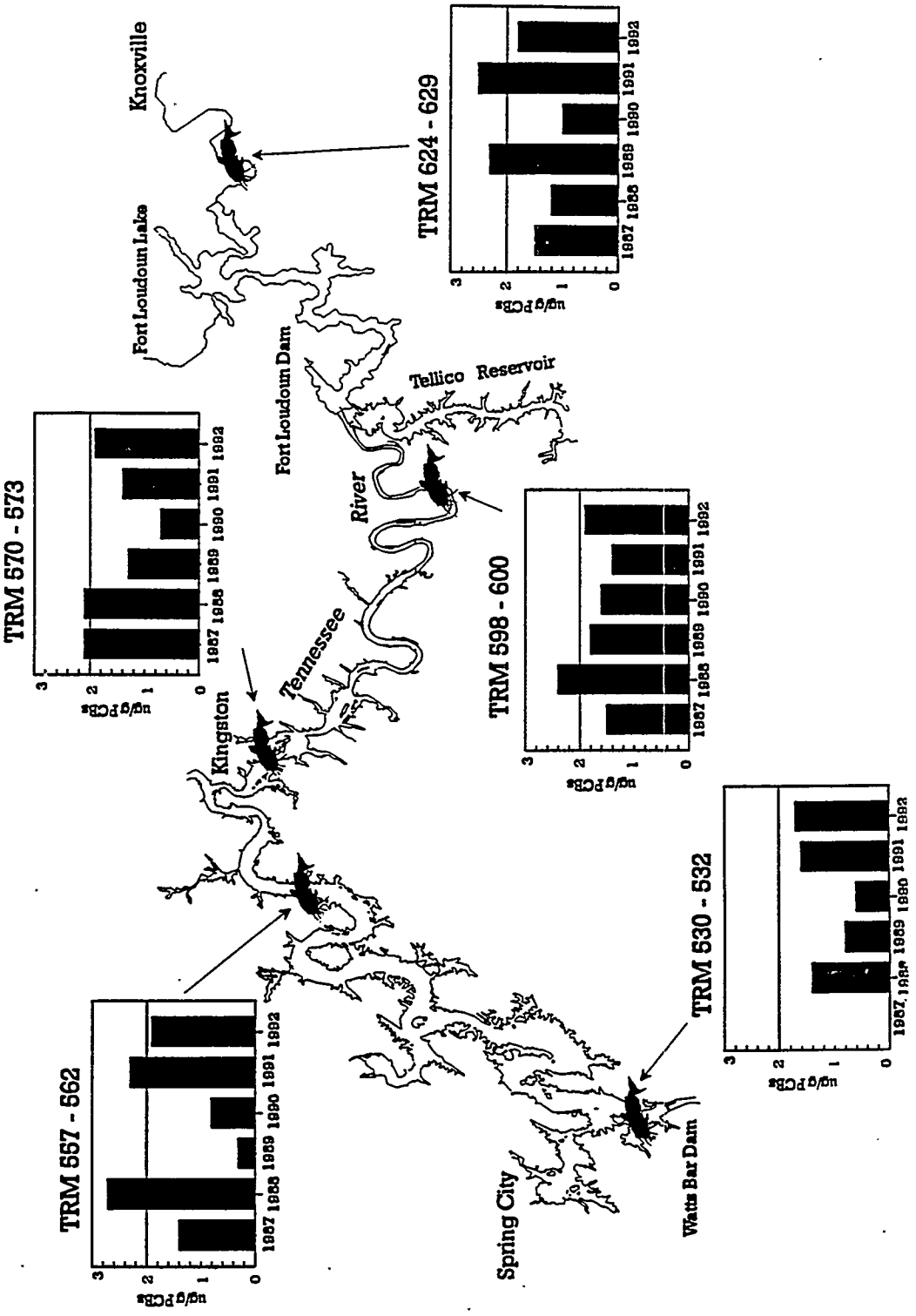


Fig. 3.48. Mean total PCB concentrations (mg/kg) in catfish at various sites throughout the Tennessee River section of Watts Bar Reservoir from 1987 to 1992.

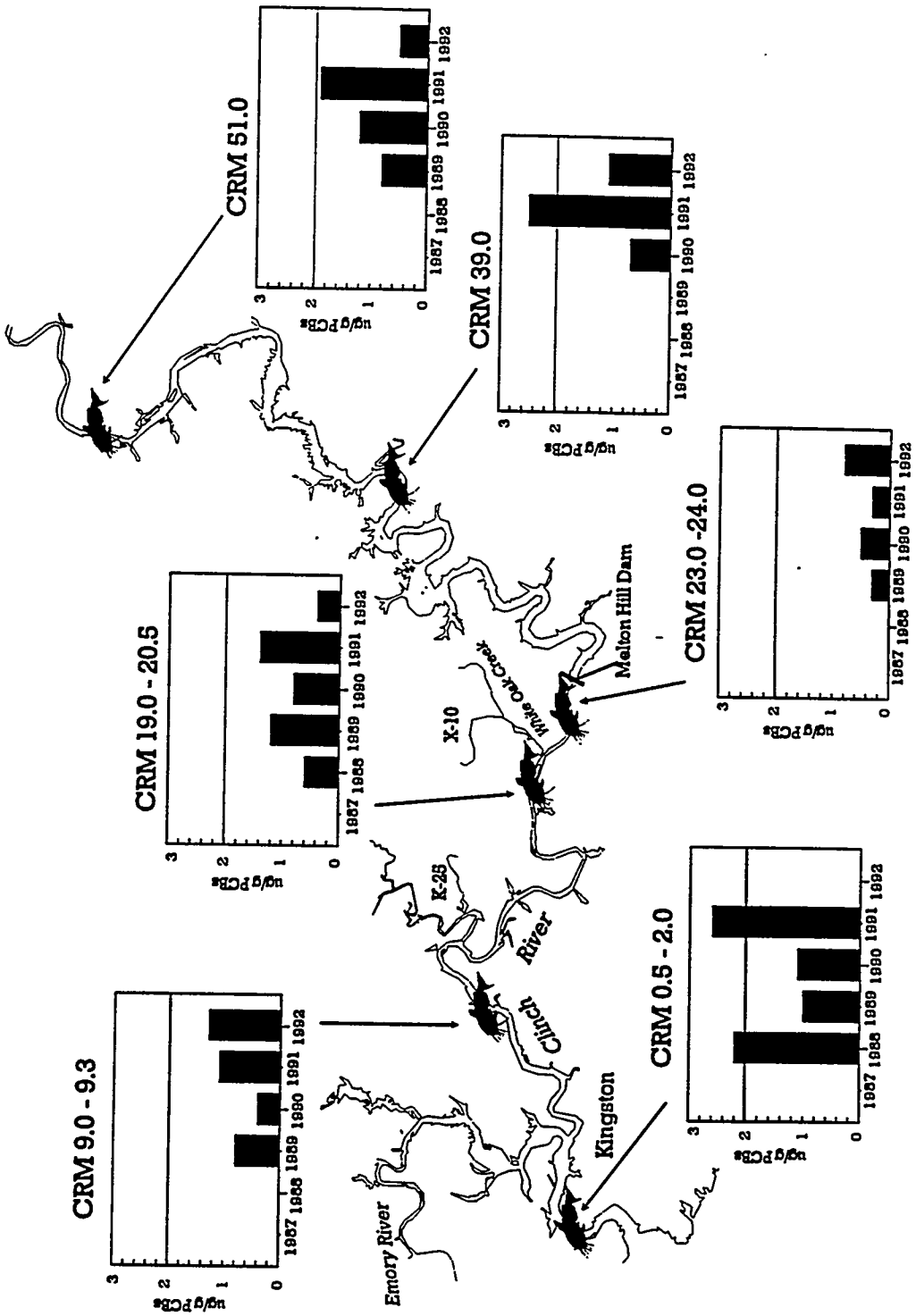


Fig. 3.49. Mean total PCB concentrations (mg/kg) in catfish at various sites throughout the Clinch River arm of Watts Bar Reservoir and Poplar Creek embayment from 1988 to 1992.

either not sampled or sampled in inadequate numbers to address among- and within-site variability and differences among species. Possible temporal trends were difficult to identify given the single collection period (fall 1989) and the limited historical data set. In addition, a detailed evaluation of the PCB problem was not possible since PCB concentrations in fish tissue were expressed as mixtures of several PCBs (e.g., Aroclor 1260 or 1254) and not as individual PCB congeners. For these and other reasons, the sampling and analysis of fish were continued during Phase 2 of the CRRI.

The primary objective of this task was to determine the extent to which key fish species in the CR/PC OU are contaminated by past and present releases from the ORR. This information will be used in the human health and ecological risk assessments and will also provide supportive data for other biota tasks. Key fish species from selected sites throughout the CR/PC system, including reference areas, were sampled and analyzed for contaminants identified in Phase 1 as those of potential human health or ecological concern. This sampling, while concentrated in the Poplar Creek embayment and the Clinch River arm of WBR, also included sites in the Tennessee River and Norris Reservoir for comparison.

3.5.1.3 Sampling and analysis

Several species of fish were collected from 1989 to 1994 at sites throughout WBR and from upstream "reference" reservoirs. Species were selected based on angling popularity, desirability for consumption, contaminant assimilation properties, and diversity (e.g., different sizes and trophic levels). Catfish (channel, *Ictalurus punctatus*, and blue, *I. furcatus*) are a popular food fish particularly for subsistence fishers and are omnivorous bottom-dwelling fish with relatively high lipid content. Fish with higher lipid contents generally accumulate higher concentrations of hydrophobic contaminants such as PCBs. Largemouth bass are an extremely popular sport fish and are mid-sized piscivores with a somewhat limited home range. Striped bass (including the striped bass X white bass hybrid, *Morone saxatilis* X *M. chrysops*), popular as a sportfish but less so as a food fish, are large piscivores that are known to range throughout the reservoir on a seasonal basis (Cheek et al. 1983). Bluegill sunfish (*Lepomis macrochirus*), also a popular food fish, are small invertivores with a relatively small home range. The contaminant body burden of fish with small home ranges should be more representative of conditions at the capture location compared to fish with larger home ranges. A limited number of carp (*Cyprinus carpio*) and white crappie (*Pomoxis annularis*) were also collected. The lengths of fish collected were within the range of the typical harvestable size for each species.

During Phase 2 (1991–1994), fish were collected by electroshocking, gill-netting, and trapping (slat-baskets) at 13 sites in WBR, at two reference sites in Norris Reservoir, in Melton Hill Reservoir, and in Chickamauga Reservoir (Fig. 3.50). Sites were selected such that all areas of the reservoir were represented, particularly those close to potential sources of contamination. Bluegill and largemouth bass were collected in the spring, while the other species were, with a few exceptions, collected in the fall.

Fish were usually analyzed as individual samples, but in some cases, five fish were combined into a composite sample. Fillets were removed from all sportfish species for analysis. Catfish were skinned, while other species were scaled and analyzed with the skin on. Samples were homogenized and analyzed by standard analytical methods identified in the sampling plan (DOE 1993). Catfish, largemouth bass, striped bass (which includes some hybrids), and a single composite sample each of carp and white crappie were analyzed for PCBs (Aroclor mixtures and individual congeners) and pesticides (primarily chlordane). Bluegill, largemouth bass, and a few catfish were analyzed for several inorganics (As, Be, Cu, Hg, and Se). In addition, catfish at one site were analyzed for a suite of semivolatile organics to confirm the general absence of this class of contaminants as documented during Phase 1.

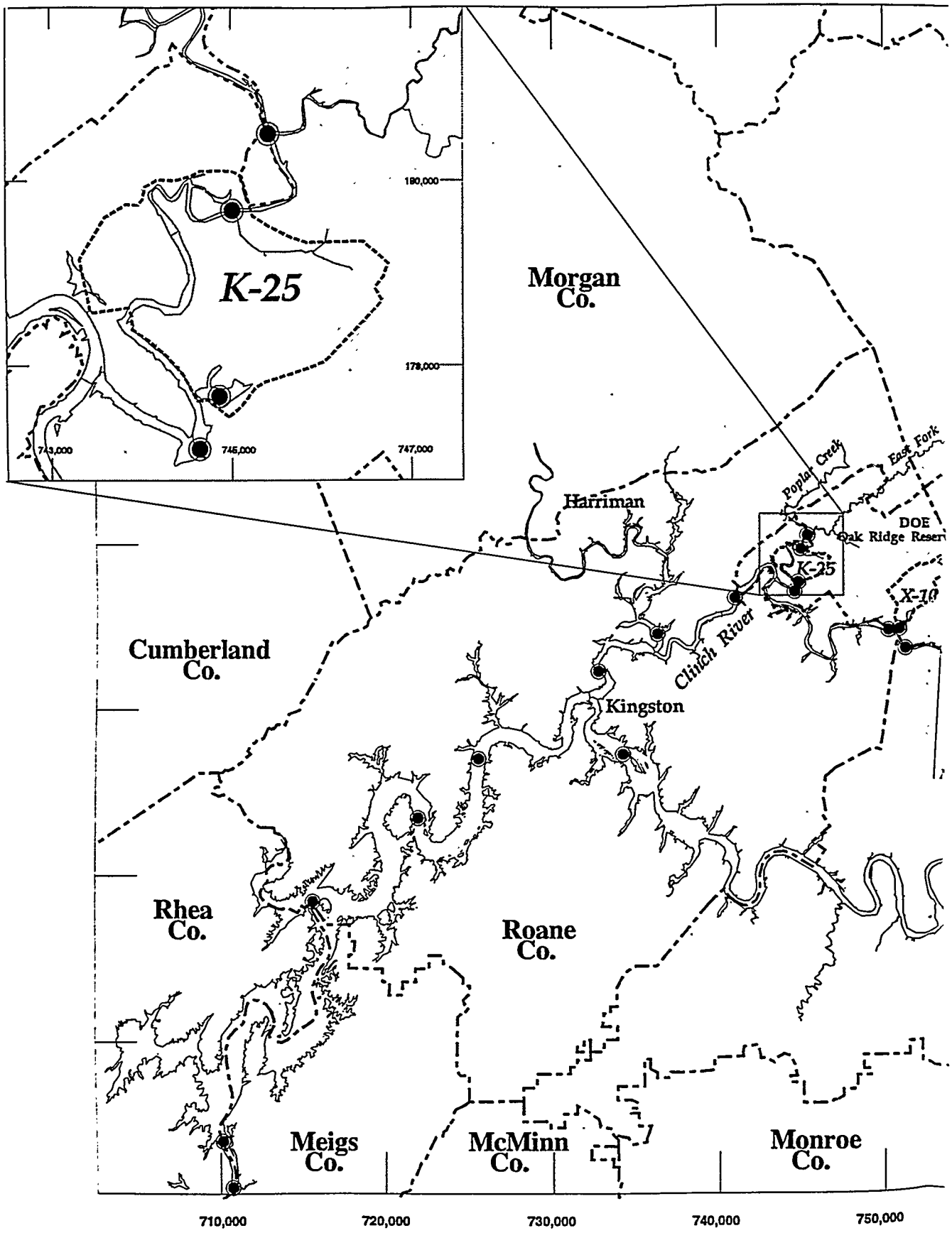
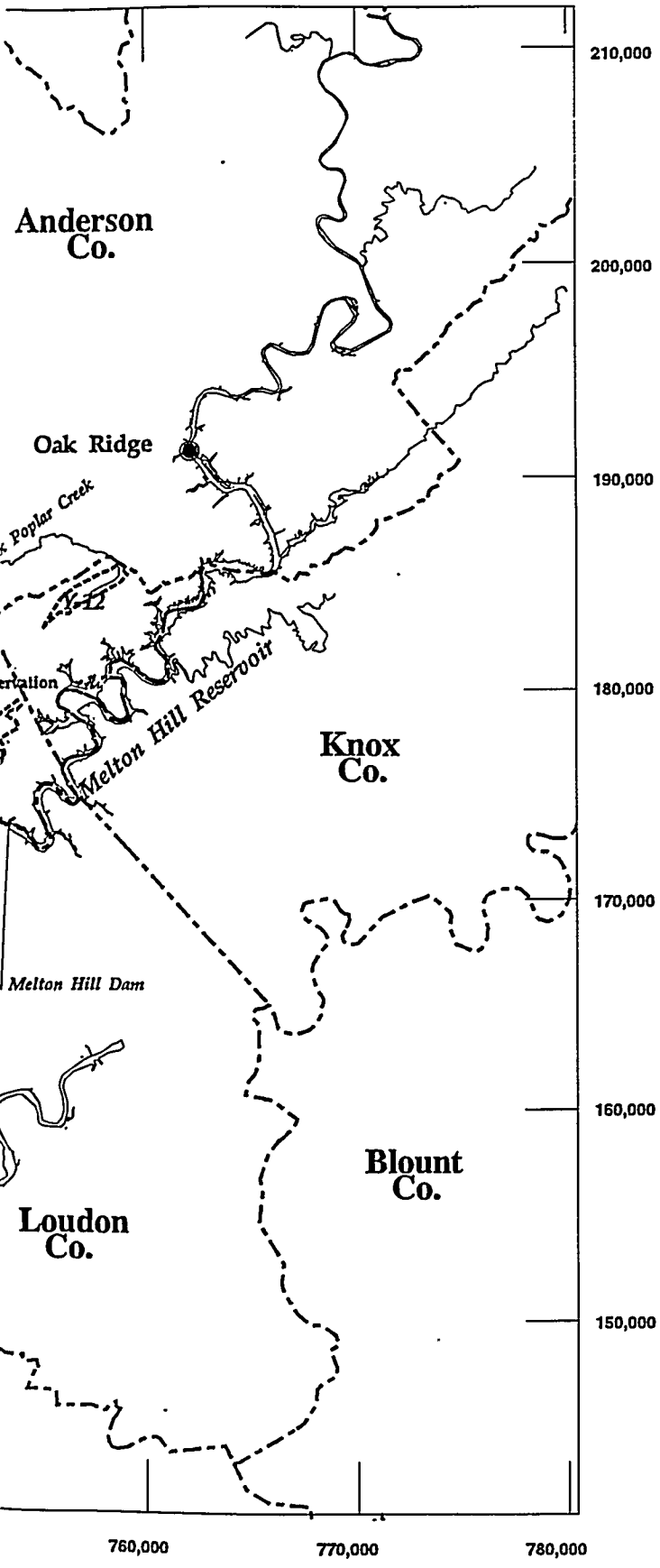
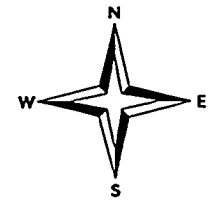


Fig. 3.50. Sampling locations for characterization of biota.



Clinch River Environmental Restoration Program

Fish Sampling Locations for Phase 2



LEGEND

◉	Sample Location
▭	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 Clinch River Environmental Restoration Program

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An unbalanced analysis of variance (GLM procedure, SAS 1989) was performed to evaluate the effects of sampling location, year, and species on contaminant concentrations. Composite samples were not included in the statistical analysis, but results from the composite samples will be presented and discussed when relevant.

3.5.1.4 Results

Inorganics

The analysis of variance (ANOVA) model included sampling site, species, and year as effects variables and contaminant concentration as the dependent variable. Scheffe's multiple comparisons test was used to identify significant differences between main effects means (e.g., between sites or species) when a significant ($\alpha<0.05$) main effect was observed. The mean concentrations for five organics and the incidence of detection are reported in Table C6 for each site-by-species combination.

Mercury was detected in 55, 76, and 87% of the 159 bluegill, 25 catfish, and 135 largemouth bass samples, respectively, collected from WBR (reference sites in Norris not included) (Table C6). Seven largemouth bass, all from Poplar Creek, had concentrations that exceeded the FDA action level of 1.0 mg/kg. This represents about 24% (7 of 29) of the largemouth bass collected from Poplar Creek in 1993 and 1994.

The results of mercury analysis on bluegill, largemouth bass, and catfish are summarized in Fig. 3.51. The ANOVA indicated a significant site effect with concentrations highest in Poplar Creek embayment as expected given its proximity to the primary source of mercury to the system (i.e., the Y-12 facility and EFPC) (Table C7). Poplar Creek sites had significantly higher concentrations than the other sights, with the highest at PCM 5.3, near the mouth of EFPC. Slightly elevated levels were also found at CRM 10, the first site downstream of the mouth of Poplar Creek in the Clinch River arm of the reservoir. The ANOVA also indicated a significant species effect with concentrations in bass greater than in catfish, which in turn were greater than bluegill.

Arsenic was detected in 36, 24, and 97% of the 159 bluegill, 25 catfish, and 135 largemouth bass, respectively, collected from WBR (Table C6). The ANOVA indicated a significant species effect (Table C7). Concentrations in largemouth bass were significantly greater than in both catfish and bluegill, which were nearly the same. A significant site effect is also indicated, but given the significant interaction between site and species effects, the site effect should be considered with some caution. No differences among sites were determined with the Scheffe's multiple comparisons test.

Beryllium was detected in 24, 16, and 33% of the 159 bluegill, 25 catfish, and 135 largemouth bass samples, respectively, collected from WBR (Table C6). The ANOVA indicated significant site, species, and year effects in the beryllium data, but highly significant interaction effects suggest that some caution should be taken in the interpretation of these results (Table C8). In general, catfish had higher concentrations than both bluegill and largemouth bass, which were similar.

Copper was detected in 36, 44, and 23% of the 77 bluegill, 25 catfish, and 62 largemouth bass samples, respectively, collected from WBR (Table C6). Copper was tested at a few select sites during Phase 2 because a small number of fish from WBR had high copper levels during Phase 1 (i.e., 3 of 110 fish exceeded 2 mg/kg). In Phase 2 samples, 6 of 164 fish had levels which exceeded 2 mg/kg. None of the main effects tested with the ANOVA were significant for copper.

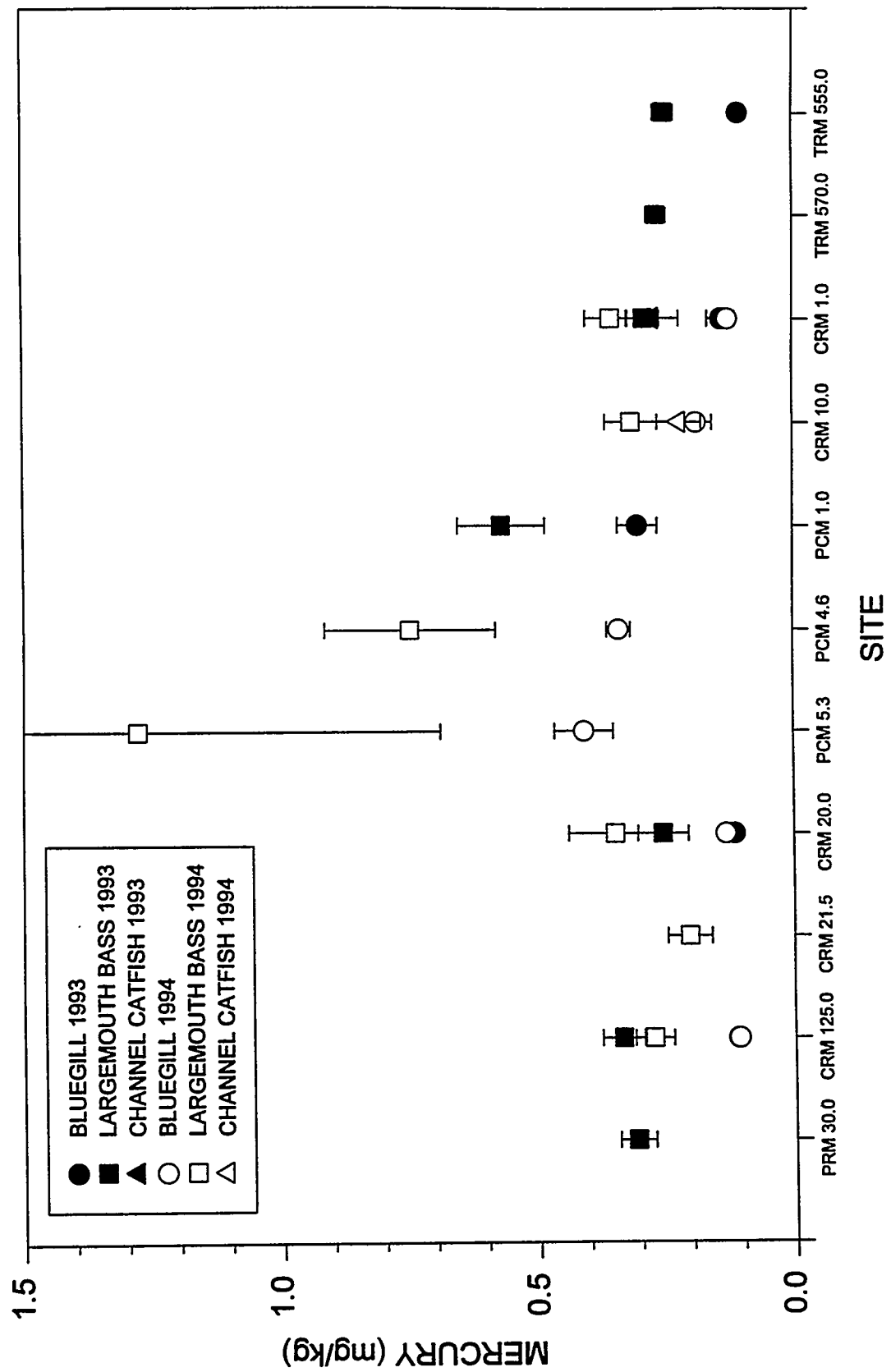


Fig. 3.51. Mercury in largemouth bass, bluegill, and catfish fillets collected from Watts Bar Reservoir and reference sites during Phase 2 sampling, 1993–1994.

Selenium was detected in 94, 48, and 87% of the 159 bluegill, 25 catfish, and 135 largemouth bass samples, respectively, collected from WBR (Table C6). The ANOVA indicated significant site and species effects, but also significant interaction effects. The three sites with the highest concentrations (PRM 30.0, PCM 5.3, and CRM 21.5) are not near each other, suggesting that surface water sources of selenium are multiple or that the primary source is atmospheric.(Table C6). The ANOVA indicated significant site and species effects, but also significant interaction effects. The three sites with the highest concentrations (PRM 30.0, PCM 5.3, and CRM 21.5) are not near each other, suggesting that surface water sources of selenium are multiple or that the primary source is atmospheric.

PCBs/Pesticides

PCBs were reported as both Aroclor mixtures and individual congeners for most samples. As initially manufactured, each Aroclor mixture (e.g., 1254 or 1260) included different combinations of the 209 possible individual PCB compounds or congeners. The higher numbered Aroclors contain a larger proportion of the highly chlorinated congeners that are suspected to be generally more toxic. Due to the degradation and transformation of PCBs in the environment as a result of natural biological processes, what the analytical laboratory quantifies as a specific Aroclor mixture likely has a substantially different composition from the Aroclor as originally manufactured. Although the amount of information from a toxicological standpoint contained in an Aroclor value is less than that obtained in measures of individual congeners, Aroclor concentrations were included in this investigation for two reasons—(1) to maintain comparability with prior years of PCB data for WBR and (2) risk assessment factors have not yet been developed for individual congeners. The analysis of individual PCB congeners was initiated to provide additional information regarding (1) the relative contribution of the most toxic PCBs to the total PCB mixture, (2) differences among species, (3) temporal variability, and (4) site differences and similarities relative to possible sources.

Analyses were conducted for PCB Aroclors 1016, 1221, 1232, 1242, 1248, 1254, and 1260, but only Aroclors 1254 and 1260 were detected (Table C6). Excluding samples collected from Norris Reservoir (CRM 125 and PRM 30), where PCB levels are low and the incidence of non-detects is high, Aroclors 1254 and 1260 were detected in 96% of the samples. The 1254 and 1260 values were summed (with non-detects assigned a value of one half the detection limit) to estimate the total PCB concentration.

The FDA action level for PCBs of 2.0 mg/kg was exceeded by only one sample mean (catfish at CRM 1 in 1991), but was exceeded in 48 individual fish from the Clinch River (23 catfish), Poplar Creek (6 catfish), the upper Tennessee River (6 catfish), the Emory River (1 catfish), and lower Watts Bar (12 catfish and 1 striped bass). Mean total PCBs exceeded 1.0 mg/kg in 15 instances (Table 3.7).

Table 3.7 CRRI sampling events and locations in which mean total PCB concentrations in catfish exceeded 1.0 mg/kg

Site	Catfish	Striped bass	Largemouth bass
CRM 20	1993		1994
PCM 1.0	1992		
ERM 1	1993		
CRM 1	1991–1993	1992	
TRM 570	1991–1992		
TRM 561	1991		
TRM 545	1992	1992	
TRM 530	1991, 1993		

An unbalanced ANOVA was performed to evaluate the effects of sample location, species, and year on the total PCB concentration in fish (Table C8). The results suggest significant site and species effects, but significant interaction effects cast some uncertainty on this conclusion. Figure 3.52 clearly indicates that mean PCB levels in WBR catfish are higher than those at the two reference sites in Norris Reservoir. Several sites downstream of the ORR (such as CRM 20, PCM 1, CRM 1, TRM 561) have elevated levels of PCBs, as do two sites (TRM 570 and ERM 1) located upstream of the influence of ORR operations, suggesting there are numerous sources of PCBs to WBR. Similar among-site differences and similarities are seen in the data for the other species (Figure 3.53).

The among-species differences identified in the ANOVA (Table C8) are confirmed by a closer look at the means for each species by site and year. A comparison of mean PCBs among catfish and the other species at sites where both were sampled suggests catfish are likely to have among the highest PCB concentrations in the fish community. In 13 cases where both largemouth bass and catfish were sampled at a given site in the same year, the largemouth bass mean PCB concentration ranged from 19 to 84% that of catfish (mean=37%). In 3 cases where both striped bass and catfish were sampled, the striped bass mean PCB concentration ranged from 54 to 105% that of catfish (mean=78%). A composite sample of five carp at CRM 1 had a PCB concentration that was 45% of the mean concentration of catfish collected at the same time and place. Similar composites of bluegill and white crappie had PCB concentrations that were 38 and 17%, respectively, of the concentration in catfish. Some of the variation among species may be explained by differences in lipid content of the fish although there are obviously other factors as well. The average lipid content of the sportfish collected during Phase 2 was 9.7% for striped bass, 5.9% for catfish, 1.7% for largemouth bass, 6.9 for bluegill, 24.0% for carp, and 0.6% for white crappie.

No year effect was found in the ANOVA (Table C8) even when the analysis was performed on catfish data only. There appear to be significant year-to-year differences within some sites, but these trends are 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-laden sediments, or changes in the aquatic community and food-chain dynamics. An evaluation of PCB data collected since 1987 by several agencies shows no consistent trends in PCB concentrations throughout WBR (Figs. 3.48 and 3.49).

PCB congeners were measured in samples collected from fall 1992 through spring 1994. Forty-four different congeners were selected for analysis based on their occurrence in other aquatic-related samples (water, sediment, or biota), from other studies throughout the country, or on their supposed high toxicity. The list included those congeners with a co-planar configuration that are geometrically similar to dioxin molecules and are suspected to be more toxic than many of the other PCB congeners.

All of the 44 congeners were found in at least one of the samples from the CR/PC OU (Table C9). Twenty congeners were found in more than 50% of the samples, and eight were found in more than 90% of the samples. Those congeners with the highest mean concentration across all samples were PCB numbers 153, 180, 138, 118, and 99 (Table C9).

One reason for analyzing congeners was to provide more detailed information on those congeners with suspected greater toxicity. The proportion of the total PCB (sum of the congeners) that can be attributed to coplaner congeners (PCB congeners 37, 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189) averaged about 27% for all samples from the CR/PC OU, with little variation among sites or species.

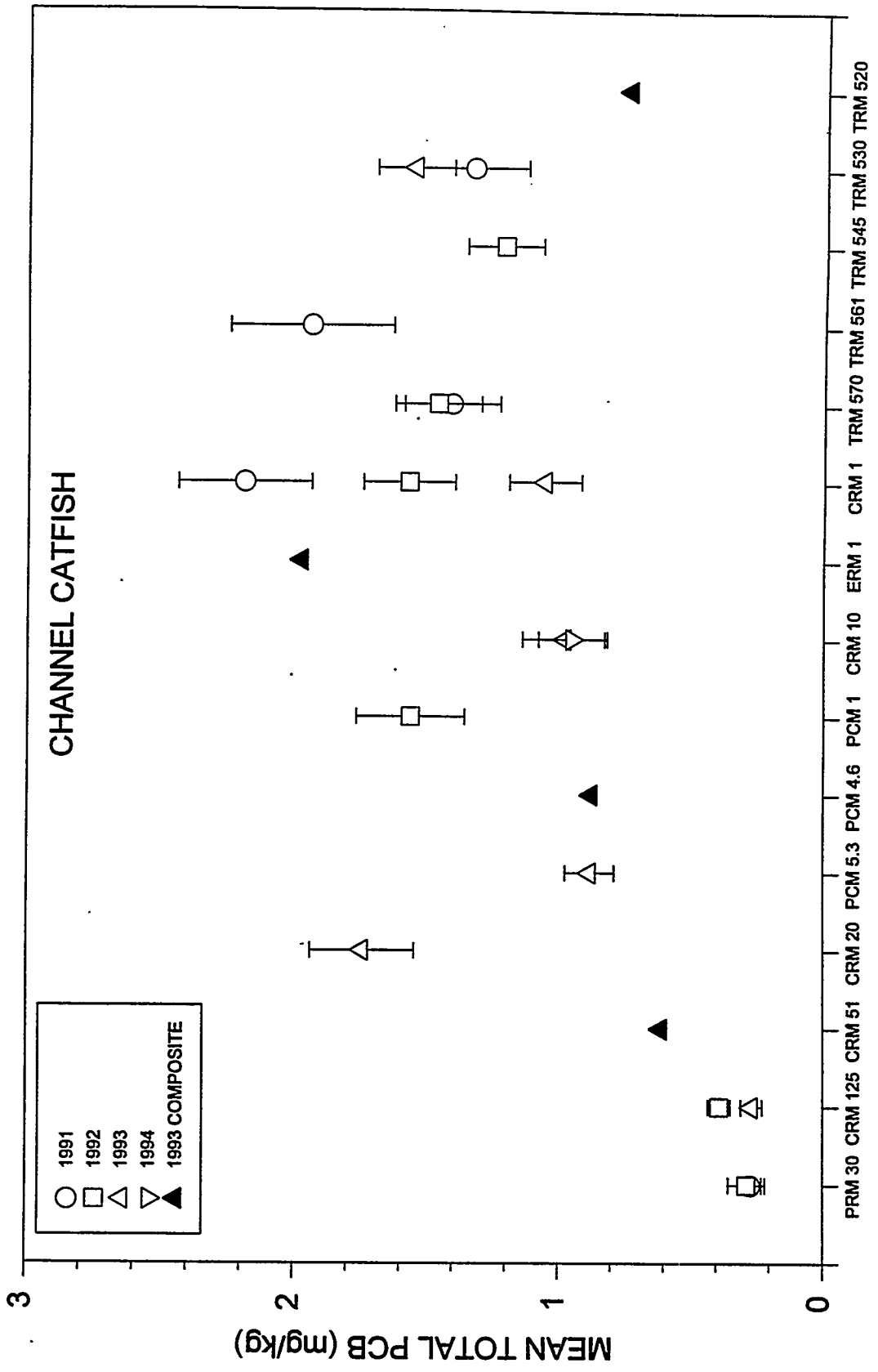


Fig. 3.52. Mean total PCB concentrations (mg/kg) in catfish collected from Watts Bar Reservoir and reference sites during Phase 2 sampling, 1991–1994.

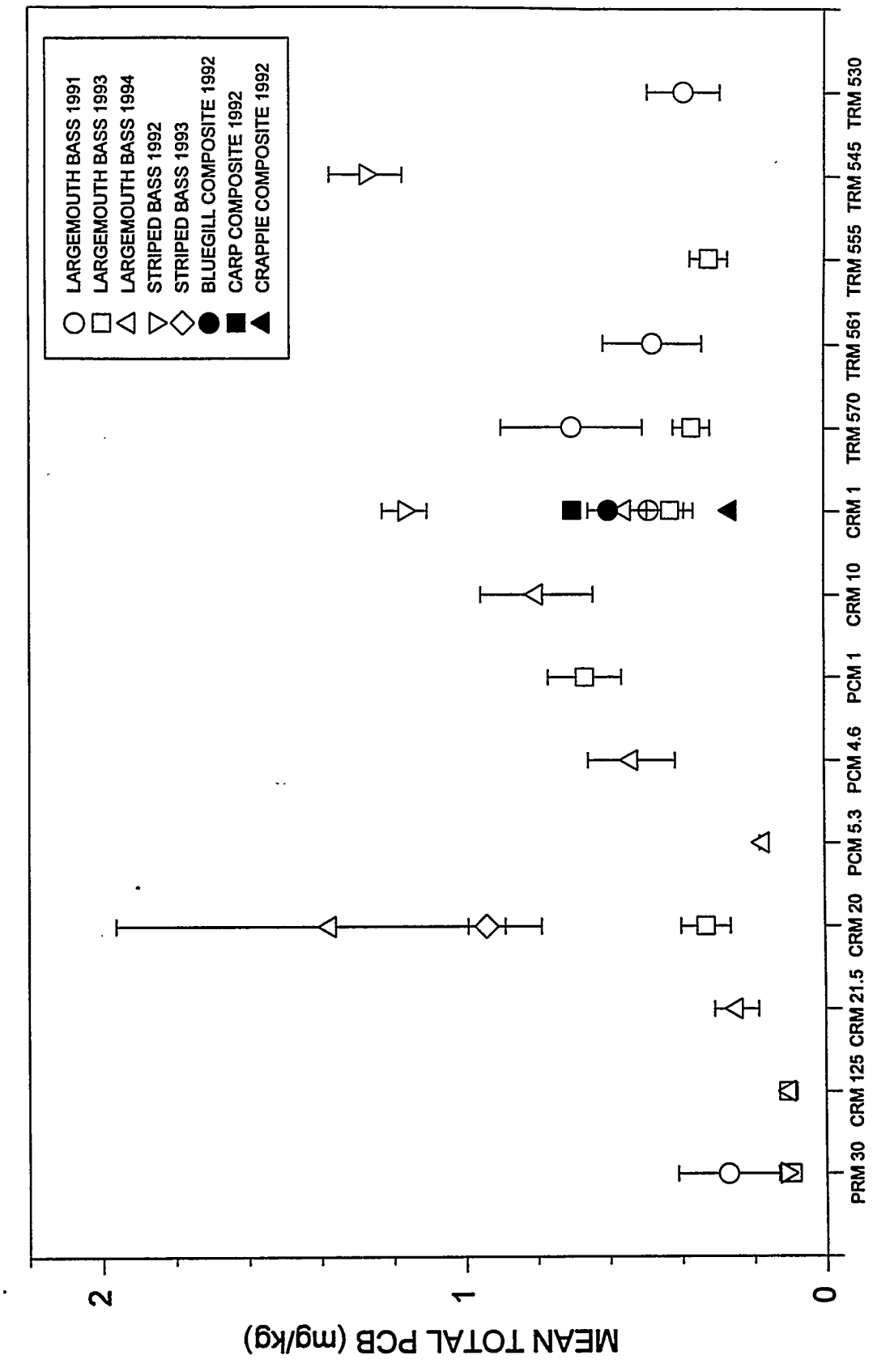


Fig. 3.53. Mean total PCB concentrations (mg/kg) in striped bass, largemouth bass, other species collected from Watts Bar Reservoir and reference sites during Phase 2 sampling, 1991–1994.

The temporal variation in PCB concentrations in WBR that has been observed in Aroclor and total PCB data over the last several years is also evident in the congener data. Table C10 shows how the incidence of detection of each congener changed from fall 1992 to spring 1994. For example, the detection of PCB congeners 28, 31, 99, and 170 decline noticeably during this time. Conversely, the detection of congeners 101 and 105 increased since the first sample in fall 1992. Some congeners, such as 153, had a high incidence of detection throughout the study, but showed a noticeable increase in concentration in catfish tissue in the second year of sampling (Fig. 3.54). This increase in the concentration of congener 153 was also observed in largemouth bass and shad samples. Figure 3.54 illustrates not only the year-to-year variation in congener concentrations (fall 1992 versus fall 1993), but also shows a marked difference between spring and fall samples of the same species from the same site.

Differences among species are also obvious in the congener data (Fig. 3.55). Differences in the magnitude of the contamination among species are similar to those observed in the Aroclor data, but there is little difference in the congeners present among the various species. Most of the congeners detected in those species with the highest PCB concentrations (i.e., catfish and striped bass) are also detected in the less contaminated species.

Total chlordane is calculated as the sum of six chlordane constituents (gamma-chlordene, alpha-chlordene, gamma-chlordane, cis-nonachlor, trans-nonachlor, and chlordane). Non-detects were not included in the sum. At least one of these constituents was detected (and thus a total chlordane value calculated) in 95% of the catfish, 95% of the striped bass, and 2% of the largemouth bass samples from WBR (Table C6). The incidence of detection for total chlordane was 85 and 77%, respectively, for catfish and largemouth bass at the reference site, Norris Reservoir.

The ANOVA indicated significant species and site effects, but also significant interaction terms (Table C8). The concentration of total chlordane in catfish is roughly twice that in largemouth bass. Site means for catfish in WBR ranged from 0.08 to 0.22 mg/kg, with the highest concentrations at CRM 20, CRM 1, TRM 570, and TRM 561 (Table C6). Largemouth bass means ranged from 0.06 to 0.085 mg/kg with little difference among sites; striped bass means were similar to catfish means, ranging from 0.10 to 0.14 mg/kg.

3.5.1.5 Semivolatile organics

During Phase 1, over a hundred samples were analyzed for a total of 65 semivolatile organic compounds. Many of the analyses were rejected due to quality control (QC) problems, but of those that were acceptable, very few had detectable levels of any compound. During Phase 2, the concentrations of semivolatile organics were measured in ten catfish from Poplar Creek to confirm these low levels. As in Phase 1, very few semivolatiles were detected (Table C6). Diethylphthalate, di-n-butylphthalate, and naphthalene were detected in three samples each, and bis(2-ethylhexyl)phthalate was detected in six.

3.5.1.6 Summary

The only metal for which a meaningful spatial pattern was identified was mercury; concentrations were highest in fish from Poplar Creek. However, mean values in Poplar Creek were below the FDA action level of 1 mg/kg for all species. Concentrations were highest in largemouth bass, second highest in catfish, and lowest in bluegill.

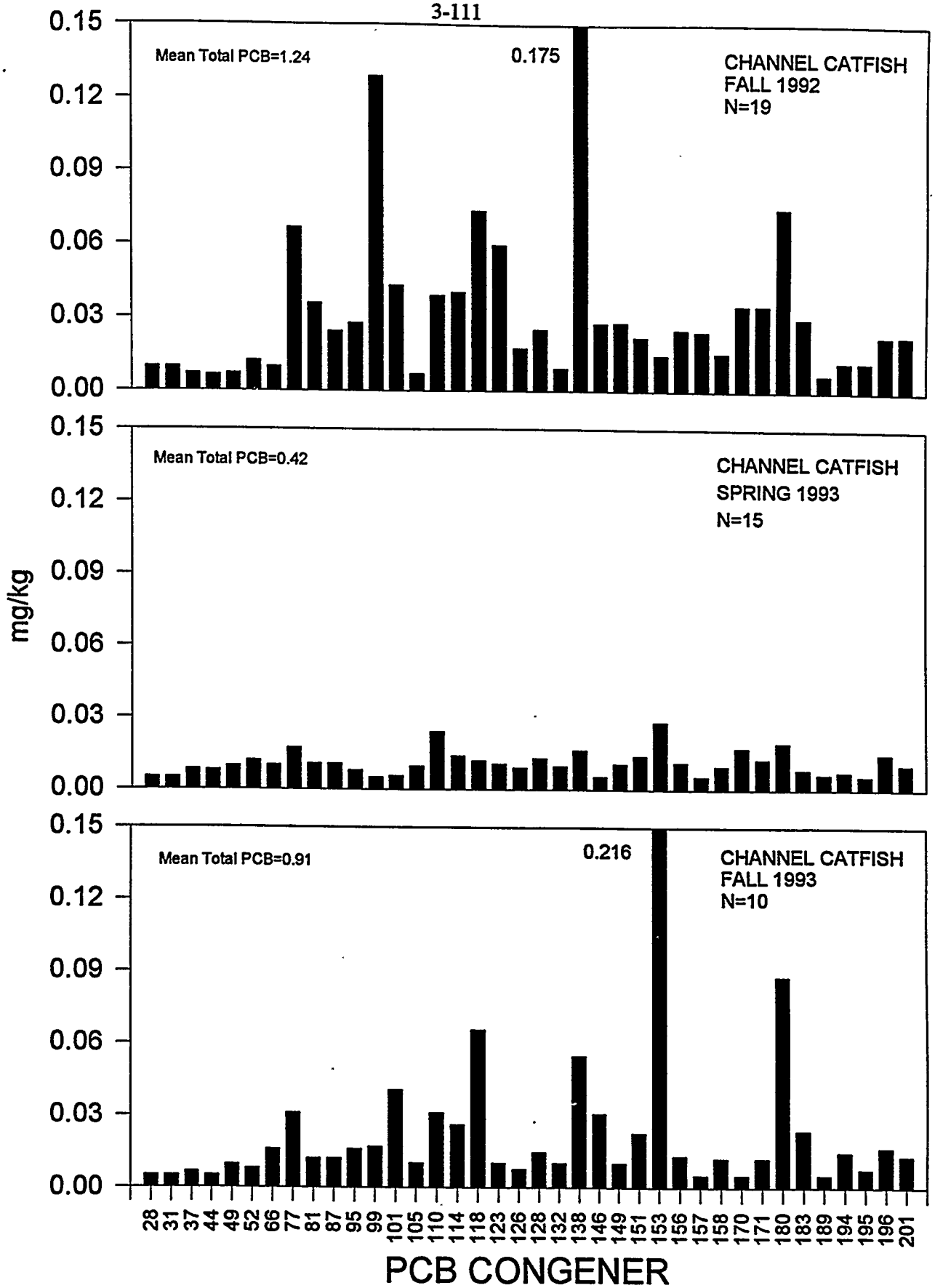


Fig. 3.54. Congener concentrations in catfish, fall 1992 to fall 1993.

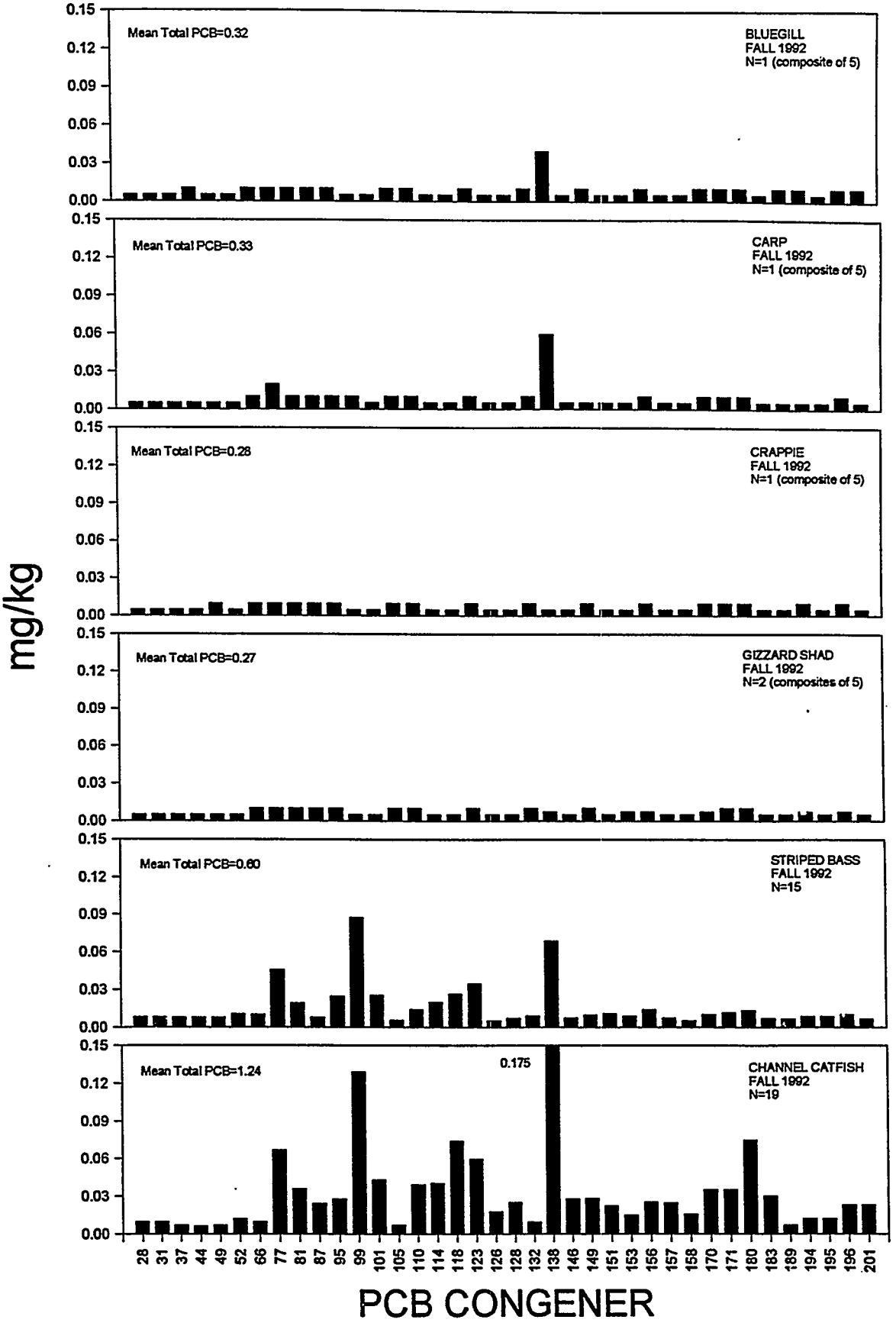


Fig. 3.55. Congener concentrations in sportfish.

Aroclors 1260 and 1254 were detected in 96% of all fish sampled from the study area (excluding the Norris reference site); no other Aroclors were detected in any sample. In only one case (catfish collected from CRM 1.0 in 1991) did the mean total PCB concentration exceed the FDA action level of 2.0 mg/kg. Concentrations were highest in catfish and lowest in bluegill and crappie. Spatial patterns suggest several sources of PCBs to the river system. No temporal pattern was evident, as mean values of both Aroclors and individual congeners varied widely over time.

Chlordane was detected in most fish from the study sites. Concentraions were highest in catfish, but mean levels were well below the FDA action level of 1.0 mg/kg. No meaningful spatial patterns were evident.

The analyses of catfish from Poplar Creek confirmed the expected low levels of other semivolatile organic compounds in fish flesh. The only compounds detected were several phthalates and one PAH, all of which are commonly detected in environmental samples.

3.5.2 Contaminants in the Food Web

3.5.2.1 Introduction

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. These data are also used in the ecological risk assessment.

3.5.2.2 Sampling and Analysis

Original plans called for the collection of crayfish and dragonfly larvae at three sites in WBR (DOE 1993). Due to their greater abundance and ease of collection, adult emergent mayflies (*Hexagenia* sp.) were collected instead of dragonfly larvae. Mayflies were collected at three locations—PCM 2, just upstream of the K-1007B pond outfall (1993 and 1994); CRM 1, at the Kingston hardware store (1993 and 1994); and TRM 570, at the Riley Creek recreation center boat dock (1993). Several dozen mayflies were composited to produce three samples from each site. The samples were analyzed for five metals (As, Pb, Hg, Ag, and Zn) and a suite of pesticides and PCBs. Several methods were attempted to collect crayfish in the reservoir, including baited traps and electroshocking, but none were successful.

Shad (gizzard, *Dorosoma cepedianum*, or threadfin, *D. petenense*) are most likely the primary food item for striped bass and largemouth bass in Watts Bar, but are not consumed by humans. Shad were collected at nearly all the same sites and times as were catfish and analyzed whole for PCBs/pesticides. Most (114; 72 individual and 42 composite samples) of the 119 shad samples were of yearling fish about 1.5 years old, though there were five samples that were composites of young-of-year fish about 0.5 years old.

3.5.2.3 Results and Interpretation

No pesticides (including DDT, DDD, DDE, aldrin, dieldrin, total chlordane, and others) were detected in mayfly samples, and of the PCB Aroclors, only 1254 was detected (Table C11). The mean values at all sites were less than 0.15 mg/kg, with concentrations in the Poplar Creek samples being slightly lower than the other two sites. Total PCB concentrations (with non-detects added at half the detection limit) in mayflies are less than in the fish species at the same sites except when compared to shad at TRM 570 in 1993 (Fig. 3.56).

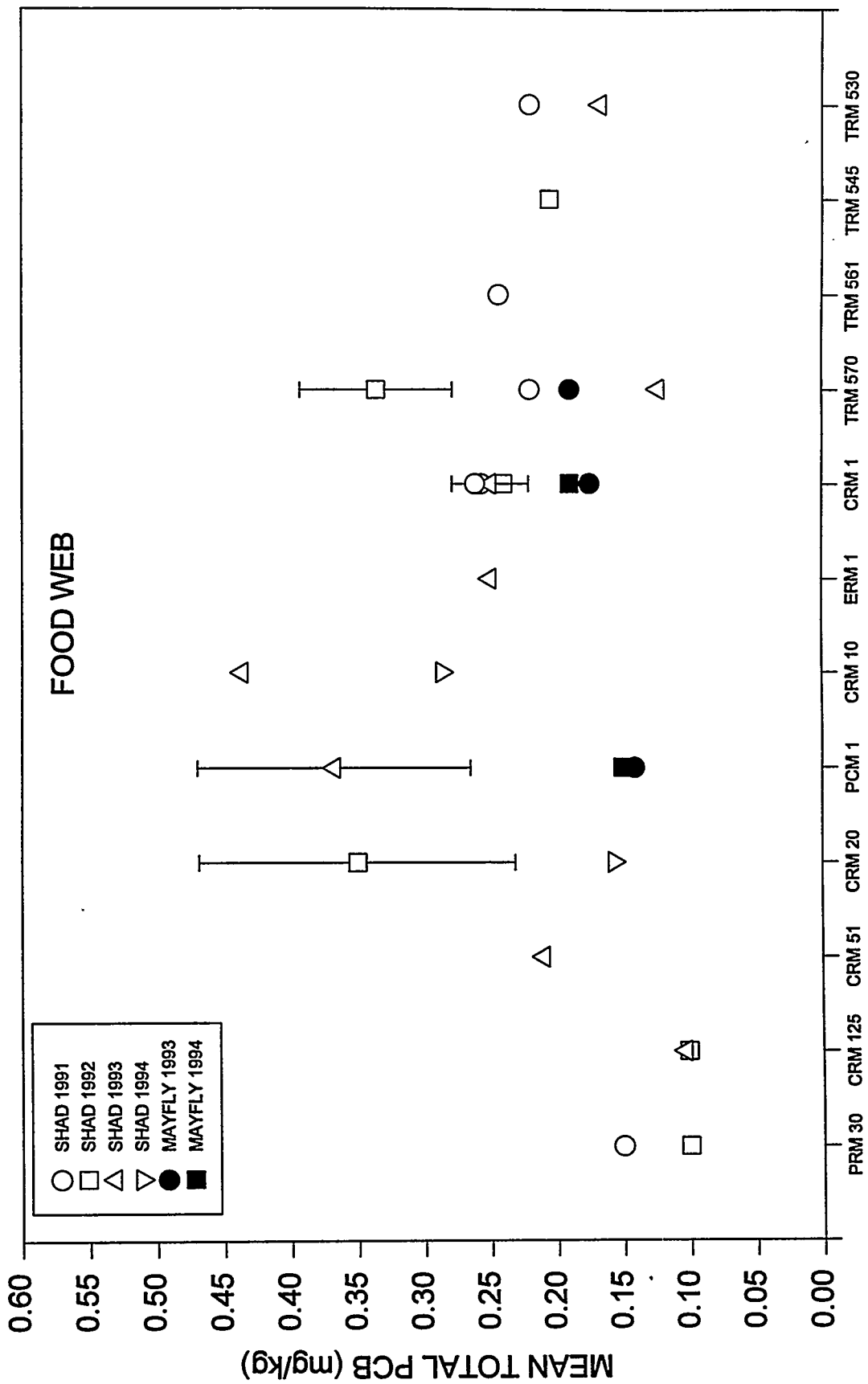


Fig. 3.56. Mean total PCB concentrations (mg/kg) in shad and mayflies collected from Watts Bar Reservoir and reference sites during Phase 2 sampling, 1991–1994.

Of the five metals for which mayflies were analyzed, As, Hg, and Zn were detected in all nine samples, Pb in seven of nine, and Ag in none (Table C11). Mercury concentrations in mayflies are roughly 10–20% of that found in bluegill fillets at the same sites and about 5–10% of that found in largemouth bass fillets (Table C6). Likewise, arsenic concentrations in mayflies are roughly 30 and 15% of the levels found in bluegill and largemouth bass filets, respectively. Zinc concentrations in mayflies are roughly 10 times greater than those found in fish during Phase 1 sampling, while lead was not detected at all in fish during Phase 1.

PCBs were detected in three of the five young-of-year composite samples. Total PCBs at CRM 1, TRM 561, and TRM 570 ranged from 0.13–0.19 mg/kg, while PCBs were not detected at PRM 30 and TRM 530.

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. 3.56). At only one site (PCM 1.0, 1992) did total PCBs exceed 1.0 mg/kg; the next highest mean was only 0.44 mg/kg. On average, shad PCB concentrations were roughly 25% that of catfish and striped bass fillets, and 70% that of largemouth bass fillets.

3.5.2.4 Summary

Mayflies were found to contain several of the contaminants found in the sediment and in fish (particularly, arsenic, mercury, zinc, and PCBs) and are likely to provide one of many mechanisms for transport of these contaminants into fish from the sediment. Shad (even young-of-year fish) had detectable levels of PCBs and are likely the most important pathway for PCB contamination of piscivorous fish.

3.5.3 Contaminants in Piscivorous Wildlife (Great Blue Heron and Mink)

Suter (1990) identified piscivorous wildlife along the Clinch River as being potentially at risk from toxicant releases from the ORR. This section evaluates the potential for biological effects in great blue heron and mink, two piscivorous species known to occur along the Clinch River.

3.5.3.1 Contaminant accumulation and effects in great blue heron

Introduction. This study seeks to assess the general health and fecundity of the ORR great blue heron population and document any contaminant-induced effects, particularly with respect to mercury and PCB exposure. Reproductive health is of particular concern, and the potential for off-site transportation of contaminants through heron body burdens is addressed.

Monitoring of great blue heron (*Ardea herodias*) was begun in 1991. 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 toxicological literature; and, (5) satisfies necessary logistical sampling considerations. Important logistical considerations were the presence of study and reference colonies on or in close proximity to ORR and population densities to meet sampling requirements. In addition, great blue heron are highly visible, facilitating direct observation and site location.

Study Sites

Four colonies were used for heron chick and egg collection. Two colonies (K-25 and Melton Hill colonies) are located within 1.8 miles of the ORR, and herons using these colonies are potentially exposed to contaminants occurring on the reservation. The remaining two colonies (Long Island and Looney Island colonies) are located >6 miles from the ORR, and heron using these colonies are assumed not to be exposed to contaminants that occur on the ORR. In general, there has been an increase in heron colonies in eastern Tennessee resulting from recent range expansion of the great blue heron in the upper Tennessee Valley (Pullin 1990).

The K-25 colony is located within the boundaries of the K-25 Site, on the west bank of Poplar Creek approximately 1.2 miles from its confluence with the Clinch River. Areal surveys by the TVA documented nesting activity at K-25 in 1986, and an estimated 31 nests were active in 1988 (Pullin 1990). The number of nests at this colony have remained at 25 to 50 during the 4 years of this study.

The second colony is located among three small islands in Melton Hill Lake approximately 1.2 miles above Melton Hill Dam. This colony has been active for at least 4 years (personal communication, Jim Evans, TWRA, Oak Ridge National Laboratory), and is composed of approximately 40 active nests.

Colonies on Long Island and Looney Island are each greater than 6 miles from the ORR and serve as off-site reference locations. Long Island is approximately 1.8 miles from the confluence of the Tennessee and Clinch rivers in the Tennessee River arm of WBR. The colony has been active since at least 1983, and 141 great blue heron nests were active in 1988 (Pullin 1990). Approximately 200 active nests have been observed during each year of the current study.

Looney Island is located in the Tennessee River approximately 18 miles upstream of Long Island in Fort Loudon Reservoir. The colony has been active since at least 1992, and an estimated 100 active great blue heron nests were observed on the island during the 1993 and 1994 nesting seasons. Black-crowned night herons also were found nesting on this island during both seasons.

Methods

For three years (March to June, 1992–1994) eggs and chicks were collected from great blue heron colonies. Egg and chick collection and processing techniques are described in ER/CR-SOPs 2180 and 2190.

To assess heron food preference and availability, crop contents were examined, and fish regurgitated from chicks or found beneath nests were collected. Recognizable fish samples were archived prior to PCB and mercury analysis. Abnormalities were noted and organ somatic indices were computed.

Results of contaminant analyses, physiological measurements, and reproductive parameters were statistically evaluated for differences between colonies located on and off the ORR. Aroclor 1260 and individual congener differences were evaluated when concentrations in at least one location (on or off ORR) were greater than 30 ppb. For congeners that were evaluated, any concentrations below detection limits were assigned a value equal to half the detection limit. Differences between locations were evaluated using a Student's t-test and, when appropriate, differences among colonies were evaluated using an ANOVA or Kruskal-Wallis test.

Results

Contaminant concentrations in fish collected from heron colonies

Mercury and chromium concentrations were significantly greater in fish collected from colonies located on the ORR than in colonies located off the ORR (Table C12). No significant differences in Aroclor 1260 concentrations existed between on- and off-ORR colonies and all PCB congener concentrations were less than 20 μg/kg.

Egg and chick metal concentrations

Thirty-three and 34 eggs, and 38 and 35 chicks, respectively, were collected from heron colonies on and off the ORR. Extreme mercury concentrations were quantified in one egg from the Melton Hill colony (KE022, Hg concentration = 0.601 mg/kg) and one egg from the Long Island colony (LE513, Hg concentration = 0.596 mg/kg). The egg from the Melton Hill colony was collected from a nest with 2 chicks approximately 2 weeks of age and this egg did not show signs of development. Egg LE513 was collected from a nest that contained 3 eggs and was in an early stage of development. Both concentrations were rejected as outliers by the Dixon and Grubbs test for outlying observations and were eliminated from the data set (Taylor 1987).

No difference existed between mean mercury concentrations in eggs collected from the Long Island and Looney Island colonies (Table C13); therefore, data from both colonies were used in calculating differences between on- and off-ORR mercury concentrations in eggs. However, mean mercury concentration in eggs collected from the K-25 colony ($\bar{x} \pm SE = 0.17 \pm 0.02$, mg/kg) was significantly greater ($P < 0.001$) than the mean concentration in eggs collected from the Melton Hill colony ($\bar{x} \pm SE = 0.07 \pm 0.01$, mg/kg). Therefore, only mercury data from eggs collected from the K-25 colony was used to test for differences between on- and off-ORR mercury concentrations in eggs.

Mean mercury concentration in eggs collected on the ORR was significantly greater than the mean concentration in eggs collected off the ORR (Table C13). Mean concentrations were greatest in eggs collected from the K-25 colony and least in the Melton Hill colony.

Mean chromium concentration in eggs collected on the ORR was significantly greater ($P = 0.046$) than the mean concentration in eggs collected off the ORR (Table C13). Concentrations were greatest in eggs collected from the Melton Hill colony ($\bar{x} = 0.22$ mg/kg) followed in decreasing order by concentrations in the K-25 ($\bar{x} = 0.15$ mg/kg), Long Island ($\bar{x} = 0.11$ mg/kg), and Looney Island ($\bar{x} = 0.11$ 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 compared to those collected off the ORR (Table C14). Mean feather mercury concentration was significantly greater in chicks collected from the K-25 colony ($\bar{x} = 2.02$ mg/kg) compared to feathers from chicks collected from the Melton Hill ($\bar{x} = 1.02$ mg/kg), Looney Island ($\bar{x} = 0.97$ mg/kg), and Long Island ($\bar{x} = 0.87$ mg/kg) colonies. Mean liver mercury concentration was not significantly different between chicks collected from the K-25 colony ($\bar{x} = 0.29$ mg/kg) and the Looney Island colony ($\bar{x} = 0.25$ mg/kg). However, mean liver mercury concentration was significantly greater in chicks collected from the K-25 colony compared to chicks collected from the Melton Hill ($\bar{x} = 0.15$ mg/kg) and Long Island ($\bar{x} = 0.12$ mg/kg) colonies. No significant differences existed in mean muscle concentrations between colonies located on or off the ORR. Mean muscle mercury concentration was significantly greater in chicks collected from the K-25 colony ($\bar{x} = 0.09$ mg/kg) compared to chicks

collected from the Melton Hill colony ($\bar{x}$ = 0.05 mg/kg). No significant differences in mean liver, muscle, or feather concentrations of arsenic or chromium were detected between chicks collected on and off ORR, or among colonies (Table C14).

Egg and chick PCB concentrations

Mean concentrations of Aroclor 1260 and 76% of the quantified PCB congeners were significantly greater ($P < 0.05$) in eggs collected from the K-25 colony compared to those collected from the Melton Hill colony. No significant differences were detected in Aroclor 1260 or congener concentrations between Long Island and Looney Island eggs. Therefore, for statistical analysis between on- and off-ORR colonies, PCB concentrations in K-25 eggs were compared to concentrations in Long Island and Looney Island eggs combined. Mean concentrations of Aroclor 1260 and 10 congeners were significantly greater in eggs collected on the ORR compared to concentrations in eggs collected off the ORR (Table C15). Concentrations of 47% of the 30 congeners evaluated in egg homogenates were below 30 μg/kg. Concentrations of congeners 123 and 167 were the only coplanar congeners that differed between eggs from colonies located on and off the ORR.

Aroclor 1260 and a majority of congener concentrations were significantly greater ($P < 0.05$) in fat, liver, and muscle tissue from heron chicks collected from the K-25 colony compared to those collected from the Melton Hill colony. No differences existed between chicks collected from the Long Island and Looney Island colonies. Therefore, comparison of PCB results between on and off ORR was between the K-25 colony and the Long Island and Looney Island colonies combined. Aroclor 1260 concentration was significantly greater ($P < 0.05$) in fat, liver, and muscle tissue from chicks collected on the ORR compared to concentrations in chicks collected off the ORR. 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 (Table C16). Congener 156 was the only coplanar congener that was significantly different in fat tissue between on and off ORR colonies.

Of the 30 congeners evaluated, 77 and 73% of the concentrations in chick muscle and liver tissue, respectively, were below 30 μg/kg. In chick liver tissue, concentrations of congeners 110, 118, 151, 153 and 180 differed significantly between on and off ORR colonies (Table C17). In chick muscle tissue, concentrations of congeners 110, 118, 138, 153, and 180 differed significantly between on- and off-ORR colonies (Table C18). Congener 110 and 118 were the only coplanar congener that differed between on- and off-ORR colonies in liver and muscle tissue, although concentrations of coplanar congeners ranged from below detection to 18 mg/kg in chick fat tissue.

Physiological and reproductive effects

No significant differences were observed in the number of eggs or chicks per nest between on- and off-ORR colonies (Table C19). The mean weight of eggs collected from colonies on the ORR was significantly heavier than eggs collected off the ORR; however, there was no difference in shell thickness.

Chick weight/length ratios, liver somatic indexes, and hematocrit measurements were not different in chicks collected on and off the ORR. However, liver EROD 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 C19).

Summary

Although herons occupying the K-25 colony have elevated body burdens of mercury and PCBs compared to herons occupying colonies off the ORR, the contaminant levels in tissues do not appear to affect the number of eggs laid or survival of chicks to fledging. Effects on chick survival from fledging to reproductive maturity were not determined. Contaminant data from one adult heron collected on the ORR in August 1992 suggest that body burdens are much greater in adults than in chicks.

3.5.3.2. Reproductive performance of mink

Introduction

Mink (*Mustela vison*) have been shown to be one of the mammals most sensitive to PCB exposure (Aulerich and Ringer 1977). They are also sensitive to other toxicants, including mercury (Calabrese et al. 1992).

Because mink are sensitive to PCB and mercury toxicity and inhabit wetland areas, they are potentially good indicators of environmental effects of these chemicals in aquatic habitats. However, mink are secretive and population densities tend to be low, making assessment of environmental contaminant effects in natural populations difficult. The objectives of this study were to evaluate contaminant bioaccumulation and correlated reproductive effects in mink fed fish collected from three locations: (1) on the ORR (Poplar Creek), (2) the Clinch River above the ORR, and (3) the ocean.

Materials and Methods

Fish collection and diet preparation

Fish used in mink diets were collected from (1) Poplar Creek between its mouth and the mouth of EFPC and (2) the Clinch River above Melton Hill Dam (above the ORR). Ocean fish (mackerel) were obtained from a commercial supplier. Fish from each location were homogenized and ten aliquots of homogenate collected for contaminant analyses.

Five diets composed of 75% fish and 25% normal mink diet (eggs, liver, vitamin and mineral premix, d-biotin, and cereal) were prepared. All diets were formulated to meet the nutrient requirements of the mink (NRC 1982). The fish portion of two reference diets contained ocean fish only (Diet A) and fish from the Clinch River only (Diet B). The fish portion of the remaining 3 diets (C, D, and E) contained 33, 67, and 100% fish collected from Poplar Creek and 67, 33, and 0% ocean fish, respectively. Diet proximate analyses were determined by MSU. Ten aliquots of each diet were collected for contaminant analyses.

Mink feeding experiment

Fifty adult, natural dark mink from the MSU Experimental Fur Farm were uniquely identified and randomly divided into 5 groups of 2 males and 8 females per group. Mink were housed individually in wire cages (61 x 76 x 46 cm) with attached nest boxes (38 x 28 x 27 cm). Cages were suspended above the ground in open-sided sheds. Throughout the study, mink were provided food and drinking water *ad libitum* and exposed to ambient temperature and photoperiod. Mink were immunized against canine distemper, viral enteritis, infectious pneumonia, and botulism, and provided thiamine daily to prevent thiamine deficiency resulting from thiaminase in fish. Mink were acclimated to the test facilities for at least 1 week prior to the definitive test.

Each mink group was fed one of the prepared diets from December 1, 1993 (approximately 3 months prior to breeding) through approximately June 30, 1993 (6 weeks postpartum). Mink were weighed at the beginning of the feeding trials and at monthly intervals thereafter (except during the gestation period). They were observed daily and any behavioral changes or clinical signs of toxicity recorded.

Mating began March 1, 1994, and was confined within the respective groups. Females were given an opportunity to mate every fourth day until a confirmed mating (presence of motile spermatozoa in vaginal aspirations) was obtained. The mated females were given an opportunity for a second mating the day following the initial mating or eight days later (a standard commercial mink ranch practice). The bred females were checked daily during gestation for evidence of whelping. The gestation period of mink averages 51 days but is highly variable (42 to 65 days) due to delayed implantation. Whelped kits were counted, sexed, and weighed on the day of birth and at 3 and 6 weeks of age. Mink kits begin to consume solid feed at 21 to 24 days of age and are weaned at 6 weeks. Thus, kit body weight at 3 weeks of age provides a good indication of the lactational performance of the female. 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 and 6 weeks of age.

Adult mink with > 30 % decrease in original body weight and all adults at the termination of the study were euthanized (CO_2). All adult mink were necropsied, organ (brain, liver, kidneys, heart, lungs, gonads, and adrenal glands) weights were recorded and samples of adipose, liver, kidney, and hair collected for residue analysis. At 6 weeks of age, 3 kits each from 3 randomly selected females from dietary groups A, B, C, and E were euthanized; liver, spleen, and kidneys weighed; and samples (liver, kidney, and remaining carcass) collected for residue analyses. Tissue samples (brain, liver, kidney, heart, lungs, and adrenal gland) also were collected from adult mink and preserved in 10% formalin for histopathologic examination.

Ten aliquots each of homogenized fish and mink diets and the collected mink tissues were analyzed for PCB congeners, Aroclors, and mercury. Adult liver and fat tissues were analyzed for PCBs; the kidney, liver, and hair tissues were analyzed for mercury. Liver tissue was also analyzed for ethoxyresorufin-o-deethylase (EROD) activity. Kit whole carcasses were analyzed for PCBs and mercury, liver tissue for PCBs only, and kidney tissues for mercury only. Lipid concentration (% lipid) was determined for all samples analyzed for PCBs.

PCB results were adjusted for lipid concentration (tissue concentrations divided by the % lipid) prior to statistical analysis. Results of contaminant analyses, physiological measurements, and reproductive parameters were statistically evaluated for differences among diet groups and between diet groups A or B and E using non-parametric (Wilcoxon or Kruskel-Wallis) and gausian (t-test or ANOVA) tests.

Results

Mercury concentrations in fish, diets, and mink

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 C20). Mean mercury concentrations were significantly different among fish collected from the ORR, Clinch River, and the ocean (Table C21). Mean mercury concentrations in mink diets increased progressively from diet A through diet E (Table C21). Correspondingly, mercury concentrations in liver, kidney, and hair of adult female mink

increased progressively in mink fed diets A through E (Table C22). 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 C22).

PCB and congener profiles in fish and mink diets

Aroclor 1260 was the dominant Aroclor detected in aliquots of homogenized fish, mink diets, and tissues of adult mink and kits, although Aroclor 1254 also was quantified in several tissue samples. Aroclor 1260 was quantified in aliquots of all diets except for diet A (75% ocean fish), in fat tissue from mink in all diet groups except group C, and in liver tissue of mink from diet groups D and E. Twenty-three specific congeners, including 8 coplanar congeners (non-ortho or mono-ortho congeners), were evaluated in homogenized fish, diets, liver and fat tissues of adult mink, and liver tissue and carcass of kits. Ninety-six percent of congener concentrations were significantly different among ocean, ORR, and Clinch River homogenized fish, including all but one coplanar congeners (Table C23). In all cases where significant differences existed in homogenized fish congener concentrations among collection sites, the greatest concentrations were in fish collected from the ORR. Aroclor 1260 and 87% of the congener evaluated also were quantified in mink diets (Table C24). Mean Aroclor 1260 concentration was significantly greater ($P < 0.01$) in diet E compared to concentrations in other diets. Coplanar congeners 126 and 189 were not detected in any diets; congener 156 was detected in low concentrations in diets B, C, and D; congeners 77 and 81 were quantified in low concentrations in diets C, D, and E; and congener 167 was only quantified in diets D and E. Concentrations of coplanar congener 118 were significantly greater ($P < 0.01$) in aliquots of diet E compared to aliquots of the other diets. Mean concentrations of congener 123 progressively increased from diet C to D, however, quantified concentrations in diet E were surprisingly lower than concentrations in diet C. Low concentrations of this congener in diet E are thought to result from matrix interference in this diet. Similarly, lower than expected concentrations were quantified for congeners 99, 101, 156, and 171 in diet E. The remaining non-planer congener concentrations were generally greatest in aliquots from diet E (Table C24).

PCB and congener profile in mink tissues

Mean Aroclor 1260 concentration was significantly greater in liver tissue of female mink fed diet E (Table C25). Coplanar congener 189 was quantified at low concentrations (<21 μg/kg) in liver tissue from all mink fed diet E, 50% of those fed diet D, and less than 10% of those fed diets A, B, or C. Liver concentrations of congener 126 were significantly greater ($P < 0.01$) in samples collected from female mink fed diet E. Low concentrations (<32 μg/kg) of congener 126 also were quantified in liver tissue from female mink fed diets B and D. Mean liver concentrations of coplanar congeners 77 and 81 were ≤ 6 μg/kg in all female mink regardless of diet. Mean concentrations of congeners 156 and 167 increased progressively in liver tissue from female mink fed diets A–E and were significantly greater ($P < 0.01$) in mink fed diet E compared to those fed diets A or B. Mean concentration of coplanar congeners 118 and 123 were significantly greater in liver tissue from female mink on diet E compared to concentrations in liver tissue from mink fed diets A or B. There were significant differences ($P < 0.05$) in all female mink liver tissue non-planar congener concentrations among diet groups. For all non-planar congener in female liver tissue, concentrations were significantly greater ($P < 0.05$) in tissues from diet E mink compared to diet A mink, except for concentrations of congeners 195 and 196, which were not significantly different. Similarly, mean female mink liver concentrations of all non-planar congeners were significantly greater in female mink liver from diet group E mink compared to diet group B mink, except for congeners 146, 153, 170, 180, 183, and 201, which were not significantly different between these diet groups.

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 C26). Mean concentrations of all coplanar congeners were significantly greater ($P < 0.05$) in fat tissue from female mink fed diet E compared to mean concentrations in mink fed diets A or B, except that there was no difference in congener 126 fat concentration between female mink fed diets E and B. Non-planar congener mean concentrations were significantly greater ($P < 0.05$) in fat tissue from female mink fed diet E compared to mean concentrations in mink fed diets A or B, except congeners 170, 194, and 195. There was no difference between congener 170 or 194 mean fat concentrations between female mink fed diets B or E, and mean fat concentration of congener 195 in female mink was significantly greater ($P < 0.05$) in female mink fed diet B compared to diet E.

Concentrations of coplanar congeners 77, 81, 123, 156, and 189 were < 20 μg/kg in liver tissue from kits in all diet groups (Table C27). Concentrations of coplanar congeners 118, 126, and 167 were significantly greater ($P < 0.05$) in liver from diet E kits compared to those from diets A, B, or C. Concentrations of non-planar congeners 101, 151, 171, and 195 were less than 15 μg/kg in liver tissue from kits in all diet groups. Concentrations of the remaining non-planar congeners quantified were greater in liver tissue of diet E kits. The pattern of concentrations of coplanar congeners in kit carcasses homogenates from the various diets were similar to those observed in kit liver tissue (Table C28), except that the concentration of coplanar congener 126 was less than 25 μg/kg in kit carcass homogenates and the concentration of congener 156 was significantly greater ($P < 0.05$) in kit carcass homogenates from diet group E compared to other diet groups. Concentration patterns of non-planar congeners in the various diet groups also were similar in carcass homogenates compared to concentrations in liver tissue, except concentrations of congeners 151 and 196 were the only congeners with concentrations less than 25 μg/kg. Concentrations of all other congeners quantified were greater in homogenized carcasses of kits from diet group E.

Physiological and reproductive effects

Two mink from diet group A, one male and one female, died during the experimental period. The male died on March 18, 1994, from hemorrhagic and necrotizing cystitis, and the female died on April 24, 1994, due to complications during parturition. In addition to these adult mink, kits from one female in diet group B developed staph infections, and all but one died prior to 6 weeks of age.

One, 2, and 4 females did not whelp in diet groups A and B, D, and C, respectively (Table C29). Two females that did not whelp (1 from diet group A and 1 from diet group C) had cysts in the reproductive tracts that probably interfered with normal reproduction. Two females that did not whelp (1 from diet group C and 1 from diet group D) had no placental scars in the uterus, and, therefore, probably were not pregnant. The reason that the remaining four females did not whelp is unknown.

Mean whole body weights of female mink were not significantly different among diet groups at the beginning of the experimental period; however, mean weights of females in diet group E were significantly less ($P=0.03$) than mean weights of females in diet group A at the end of the experimental period (Table C29). 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 significantly lower ($P=0.004$) in male kits from diet group E compared to 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 ($\bar{x}$=1.0 g) compared to those from diet group E ($\bar{x}$=1.3 g). 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 compared to diet groups A, B, and C but not diet group D (Table C29). Liver EROD activity was significantly increased in adult female mink from diet groups D and E compared to those from diet group A (Table C29).

Summary

Although concentrations of mercury and PCBs were greater in fish collected from streams located on the ORR and these contaminants were higher in diets fed to mink with increasing percentage of ORR fish, reproductive effects were only noted in mink fed 75% ORR fish. Liver EROD activity, a sensitive biomarker of exposure to PCBs, increased in mink fed diets containing 50% and 75% fish collected from the ORR. Although fish are a major food item in the diet of wild mink, the proportion of fish in their diets normally does not exceed 4–60%. However, in addition to fish, concentrations of contaminants in other food items (crayfish, frogs, muskrat, ducks, and rodents) need to be evaluated in assessing effects of contaminants on mink living on the ORR.

3.5.4 Bioindicator Analysis

In this task, bioindicator analysis was conducted on largemouth bass and bluegill to (1) determine if the health of these species is being adversely affected by contaminants in the Clinch River/Poplar Creek system, (2) establish baseline conditions for evaluating the effectiveness of any future remedial actions, and (3) 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.

3.5.4.1 Methods and approach

Field collection and processing methods and laboratory processing approaches are described in the sampling and analysis plan (DOE 1993). Histopathological analyses were based on procedures provided in Hinton and Lauren (1990) and Hinton and Couch (1984). Detoxification enzyme analysis and blood chemical analysis involving indicators of contaminant exposure and organ dysfunction, respectively, were performed according to Adams and Ryon (1994) and Adams et al. (1990). Indicators of overall condition (condition indices) and bioenergetic and nutritional status were determined based on the methods given in Adams and Ryon (1994), Adams et al. (1992a), and Adams et al. (1992b). The health assessment index (HAI) was also applied to individual bluegill and largemouth bass collected at each site (Adams et al. 1993). This index is a field necropsy method that provides a health profile of a fish population based on the incidence of anomalies observed in tissues and organs of individual fish. This method is used to detect gross changes in the health of fish populations but is not diagnostic in character.

An ANOVA was used to determine statistical significance of variables among sites, including detoxification enzymes, indicators of organ dysfunction, histopathological responses, condition indices, and indicators of bioenergetic and nutritional status. If the ANOVA rejected a multisample null hypothesis of equal means ($P<0.05$), then Dunnett's test (Zar 1984) was used to determine significance for individual variables among sites. Tests for homogeneity of variance of individual response variables between sites were conducted with Levene's test (Sokal & Rohlf 1981).

To determine the integrated (multivariate) response of fish to the environmental conditions at each sampling site, all the bioindicator variables were considered jointly within a multivariate context using a canonical discriminant analysis procedure (SAS 1985; Adams et al. 1994). This method provides a graphical representation of the positions and orientations of the integrated site responses of the fish at each site relative to responses at the other sites. In addition, this procedure computes linear combinations

of all the variables under consideration and can detect statistical differences among means for the sites even if individual variables do not indicate such differences. A variable selection procedure (SAS 1985) was used to identify which variables contributed most to the discrimination among the integrated site responses. This variable selection procedure considered all possible combinations of the individual variables and, for any specified subset size, selected those with the best discriminating power.

In evaluating the health status of fish at each sample site, comparisons of the individual bioindicator response groups and the integrated effects were made between each of the sample sites in Poplar Creek and the Clinch River with two reference areas, Norris reservoir (CRM 125) and lower Watts Bar (TRM 530). Two reference sites are required in this study because Norris is a headwater system characterized by extremely low contaminant loading and low biological productivity. Lower WBR, however, also has relatively low contaminant loading but has much higher levels of biological productivity. Since some of the bioindicators responses measured in these fish can be modified or influenced by the level of biological productivity in the system (as reflected in feeding, nutrition, etc.), it is necessary to also compare the bioindicator responses of fish from sites in the Clinch River and Poplar Creek to an internal reference (TRM 530) that has somewhat similar trophic dynamics and productivity.

3.5.4.2 Functional response groups

For the purpose of analysis and evaluation, individual bioindicator responses were grouped into five functional categories representing indicators of (1) detoxification enzymes, (2) organ dysfunction, (3) histopathology, (4) overall fish health (condition indices), and (5) nutrition and feeding status. In addition the 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). Those variables in categories (1) and (2) are primarily short-term response indicators and have relatively low ecological relevance, while those variables in groups (3) through (5) are longer-term response indicators and are characterized by lower toxicological but higher ecological relevance.

For each sample site, the means and standard error for each bioindicator variable are presented in Tables C30 and C31 for female and male bluegill and Tables C32 and C33 for female and male largemouth bass. In this section on bioindicator analysis, only data for males of each species are discussed because information on females will be presented in Sect. 3.5.5 (Reproductive Analysis). Also, sample sizes for males are generally higher at each site and, therefore, more amenable to statistical analysis.

Detoxification Enzymes

The activity or level of liver detoxification enzymes is used to indicate exposure to and possible effects of various xenobiotics, such as hydrocarbons, PAHs, PCBs, and pesticides (Payne and Penrose 1975). Many of these compounds commonly occur in industrial and municipal effluents and are accumulated within living tissues (Ahokas et al. 1976; Neff 1978; Brown et al. 1986). As has been demonstrated in studies of other aquatic systems, EROD enzyme activity has been one of the best indicators of fish contaminant exposure in field populations (Adams et al. 1992a; 1992b).

Increased activity of EROD at the two lower Poplar Creek sites and in the Clinch River immediately downstream of the mouth of Poplar Creek (CRM 10), indicated that largemouth bass sampled from these sites had been exposed to elevated levels of contaminants relative to bass from the two reference sites (Fig. 3.57). Even though elevated activity was indicated at PCM 5.3, only one bass was collected at this site due to their low abundance in the area of the creek. Bluegill, however, did not show the same pattern

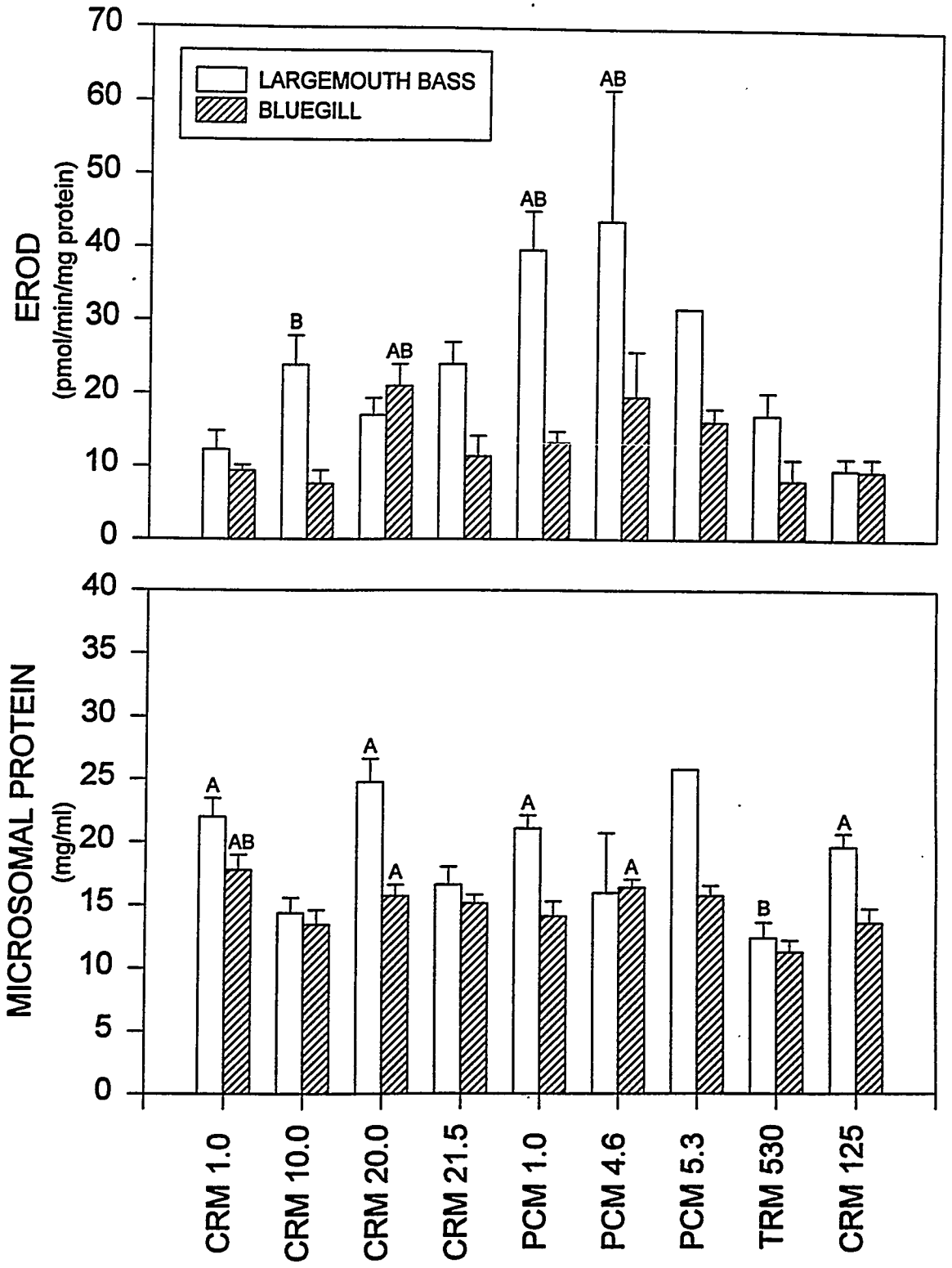


Fig. 3.57. Means + SE for detoxification enzyme indicators (EROD and microsomal protein) for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site 1993-1994. A = significantly different from TRM 530, B = significantly different from CRM 125.

of enzyme induction as did largemouth bass. Only bluegill at CRM 20 near the mouth of WOC demonstrated a significant increase in enzyme activity.

Microsomal protein, which is not only used as a standard for quantifying EROD activity but also as an indicator of microsomal proliferation in the cell, was higher at two Clinch River sites and two Poplar Creek sites compared to the internal reference (TRM 530). Microsomal protein in bluegill, however, displayed a similar trend to EROD activity in that only one site (CRM 10) was significantly higher than the reference.

There are two possible explanations for the higher EROD activity and microsomal protein values in largemouth bass compared to bluegill at some of the sample sites. Bass are longer lived species and, therefore, may have been exposed to contaminants for a longer period of time than bluegill. The most feasible explanation for these species differences, however, may be related to the nature of their respective diets. Shad are the main component in the diet of largemouth bass and these prey have higher body burdens of some contaminants such as PCBs than the benthic macroinvertebrates that are typically ingested by bluegill. In fact, levels of PCBs in shad are higher than the PCB levels measured in mayflies (Sect. 3.5.2).

PCBs have been shown to induce the mixed function oxidase (MFO) system in fish (Spies et al. 1982; Ankley et al. 1986; Melancon and Lech 1983). Even at relatively low levels, which approximate the body burdens of PCBs measured in some fish collected from the Clinch River and Poplar Creek, PCBs have been shown to induce MFO activity (Nasci et al. 1991). Studies by Greeley et al. 1994 demonstrated significant EROD induction in largemouth bass exposed to PCBs in Hartwell Reservoir (Georgia/South Carolina).

Histopathology

The histopathological condition of largemouth bass and bluegill were assessed both quantitatively, to determine the percentage of the total liver damage caused by various pathologies, and qualitatively, to assess the frequency occurrence of the various lesions present in the liver.

Quantitative evaluation. Indicators of histopathological condition demonstrated few changes in largemouth bass compared to bass collected from reference areas, but a variety of pathologies were evident in bluegill (Fig. 3.58). Macrophage aggregates, which are indicators of diseased or infected areas in tissue, were significantly higher than reference values for largemouth bass sampled from CRM 1. Even though increases in this pathology were indicated for bluegill in the upper two sites of Poplar Creek, values were not statistically different due to the high variability among sample means. Necrotic liver parenchyma was also somewhat elevated in bluegill from Poplar Creek and CRM 10. Also, the abundance of liver parasites was higher at two Clinch River and two Poplar Creek sites compared to the references. This increased frequency of pathologies in the liver tissue of bluegill collected from some Poplar Creek and Clinch River sites indicates that these sunfish have less viable liver to (1) produce MFO enzymes for detoxifying contaminants, (2) store energy reserves as glycogen and lipid, and (3) manufacture vitellogenin for oocyte and egg development.

Qualitative evaluation. The major lesions observed in the liver and spleen tissue of largemouth bass and bluegill were recorded by their severity. Six major types of pathologies were observed in the liver of bass and bluegill, while two and four types of lesions were found in the spleen of bluegill and bass, respectively (Tables C34 and C35). The average rank of all lesions served as an indicator of the severity of histopathological damage to the major organs of fish at each of the sample sites.

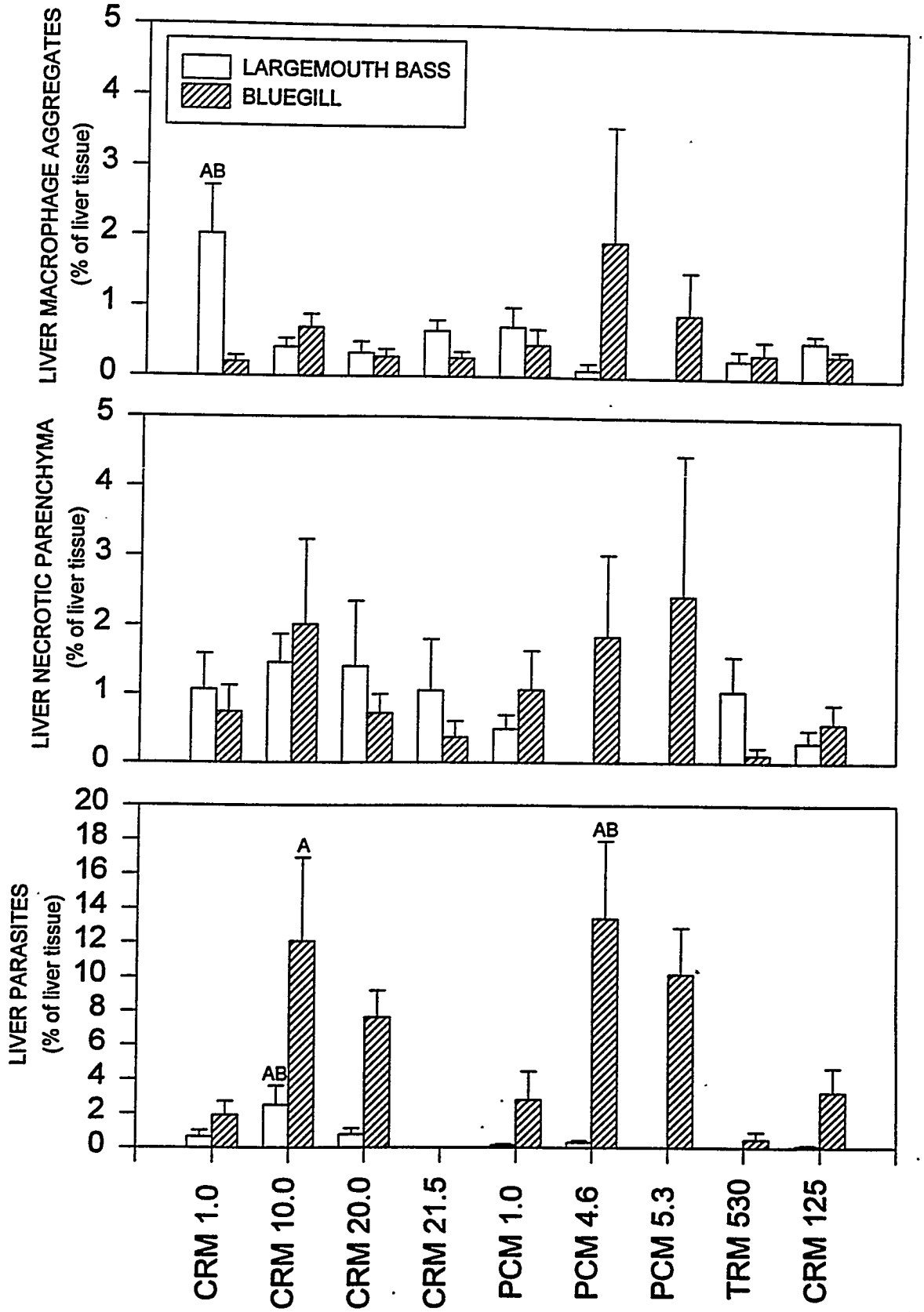


Fig. 3.58. Means + SE for histopathological indicators for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site 1993–1994. A = significantly different from TRM 530, B = significantly different from CRM 125.

The upper two sites in Poplar Creek (PCM 5.3 and 4.6), corresponding to the mouth of EFPC and Mitchell Branch, had the lowest average pathology rank and, therefore, represented the bluegill in the poorest histopathological condition (Table C34). Sites in the Clinch River—CRM 10.0, CRM 21.5, and CRM 1.0—ranked 3, 4, and 5, respectively, in their severity of liver and spleen lesions. Fish from lower Watts Bar (TRM 530) had the fewest lesions and were considered to be in the best histopathological condition. Since bluegill have restricted home ranges and movement patterns (Gatz and Adams 1994), the condition of these fish basically reflect the environmental and water quality conditions at these sites. It was somewhat surprising, however, that bluegill from CRM 20 (near the mouth of WOC) were in better condition than those bluegill 1.5 miles upstream of the WOC mouth.

Even though largemouth bass bass from PCM 4.6 demonstrated the poorest histopathological condition, the other two sample sites in Poplar Creek did not show such anomalies (Table C34). One potential explanation for the higher variability in histopathological condition of largemouth bass in Poplar Creek compared to bluegill is that largemouth bass are more transient and those individuals sampled from various sites in Poplar Creek may have migrated in from other areas of the Watts Bar system such as from the Clinch River. In contrast to bluegill, bass collected from near the mouth of WOC (CRM 20) had the second highest incidence of liver and spleen lesions. Further downstream at CRM 10, largemeouth bass from the Brashear Island area had the third poorest pathological condition, followed by those individuals collected from the lower Clinch River (CRM 1.0). In the Clinch River, therefore, there appears to be a downstream gradient in the pathological condition of largemouth bass which extends from CRM 20 downstream to CRM 1.0.

In several studies, PCBs have been implicated in the liver pathologies of fish. Exposure to PCB was found to increase the occurrence of hepatocellular necrosis and liver lesions in flatfish (Rhodes et al. 1985), and necrotic changes in hepatocytes were reported after fish were exposed to the PCB Aroclor 1254 (Hacking et al. 1978). In a river contaminated by PCBs, Peters et al. (1987) reported an increase in abundance of macrophage aggregates in fish liver tissue. In the Clinch River/Poplar Creek study, liver necrosis and macrophage aggregates were found at higher frequencies in liver tissue in those largemouth bass collected from the Clinch River and Poplar Creek sites than in fish from both references. Hinton and Lauren (1990) state that hepatocellular necrosis is a good biomarker of anthropogenic toxicant exposure, and is useful for identifying point sources of toxicant emissions. In the case of fish collected from the Clinch River and Poplar Creek, PCBs are the main contaminants of concern; therefore, it is reasonable to assume that the hepatic necrosis observed in these fish may have been caused by this contaminant.

Organ Dysfunction

Elevated creatinine levels are typically used as an indicator of kidney damage or malfunction (Tietz 1986). For largemouth bass, creatinine values at all Clinch River and Poplar Creek sites were elevated above the reference values, but only those fish from PCM 1.0 had significantly higher concentrations (Fig. 3.59). Creatinine levels in bluegill were generally similar to reference values with only those sunfish from PCM 1.0 being higher. Therefore, some possible kidney damage in largemouth bass and bluegill is implicated, particularly at the lower Poplar Creek site.

The transferase enzyme ALT (alanine aminotransferase) is generally used as an indicator of liver damage, and elevated levels in humans is a sign of liver cirrhosis, hepatitis, or disease which affects liver cell integrity. Levels of this enzyme in largemouth bass did not suggest liver damage, but in bluegill liver damage is implicated in individuals sampled from PCM 1.0. Histopathological examination of bluegill did in fact reveal a relatively high occurrence of liver pathologies for fish collected from Poplar Creek.

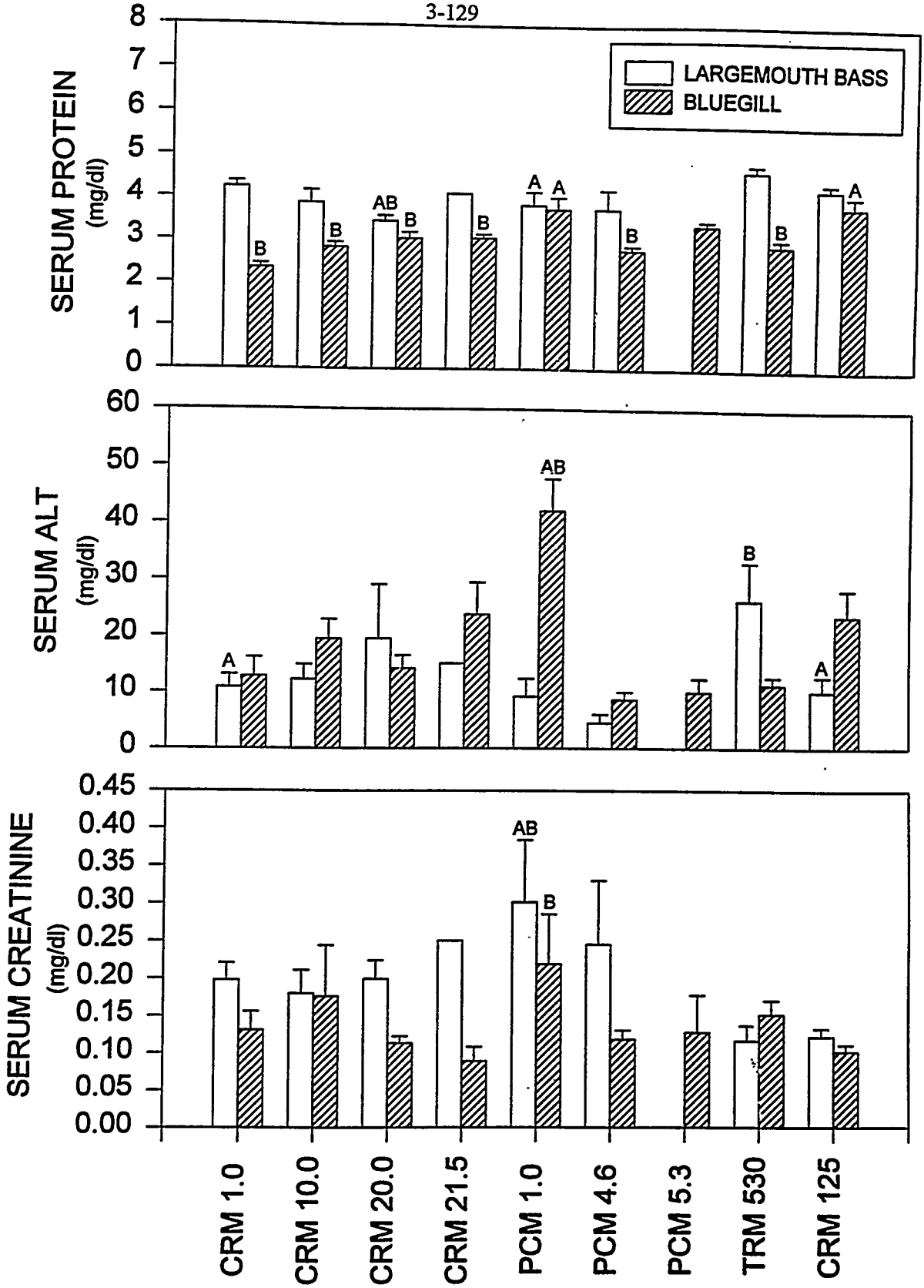


Fig. 3.59. Means + SE for organ dysfunction indicators (serum creatinine, ALT, protein) for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site 1993-1994. A = significantly different from TRM 530, B = significantly different from CRM 125.

Particularly evident among these pathologies is the incidence of focal necroses, which are direct indicators of liver damage.

Total serum protein is used as a general indicator of protein metabolism in organisms. Largemouth bass from CRM 20 and PCM 1 had significantly lower protein concentrations than did the reference fish. Bluegill collected from the Clinch River sites and PCM 4.6 all had lower values compared to 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).

Condition Indices

Various condition indices were measured as indicators of the general health of fish. 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 metabolic stress due to contaminants and nutritional status. For largemouth bass, the condition factor was significantly higher at CRM 1 and PCM 1 than in the Norris reference bass, while the condition factor of CRM 20 fish was lower (Fig. 3.60). Because Norris is basically an oligotrophic system, the more appropriate reference site for comparison on a bioenergetic basis is the lower WBR site (TRM 530). For this comparison, only largemouth bass from CRM 20 were lower than the reference. Bluegill from all sites, however, had similar condition factors among sites.

The visceral-somatic index (VSI) reflects energy stored as lipids in the mesenteries of the viscera. The VSI generally reflects the level of total body lipids, which is used to indicate the overall fat storage and bioenergetic condition of fish. Largemouth bass sampled from all Poplar Creek and Clinch River sites except CRM 21.5 and PCM 4.6 had significantly higher VSI values than did the Norris reference, while values were all similar to the lower Watts Bar reference (Fig. 3.60). In contrast, the visceral-somatic indices for bluegill from CRM 20 and 21.5 and lower Poplar Creek were significantly lower than TRM 530, with only CRM 1 being higher than the Norris reference. Because of the lower lipid levels in bluegill from the upper Clinch River sites and the lower Poplar Creek area, some level of bioenergetic dysfunction in fish from these sites is indicated.

The liver-somatic index (LSI) is an indicator of metabolic energy demands (Adams and McLean 1985) and is sensitive to toxicant stress with both liver enlargement due to hyperplasia and hypertrophy being reported in fish exposed to toxic compounds (Fletcher et al. 1982, Addison 1988, Heath 1987). Largemouth bass sampled from all Clinch River sites except CRM 10 and all Poplar Creek sites had significantly lower LSI values than the Watts Bar reference bass (Fig. 3.60). Only at CRM 10 was the LSI significantly higher than the Norris reference. Because the LSI in largemouth bass is generally lower than that from Norris fish, the LSI in this case does not reflect changes in liver size due to toxicant exposure but probably is a response to nutritional and bioenergetic factors. For bluegill, the LSI was similar among all sites.

Nutritional and Feeding Status

In evaluating the effects of contaminant stress on fish, the nutritional status of the organism is important in interpreting the nature of these effects. For example, poor nutrition can weaken fish and render them more susceptible to other stresses (Shulman 1974, Adams et al. 1985). As an indicator of short-term feeding intensity, the percentage of food in the intestine was evaluated among sites for largemouth bass and bluegill (Fig. 3.61). Feeding by largemouth bass was generally low at all sites, while feeding intensity at CRM 1.0 and 21.5 and the lower two Poplar Creek sites indicated significantly lower

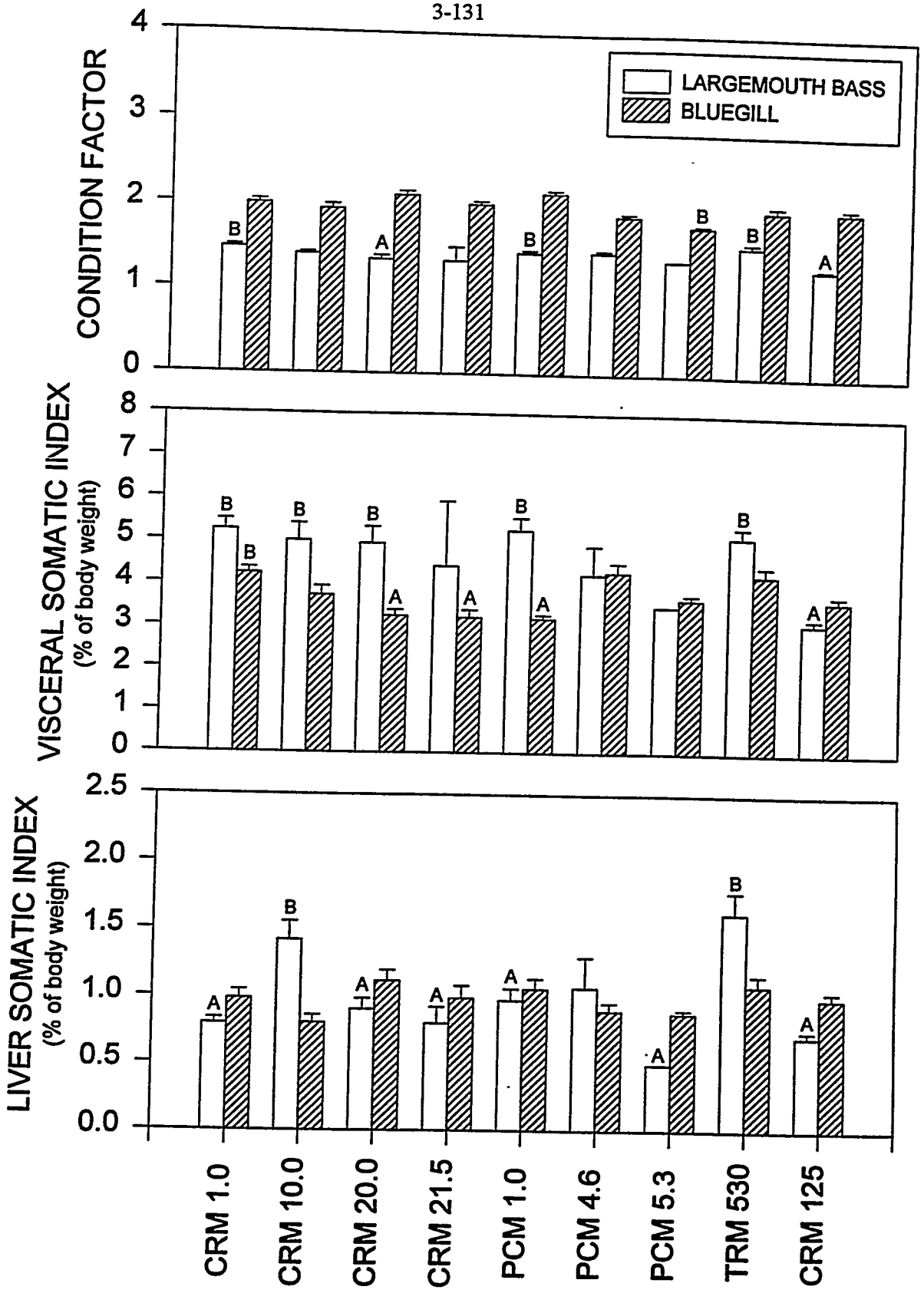


Fig. 3.60. Means + SE for three condition indices for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site 1993-1994. A = significantly different from TRM 530, B = significantly different from CRM 125.

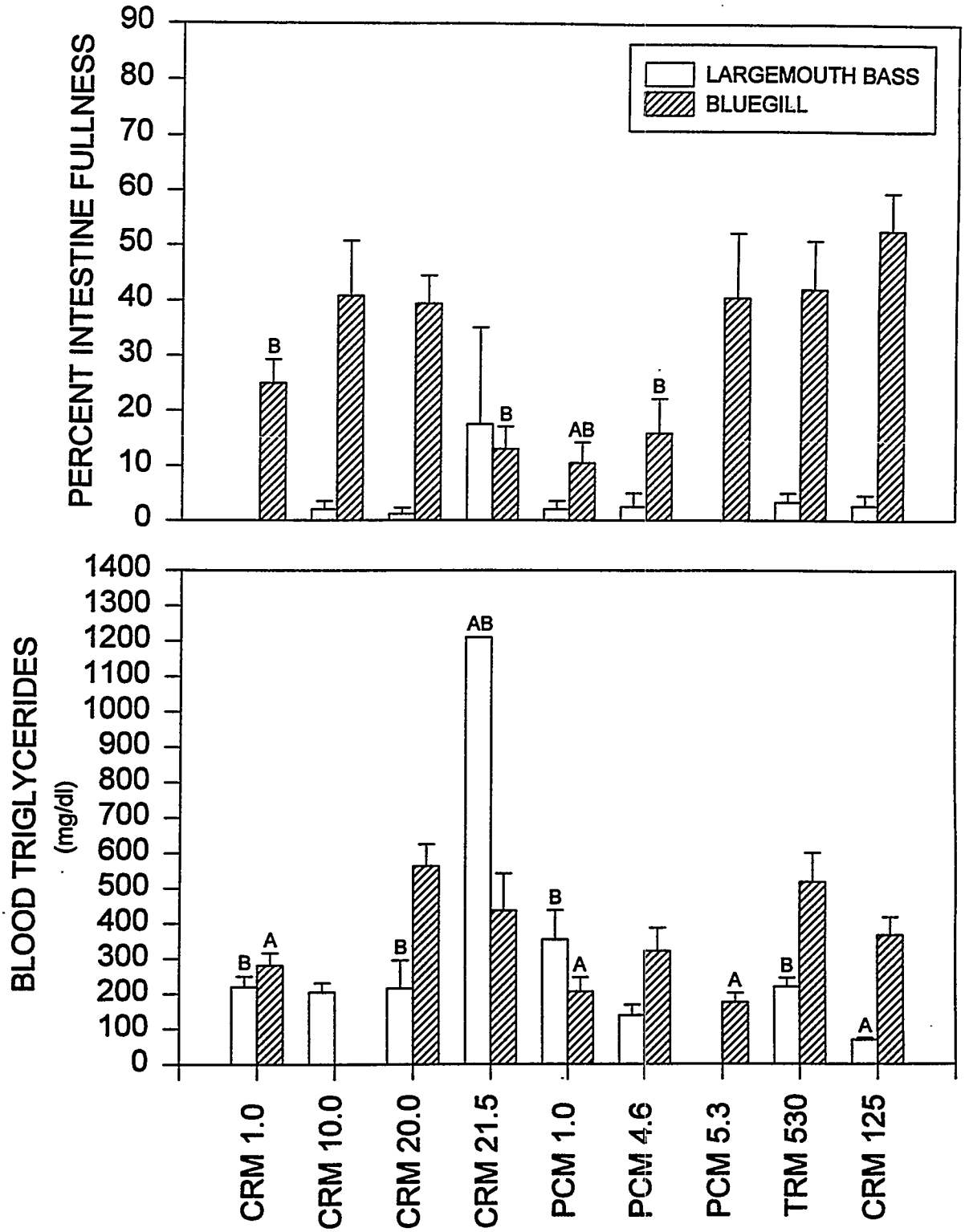


Fig. 3.61. Means + SE for feeding and nutritional indicators for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site 1993–1994. A = significantly different from TRM 530, B = significantly different from CRM 125.

feeding compared to the references. Serum triglycerides, which reflect a more long-term feeding status, were extremely low in the Norris reference bass and intermediate in Norris bluegill. Only bass collected from CRM 10 and the upper two Poplar Creek sites were not different from the Norris reference, while only bass from CRM 21.5 had higher triglyceride levels than the Watts Bar reference. Triglyceride levels in bluegill, however, were significantly lower than the Watts Bar reference at CRM 1.0 and PCM 1.0.

3.5.4.3 Health assessment index

The HAI, which provides an overall health profile of a fish population, indicated that even though bluegill sampled from the Clinch River and Poplar Creek had higher HAI scores than fish from other sites (and therefore poorer health), these values were not statistically different from the Norris reference site (CRM 125) (Fig. 3.62). Only bluegill collected from PCM 4.6 (downstream of Mitchell Branch) were in significantly poorer health than the reference fish. Anomalies in three of the index variables (kidney, parasites, and levels of serum protein) were primarily responsible for the elevated HAI values at this site. Bluegill collected from PCK 4.6 had a high incidence of swollen kidneys, which is a general indicator of organ pathology (Goede and Barton 1990). Abnormally low values of blood protein observed in these fish can reflect anomalies in protein metabolism or nutritional stress (Lockhart and Metner 1984).

The HAI demonstrated a more dynamic response in largemouth bass than it did in bluegill. Bass from two sites in Poplar Creek (PCM 01 and 5.3) and downstream of the mouth of WOC (CRM 20) had significantly higher HAI scores than did the other sites. Anomalies in the gills, spleen, liver, and blood protein were the primary variables responsible for the elevated HAI at these sites. Reduced condition factors, which reflect overall condition and the "well-being" of fish, also contributed to the poorer health of fish at PCM 5.3 and CRM 20. Because the spleen serves a hemopoietic function, abnormalities in this organ could indicate bacterial infection or disease. Gill damage in fish collected from PCM 5.3 and 1.0 could indicate potential impairment of gas exchange, osmoregulation, and excretory function in these fish. For bass collected from CRM 20, some liver damage was implicated. Liver damage in fish can result in reduced production of detoxification enzymes, impairment of glycogen and lipid dynamics within this organ, and decreased ability to produce vitellogenin necessary for proper egg development.

3.5.4.4 Integrated site analysis

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 the multivariate analysis. This holistic approach for evaluating the effects of water quality and other environmental factors on the health of fish has the primary advantage of providing an integrated assessment of fish health based on multiple rather than individual bioindicators. As a measure of the quantitative differences between site means, the Mahalanobis statistic was used to measure the linear distance between these means. Because of relatively low sample sizes, some sites could not be included into the discriminant analysis procedure. For the largemouth bass analysis, the two upper Poplar Creek sites and CRM 21.5 had to be excluded because bass were either not present or in low abundance at these sites. For bluegill, only PCM 5.3 and CRM 21.5 had to be excluded from the integrated site analysis because of sample size limitations.

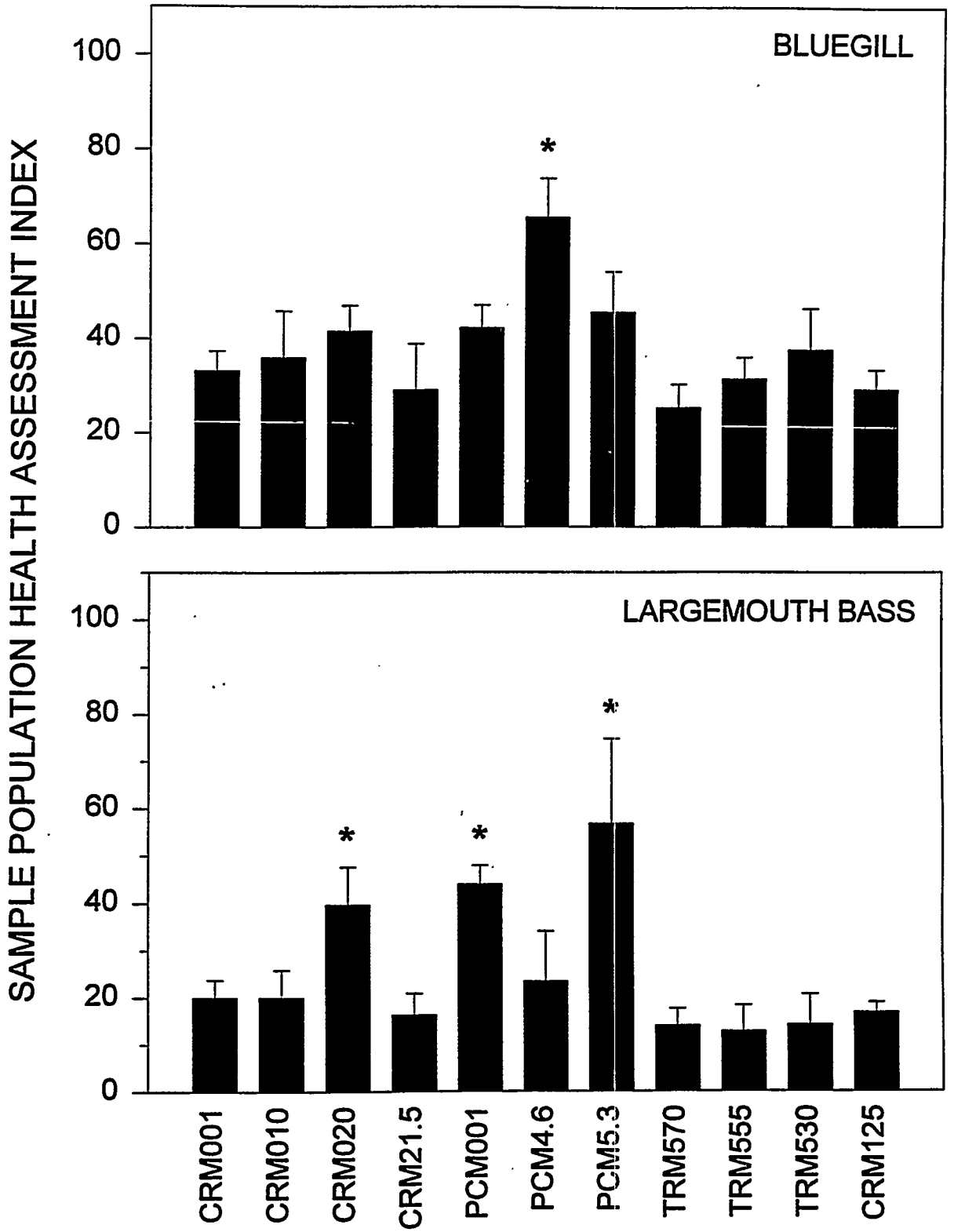


Fig. 3.62. Means + SE for the health assessment index (HAI) for male largemouth bass (clear bars) and bluegill sunfish (hatched bars) at each sample site 1993–1994. * = significantly different from CRM 125.

Results of the discriminant analysis revealed several patterns relative to the health of largemouth bass populations at the two reference sites compared to fish sampled from the Clinch River and lower Poplar Creek (Fig. 3.63). Norris bass were least similar to those fish sampled from lower Poplar Creek (PCM 1) and most similar to the lower Watts Bar reference site. Even though there are major differences in the trophic dynamics of biological productivity of these two reference sites, fish from these areas were more similar to each other than they were to the Clinch River and Poplar Creek fish. Bass sampled from PCM 1.0 were most similar to fish from CRM 10 and 20. The functional group analysis also demonstrated that bass from PCM 1.0 were in the poorest health as indicated by the detoxification enzyme, histopathological, and bioenergetic responses.

The individual response variables that best discriminated among the health status of fish from all sample sites were identified by a multivariate selection procedure. These variables were used to generate the final integrated site responses shown in Figs. 3.63 through 3.65. For largemouth bass, four variables were significant in discriminating among sites. These were, in order of importance, the VSI, EROD, microsomal protein, and the condition factor. Two of these variables, EROD and microsomal protein, are indicators of contaminant exposure, while the VSI and condition factor reflect the bioenergetic and overall condition of these fish (see Sect. 3.5.4.2). The primary factors, therefore, that are important for characterizing the health status of largemouth bass in the Clinch River and Poplar Creek are those that demonstrate that these fish have not only been exposed to contaminants, but bioenergetic function of these fish may have been impaired by this exposure.

The integrated site response of bluegill sampled from the Clinch River and Poplar Creek also demonstrated distinct patterns (Figs. 3.64 and 3.65). The two Poplar Creek sites segregate together as do the reference areas. The two reference sites were more similar to each other than they were to any of the Clinch River or Poplar Creek sites. Interestingly, the two Poplar Creek sites were more similar to CRM 20 than the other two Clinch River areas, even though CRM 10 is immediately downstream of the mouth of Poplar Creek.

In Fig. 3.65 the three Clinch River sites segregate as a group as do the two references. The upper Clinch River site (CRM 20) is the least similar to the two references, while CRM 10 is more like TRM 530 than the Norris reference. All three Clinch River sites are equally similar to each other implying that largemouth bass at these sites are equally affected by environmental conditions including contaminant exposure and the biological effect responses resulting from this exposure.

The variables most responsible for discriminating among sites for bluegill were EROD, the VSI, and microsomal protein. These are three of the same four variables that were also involved in site discrimination for largemouth bass. As was the case for largemouth bass, bluegill are exposed to contaminants in the CR/PC system, and one potential implication of this exposure is impairment of bioenergetic function in these fish.

3.5.4.5 Summary and conclusions—bioindicators task

The bioindicator data indicate an apparent gradient in biological effects extending from upper Poplar Creek down the Clinch River to lower WBR. Some of the major health effects responses observed in fish, such as detoxification enzyme induction and poor histopathological condition, are consistent with PCB exposure. Based on the evidence presented in this study, therefore, it appears reasonable to conclude that PCBs may be adversely affecting the health of fish in some areas of the CR/PC system.

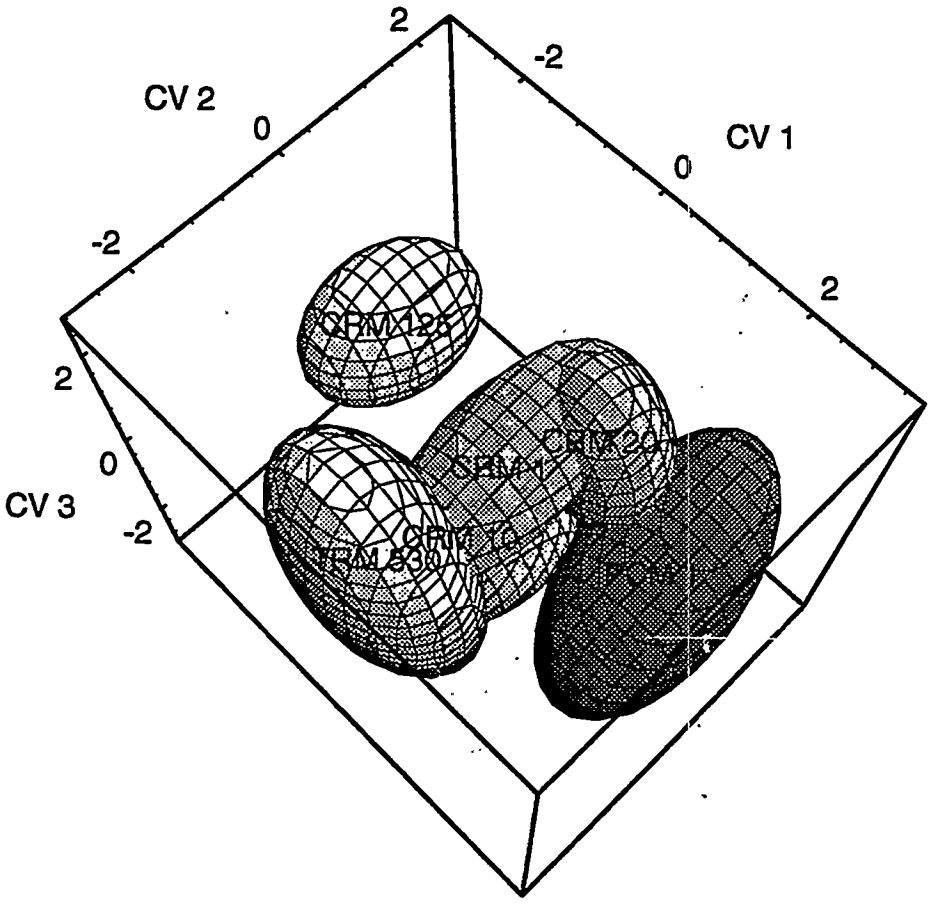


Fig. 3.63. Segregation of integrated health responses for male largemouth bass collected from two reference sites (CRM 125 and TRM 530), three Clinch River sites (CRM 1, 10, 20) and the lower Poplar Creek site (PCM 1.0). Ellipsoids represent the mean integrated responses of bass at a site for 1993–1994.

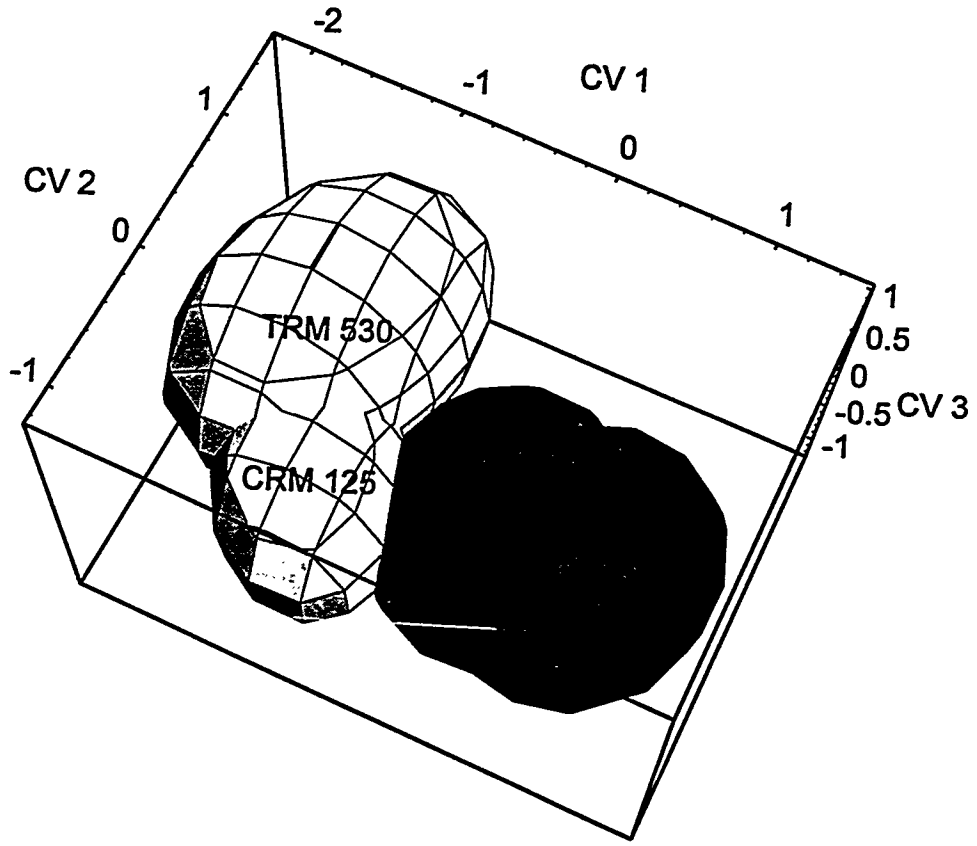


Fig. 3.64. Segregation of integrated health responses for male bluegill sunfish sampled from two reference sites (CRM 125 and TRM 530) and two Poplar Creek sites (PCM 4.6 and 1.0). Ellipsoids represent the mean integrated responses of bass at a site for 1993–1994.

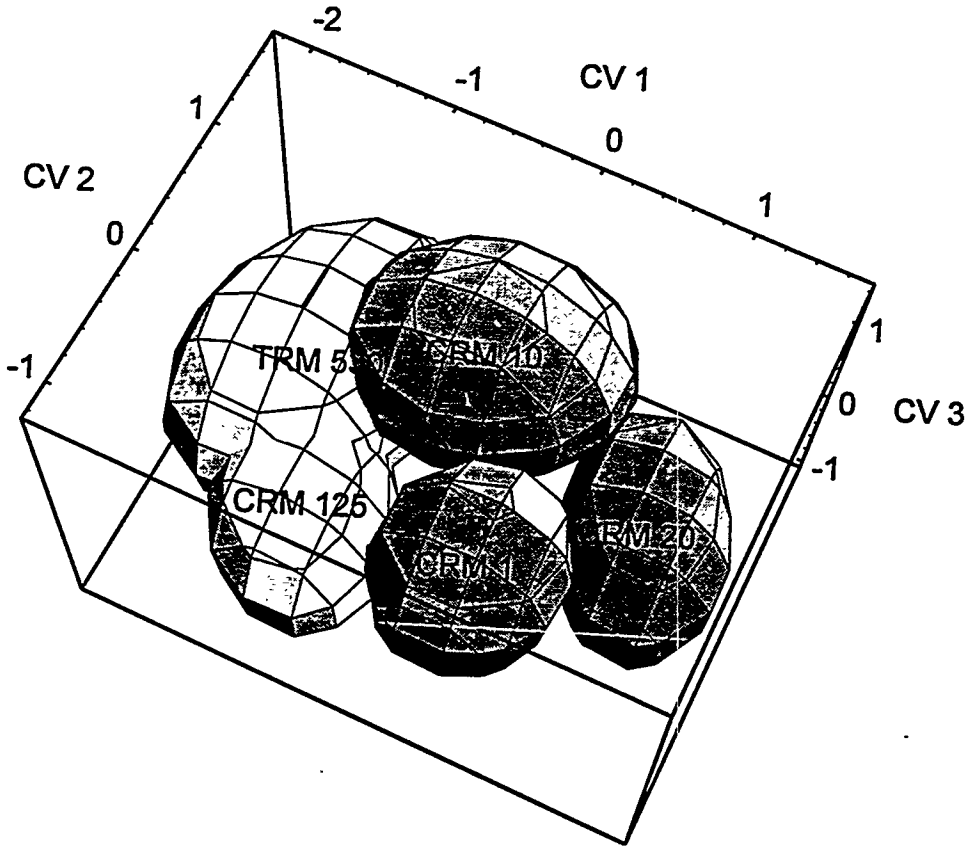


Fig. 3.65. Segregation of integrated health responses for male bluegill sunfish sampled from two reference sites (CRM 125 and TRM 530) and three Clinch River sites (CRM 20, 10, 1). Ellipsoids represent the mean integrated responses of bass at a site for 1993–1994.

3.5.5 Reproductive Effects Assessment

The establishment and maintenance of fish populations is intimately associated with the ability of individuals within a population to reproduce. Reproduction is thus generally considered to be the most critical life function affected by environmental contamination. From a regulatory perspective, the issue of potential contaminant-related effects on fish reproduction has a particular significance because the growth and propagation of fish and other aquatic life is one of the classified uses of the CR/WBR as determined by the state of Tennessee.

Common environmental contaminants, many of which are found in the CR/WBR, are known to have a variety of effects on the reproductive processes of fish, including decreases in the gonadal-somatic index (GSI) upon exposure of fish to PCBs (Monosson et al. 1994) and mercury compounds (Dey 1989, Kirubagaran and Joy 1992). Levels of testosterone and estradiol have been shown to decrease in fish blood following laboratory or environmental exposure to a number of potential environmental pollutants, including PCBs (Sivarajah et al. 1978, Thomas 1990), aromatic hydrocarbons (Johnson et al. 1988), paper and pulp mill effluents (McMaster et al. 1991, Munkittrick et al. 1992), lead (Thomas 1990), cyanide (Ruby et al. 1993), organochlorine or organophosphorus pesticides (Singh and Singh 1987), and crude petroleum (Truscott et al. 1983). Oocyte atresia has been associated with exposure to a number of chemical toxicants, including PAHs (Krarup 1969, Mattison et al. 1983, Takizawa et al. 1984), metals such as cadmium (Mattison et al. 1983), and mixed environmental contaminants (including PCBs and PAHs) (Johnson et al. 1994). Relatively high incidences of oocyte atresia have been previously observed in breeding redbreast sunfish from both upper EFPC (Hinzman 1993) and the Pigeon River, a stream heavily impacted by effluents from a paper and pulp mill (Adams et al. 1992). Abnormally high numbers of atretic oocytes have also been observed in fathead minnows undergoing acid-induced stress (McCormick et al. 1987).

The goal of this task was to evaluate the reproductive health of fish in the CR/WBR downstream of the ORR. This goal was addressed through a comparison of selected reproductive criteria in key fish species collected from the CR/WBR and reference sites. Largemouth bass and bluegill sunfish, both important sport fishes in the CR/WBR system, were the target organisms examined in the field portion of this task.

3.5.5.1 Methods and approach

Largemouth bass and bluegill sunfish were collected from seven sites in Poplar Creek, Watts Bar Lake and Norris Lake at the beginning of the 1993 reproductive season to assess the comparative reproductive status of fish populations at these sites. General measures of reproductive condition examined in both male and female fish included the GSI, a measure of gonad size relative to body size, and plasma concentrations of important reproductive hormones such as estradiol and testosterone. Hormones were measured by standard radioimmunoassay procedures (ER/CR-SOP2170) as adapted from Greeley et al. (1988). At least one testis from each male fish was also examined histologically for evidence of histopathological abnormalities, as well as to assess the developmental status of spermatogenic tissues within the organ. For female fish, reproductive status was quantitatively evaluated by sizing, staging and enumerating all oocytes above a certain threshold size contained within a weighed piece of ovary (ER/CR-SOP2160, as adapted from Greeley et al. 1987, Lin et al. 1989). Measures of reproductive condition derived from this exercise include one or more estimates of fecundity, total counts of vitellogenic ("yolk-laden") oocytes within an ovary, and the prevalence of parasitic lesions and oocyte atresia (oocyte death and/or reabsorption).

Data was log-transformed before statistical analysis. Homogeneity of variance was verified by a Levene's Test (Sokal and Rohlf 1981), an F-distribution test that compares the ratios of the variances from two independent sample populations. If variances proved to be homogeneous, a one-way ANOVA was used to test for site effects for the various reproductive parameters. If the ANOVA rejected a multisample null hypothesis of equal means, then Dunnet's test (Zar 1984) was used to test for significant site differences from the reference in response parameters. For data with non-homogeneous variances, difference between means were examined by the Kruskal-Wallis Test. In all cases, the significance level for rejecting the hypothesis of equal means between sites was set at $\alpha = 0.05$.

3.5.5.2 Results and discussion

Male Largemouth Bass

For male largemouth bass, the study sites with the highest mean GSIs proved to be the two reference sites in the upper reaches of Norris Lake (Table C36: CRM 125 and PRM 30). In contrast, the lowest GSIs were observed at PCM 1 and CRM 20, the two study sites adjacent to the ORR. Since the mean developmental stage of the testes were similar between sites, the smaller relative size of the gonads at PCM 1 and CRM 20 apparently represented a true decrease in male reproductive effort at these sites as compared with reference populations.

Plasma concentrations of the androgenic steroid hormone, testosterone, were also particularly low in male largemouth bass from PCM 1 and CRM 20, averaging approximately half the levels measured in fish from the reference sites (Table C36). The estrogenic steroid hormone, 17β-estradiol, which functions in female fish to regulate the production of yolk proteins, is generally not detectable in male fish. However, production of this typically female reproductive hormone may be stimulated in males upon exposure to compounds that act as either endocrine mimics or blockers. As expected, the mean concentrations of estradiol in plasma samples from the male bass collected for this study were very low at all study sites (sensitivity limit of the assay was 100 pg/mL; nondetects were assigned a value of one-half the sensitivity limit for the purposes of calculating the means). However, the frequencies with which detectable levels of estradiol occurred in male bass did differ markedly between sites. Estradiol was low but measurable in 88% of the males collected from CRM 20, 69% from CRM 0.5, and 41% from CRM 557. Corresponding incidences of detectable levels of estradiol in male bass at other sites were 36% for PRM 30, 11% for CRM 125, and 0% for TRM 570 and PCM 1. 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.

Female Largemouth Bass

The mean GSIs of the female largemouth bass populations examined in this study tended to separate into two groupings (Table C37). One grouping was composed of sites with relatively low GSIs, and included PCM 1, CRM 20, and the two reference sites, CRM 125 and PCM 30. A second grouping was composed of sites with relatively higher mean GSIs, and included TRM 557, TRM 570, and CRM 0.5.

Estimates of fecundity were derived for female largemouth bass in the present investigation by two very different methods (Table C37). The sum of all obviously maturing oocytes, mature eggs, and/or shed follicles (the remnants of eggs left in the ovary following a spawn) in the ovaries of females collected either just prior to or immediately after the first spawn of the breeding season yielded one estimate of fecundity. The advantage of this method is that it probably best represents the actual numbers of eggs spawned by a female fish during any single spawning episode. The primary shortcoming of this method of estimation is that determining which oocytes are "obviously" maturing is inherently subjective.

Despite this limitation, the mean batch fecundities derived from this method of estimation proved to be very similar across all the study sites.

Fecundity was also estimated by a more widely accepted method of enumerating all oocytes greater than 0.75 mm in diameter in female bass collected at the beginning of the breeding season (Table C37). However, there are some potentially significant drawbacks to this method. The choice of oocyte sizes that are included in the estimate are based upon somewhat arbitrary assumptions. Furthermore, this method almost certainly underestimates the actual number of eggs to be spawned during the entire breeding season, for smaller oocytes appear to undergo rapid growth into this size range during the entire duration of the short breeding season (personal observations). Regardless of these shortcomings, however, estimates of fecundity derived by this method are probably the most suitable for comparing different bass populations, because the estimates (although admittedly flawed) are less subjective than those derived by alternative procedures. Using this method of estimation, the mean fecundities of female largemouth bass populations at PCM 1 and CRM 20 were determined to be approximately 20% lower than at any of the other study sites, although there were no statistically significant differences in this parameter between sites.

Oocyte death (atresia) prior to the completion of development is a natural occurrence (Saidapur 1978), but also is associated with exposure of an organism to chemical toxicants (Krarup 1969, Mattison et al. 1983, Takizawa et al. 1984, Johnson et al. 1994). In the present investigation, highest incidences of oocyte atresia were observed in female bass collected from PCM 1.0 and CRM 20 (Table C37), although only for the latter site did the prevalence prove to be statistically greater than in a reference population.

17β-Estradiol has the critical reproductive function in female fish of regulating the production of yolk proteins by the liver for incorporation into vitellogenic oocytes. Plasma concentrations of estradiol were lowest in female bass from CRM 20 and CRM 0.5 (approximately half the values at the reference sites), but again there were no clear statistically significant differences between sites (Table C37).The primary role of testosterone in female fish is less clear, although it serves as a precursor in the production of 17β-estradiol and may also function in regulating spawning behavior (Greeley et al. 1988b). Plasma testosterone concentrations were significantly lower in female bass from CRM 0.5 (compared with both reference sites), CRM 20 (compared with PRM 30), and TRM 557 (compared with PRM 30).

Male Bluegill Sunfish

The GSIs of male bluegill sunfish collected at the beginning of the 1993 breeding season did not differ significantly between sites (Table C38) . However, lowest GSIs were observed at the two Clinch River sites downstream of the ORR (CRM 20 and CRM 0.5), intermediate GSIs at the two Norris Lake reference sites (CRM 125 and PRM 30), and the highest GSIs at PCM 1 and the two Tennessee River sites (TRM 557 and TRM 570). The apparent trend for lower GSIs at CRM 20 and CRM 0.5 is probably due simply to these populations lagging slightly behind populations at the other sites in the progress of testicular development, as suggested by the lower mean testes stages at these sites. No obvious abnormalities in the spermatogenic composition of testicular tissue were observed at any site.

Testosterone concentrations were significantly higher in male bluegill sunfish at PCM 1 and TRM 557 than at one reference site, CRM 125 (Table C38). Otherwise, there was little significant variation between sites in the mean plasma concentrations of either testosterone or estradiol. Measurable estradiol was found in blood samples from male bluegill sunfish collected from TRM 557 (45% incidence), CRM 20 (15%), PCM 1 (18%), and CRM 125 (10%). Estradiol was not detected in male bluegill sunfish at any of the remaining study sites.

Female Bluegill Sunfish

As with male sunfish, the mean GSIs of female bluegill sunfish tended to be higher at PCM 1.0 than at any of the other study sites (Table C39). Similarly, mean fecundity (estimated in the bluegill from the size of an obvious clutch of simultaneously maturing oocytes) was also highest at PCM 1.0, although there were no statistically valid differences in fecundity between any of the study sites.

Oocyte atresia was relatively prevalent at all study sites other than the two Norris Lake reference sites (Table C39: CRM 125 and PRM 30). Highest incidences of atresia were observed in the ovaries of female sunfish from the three study sites nearest the ORR (PCM 1, CRM 20, and CRM 0.5).

Ovarian parasites were more prevalent in bluegill sunfish than in bass (Table C39). Although the highest incidences of parasites were again observed in samples from PCM 1.0, ovarian parasites were also observed in fish from the two reference sites as well as TRM 557.

The lowest mean concentrations of plasma estradiol occurred in female sunfish from PCM 1.0 and CRM 0.5 (Table C39). Plasma concentrations of testosterone were lowest in female sunfish from TRM 557 and PCM 1.0 and significantly greater than reference values at TRM 570.

3.5.6 Summary and Conclusions

An apparent pattern in the relative reproductive condition of largemouth bass populations at the various study areas is evident upon simultaneous examination of site-differences in both male and female reproductive indicators. In both sexes, mean GSIs and plasma levels of testosterone are depressed at PCM 1 and CRM 20 as compared with the corresponding values at either of the two upstream reference sites. Prevalences in male bass of measurable amounts of the typically female sex steroid, 17β-estradiol, are highest at CRM 20. In female bass, fecundity tends to be suppressed at PCM 1 and CRM 20, and plasma concentrations of both estradiol and testosterone are lower than in reference populations, especially at CRM 20. Highest incidences of oocyte atresia also occur at these two sites, again most markedly at CRM 20. The sum of these disparate observations strongly suggest that largemouth bass reproduction is adversely impacted at the two study sites adjacent to the ORR.

A similar trend is not readily apparent in the case of the bluegill sunfish populations examined in this study. In fact, for both males and female sunfish, highest mean GSIs were actually observed at a study site close to the ORR (PCM 1). However, as with the largemouth bass, the highest mean incidences of oocyte atresia in female sunfish were observed at PCM 1, although atretic oocytes were also relatively common at all non-reference study sites.

3.6 SUMMARY OF SITE CHARACTERIZATION

Historical studies indicated that contamination in the CR/PC OU consisted primarily of radionuclides (mostly ^{137}Cs) in sediment downstream of WOC, with greatest inventories being found in areas of sediment accumulation downstream of Poplar Creek. Levels of ^{137}Cs and ^{60}Co have been shown to be much lower in near-shore areas than in areas where water depth and sediment accumulation are greater. Mercury contamination of sediment and biota in Poplar Creek downstream of EFPC was well documented. Arsenic contamination of sediment in Poplar Creek and McCoy Branch was also known, the result of historical fly ash disposal from the K-25 and Y-12 steam plants, respectively. PCB and pesticide contamination was well documented in several fish species in the CR/PC system, WBR, and other East Tennessee waters.

Water. In the current study, mean concentrations of all analytes in surface water were below AWQC at all sites. Maximum concentrations of some analytes exceeded these criteria, but determining the duration of such events was beyond the scope of this study. In general, concentrations of most analytes were highest in summer, the period of least flow and dilution. Organic compounds were rarely detected and consisted entirely of phthalates and PAHs, classes of compounds that are relatively ubiquitous environmental contaminants.

Few metals were detected at levels that exceeded upstream reference values, but the use of sensitive analytical techniques allowed documentation of mean total mercury and total methyl mercury values that were elevated in Poplar Creek downstream of EFPC. Interestingly, the greatest concentrations were found at PCM 1.0 rather than immediately below EFPC; resuspension of sediment in Poplar Creek is the possible cause. Mean concentrations of nitrite/nitrate were significantly higher in Poplar Creek immediately below Mitchell Branch. The same is true of mean gross alpha and gross beta levels in winter.

Although the study design in McCoy Branch did not allow for statistical inference, arsenic concentrations there were the highest detected at any site in the study. Ford et al. (1995) had previously documented elevated levels of arsenic in surface water at McCoy Branch and the migration of dissolved arsenic from sediment during conditions of anoxia at the bottom of the water column.

The toxicity of surface water was evaluated for a number of sites in the CR/PC system. Several aquatic organisms were used in the assessment. The majority of tests revealed no consistent toxic response. However, tests using Japanese medaka to assess embryo-larval survival and the developmental abnormalities revealed consistent toxic responses at PCM 1.0 and 4.3; the effect was also observed in similar tests using redbreast sunfish.

Sediment. The characterization of contaminants in sediment documented increased levels of one or more analytes above reference values at several locations in the CR/PC OU. As expected, concentrations of ^{60}Co in Melton Hill Reservoir were elevated as a result of historical releases from a known source on Braden Branch, unrelated to DOE operations. Several metals (arsenic, selenium, vanadium) associated with coal ash were found at elevated levels in McCoy Branch.

The concentration of ^{137}Cs was elevated throughout the Clinch River downstream of WOC. As expected, highest concentrations were found immediately below WOC and highest inventories were found downstream of Poplar Creek. Previous studies have demonstrated that peak ^{137}Cs concentrations occur at depth in the sediment profile; therefore, contaminant profiles in deep sediment were not measured in this study. Selenium and manganese concentrations were elevated immediately below Grassy Creek. Near-shore concentrations of ^{137}Cs and ^{60}Co were low throughout the River. As expected, highest concentrations were found near the mouth of WOC and declined downstream. Mercury levels are elevated below the mouth of Poplar Creek; previous studies have demonstrated that peak concentrations of mercury also occur at depth in the sediment profile. Organic compounds, including PCBs, were detected infrequently in sediment; the most common analytes detected were again phthalates and PAHs.

Levels of mercury, chromium, copper, and cadmium were elevated throughout Poplar Creek below the mouth of EFPC. Levels of several metals, uranium, and ^{99}Tc were increased. Near the fly ash disposal area in lower Poplar Creek (subreach 3.04), mean concentrations of arsenic, boron, and vanadium were elevated.

Peak concentrations of mercury and methyl mercury in surface sediment were found between PCM 1.0 and the mouth of Poplar Creek, coinciding with peak surface water values. The presence of

elevated concentrations in surface sediment may indicate the recent deposition of mercury contaminated sediment from upstream.

The toxicity of surface sediment to several endpoint species revealed little evidence of toxic effects at any site. The sites at which effects were observed most often were PCM 4.3, CRM 9.0, and CRM 19.0.

Modelling of sediment and contaminant (^{137}Cs) transport and fate in the Clinch River evaluated future (30-yr) contaminant inventories in the river under several flow conditions. Results indicated that only under conditions of a local storm in the WOC watershed would the ^{137}Cs inventory increase in the Clinch River. This increase would be temporary, as sediment deposited by the flood event would quickly be scoured and transported downstream to Watts Bar Reservoir. A regional storm would actually decrease the ^{137}Cs inventory immediately as a result of scouring of existing sediment in the Clinch. Previous modelling had indicated that even with the addition of ^{137}Cs from the Clinch, over the 30 year model period inventories decrease as a result of radioactive decay (DOE 1995).

Biota. Three species of sportfish (largemouth bass, catfish, and bluegill) in the CR/PC system were analyzed for selected metals, PCBs, and pesticides. Radionuclide analyses were not conducted, as previous studies had indicated only low levels of radionuclides in fish flesh. Mercury was found to be elevated above reference values in all three species of fish from Poplar Creek, with concentrations in largemouth bass being highest. However, mean levels in all species were below the FDA action level. As expected, PCBs and chlordane were detected in each species at most sites during the study. Mean concentrations of both contaminants varied widely both spatially and temporally, with multiple sources of each contaminant to the CR/PC being likely.

A suite of biological parameters in largemouth bass and bluegill were measured to assess patterns in the health response of fish to contaminant exposure. In general, fish from Poplar Creek and the in Clinch River immediately downstream of WOC showed signs of toxic effects as measured by one or more of the biological indicators. For example, EROD activity in largemouth bass, histopathology in bluegill, and indicators of organ dysfunction in both bass and bluegill were highest in Poplar Creek. An assessment of overall health using a mulivariate analysis procedure rated bluegill from PCM 4.3 and bass from PCM 1.0, 5.3, and CRM 20 as being in the poorest condition. An analysis of the reproductive health of largemouth bass and bluegill indicated that bass from PCM 1.0 and CRM 20 were in poorer condition than fish from other sites. Bluegill, however, from these sites were generally in good reproductive health.

Analyses of mayflies collected from several sites in the CR/PC system demonstrated the presence of PCBs and several metals, including mercury, in most samples. Shad were found to contain PCBs, although at levels generally less than in sportfish collected at the same sites. Mercury concentrations were found to be significantly elevated in bottom dwelling fish from Poplar Creek. Great blue heron eggs and chicks collected from a rookery on Poplar Creek near the K-25 site had elevated levels of mercury and PCBs relative to those from reference colonies. In a seven month feeding study, mink fed a diet of fish from Poplar Creek adjacent to the ORR developed increased body burdens of mercury and PCBs compared to mink feeding on fish collected from reference sites. These mink also had significantly smaller average litter sizes, evidence of decreased reproductive success. Taken together these results demonstrate that organisms at lower trophic levels (mayflies, shad, forage fish) can serve as a pathway for contaminant transfer to insectivores (e.g., bluegill) and piscivores (e.g., bass, heron, mink).

In summary, increased levels of mercury are known to occur in surface water, sediment, and at several trophic levels of biota in Poplar Creek below EFPC. Toxicity of these waters to Japanese medaka and redbreast sunfish has been demonstrated, and measures of several indices of biological and reproductive health in fish demonstrated that fish from Poplar Creek. It is unclear whether the mercury found in these media is primarily entering from EFPC (and is the focus of upstream remedial action) or whether it is being physically or chemically mobilized from sediment in Poplar Creek. In addition, elevated levels of radionuclides in sediment are found in the Clinch River immediately downstream of WOC, and several of the biological indicators in fish at this location were consistent with contaminant exposure and effects. However, none of the aqueous or sediment toxicity tests clearly demonstrated toxicity in the Clinch River near the mouth of WOC.

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 D 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. The only medium under consideration in the FS is contaminated sediment. However, ARARs for cleanup of contaminated surface water are presented in Appendix D for comparison with surface water sampling results to confirm this decision. There are no ARARs for cleanup of contaminated sediments, although TBC guidance is available for soil/sediment contaminated with polychlorinated biphenyls.

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 area along the Clinch River, Poplar Creek, and McCoy Branch contains several sensitive resources that are protected either by federal or state regulations, including wetlands, floodplains, cemeteries, and state-designated natural areas. Preconstruction activities performed for alternative 3, such as building of access roads or removal of bank vegetation, as well as actual containment activities may trigger certain location-specific ARARs. The location-specific ARARs summarized in Appendix D will not apply to either the no action or the institutional controls alternatives since no such activities will be performed.

The TVA, U.S. Army Corps of Engineers (COE) and the TDEC regulate activities under §26(a) of the TVA Act, the COE 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 D 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). Selection of a particular remedial action at a site will invoke the appropriate action-specific ARARs that 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 D and summarized in Sect. 9.1.2.

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 and 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 in Chap. 8 to develop remedial alternatives for the Clinch River and Poplar Creek.

The Environmental Protection Agency, Region IV, Tennessee Department of Environment and Conservation, and Department of Energy-Oak Ridge agreed in the data quality objectives working group meetings held prior to 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 will be a focused risk assessment. The focus is on contaminants of potential concern (COPCs) that were identified in previous human health risk assessment reports (Table 5.1).

Prior to the Phase 2 data collection that occurred in 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 via the evaluated dredging pathways. Cook et al. (1992) confirmed these contaminants and exposure pathways of potential concern based on CRRI Phase 1 data and also identified chlordane in fish as a COPC. Most recently, the baseline human health risk assessment for the lower WBR OU (DOE 1995) identified PCBs, pesticides, and chlordane in fish and external exposure to ^{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 prior to 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, prior to the compilation of the risk assessment database, existing data were compared to 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	Contaminants of Concern Identified by Hoffman et al. (1991)[a] and Associated Exposure Pathway	Contaminants of Potential Concern Identified by Cook et al. (1992)[b] and Associated Exposure Pathway	Contaminants of Concern Identified In DOE/OR/01-1282&D4 (1995) and Associated Exposure Pathway
Arsenic	Dredging – Agricultural Exposure	Fish Consumption Dredging – Direct Exposure Dredging – Agricultural Exposure	Fish Consumption[d]
Beryllium	Dredging – Agricultural Exposure	Fish Consumption Dredging – Direct Exposure Dredging – Agricultural Exposure	Shoreline use - sediment ingestion[e] Fish Consumption[e] Dredging – Sediment Ingestion[e] Dredging – Meat and Vegetable Ingestion[e]
Cadmium		Dredging – Agricultural Exposure	Dredging – Milk and Vegetable Ingestion[d]
Chromium	Dredging – Agricultural Exposure	Dredging – Direct Exposure	Shoreline use – Sediment Inhalation[e] Dredging – Sediment Inhalation[e] Dredging – Milk and Meat Ingestion[d]
Copper		Fish Consumption Dredging – Agricultural Exposure	
Manganese			Drinking Water Ingestion[d]
Mercury	Dredging – Agricultural Exposure	Fish Consumption Dredging – Agricultural Exposure	Fish Consumption[d] Dredging – Milk, Meat, and Vegetable Ingestion[d]
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[e]

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

Analyte	Contaminants of Concern Identified by Hoffman et al. (1991)[a] and Associated Exposure Pathway	Contaminants of Potential Concern Identified by Cook et al. (1992)[b] and Associated Exposure Pathway	Contaminants of Concern Identified In DOE/OR/01-1282&D4 (1995) and Associated Exposure Pathway
Aroclor-1254[c]	Fish Consumption	Fish Consumption	Fish Consumption[d]
Aroclor–1260[c]	Fish Consumption	Fish Consumption	Fish Consumption[d]
Aldrin		Fish Consumption	Fish Consumption[d]
Chlordane		Fish Consumption	Fish Consumption[d]
4,4'DDE			Fish Consumption[e]
4,4' DDT			Fish Consumption[e]
Dieldrin			Fish Consumption[e]
Lindane			Fish Consumption[e]
^{137}Cesium	Dredging – Direct Exposure Dredging – Agricultural Exposure	Near-shore Direct Exposure Dredging – Direct Exposure	Drinking Water Ingestion[e] Shoreline Use – External Exposure[e] Fish Consumption[e] Irrigation – External Exposure[e] Irrigation – Milk and Meat Ingestion[e] Dredging – External Exposure[e] Dredging – Milk, Meat, and Vegetable Ingestion[e]
^{60}Cobalt	Dredging – Direct Exposure	Dredging – Direct Exposure	Dredging – External Exposure[e] Irrigation – External Exposure[e]
^{90}Strontium		Dredging – Agricultural Exposure	
Uranium, total			Drinking Water Ingestion[e]

Table 5.1. Contaminants of potential concern in the Clinch River as identified in previous screening-level risk assessments (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 $10^{-6} - 10^{-4}$ and hazard quotients of 0.1 – 1.0 in a conservative and nonconservative screening, respectively, as an indicator of COPCs in the Phase 1 CRRI.

[c]Identified via nonconservative screening in each study.

[d]Calculated risk/hazard for these pathways were either $>10^{-4}$ for carcinogens or >1.0 for noncarcinogens (DOE 1995).

[e]Calculated carcinogenic risk for these pathways was between 10^{-5} and 10^{-6} (DOE 1995).

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 for using historical data quantitatively in this risk assessment were implemented. 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 the Oak Ridge National Laboratory (ORNL) and the other DOE facilities have established criteria for the publication of reports that include a specified amount of peer review prior to the release of information to the public, this second criterion was implemented to ensure that only data that had undergone extensive peer review were used.

A brief summary of the QA procedures employed in each study from which data were used in the risk assessment 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 datasets evaluated in this baseline risk assessment, tables of summary statistics have been prepared and are presented in Tables E.1– E.7. 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 dataset 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 dataset 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 PLE mean) is defined as the mean of the sample dataset calculated using 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

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	*Oak Ridge Reservation Environmental Monitoring Report for 1992*, EH/ESH–31/V1. Martin Marietta Energy Systems, Inc., Oak Ridge National Laboratory, Oak Ridge, Tennessee, 1993. (Energy Systems 1993)	• Conducted extensive internal QC programs, participated in several external QC programs (including EPA'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 (Cook 1992) and Phase 2	Phase 1: *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation*, ORNL/ER–152, Holladay, S. K., et al., Oak Ridge National Laboratory, Oak Ridge, Tennessee, 1993. (Holladay 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

greater than zero. Please note, in the case where the frequency of detection is equal to zero, it is not possible to calculate a sample mean.

- 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 or less. 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 the Oak Ridge Reservation Environmental Monitoring Program (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 datasets were combined into one media-specific database. The following paragraphs briefly describe each sampling activity. Summary statistics for the water data are listed in Tables E.1 and E.2.

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 Sect. 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 the 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 consisted of several activities including: investigation of current contaminant sources (labelled W1.1), water-biased event study (labelled W1.1B), contaminant data for toxicity tests (labelled W1.1T), contaminant remobilization from sediments study (labelled W1.1R), and an in-stream distribution of contaminants from point sources study (labelled 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 dataset: 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. The ORREMP analyzes for specific alpha emitters only if the gross alpha counts exceeded 3 pCi/L (0.11 Bq/L). Therefore, isotopic analyses for uranium isotopes and other alpha emitters are available for a subset of the total number of samples.

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

Source	Analyses	Number of Samples per Reach				
		1	2	3	4	7
CRRI Phase I (Cook et al. 1992 and Holladay et al. 1993)	Inorganics	3	2	4	3	0
	Organics	3	2	4	3	0
	Radionuclides	3	2	4	2	0
CRRI 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
CRRI 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
CRRI 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
CRRI 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
CRRI 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

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

Source	Analyses	Number of Samples per Reach				
		1	2	3	4	7
Deep-water data						
CRRI Phase I (Cook et al. 1992 and Holladay et al. 1993)	Inorganics	2	2	2	2	0
	Organics	2	2	2	2	0
	Radionuclides	2	2	2	3	0
CRRI 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
CRRI Phase I (Cook et al. 1992 and Holladay et al. 1993)	Inorganics	2	2	2	2	0
	Organics	2	2	2	2	0
	Radionuclides	4	3	4	4	0
CRRI 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

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). The report *Quality Assurance/Quality Control Summary Report for Phase 1 of the Clinch River Remedial Investigation* (Holladay, et al. 1993) provides a written summary of the data quality.

Subsequent to 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 the Watts Bar Reservoir. 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 Watts Bar Reservoir Interagency Permitting Group (WBRIPG) 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 (labelled 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 WBRIPG (labelled S2). Chapter 3 of this document 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 dataset, which comprises all samples collected at an elevation of 733 ft or greater will be used to assess the shoreline use exposure pathways. The remainder of the cores will be used to assess the deep-water environment via the dredging exposure pathways.

Based on 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" are listed in Table E.3 and for the "deep-water" datasets are listed in Tables E.4 and E.5.

The near-shore environment has historically been defined as the area at summer pool level that is covered by no more than 6 ft of water. However, at the time this report was prepared, water depth information was not available for all of the sediment samples. Therefore, the risk assessors decided to use all grab samples collected as a means of conservatively estimating the concentration in the near-shore sediments. The representative concentrations used to evaluate the near-shore exposure pathways are considered conservative based on the previous studies conducted and the fact that some of the grab samples were collected from locations that are known to be depositional zones in the deep-water environment. Summary statistics for the "near-shore" are listed in Tables E. 3 and E.4 and for the "deep-water" datasets are listed in Tables E.5 and E.6.

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). Locations for sampling are described in Chap. 3. The CRRI Phases 1 and 2 fish samples were analyzed by the Tennessee Valley Authority (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 due to 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 per Reach*				
		1	2	3	4	7
CRRI Phase I (Cook et al.1992 and Holladay et al. 1993)	Inorganics	13	29	35	27	0
	Organics	8	22	36	18	0
	Radionuclides	13	51	54	40	0
CRRI Phase 2- (Chap. 3)	Inorganics	0	66	68	105	0
	Organics	3	61	72	142	0
	Radionuclides	0	0	0	0	0

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

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 Watts Bar Reservoir, 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 dataset due to limited sample size and like characteristics, separate risk calculations were made for each species. Summary statistics were then calculated for each species of fish (Table E.6). Although some individuals consume other parts of fish (fish patties containing bone, fish roe, etc.), 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 that were collected as part of the Oak Ridge Biological Monitoring and Abatement Program. 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 (i.e., within plant boundaries) surface water bodies. Therefore, these geese potentially represent the maximally contaminated members within the goose population. Summary statistics are listed in Table E.7.

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. 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 datasets because blank information was not available. Thus, no historical contaminants were eliminated based on 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 by the laboratory for chlordane provided concentrations for the various chlordane isomers, but did not give a measure for total chlordane. However, based on 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, oxychlordane) were summed and a "chlordane, total" entry was added to the dataset.

5.1.3.4 Radionuclide considerations

The ORREMP surface water monitoring data contained entries for uranium-radioactive while 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 dataset, 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, then that contaminant was eliminated from consideration in the risk assessment. All nondetected contaminants are flagged in each of the media-specific summary statistics tables (Tables E.1–E.7) with an "ND" in the Use Column.

5.1.3.6 Essential nutrients

Essential nutrients (including calcium, copper, iron, magnesium, phosphorus, potassium, and sodium (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 prior to the calculation of risk/hazard. Those chemicals that had a frequency of detection of less than 5% (i.e., one detect in 20 samples) were eliminated from further consideration in the risk assessment due to the uncertainty associated with the accuracy/precision of the available analytical measurements. Chemicals that exhibited a frequency of detection of less than 5% are flagged in the summary statistics tables (Tables E.1– E.7) with a flag of "FD."

5.1.3.8 Summary statistics

The result of the data manipulations described in the preceding sections is a final dataset for each medium that will be evaluated in the baseline human health risk assessment. Summary statistics are provided for all contaminants in Tables E.1–E.7. These tables provide a listing of contaminants by exposure unit (reach) along with their frequency of detection; minimum, maximum, mean, and representative concentration (see Sect. 5.2.3); and any associated use flag. The use flags (ND, +, and -) are provided as an indication of those contaminants that will be 1) evaluated quantitatively (i.e., use flag of +), 2) evaluated qualitatively (i.e., use flag of -), or 3) eliminated from consideration (i.e., use flag of ND or FD). All contaminants that remain after the elimination of the contaminants that were 100% nondetects or were detected in less than 5% of the samples are considered to be 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 which serves as either (1) a point of exposure or (2) 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, or external exposure) or indirect contact (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 pathways based on 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· day) and a radionuclide-specific dose typically measured in (risk per pCi of contaminant) for each exposure pathway.

Based on 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 via dermal contact with the sediment but also via inhalation of any resuspended particulates, inadvertent ingestion, or if the dredge soils were used as garden soil, via ingestion of homegrown produce. Therefore, the exposure assessment must also evaluate 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 based on 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, topography, etc. 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 prior to the initiation of Phase 2 sampling to divide the OU into smaller sections or reaches. The boundaries of each reach were delineated based on the location of a specific physical/topographic feature (i.e., dams, streams or river confluences, etc.). Each of the reaches is described in detail in Chap. 3. Furthermore, each reach was then divided into subreaches based on information about zones of sediment deposition.

Prior to 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 (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 occurs in the summer, but because the area has a temperate climate, some activities are pursued year round. The area also draws a substantial number 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 or 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). The 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, fish consumption, ingestion of waterfowl, exposure to near-shore sediment via shoreline use, exposure to surface water via swimming, irrigation, and the use of deep-water dredged sediments for agricultural purposes. Most of the scenarios included in this risk assessment were previously evaluated in thescreening 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 the 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.

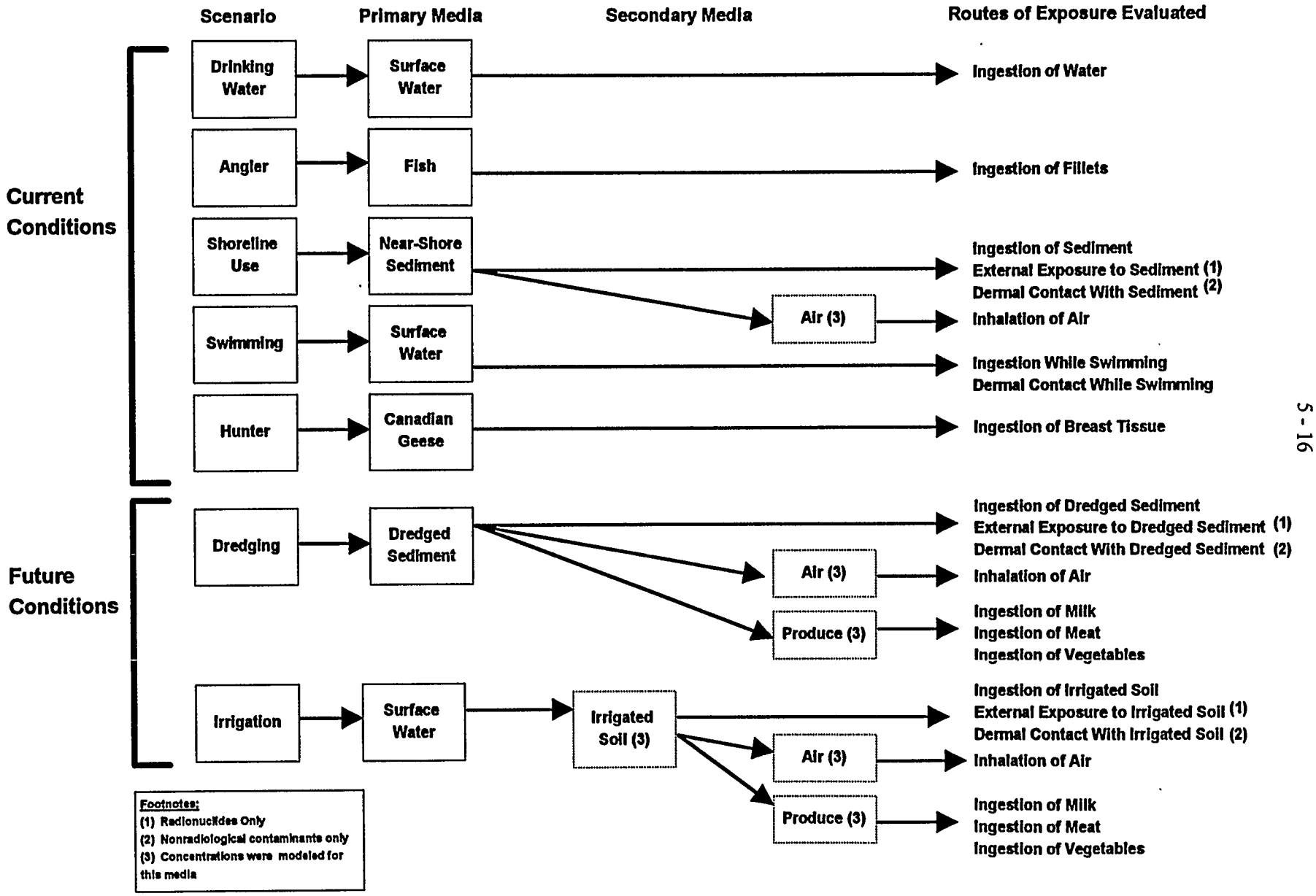


Fig. 5.1. Site model for human health risk assessment.

Exposure scenarios were designated as either current or future in the present risk assessment. Current exposure scenarios evaluate those exposure pathways that are considered to occur or have the potential to occur using measured contaminant concentrations for the media of interest. These scenarios include fish consumption, ingestion of waterfowl, exposure to near-shore sediments via shoreline use, exposure to surface water via 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. This assessment, the deep-water dredging and irrigation scenarios are considered future exposure scenarios which use modelled 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 in the risk assessment.

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, one must determine a representative concentration for each analyte and then calculate chemical intakes for the various exposure pathways identified for the site.

5.2.3.1 Derivation of an exposure concentration

Prior to 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 using 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 95% upper confidence limit (UCL) of the mean is used 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 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. 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 will be revisited in the uncertainty analysis (Sect. 5.5).

Prior to 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 product limit estimate (PLE). The purpose of this evaluation was to determine the most appropriate statistical method to use to derive summary statistics for the Clinch River datasets since the datasets 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, Beauchamp, Brandt, and Hoffman (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 anti-conservative 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 there is no physical basis 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 via 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:

$$Intake = \frac{C \times IR \times EF \times ED}{BW \times AT} \qquad (5.1)$$

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,
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 and EPA 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 based on 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 the chemicals of concern. The pathway-specific equations that were used in this baseline human health risk assessment are listed in Tables E.8–E.12, E.14–E.25, E.27–E.31 and E.34. 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 in 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/day and 1 L/day, 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 E.8.

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, wading, etc. 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 E.3.

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). 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 E.9); (2) dermal contact with near-shore sediment (Table E.10); (3) inhalation of resuspended sediment (Table E.11); (4) external exposure to radiation from gamma-emitting radionuclides in near-shore sediments (Table E.12). 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 E.9 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 per year or half of the standard exposure frequency of 350 days per year. Pathway-specific intake parameters are listed in Table E.10.

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 E.11.

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 E.12. 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 1991b). 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 dataset. The exposure concentrations detected in the surface water samples are listed in the representative concentration column in Tables E.1 and E.2.

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 E.14.

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 E.13). 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 E.15.

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 fisherman (or woman) who regularly catches fish in the Clinch River or Poplar Creek and uses the fish to feed the family. Because many fishermen 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 dataset due to 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 E.16. The contaminant exposure concentrations for this scenario are listed in the representative concentration column in Table E.6.

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, based on personal communication with the 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 1992–1993, 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 days per year or 2 weeks.

A Canada goose on average weighs approximately 12 pounds. Individuals typically eat only the breast tissue which is approximately 1/3 of the weight of the goose or 4 pounds. 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 approximately 4 pounds of goose per year (equals 12

pounds/goose × 2 geese/year × 1/3 × 0.5). The exposure equation and parameter values for the ingestion of Canada geese are listed in Table E.17.

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 using deep sediments for agricultural purposes.

Dredging activities in the Clinch River and Poplar Creek are regulated by the TVA and the U.S. Army's Corps of Engineers 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 the 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 lake-front property owner will request permission from the 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 on shore 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 via 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 via 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 (Tables E.4 and E.5). 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 E.18–E.21).

Dredging scenario: inadvertent ingestion of dredged sediment. Table E.22 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 (Tables E.4 and E.5). 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 E.23.

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 E.24.

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 E.25 . The pathway-specific exposure equations and parameter values listed in Table E.25 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 E.18–E.21). Modelled contaminant concentrations in milk, meat, and vegetables are given in Table E.26. 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 E.27.

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 due to 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 modelled values representing 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 E.28); (2) dermal contact with

irrigated soil (Table E.29); (3) inhalation of resuspended irrigated soil (Table E.30); (4) external exposure to radionuclides in irrigated soil (Table E.31) ; and (5) ingestion of vegetables, milk, and meat produced on irrigated soil (Table E.34).

The irrigation rate was determined from personal communication with the Roane County Extension Agent. The rate of irrigation in Roane County is approximately 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 (1970); however, it is assumed that irrigation in the Clinch River or Poplar Creek area occurs for only 3 months per year, and the value reported by the USDA is normalized for year-round irrigation.

The equation for calculating contaminant concentrations in the irrigated soil after a 30-year build–up period are given in Tables E.18–E.21. 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 E.32 and E.33.

Irrigation scenario: inadvertent ingestion of irrigated soil. Table E.28 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 E.29.

Irrigation scenario: inhalation of resuspended particles from irrigated soil. Table E.30 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 modelled 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 E.31. 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 E.18 through E.21. For inorganics and radionuclides, contaminant–specific uptake coefficients were used to estimate the contaminant concentrations in produce based on 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 E.35). 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 using the regression equation:

$$\log B_v = 1.588 - 0.578 \log K_{ow} \quad (5.2)$$

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 E.33 and 34. It is to be noted 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 modelling equations.

5.3 TOXICITY ASSESSMENT

The purpose of 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 comprises 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 the Appendix 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 exceeds one (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.

Table 5.6. Toxicity information for the Clinch River and Poplar Creek noncarcinogenic COPCs

Chemical	Chronic oral RfD[a,b] (mg/kg-d)	Confidence level[a]	%GI absorp.	%GI source	Chronic oral RfD absorbed[c,d] (mg/kg-d)	Chronic inhal. RfD[a,b,e] (mg/kg-d)	RfD basis (vehicle)[f]	Critical effect[f]	Uncertainty factor[a,f]; Modifying factor
Inorganics									
Antimony	4.0E-04	Low	2.0	BEIAS 1994	8.0E-06	NA	oral	gastrointestinal disorders	UF=1000; MF=1
Arsenic, Inorganic	3.0E-04	Medium	41	BEIAS 1994	1.2E-04	NA	oral	CNS, cardiovascular system, skin	UF=3; MF=1
Beryllium	5.0E-03	Low	1.0	BEIAS 1994	5.0E-05	NA	oral	reduced weight; rickets, lung, skin	UF=100; MF=1
Boron and Borates Only	9.0E-02	Medium	90	BEIAS 1994	8.1E-02	5.7E-03	NA	testicular atrophy, spermatogenic arrest (dog)	UF=100; MF=1
Cadmium (diet)	1.00E-03	High	1.0	BEIAS 1994	1.0E-05	NA	oral	renal toxicity, osteomalacia, osteoporosis	UF=10; MF=1
Cadmium (water)	5.0E-04	High	1.0	BEIAS 1994	5.0E-06	NA	oral	renal toxicity, osteomalacia, osteoporosis	UF=10; MF=1
Chromium VI	5.0E-03	Low	2.0	BEIAS 1994	1.0E-04	NA	oral water	hepatotoxicity, nephrotoxicity, dermatitis	UF=500; MF=NA
Manganese (diet)	1.4E-01	Medium	4.0	BEIAS 1994	5.6E-03	1.4E-05	oral	lethargy, tremors mental disturb-ance, muscle tonus, CNS	UF=1; MF=1

Table 5.6 Toxicity information for the Clinch River and Poplar Creek noncarcinogenic COPCs (continued)

Chemical	Chronic oral RfD[a,b] (mg/kg-d)	Confidence level[a]	%GI absorp.	%GI source	Chronic oral RfD absorbed[c,d] (mg/kg-d)	Chronic inhal. RfD[a,b,e] (mg/kg-d)	RfD basis (vehicle)[f]	Critical effect[f]	Uncertainty factor[a,f]; Modifying factor
Manganese (water)	5.0E-03	Medium	4	BEIAS 1994	2.0E-04	1.4E-05	oral	lethargy, tremors mental disturb-ance, muscle tonus, CNS	UF=1; MF=1
Mercury, Inorganic	3.0E-04	NA	7	BEIAS 1994	2.10E-05	8.6E-05	oral	CNS, kidney, gastrointestinal tract toxicity	UF=1000; MF=NA
Nickel, Soluble Salts	2.0E-02	Medium	27	BEIAS 1994	5.4E-03	NA	oral	nephrotoxicity; reduced weight	UF=100[f]; MF=3[f]
Selenium	5.0E-03	High	44	BEIAS 1994	2.2E-03	NA	oral	methemoglo-binemia, dermatitis	UF=3; MF=NA
Silver	5.0E-03	Low	18	BEIAS 1994	9.00E-04	NA	oral	Argyria	UF=3; MF=1
Vanadium, Metallic	7.0E-03	NA	1	BEIAS 1994	7.0E-05	NA	oral, oral-water	reduced pup weight (rat)	UF=100
Zinc, Metallic	3.0E-01	Medium	20	BEIAS 1994	6.0E-02	NA	oral	gastrointestinal system, kidney, renal function	UF=10; MF=NA
Organics									
Aldrin	3.0E-05	Medium	50	BEIAS 1994	1.5E-05	NA	oral	Liver toxicity (rat)	UF=1000; MF=1
Aroclor-1254	2.0E-05	NA	90	BEIAS 1994	1.8E-05	NA	NA	NA	NA

Table 5.6 Toxicity information for the Clinch River and Poplar Creek noncarcinogenic COPCs (continued)

Chemical	Chronic oral RfD[a,b] (mg/kg-d)	Confidence level[a]	%GI absorp.	%GI source	Chronic oral RfD absorbed[c,d] (mg/kg-d)	Chronic inhal. RfD[a,b,e] (mg/kg-d)	RfD basis (vehicle)[f]	Critical effect[f]	Uncertainty factor[a,f]; Modifying factor
Aroclor-1260	NA	NA	90	BEIAS 1994	NA	NA	NA	NA	NA
Benzo(a)anthracene	NA	NA	31	BEIAS 1994	NA	NA	NA	NA	NA
Benzo(a)pyrene	NA	NA	31	BEIAS 1994	NA	NA	NA	NA	NA
Benzo(b)fluoranthene	NA	NA	31	BEIAS 1994	NA	NA	NA	NA	NA
bis(2-Ethyl- hexyl)Phthalate	2.0E-02	Medium	19	BEIAS 1994	3.8E-03	NA	oral	increased liver and kidney weight in animals	UF=1000; MF=1
Chlordane	6.00E-05	Low	50	BEIAS 1994	3.0E-05	NA	Oral	Regional liver hypertrophy (rat)	UF=1000; MF=1
Cresol, p-	5.0E-03	Low	65	BEIAS 1994	3.3E-03	NA	Oral	Maternal death, hypactivity, cyanosis, eye discharge, and respiratory system distress (rabbit)	UF=1000;
DDD	NA	NA	70	BEIAS 1994	NA	NA	NA	NA	NA
DDE	NA	NA	70	BEIAS 1994	NA	NA	NA	NA	NA
DDT	5.0E-04	NA	70	BEIAS 1994	3.5E-04	NA	NA	NA	NA

Table 5.6 Toxicity information for the Clinch River and Poplar Creek noncarcinogenic COPCs (continued)

Chemical	Chronic oral RfD[a,b] (mg/kg-d)	Confidence level[a]	%GI absorp.	%GI source	Chronic oral RfD absorbed[c,d] (mg/kg-d)	Chronic inhal. RfD[a,b,e] (mg/kg-d)	RfD basis (vehicle)[f]	Critical effect[f]	Uncertainty factor[a,f]; Modifying factor
Dibenzo(a,h)anthracene	NA	NA	31	BEIAS 1994	NA	NA	NA	NA	NA
Indeno(1,2,3-cd)pyrene	NA	NA	31	BEIAS 1994	NA	NA	NA	NA	NA

ATSDR = Agency for Toxic Substances and Disease Registry, Public Health Services (1987-1990); NA = No data available or data inconclusive; GI = Gastrointestinal (%GI = percent gastrointestinal absorption); RfD = Reference dose; CNS = Central nervous system.

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

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

[c] The absorbed RfD = (RfD X %GI absorption); the absorbed RfD is used for dermal pathway calculations. RfD absorbed = RfD (i.e, %GI = 100) when the %GI absorption value is unknown (NA), and when the %GI is greater than 80.

[d] The chronic oral absorbed RfD is used for the dermal contact pathway calculations (RfD oral absorbed = RfD oral x %GI).

[e] RfCs were converted to units of mg/kg-d (i.e., RfD units) by using the inhalation rate and body weight of an adult [i.e., RfC x 20m^3/d x (1/70kg) = RfD] (EPA 1989a).

[f] BEIAS (Biomedical and Environmental Information Analysis Section) Toxicity Profiles (BEIAS 1993).

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 less than 7 years, subchronic RfDs were not used to evaluate exposure to children because the subchronic RfDs are derived based on 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 SF, which is a chemical-specific value based on carcinogenic dose-response data]. Because the SFs are the 95th percentile upper confidence limit 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 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 BHHRA, no adjustments were made to chemicals with %GI $\geq$ 80%.

Table 5.7. Toxicity information for the Clinch River and Poplar Creek carcinogenic COPCs

Chemical	Oral slope factor[a] (kg-day/mg)	%GI absorp.	%GI source	Oral slope factor absorbed[b] (kg-day/mg)	TEF[c]	Inhalation slope factor[a] (kg-day/mg)	EPA class[d]	Type of cancer
Inorganics								
Antimony	NA	2.0	BEIAS 1994	NA	NA	NA	NA	NA
Arsenic, Inorganic	NA	41	BEAIS 1994	NA	NA	5.0E+01	A	skin, liver, bladder, respiratory, gastrointestinal
Beryllium	4.3E+00	1.0	BEIAS 1994	4.3E+02	NA	8.4E+00	B2	breast, bone, uterus, lung
Boron and Borates Only	NA	90	BEIAS 1994	NA	NA	NA	NA	NA
Cadmium (diet)	NA	1.0	BEIAS 1994	NA	NA	6.1E+00	B1	respiratory tract, lung
Cadmium (water)	NA	1.0	BEIAS 1994	NA	NA	6.1E+00	B1	respiratory tract, lung
Chromium VI	NA	2.0	BEIAS 1994	NA	NA	4.1E+01	A	tumors, lung
Manganese (diet)	NA	4.0	BEIAS 1994	NA	NA	NA	D	NA
Manganese (water)	NA	4.0	BEIAS 1994	NA	NA	NA	D	NA
Mercury, Inorganic	NA	7.0	BEIAS 1994	NA	NA	NA	D	NA
Nickel, Soluble Salts	NA	27	BEIAS 1994	NA	NA	NA	D	NA
Selenium	NA	44	BEIAS 1994	NA	NA	NA	D	NA
Silver	NA	18	BEIAS 1994	NA	NA	NA	D	NA
Vanadium, Metallic	NA	1	BEIAS 1994	NA	NA	NA	D	NA
Zinc, Metallic	NA	20	BEIAS 1994	NA	NA	NA	D	NA
Organics								
Aldrin	1.7E+01	50	BEIAS 1994	3.4E+01	NA	1.7E+01	B2	liver carcinoma

Table 5.7. Toxicity information for the Clinch River and Poplar Creek carcinogenic COPCs (continued)

Chemical	Oral slope factor[a] (kg-day/mg)	%GI absorp.	%GI source	Oral slope factor absorbed[b] (kg-day/mg)	TEF[c]	Inhalation slope factor[a] (kg-day/mg)	EPA class[d]	Type of cancer
Aroclor-1254	NA	90	BEIAS 1994	NA	NA	NA	B2	liver, biliary tract, gall bladder
Aroclor-1260[e]	7.7E+00	90	BEIAS 1994	8.6E+00	NA	NA	B2	liver, biliary tract, gall bladder
Benzo(a)anthracene	7.3E-01	31	BEIAS 1994	2.43E+00	0.1	NA	B2	tumors
Benzo(a)pyrene	7.3E+00	31	BEIAS 1994	2.4E+01	1.0	NA	B2	stomach tumors
Benzo(b)fluoranthene	7.3E-01	31	BEIAS 1994	2.4E+00	0.1	NA	B2	tumors
bis(2-Ethylhexyl)-phthalate	1.4E-02	19	BEIAS 1994	7.4E-02	NA	NA	B2	Liver neoplastic nodule and hepatocellular carcinoma
Chlordane	1.3E+00	50	BEIAS 1994	2.6E+00	NA	1.3E+00	B2	(O): Liver carcinoma (mouse); (I): CNS depression
Cresol,p-	NA	65	BEIAS 1994	NA	NA	NA	C	NA
DDD	2.4E-01	70	BEIAS 1994	3.4E-01	NA	NA	B2	Liver tumors (mouse)
DDE	3.4E-01	70	BEIAS 1994	4.9E-01	NA	NA	B2	Hepatocellular carcinoma and hepatoma (mouse)
DDT	3.4E-01	70	BEIAS 1994	4.9E-01	NA	3.4E-01	B2	(O): Liver tumors (mouse, rat)
Dibenzo(a,h)anthracene	7.3E+00	31	BEIAS 1994	2.4E+01	1.0	NA	B2	tumors

Table 5.7. Toxicity information for the Clinch River and Poplar Creek carcinogenic COPCs (continued)

Chemical	Oral slope factor[a] (kg-day/mg)	%GI absorp.	%GI source	Oral slope factor absorbed[b] (kg-day/mg)	TEF[c]	Inhalation slope factor[a] (kg-day/mg)	EPA class[d]	Type of cancer
Indeno(1,2,3-cd)pyrene	7.3E-01	31	BEIAS 1994	2.4E+00	0.1	NA	B2	tumors

NA = No data available or data inconclusive.
TEF = Toxicity equivalency factor.
[a] Based on Integrated Risk Information System (IRIS) (EPA 1994) or Health Effects Assessment Summary Tables (HEAST), (EPA 1993b).
[b] The absorbed oral slope factor is used for the dermal contact pathway calculations; the absorbed oral SF = (SF/%GI).
[c] The oral SFs for these PAHs are derived by multiplying the benzo(a)pyrene oral SF (7.3E+00) by the chemical specific toxicity equivalency factor (TEF) (EPA 1993a).
[d] EPA Weight of Evidence Classification System for Carcinogenicity-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.
[e] For these Aroclors, the oral SF (7.7E+00) for polychlorinated biphenyls was used.

Table 5.8. Toxicity information for exposure to the Clinch River and Poplar Creek radionuclide COPCs

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
Cesium-137+D	2.0E-06	2.8E-11	1.9E-11	D	A	various
Cobalt-60	8.6E-06	1.5E-11	1.5E-10	Y	A	various
Strontium-90+D	0	3.6E-11	6.2E-11	D	A	various

[a] Based on Health Effects Assessment Summary Tables (HEAST) (EPA 1993b).
[b] The radionuclide slope factors include contributions from daughter products.
[c] Lung clearance classification recommended by the International Commission on Radiological Protection (ICRP): Y = year; W = week; D = day; G = gas.
[d] EPA Weight of Evidence Classification System for Carcinogenicity-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.

Minor adjustments to the oral RfDs and SFs, used in the dermal assessments only, which favor conservatism, were made for this BHHRA. 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 prior to making decisions based on 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 endpoint measured (unless adequate human health data are available). From these studies, the experimental exposure representing 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 NOAEL (or LOAEL) for the critical toxic effect by dividing the NOAEL (or LOAEL) by uncertainty factors. These factors usually are in multiples of 10, 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. Based on weight-of-evidence studies using human and laboratory animal research, chemicals fall into one of five groups (EPA 1989a, 1993b): (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 95th percentile upper confidence limit 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 contaminants of potential concern 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 chemicals of concern 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 resulting 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 excess cancer risk 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 1989). The dose response information used for carcinogens in the present risk assessment is in the form of EPA–recommended slope factors (SFs) provided in the HEAST and the Integrated Risk Information System (IRIS) and supplied to the CR–ERP by the Biomedical Information and Analysis Section, Health Sciences Research Division, ORNL. 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 (OSWER 1991). 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 excess cancer risk for carcinogens is calculated by multiplying the calculated intake/dose for each contaminant by the appropriate EPA–approved slope factors. This estimate of excess cancer risk 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 via summing excess cancer risks 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 using reasonable maximum exposure assumptions for either current or future land use exceeds the 10^{-4} lifetime excess cancer risk 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 reasonable maximum exposure for both current and future land use is less than 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 of greater than 10^{-4}. Within the pathways that have a total risk of greater than 10^{4}, the carcinogenic chemicals of concern are defined as those contaminants whose excess cancer risk is greater than 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 toxicity values referred to as 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 1989). RfDs used in this document were obtained from the Biomedical and Information Analysis Section, Health Sciences Research Division, Oak Ridge National Laboratory, who in turn, obtained the toxicity values from an on–line search of HEAST and IRIS. 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 via ingestion, inhalation, and dermal contact. For this

risk assessment, pathways of concern for noncarcinogenic effects are defined as those with a HI > 1.0. Within the pathways that have a 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 data bases used in each scenario are given in the following text. Prior to 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 excess cancer risk of greater than 10^{-6} or a hazard quotient of >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 of >0.1 for one pathway in a given scenario, the excess cancer risk 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. The methods, equations, and parameter values in the human health risk assessment for radiological and nonradiological contaminants in drinking water are listed in Table E.8.

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 E.36. For the drinking water ingestion pathway, none of the reaches evaluated exhibited a pathway excess cancer risk of greater than 10^{-4}. Thus, no carcinogenic chemicals of concern were identified for the drinking water ingestion pathway for reaches 1, 2, 4, or 7.

Noncarcinogenic hazard indices were calculated for both an adult and a child for the drinking water ingestion pathway. As is evidenced in Table E.37, 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 E.38). Several inorganic contaminants (arsenic, antimony, manganese, and nitrate) were found to have an individual hazard quotient of greater than 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 more than 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 due to 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 excess cancer risk of greater than 10-4 (Table E.39). Therefore, no carcinogenic chemicals of concern were identified.

Noncarcinogenic contaminants whose hazard indices (i.e., the sum of the hazard quotients for a particular pathway) exceeded 1.0 are listed in Table E.40 for adults and E.41 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 less than 1.0 and no chemicals of concern were identified. In subreach 3.04, the HI = 1.0 with manganese contributing ~50% of the total hazard. For the child drinking water ingestion pathway, several inorganic contaminants (arsenic, antimony, manganese) and one organic contaminant (PCB-1254) were found to have an individual hazard quotients of greater than 0.1 for this pathway. In addition, within subreach 3.03, the child's hazard index for nitrate was 0.2 for a child. 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 PCB-1254 (40% and 29%, respectively). For subreach 3.02, manganese contributed 53%, arsenic 31%, and antimony 15% of the HI. For subreach 3.03, manganese contributed 65%, arsenic 18%, and nitrate 12% of the HI. Each of these chemicals (arsenic, antimony, manganese, nitrate, and PCB-1254) are considered to be chemicals of concern 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" dataset consists of samples collected at elevations greater than 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 E.42-E.44.

For the subreaches evaluated, the only subreach with a pathway excess cancer risk of greater than 10^{-4} was subreach 3.02 (Table E.42). The ECR for the inhalation pathway for subreach 3.02 was 1.5×10^{-4}. The following chemicals had an ECR $>10^{-6}$: chromium, arsenic, ^{234}U, ^{238}U, and ^{235}U.

The only chemical identified as having a HQ of greater than 1.0 for the adult shoreline use sediment pathways was manganese via the inhalation of resuspended sediment. As is evidenced in Table E.43, the HQ for the inhalation of resuspended manganese is greater than 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 a 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 greater than 97% of the HI.

Barium was also found to have a HQ >0.1 for the child inhalation pathway (Table E.44) for all subreaches except 3.01, 4.01, and 4.02, but its relative contribution to the HI was less than 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 E.14 and E.15.

Carcinogenic and noncarcinogenic chemicals of concern for the Clinch River swimming pathways. No carcinogenic or noncarcinogenic chemicals of concern 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 chemicals of concern 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 a HQ of 1.6 for an adult and a HQ of 2.4 for a child via dermal contact (Table E.45 and E.46). 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 (Patnaik, 1992). As a result, the phthalate esters are commonly found as environmental contaminants and are, therefore, not considered to have been introduced to the environment solely as a result of activities on the Oak Ridge Reservation. Therefore, for this baseline risk assessment this phthalate ester is not considered to be chemical of concern. In subreach 3.01, one Aroclor, PCB-1254, was determined to have a hazard quotient of 1.1. This PCB will be considered a contaminant of concern.

5.4.3.4 Fish consumption scenario

Nine contaminants detected in fish fillets produced cancer risks $>10^{-6}$ (Table E.47). 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 excess cancer risk 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 condern for fish include two inorganics (arsenic and beryllium) 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 chemicals of concern (Tables E.48 and E.49) that contributed to a pathway hazard index of greater than 1.0 included several inorganic contaminants (arsenic, mercury, and selenium), two pesticides (chlordane and 4,4-'DDT), and PCB-1254. The highest HQs for the fish pathway were for the ingestion of PCB-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. One composite sample of bluegill, a species of fish previously not considered to be an effective bioindicator of PCB contamination based on their feeding habits and relatively low lipid content, was collected and analyzed for PCBs in reach 4 only. The excess cancer risk calculated for PCB-1260 in these bluegill was 9.8×10^{-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 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 will be performed for the chemicals of concern for this pathway. The results of the uncertainty analysis are contained in Sect. 5.5.

5.4.3.5 Waterfowl consumption

As was 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 E.50–52) indicate that no carcinogenic or noncarcinogenic chemicals are of concern for this pathway.

5.4.4 Risk Characterization Results for Future Conditions

5.4.4.1 Irrigation scenario

As was previously 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 excess cancer risks calculated for each pathway for each reach within the Clinch River (Table E.53) and all subreaches within Poplar Creek (Table E.56) were all less than 10^{-4}. As a result, no carcinogenic chemicals of concern 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 E.54) via 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 E.55) are (1) nitrate via ingestion of milk, meat, and vegetables, (2) arsenic and antimony via ingestion of leafy vegetables, (3) nickel via ingestion of milk, and (4) acetone via 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 previously 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 as contaminants of concern in this risk assesment. Noncarcinogenic chemicals identified as having a HI >1.0 for the child irrigation pathways for the Poplar Creek subreaches are shown in Table E.58. The hazard index for the ingestion of milk by a child for Subreaches 3.01 and 3.04 of Poplar Creek indicated that the current concentrations of PCB-1254 when modelled to a concentration in milk produced by cattle grazing on irrigated soil and two inorganics, nitrate and nickel would result in a hazard quotient of greater than 1.0. The hazard index for the ingestion of vegetables grown in irrigated soil indicated that two inorganics (arsenic and nitrate) and one organic (acetone) would result in a 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 a HQ>1.0.

5.4.4.2 Dredging scenario

As was previously discussed in Sect. 5.2, seven exposure pathways were evaluated in the dredging scenario. The deep-water sediment data were aggregated into two datasets for risk assessment purposes. The first dataset evaluates the Clinch River reaches 1 and 7, while the second evaluates the following subreaches: 2.04, 3.01, 3.02, 3.03, 3.04, 4.01, 4.02, 4.03, and 4.04.

The excess cancer risks calculated for direct dredging pathways (ingestion, inhalation, external, and dermal) for Clinch River reaches 1 and 7 were less than 10^{-4} for all pathways except external exposure.

In reach 1, ^{60}Co and ^{137}Cs had excess cancer risks via external exposure of 1.3E-04 and 1.2E-05, respectively (Table E.59). 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. The ANC was under the jurisdiction of the U.S. Atomic Energy Commission until 1965 at which time the state of Tennessee assumed jurisdiction. The 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 excess cancer risks calculated for the meat and vegetable ingestion pathways were less than 10^{-4} for both Clinch River reach 1 and 7. The ingestion of milk was greater than 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 excess cancer risk of $>10^{-6}$ (Table E.59).

The noncarcinogenic hazards calculated for the Clinch River reaches 1 and 7 (Table E.60) indicate that for the adult dredging pathways exposure to manganese via the sediment inhalation and vegetable ingestion for reach 7 has a 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 is discussed in Chap. 3., the highest average surface sediment concentrations of manganese are found in Reaches 7 (McCoy Branch) and 8 (Walker Branch) suggesting 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 chemical of concern, 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 E.61) indicate that exposure to children via the ingestion of meat, milk, and vegetables would pose a potential threat of noncarcinogenic health effects. The chemicals of concern for these pathways are: 4,4'-DDT (milk), arsenic (milk, meat , and vegetables), mercury (vegetables), chromium (meat and vegetables), nickel (milk and vegetables), selenium (milk and meat), zinc (milk, meat, and vegetables), vanadium (milk), manganese (milk and vegetables).

The excess cancer risks calculated (Table E.62) 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), ^{238}U (subreach 3.04), and ^{90}Sr (subreach 4.04) would result in an pathway excess cancer risk of $>10^{-4}$. In addition, inhalation of resuspended sediments for subreach 3.04 indicate that concentrations of arsenic, chromium, Cadmium, ^{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 excess cancer risks (Table E.62) are PAHs. Polycyclic aromatic hydrocarbons [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 PAHs are considered to be indicative of background and are not considered to be chemicals of concern. Therefore, the only carcinogenic chemicals of concern identified for the indirect pathways (Table E.62) in the dredging scenario are beryllium (vegetables), ^{137}Cs (milk), ^{90}Sr (milk, meat, and vegetables), ^{99}Tc (milk and vegetables), ^{234}U (milk), ^{238}U (milk), arsenic (milk, meat, and vegetables), and PCB-1260 (milk, meat, and vegetables).

The noncarcinogenic chemicals of concern for the adult and child dredging pathways for the Clinch River and Poplar Creek subreaches are listed in Tables E.63 and E.64. Twelve inorganics and two organics were found to have hazard quotients greater than 0.1 within those pathways having a hazard index of greater than 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 E.5 and E.6); 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 (Energy Systems 1993), lead was included in the risk assessment because of the public's concern with lead toxicity. An EPA uptake/biokinetic model (EPA 1991c) 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, modelled 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 using 44% as the average proportion (EPA, 1990) and a reasonable maximum proportion of 75% (EPA, 1993) 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, 1993). 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 using 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 chemical of concern for Reach 1.

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]	
								Avg.	Max.	Avg.	Max.
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 i 25%; the reasonable maximum(worst case) consumption is 40% (EPA 1993).
[c]Average consumption of home-grown beef is 44% (EPA 1989a); the reasonable maximum (worst case) consumption is 75% (EPA 1993).
[d]Geometric mean of blood lead level assuming average (Avg.) or maximum (Max.) consumption of homegrown vegetables and beef.
[e]Probability of the blood lead level exceeding 10 μg/dL assuming average (Avg.) or maximum (Mas.) consumption of home-grown vegetables and beef.

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 chemical of concern for Reach 2.

5.4.6 Risk characterization summary

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 $>10^{-4}$: the PCB mixture, 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.

PCBs are known to be released from the ORR; however, it is well documented that many rivers and reservoirs of eastern Tennessee are contaminated with PCBs (TVA 1988). Although past releases from ORR have contributed to the PCB concentrations found in the Clinch River and Poplar Creek, it has not been established that ORR is the major contributor.

Other carcinogenic compounds found in fish tissue with excess cancer risks within the range of concern included several pesticides (Aldrin, Chlordane, 4,4'-DDE, and 4,4'DDT). These pesticides were widely used until the 1970's when their use and production was banned. The widespread use of these pesticides prior to their being banned has resulted in residual levels of these contaminants in the environment. The source of these contaminants in the Clinch River and Poplar Creek is, therefore, unknown and these chemicals will not be considered in any risk management decisions.

Relatively large releases of radionuclides from the ORR have occurred in the past. However, the only scenarios for which radionuclides pose an unacceptable excess cancer risk are the fish consumption, dredging, and shoreline use scenarios. The excess cancer risks for two radionuclides (^{137}Cs, and ^{90}Sr) detected in fish were greater than 10^{-6}.

For the dredging scenario, several radionuclides posed a risk of greater than 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 excess cancer risk of greater than 10^{-6} for the inhalation of resuspended dredged sediment. In addition to the aforementioned radionuclides, ^{90}Sr and ^{99}Tc were also found to contribute to the excess cancer risk 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 excess cancer risk of greater than 10^{-5} for the ingestion of meat and an excess cancer risk of greater than 10^{-6} for the ingestion of vegetables pentachlorophenol was infrequently detected, but poses a greater than 10^{-6} excess cancer risk 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 into the Clinch River via EFPC and Poplar Creek, produced HQs >1.0 for the ingestion of bluegill, catfish, and largemouth bass by a child. The hazard quotients for several inorganics (arsenic, mercury, and selenium) detected in fish were also determined to pose a noncarcinogenic hazard, mainly via the ingestion of bluegill and largemouth bass.

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.

The results of the drinking water ingestion scenario indicate that multiple noncarcinogenic chemicals (arsenic, antimony, manganese, nitrate, and PCB-1254) pose an unacceptable hazard via 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 chemicals of concern were identified for the swimming pathways nor for dermal contact or external exposure to near-shore sediments. The noncarcinogenic chemicals of concern for the shoreline use scenario via the inhalation of resuspended sediments are: barium and manganese. The carcinogenic chemicals of concern are arsenic, chromium, ^{234}U, ^{235}U, and ^{238}U.

No carcinogenic chemicals of concern were determined for the irrigation scenario. Noncarcinogenic chemicals of concern for this scenario are nitrate, arsenic, antimony, nickel, and acetone.

5.5 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 non-carcinogens. 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). 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 using subjective judgment in the absence of directly relevant data. The subjective confidence intervals for the excess lifetime cancer risks and the hazard indices 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 having 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; 1981b), IAEA (1989), and Iman and Helton (1991).]

5.5.1 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 above 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 5.10 for noncarcinogens and Table 5.11 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.

Table 5.10. Noncarcinogenic chemicals of concern for the fish ingestion pathway

			Risk Estimate	
Pathway	**Contaminant**	**Reach**	**Adult**	**Child**
Bluegill	Mercury	3	1.1	2.8
		4	---	1.2
Largemouth bass	Mercury	2	---	1.8
		3	1.8	4.7
		4	---	2.2
	Chlordane	2	---	2.3
		3	---	2.5
		4	---	2.3
	Aroclor-1254	2	6.8	18
		3	14	35
		4	8.1	21
Catfish	Mercury	4	---	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

Table 5.11. Carcinogenic chemicals of concern for the fish ingestion pathway.

Contaminant	Pathway	Reach	Risk Estimate
PCB-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}Cesium	*Largemouth Bass*	2	1.5E-04

5.5.1.1 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 using 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 (LCLs), and the 95% UCLs were obtained.

The subjective confidence intervals for the excess lifetime cancer risk resulting from the contamination of ^{137}Cs are summarized in Table 5.12. 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 6.2E04.

The subjective confidence interval given in Table 5.12 provides valuable information that can be used to guide decision-making. For example, if a 5% LCL 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% UCL is above the standard, but the 50th percentile is below the standard, further study should be recommended for those parameters which dominate the overall uncertainty. 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 versus benefits analysis. Because the 95% UCL 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.

Table 5.12. Results obtained for the excess lifetime cancer risk from ^{137}Cesium in the Clinch River and Poplar Creek

Species	Radionuclide	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
Largemouth Bass	^{137}Cesium	1.5E-05	9.3E-05	6.2E-04

The subjective confidence intervals for the excess lifetime cancer risk resulting from Aroclor-1260 and chlordane are summarized in Table 5.13. 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).

Table 5.13. 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
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.

The highest potential excess lifetime cancer risk results from eating fish contaminated with Aroclor-1260. The subjective confidence intervals given in Table 5.13 provide valuable information that can be used to guide decision-making. For example, since the 5% lower confidence limit is above a 10^{-4} life-time

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} life-time 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 PCB's in animals is applicable to human populations).

5.5.1.2 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 5.14. The sensitive parameters are ranked according to priority for further study.

Table 5.14. Results obtained for the sensitivity analysis performed for excess lifetime risk from ^{137}Cs contamination associated with the human ingestion of fish from the Clinch River and Poplar Creek

Species	Sensitive Parameter	Sensitivity Index	Rank
Largemouth Bass	I_f: meal/day	35.3 %	1
	Cf: Bq/kg	26.5 %	2
	RCF: risk/Sv	23.6 %	2

The most sensitive parameters for lifetime cancer risk from ^{137}Cs are (1) the number of fish-meals per day, (2) the concentration of ^{137}Cs in largemouth bass for Reach 2, and (3) the risk conversion factor. These results indicate that the uncertainty involved with the excess lifetime cancer risk 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, etc., rather than further sampling of fish. Such surveys would aid in the development of more definitive estimates of subjective confidence intervals for the number of fish-meals per day, the exposure duration, and the amount of fish eaten per fish-meal. For example, if the number of fish-meals per day were known exactly, then the uncertainty in the overall excess lifetime cancer risk would be decreased by almost a factor of two.

The uncertainty in the overall excess lifetime cancer risk is also very sensitive to the concentration of ^{137}Cs in largemouth bass for Reach 2 and the risk conversion factor. The only method of decreasing the uncertainty of the concentration would be to obtain more fish samples. In addition, until better estimates of the cancer risk at high dose and high dose rate exposures and/or significant data at low dose and low dose rate exposures are obtained, it is unlikely that the uncertainty in the risk conversion factor for radionuclides can be lowered.

The top three sensitivity indices for the total risk resulting from Aroclor-1260 and chlordane contaminated fish are presented in Table 5.15. The sensitive parameters are ranked according to priority for further study.

Table 5.15. Results obtained for the sensitivity analysis performed for the Total Cancer Risk from select carcinogens associated with the human ingestion of fish from the Clinch River and Poplar Creek

Species	Sensitive Parameter	Sensitivity Index	Rank
Catfish (Reach 1)	I_f: meal/day	64.4 %	1
	I_m: kg/meal	12.5%	2
	Aroclor-1260 SF: (mg/kg-day)$^{-}$	11.8%	2

The most sensitive parameter for the total excess lifetime cancer risk from select carcinogenic chemicals for catfish in Reach 1 are consistent with the risks from other reaches and species. This parameter is the number of fish-meals per day. The overall estimate of uncertainty was moderately sensitive to the amount of fish eaten per meal and the Aroclor-1260 slope factor.

These results agree with the results obtained for the radionuclide analysis. The uncertainty involved with the excess lifetime cancer risk from the human ingestion of fish from the Clinch River and Poplar Creek would be most effectively lowered by taking local surveys of fish consumption patterns, etc., rather than taking further samples of fish.

5.5.1.3 Subjective confidence intervals obtained for the HI from select non-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 using 500 iterations of LHS to obtain subjective probability distributions for the health hazard associated with human (adults and children) ingestion of fish from the Clinch River and Poplar Creek which are contaminated with methyl mercury, chlordane, and Aroclor-1254. From these distributions, the medians, the 5% LCLs, and the 95% UCLs were obtained.

The subjective confidence intervals obtained for the health risk to adults from the fish are summarized in Table 5.16. The median HQ for chlordane in the catfish of Reach 1 for adults, for instance, is 1.9; there is 90% confidence that the true but unknown HQ lies between 0.1 and 22.5 (or there is 95% confidence that the true but unknown value does not exceed 22.5). The subjective confidence intervals obtained for the health risk to children from the fish are summarized in Table 5.17. The median HQ for chlordane in the catfish of Reach 1 for children, for instance, is 1.6; there is 90% confidence that the true but unknown HQ lies between 0.1 and 40.7.

The highest potential HI for children occurs from the catfish of Reach 4 with a median of 7.5. There is 95% subjective confidence, given the assumptions used in this analysis, that the true but unknown total HI for children from non-carcinogens in catfish does not exceed 66.6.

The wide ranges of the HQs for chlordane in all of the species are primarily due to the large uncertainty associated with its RfD. The subjective confidence intervals given in Tables 5.16 and 5.17 provide valuable information that can be used to guide decision-making. For example, if a 5% LCL is above a regulatory standard of concern, then remediation is most likely needed. If the 95% UCL is below the standard, remediation is most likely not required. If the 95% UCL is above the standard, but the 50th percentile is below the standard, further study should be recommended for the parameters that dominate the overall uncertainty. However, if the 50th percentile is above the standard, further study may still be recommended, although under some circumstances it may be prudent to proceed with remediation. In all cases, the 95% UCL exceeds an HQ of 1.0 for both adults and children, indicating need for further study.

Table 5.16. Results obtained for the hazard quotients for adults from select non-carcinogenic 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 HI[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 HI[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 HI[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 HI[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 HI[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 HI[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 HI[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 HI[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
	Total HI[a]		0.6	3.7	25.2

[a]The values for the total risk will not be directly additive because of the random process used for error propagation.

Table 5.17. Results obtained for the hazard quotients for children from select non-carcinogenic 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 HI[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 HI[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 HI[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 HI[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 HI[a]		1.0	5.7	35.4
	Catfish	Chlordane	0.0	0.4	8.7
	Catfish	Aroclor-1254	0.3	2.0	15.3
	Total HI[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 HI[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 HI[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 HI[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.

5.5.1.4 Results of the sensitivity analysis for select non-carcinogens

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 5.18. The sensitive parameters are ranked according to priority for further study.

Table 5.18. Results obtained for the sensitivity analysis performed for the Total HI from select non-carcinogen contamination associated with the adult ingestion of fish from the Clinch River and Poplar Creek

Species	Sensitive Parameter	Sensitivity Index	Rank
LMB (Reach 3)	I_f: meal/day	62.3 %	1
	Mercury RfD: mg/kg-day	14.2%	2
	Aroclor-1254 RfD: mg/kg-day	10.1 %	2
	Im: kg/meal	6.3%	3

The two most sensitive parameters for the total HI from large mouth bass are the number of fish-meals per day and the RfD for Aroclor-1254 and methyl mercury. These results indicate that 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 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. 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). Because of these small uncertainty estimates that are not consistent with the standard deviations which would be expected, the concentration of the contaminant in fish should also be considered as a parameter for future study.

5.5.1.5 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.

- Because the total hazard quotients for all contaminants and fish exceeds 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, further study of the toxic effects from chlordane, Aroclor-1254, and mercury is recommended. 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. Because the median values exceed 10^{-4} risk for each of the species and reaches studied for each carcinogen, further study is recommended for chlordane and Aroclor-1260.

- Results of the sensitivity analysis indicate that 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 fisherman.

5.5.2 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 greater than 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 5.19.

Table 5.19. Contaminants of concern for the dredging pathway

Pathway	Contaminant	Reach	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}Cesium	2.04	2.7E-04	
	^{137}Cesium	3.04	1.0E-04	
	^{137}Cesium	4.03	2.3E-04	

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.

5.5.2.1 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 using 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 5% LCLs, and the 95% UCLs were obtained.

The subjective confidence intervals for the adult are summarized in Table 5.20, and the subjective intervals for children are presented in Table 5.21. 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).

Table 5.20. Results obtained for the excess health risk to adults from select noncarcinogenic contamination in deep sediments of the Clinch River and Poplar Creek

Reach	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
	Non-Leafy Vegetables	Mercury	0.13	1.96	43.95

Table 5.21. Results obtained for the excess health risk to children from select noncarcinogenic contamination in deep sediments of the Clinch River and Poplar Creek

Reach	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
	Non-Leafy Vegetables	Mercury	0.24	3.57	74.18

The subjective confidence intervals given in Table 5.20 and Table 5.21 provide valuable information that can be used to guide decision-making. For example, if a 5% LCL is above a regulatory standard of concern, then remediation is most likely needed. If the 95% UCL is below the standard, remediation is most likely not required. If the 95% UCL is above the standard, but the 50th percentile is below the standard, further study should be recommended for those parameters which dominate the overall uncertainty (milk pathways). 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 versus benefits analysis.

5.5.2.2. 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 5.22. The sensitive parameters are ranked according to priority for further study.

Table 5.22. 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

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.

5.5.2.3 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 using 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 5% LCLs, and the 95% UCLs were obtained.

The subjective confidence intervals for the adult are summarized in Table 5.23. The median risk for the external exposure from ^{137}Cesium, 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 excess health risk to adults from ^{137}Cs contamination in deep sediment of the Clinch River and Poplar Creek

Reach	Pathway	Contaminant	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
2.04	External	^{137}Cesium	1.0E-05	1.1E-04	7.2E-04
3.04	External	^{137}Cesium	6.1E-06	8.1E-05	4.7E-04
4.03	External	^{137}Cesium	9.6E-06	5.2E-05	2.4E-04

The subjective confidence intervals given in Table 5.23 provides valuable information that can be used to guide decision-making. For example, if a 5% LCL is above a regulatory standard of concern, then remediation is most likely needed. If the 95% UCL is below the standard, remediation is most likely not required. If the 95% UCL is above the standard, but the 50th percentile is below the standard, further study should be recommended for those parameters which dominate the overall uncertainty (Reaches 3.04 and 4.03). However, if the 50th percentile is above the standard, further study may still be recommended (Reach 2.04). The decision as to whether or not to proceed with remediation will be driven by a cost versus benefits analysis.

5.5.2.4. 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 5.24. The sensitive parameters are ranked according to priority for further study.

Table 5.24. 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 (Reach 2.04)	C_{sed} (sediment concentration)	38.4%	1
	RCF (Risk Conversion Factor)	21.2%	2
	D_1 (dilution factor)	9.6%	3
External (Reach 3.04)	C_{sed} (sediment concentration)	38.5%	1
	RCF (Risk Conversion Factor)	22.2%	2
	λeff	10.7%	3
External (Reach 4.03)	RCF (Risk Conversion Factor)	31.7%	1
	External Exposure Frequency	15.3%	2
	λeff	14.4%	2

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. 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.

5.5.2.5. 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 contaminants of potential concern). Further study is recommended for both contaminants. 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 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 clean-up levels in the CERCLA process, but 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 based on 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 5.25–5.31. 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. 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}Cobalt	4.60E+00	4.60E-01	4.60E-02	pCi/g
	^{137}Cesium	7.84E+00	7.84E-01	7.84E-02	pCi/g
	^{238}Uranium	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}Uranium	1.27E+02	1.27E+01	1.27E+00	pCi/g
	^{238}Uranium	1.43E+02	1.43E+01	1.43E+00	pCi/g
Ingestion of milk produced by cattle pastured on dredged sediments	^{137}Cesium	4.18E+01	4.18E+00	4.18E-01	pCi/g
	^{90}Strontium	2.48E+01	2.48E+00	2.48E-01	pCi/g
	^{99}Technetium	1.58E+03	1.58E+02	1.58E+01	pCi/g
	^{234}Uranium	8.49E+02	8.49E+01	8.49E+00	pCi/g
	^{238}Uranium	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
	PCB-1260	3.10E-02	3.10E-03	3.10E-04	mg/kg

Table 5.25 (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}Cesium	7.64E+01	7.64E+00	7.64E-01	pCi/g
	^{90}Strontium	8.67E+01	8.67E+00	8.67E-01	pCi/g
	Arsenic	1.01E+02	1.01E+01	1.01E+00	mg/kg
	PCB-1260	1.10E-01	1.10E-02	1.10E-03	mg/kg
Ingestion of vegetables produced on dredged sediments	^{137}Cesium	3.50E+02	3.50E+01	3.50E+00	pCi/g
	^{99}Technetium	6.80E+01	6.80E+01	6.80E+00	pCi/g
	^{90}Strontium	2.80E+01	2.80E+01	2.80E+00	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
	PCB-1260	1.87E+01	1.87E+00	1.87E-01	mg/kg

Table 5.26. 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}Uranium	2.53E+02	2.53E+01	2.53E+00	pCi/g
	^{235}Uranium	2.73E+02	2.73E+01	2.73E+00	pCi/g
	^{238}Uranium	2.86E+02	2.86E+01	2.86E+00	pCi/g

Table 5.27. 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
	PCB-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}Cesium	5.58E+00	5.58E-01	5.58E-02	pCi/g
	^{90}Strontium	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

Table 5.28. Remedial goal options for soil for the noncarcinogenic chemicals and pathways of concern for the dredging scenario

Pathway	Chemical	Adult			Child		
		RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a HQ of 0.1 (mg/kg)	RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a 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
	PCB-1254	2.46E+00	2.46E-01	2.46E-02	2.64E-01	2.64E-02	2.64E-03

Table 5.28 (continued)

Pathway	Chemical	Adult			Child		
		RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a HQ of 0.1 (mg/kg)	RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a 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
	PCB-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 5.28 (continued)

Pathway	Chemical	Adult			Child		
		RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a HQ of 0.1 (mg/kg)	RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a 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
	PCB-1254	4.64E+01	4.64E+00	4.64E-01	9.94E+00	9.94E-01	9.94E-02

Table 5.29. 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 a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a HQ of 0.1 (mg/kg)	RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a 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

Table 5.30. Remedial goal options for the noncarcinogenic chemicals and pathways of concern for water

Pathway	Chemical	Adult			Child		
		RGOs based on a HQ of 10 (mg/L)	RGOs based on a HQ of 1 (mg/L)	RGOs based on a HQ of 0.1 (mg/L)	RGOs based on a HQ of 10 (mg/L)	RGOs based on a HQ of 1 (mg/L)	RGOs based on a 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
	PCB-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
	PCB-1254	7.30E-03	7.30E-04	7.30E-05	3.13E-03	3.13E-04	3.13E-05

Table 5.31. Remedial goal options for the noncarcinogenic chemicals and pathways of concern for fish tissue

Pathway	Chemical	Adult			Child		
		RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a HQ of 0.1 (mg/kg)	RGOs based on a HQ of 10 (mg/kg)	RGOs based on a HQ of 1 (mg/kg)	RGOs based on a 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
	PCB-1254	2.7E-01	2.7E-02	2.7E-03	1.04E-01	1.04E-02	1.04E-03

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 Operable Unit (CR OU). The CR 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 endpoints. Then, for each endpoint, 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 endpoints, and summarization of that information in terms of a conceptual model of the hazard posed by the contaminants to the endpoint 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 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.

COPECs for ecological risks have been identified in previous screening assessments (Suter 1991, Cook et al. 1992, Blaylock et al. 1993). 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 Endpoints

The problem formulation must identify both the assessment endpoints, which are explicit statements of the characteristics of the environment that are to be protected, and the measurement endpoints, which are quantitative summaries of a measurement or series of measurements that are related to effects on an assessment endpoint.

6.1.3.1 Assessment endpoints

The following assessment endpoints 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 endpoints have been selected based on DQO meetings that included representatives of the DOE, EPA Region IV, and TDEC. The selected endpoints are explained below.

1. The fish community is considered to be an appropriate endpoint 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 OU move within an area smaller than the OU.

2. The benthic invertebrate community is considered to be an appropriate endpoint 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 endpoint 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 endpoint 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 endpoint 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 endpoint populations are the little brown bat and the rough-winged swallow.

5. The terrestrial plant community is considered to be an appropriate endpoint 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 endpoint 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.
- Four 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 endpoints 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 endpoint 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 endpoints 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 endpoints

Three basic types of effects data are potentially available to serve as measurement endpoints: results of biological surveys, toxicity tests performed on media from the CR OU, and toxicity test endpoints for chemicals found in the CR OU. Measurement endpoints are presented below for each assessment endpoint.

Fish

Biological Survey Data (a) Species richness and abundance from TVA reservoir surveys and fish community surveys conducted for this assessment. These measurement endpoints are assumed to be direct estimates of that assessment endpoint.

Biological Indicators Data (b) Frequencies of gross pathologies are a direct measure of one aspect of the assessment endpoint. (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 endpoints correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to

approximate the distribution of CR 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

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 endpoints are assumed to be direct estimates of the assessment endpoint 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 endpoints correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of CR 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 endpoint. Second, thresholds for lethality in toxicity tests of contaminated sediments, which are also assumed to correspond to the assessment endpoint effect, but with greater uncertainty due to the extrapolation to the field.

Piscivorous wildlife

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 endpoint 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 endpoints 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 endpoints. After allometric scaling for the endpoint species, these test endpoints are assumed to correspond to effects on individuals that could result in exceedence of the population-level assessment endpoint. An extrapolation must be made to populations if effects on individuals are estimated to occur.

Flying insectivorous wildlife

Biological Survey Data None.
Media Toxicity Data None.
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 endpoints. After allometric scaling for the endpoint species, these test endpoints are assumed to correspond to effects on individuals that could result in exceedence of the population-level assessment endpoint. An extrapolation must be made to populations if effects on individuals are estimated to occur.

Terrestrial plants

Biological Survey Data None.
Media Toxicity Data None.
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 endpoints are assumed to correspond to the assessment endpoint 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 endpoints 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

Biological Survey Data None.
Media Toxicity Data None.
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 endpoints. After allometric scaling for the endpoint species, the eastern cottontail, these test endpoints are assumed to correspond to effects on individuals that could result in exceedence of the population-level assessment endpoint. 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

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 endpoint 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 endpoints 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 endpoint.

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 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 endpoints 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 endpoint by (1) screening all measured contaminants against toxicological benchmarks and

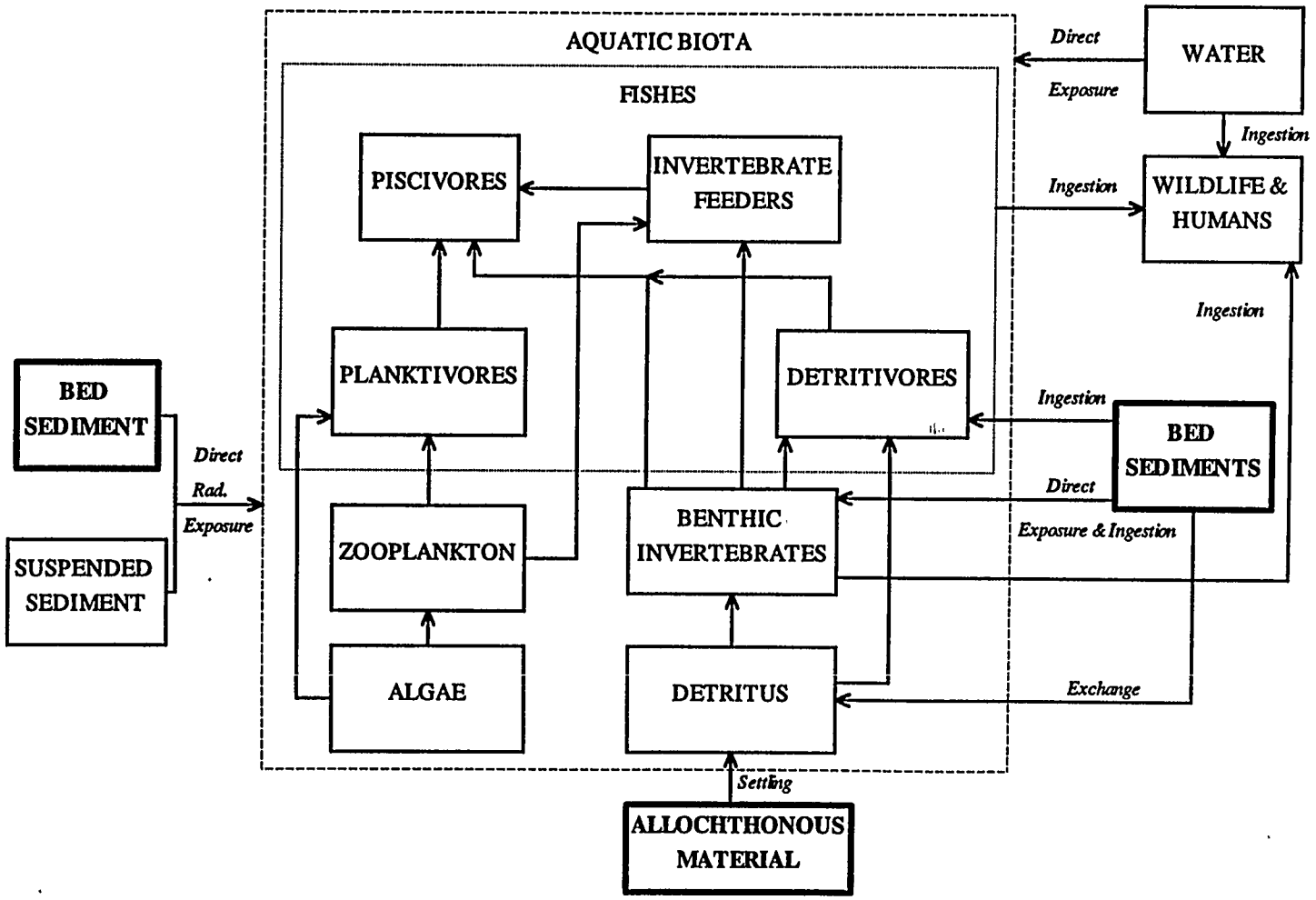


Fig. 6.1. Conceptual model of contaminant sources, transport, and exposure in the aquatic food web. Dark-lined boxes represent sources that may be remediated. Arrows represent routes of transport and exposure, described by italic labels. Unlabeled arrows represent food web transfers by ingestion.

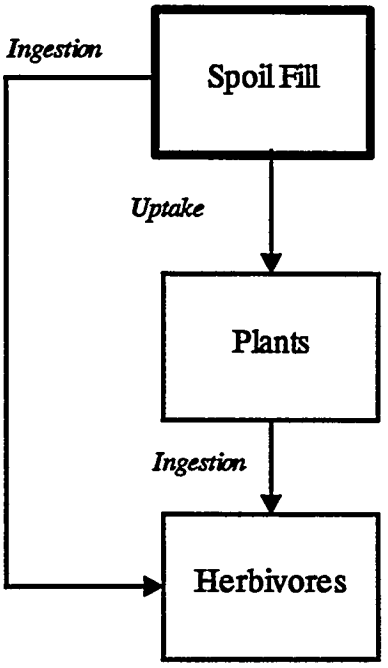


Fig. 6.2. Conceptual model of contaminant sources, transport, and exposure in the terrestrial food web for the baseline dredging case. Dark-lined boxes represent sources that may be remediated. Arrows represent routes of transport and exposure, described by italic labels.

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 endpoint biota based on the results of the biological survey data, (5) logically integrating the lines of evidence to characterize risks to the endpoint, 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 endpoints, 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 endpoint, 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 endpoint (Finney 1971). TUs are quotients of the concentration of a chemical in a medium divided by the standard test endpoint concentration for that chemical. They are similar to HQs except that a common test endpoint 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 endpoint 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 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 endpoint 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 endpoints are chosen for their consistency rather than their relationship to an assessment endpoint, 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 Reach 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

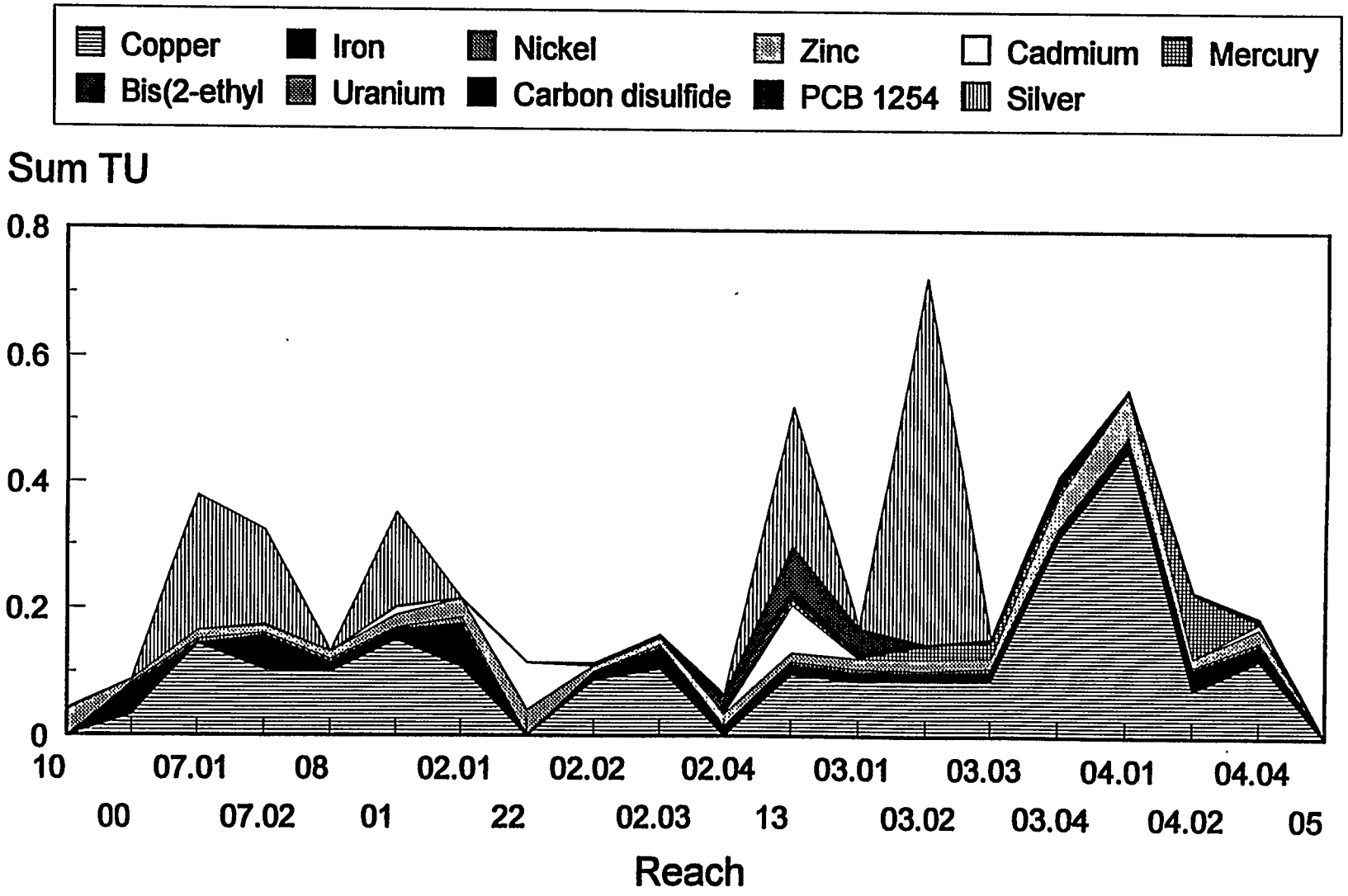


Fig. 6.3. Plot of concentration of COPECs normalized to toxic units (TU). The upper contour of the graph is the sum of TU for the reaches.

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 Reach 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 endpoint, 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 less than 20% of locations in Reach 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 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, less than 1% of the members of a rough-winged swallow colony located in Reach 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 Reach 3.02 will receive a dose rate greater than the LOEL is greater than 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 endpoint, so they are not summarized here. However, for all endpoints 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 (HECD 1992, 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 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 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 approximately 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 endpoints 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.

Table 6.1. Descriptions of the ecotoxicological screening benchmarks for aquatic biota. More details are presented by Suter and Mabrey (1994).

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 Effect Concentration (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 non-daphnid 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.

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).

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 greater than 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 endpoint 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 greater than 1, one would expect acute lethality to *Daphnia*. ΣTU values greater than 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, 13, and 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).

Table 6.2. Results of screening of chemicals that exceed benchmarks in whole or filtered water for COPECs

Chemical	COPEC	Reason for Inclusion or Rejection
Aluminum	No	Exceeds chronic NAWQC in all reaches and acute NAWQC in 18, 21, and 22 in whole water but no benchmarks in dissolved phase. Hence, 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 Ba 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 Cd 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.
PCB 1254	Yes	Exceeds the plant LCV and SCV in whole water from 3.01, 3.04, and 13.

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 endpoints 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 (Reaches 1 and 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 endpoints 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). 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 approximately 90% of the CVs, approximately 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 Reach 7.02 and in 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 endpoints 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 Reaches 3.02, 3.03, and 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 (Reach 20.1) which is the presumed source.

Nickel. The distributions of ambient nickel concentrations and of aqueous toxicity test endpoints are shown in Fig. F2.4. Unlike other COPECs, episodic high concentrations of Ni were not detected. Rather, in Reaches 3.03, 3.04, 4.04, 7.02, and 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 endpoints 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 PCB 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 endpoints 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 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 endpoints 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 approximately 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 PCB-1254 and 4.1 and 13.1 mg/kg for PCB-1260. A test of dietary exposure of channel catfish to PCB-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 PCB-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 less than 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 Reach 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 greater than 20% reduction in fecundity which was also statistically significant (Sect. 3.3). Since effects were greater in the upstream reference Reach (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 less than 10% in embryos incubated in water from Reaches 7.01, 7.02, 8, 2.01, and 4.01. However, mean rates were greater than 30% in all subreaches of Reach 3, and were greater than 50% in Reach 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 (Reach 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

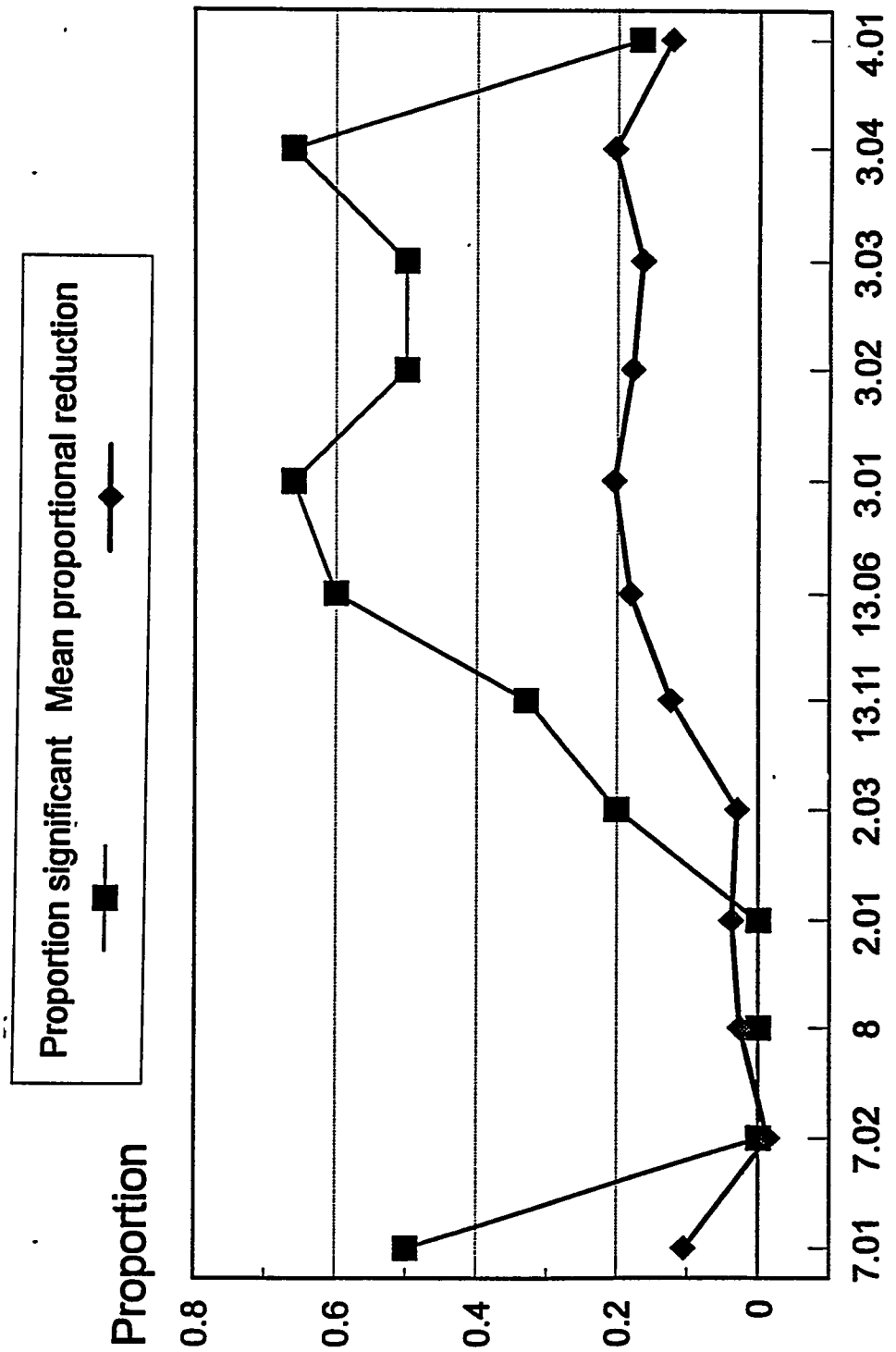


Fig. 6.4. Proportional reduction in fathead minnow larval survival relative to controls and proportion of tests in which survival was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly decreased). Note for comparison the responses in the Clinch River reference subreach (2.01), the far upstream reference reach in Poplar Creek (13.11) and the near upstream reference (13.06).

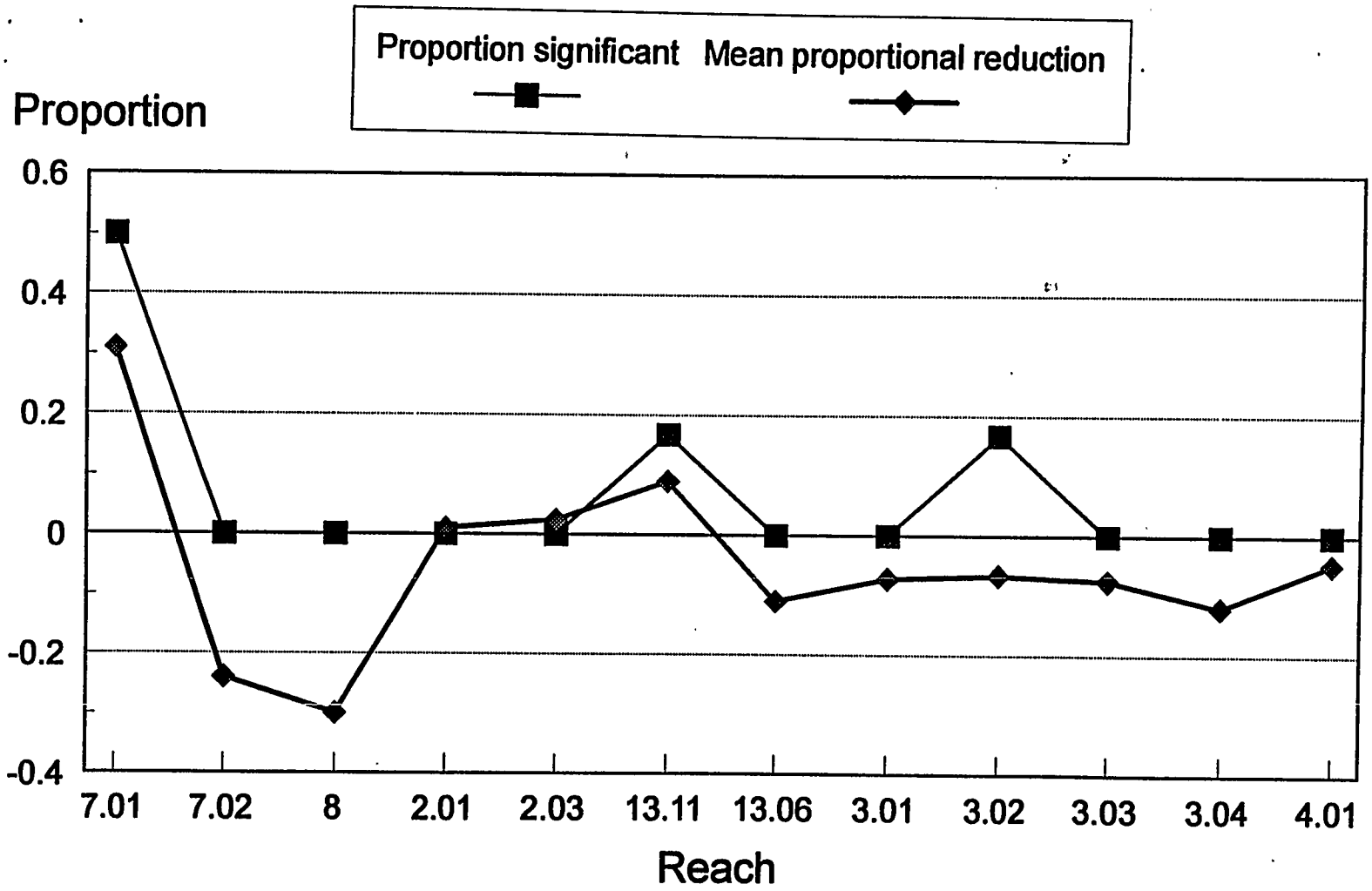


Fig. 6.5. Proportional reduction in *Ceriodaphnia dubia* fecundity relative to controls and proportion of tests in which fecundity was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly decreased). Note for comparison the responses in the Clinch River reference subreach (2.01), the far upstream reference reach in Poplar Creek (13.11) and the near upstream reference (13.06).

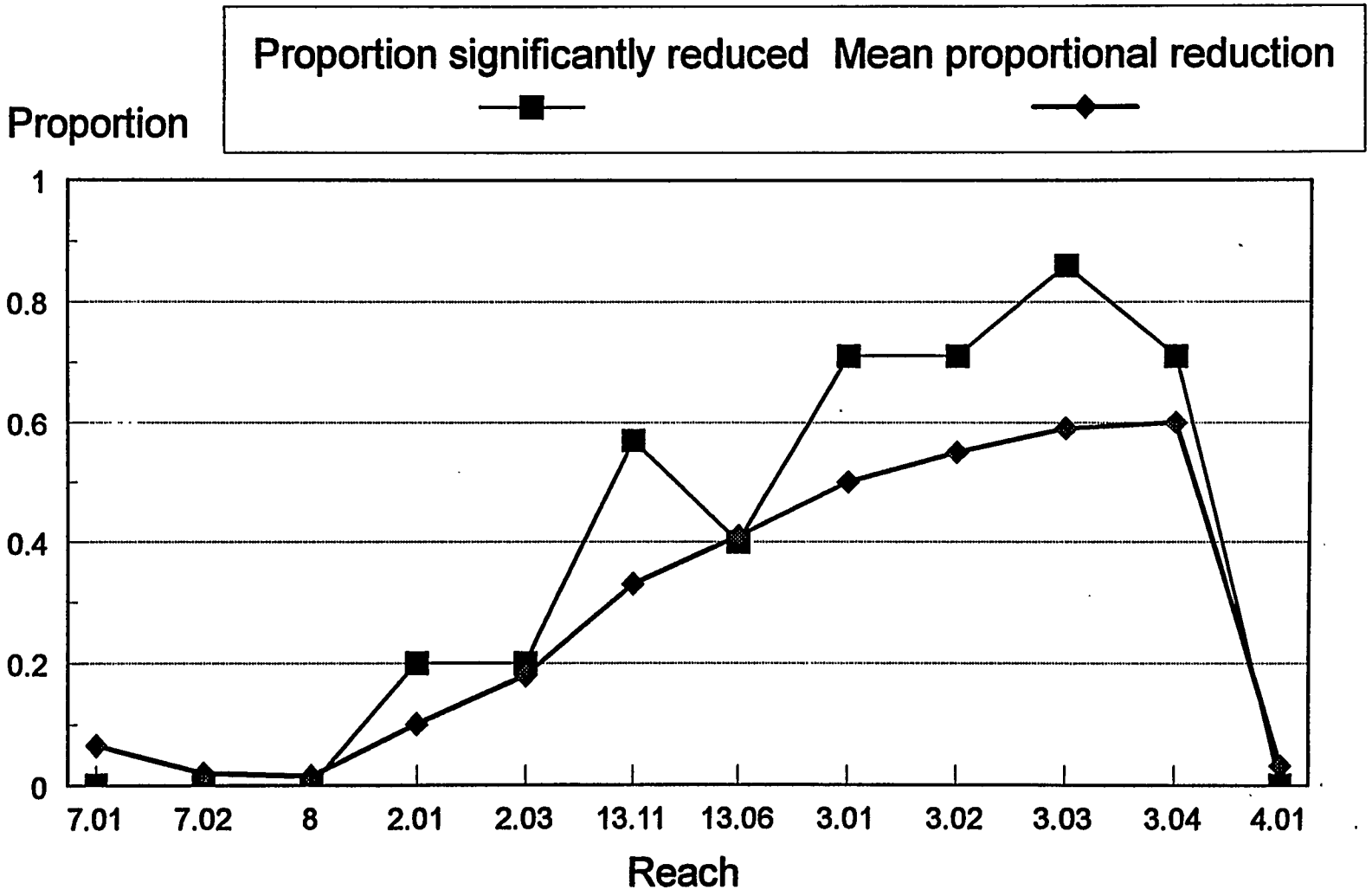


Fig. 6.6. Proportional reduction in survival of medaka eggs relative to controls and proportion of tests in which survival was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly increased). Note for comparison the responses in the Clinch River reference subreach (2.01), the far upstream reference reach in Poplar Creek (13.11) and the near upstream referenc (13.06).

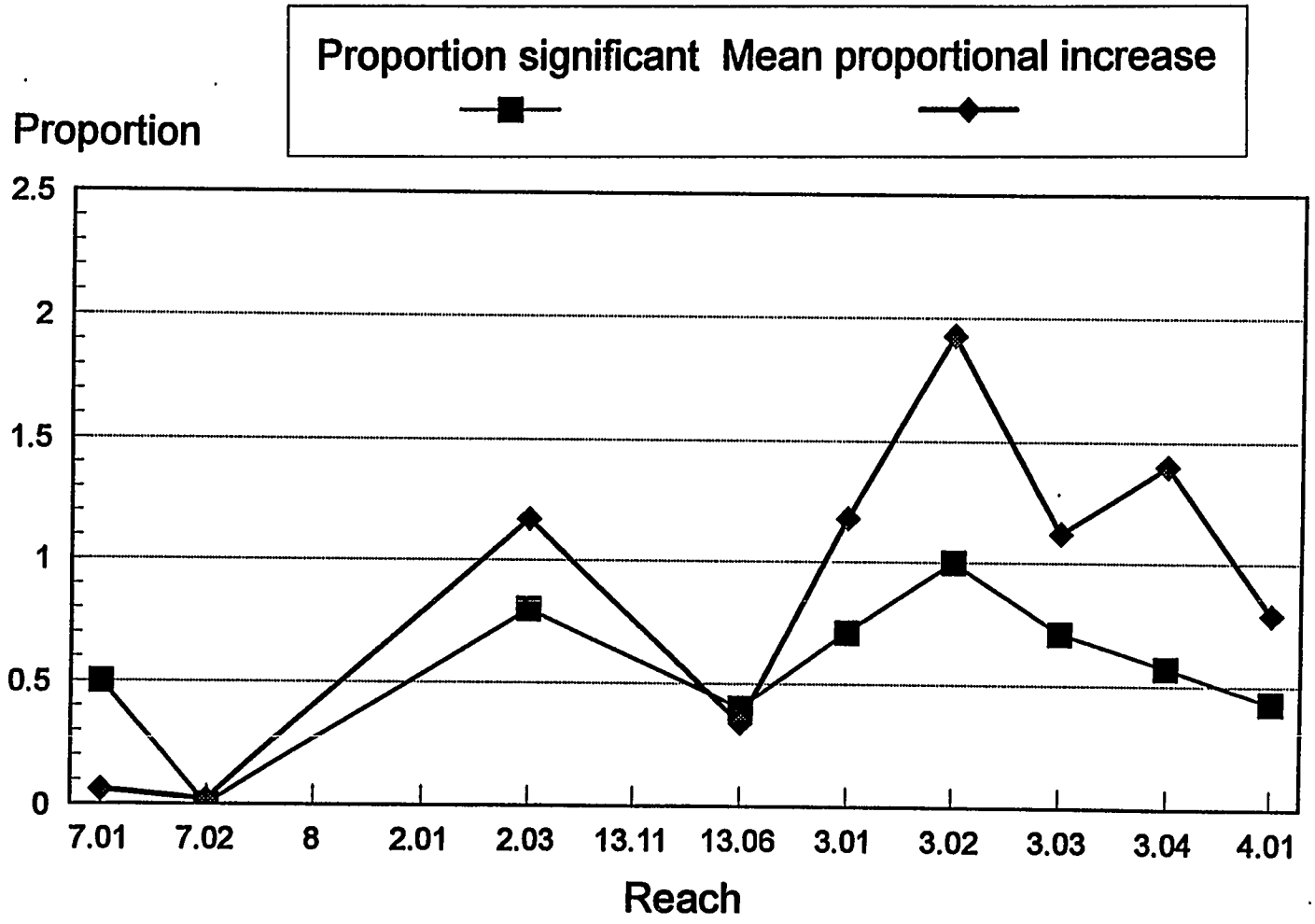


Fig. 6.7. Proportion of abnormal medaka larvae and proportion of tests in which abnormalities were statistically significantly increased relative to reference or were increased by at least 20% relative to reference (i.e., significantly increased). Note for comparison the frequencies of abnormalities in the Clinch River reference subreach (2.01), the far upstream reference reach in Poplar Creek (13.11) and the near upstream reference (13.06).

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 (Reach 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 less than 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.

Summary of aqueous toxicity tests. Of the three aqueous toxicity tests, only the medaka embryo test shows consistent evidence of toxicity. In addition, only nickel concentrations and medaka survival showed a relationship between exposure and response. The lack of response in the two standard tests does not eliminate the possibility of toxic conditions with respect to those test species. The high episodic 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 Reaches 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 Reach 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 Reaches 2.02, 3.04, and 4.04. At both Reaches 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 endpoint, they are suggestive of inhibited reproduction in individuals of one species.

6.2.3.5 Biological surveys

The data on fish species richness (Figs. 6.8 through 6.11) show no pattern that can be attributed to a contaminant effect. For both years and gear types, the sampling stations in the Clinch River (Reaches 2.02, 4.01, and 4.04) fall within the range of values for the reference sites: Norris Reservoir (Reach 10) and Watts Bar Reservoir (Reach 5 and Reach 18). As noted in Appendix F8, analysis of fish community composition for reservoirs throughout the Tennessee Valley shows that Norris and Watts Bar are ecologically distinct from each other. Fish abundance and species richness at the mouth of Poplar Creek (Reach 3.04) are similar to abundance and richness in the lower Clinch River and Watts Bar reservoir. However, richness and abundance in the two upstream subreaches, Reach 3.02 and 3.04, are significantly lower than in the Clinch or Watts Bar. Because no reference embayments were included in the survey, no conclusion regarding contaminant effects is possible from this line of evidence. Because of habitat differences, the fish community in Poplar Creek embayment would be expected to differ to some extent from the fish community present in a mainstem river or reservoir.

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). Although the evidence is not strong, it is more likely than not that the fish community endpoint is violated. 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. In addition, it is not clear what chemicals from what sources are the cause of the observed effects. The conclusion is mitigated by the

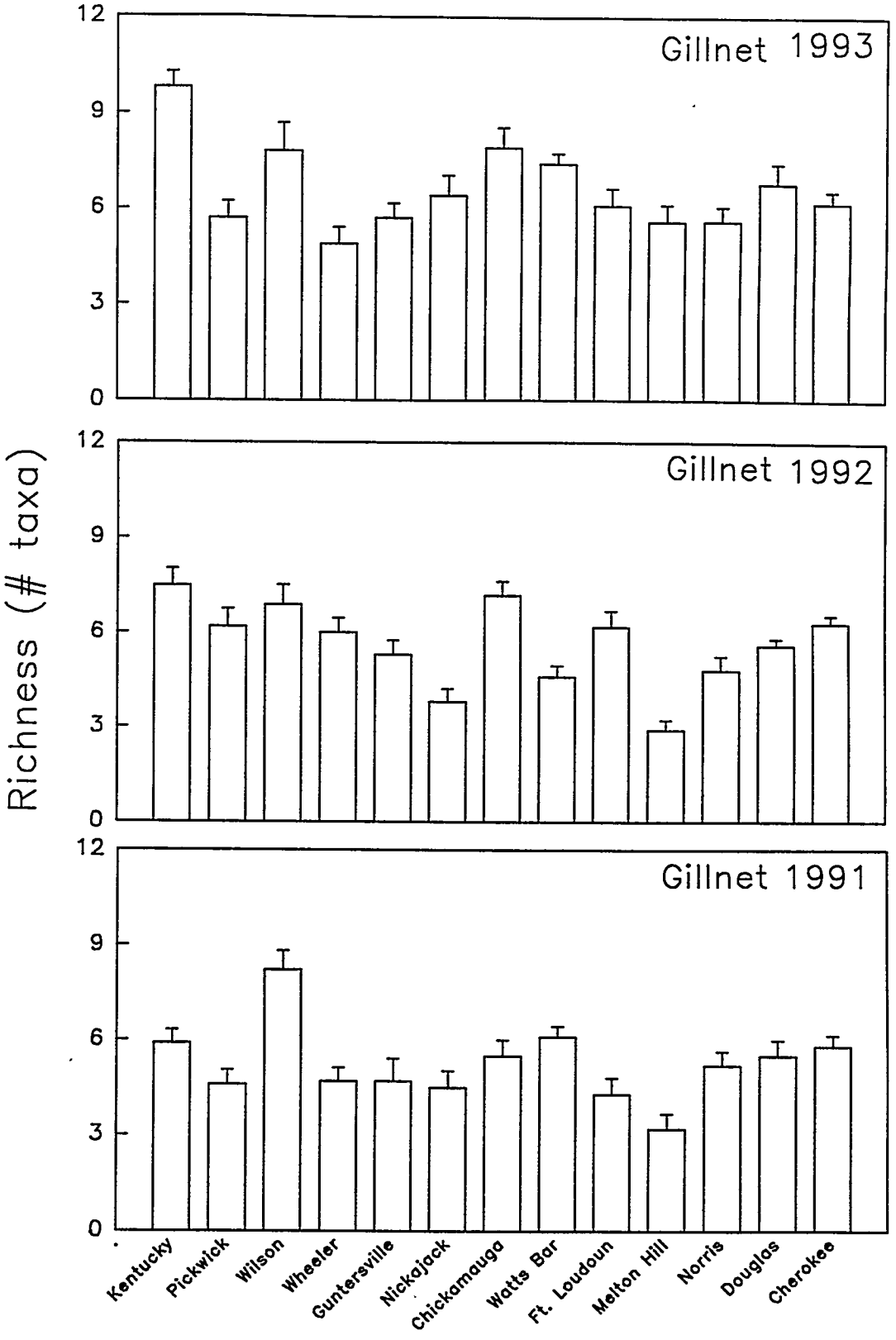


Fig. 6.8. Species richness of fish sampled by gill netting in individual reservoirs in 1991–1993.

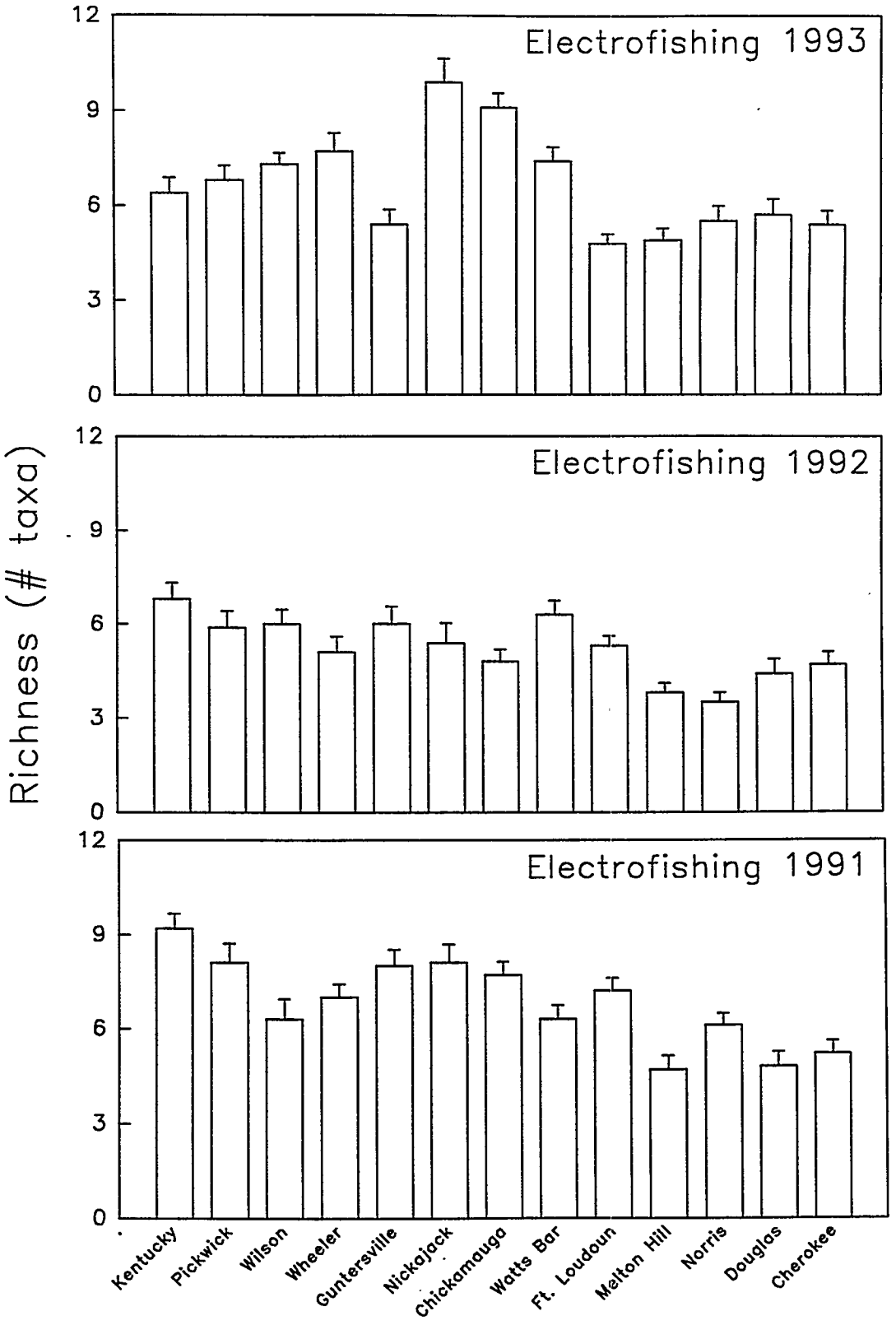


Fig. 6.9. Species richness of fish sampled by electrofishing in individual reservoirs in 1991–1993.

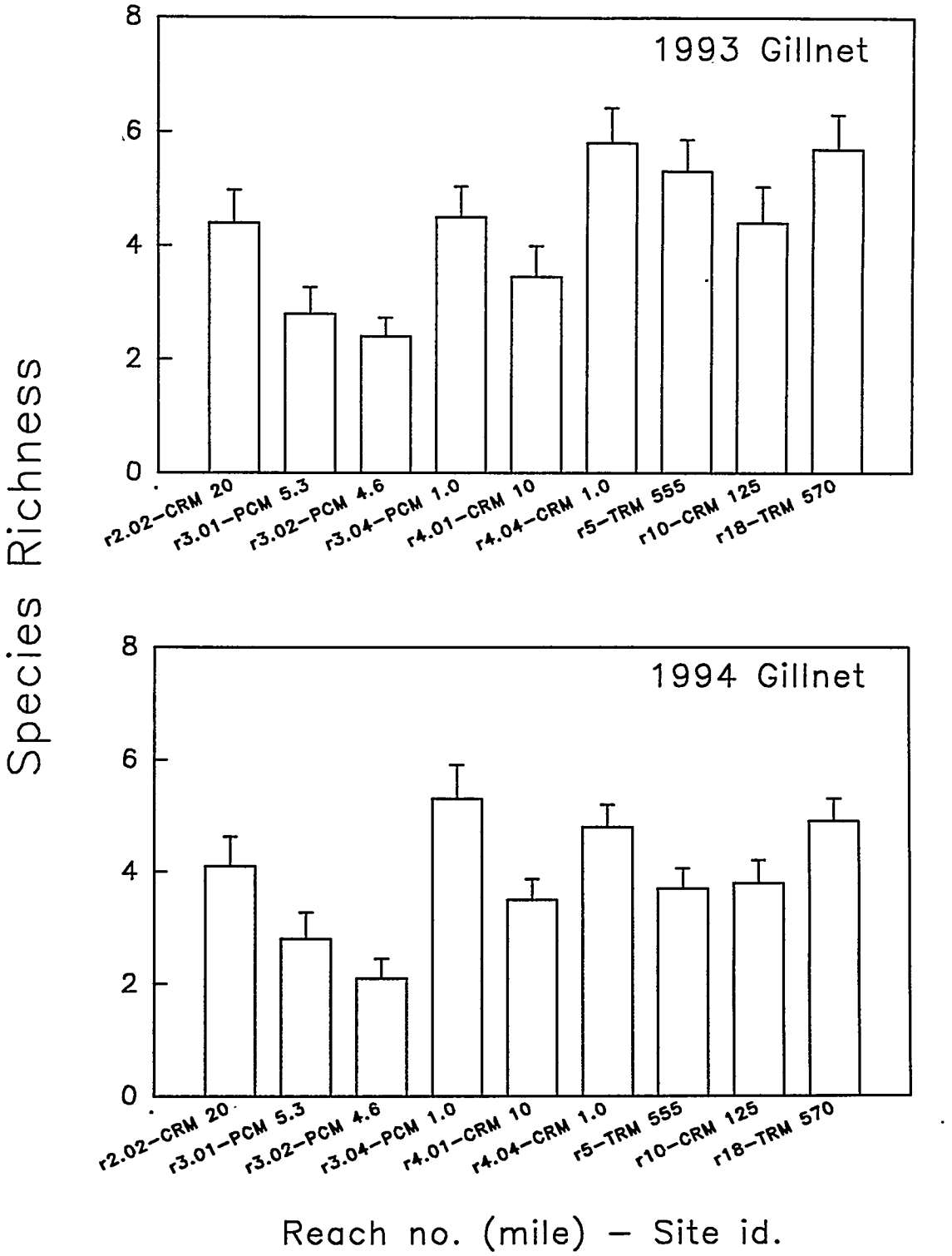


Fig. 6.10. Species richness of fish sampled by gill netting in reaches of the Clinch River/Poplar Creek system and reference reaches in 1993–1994.

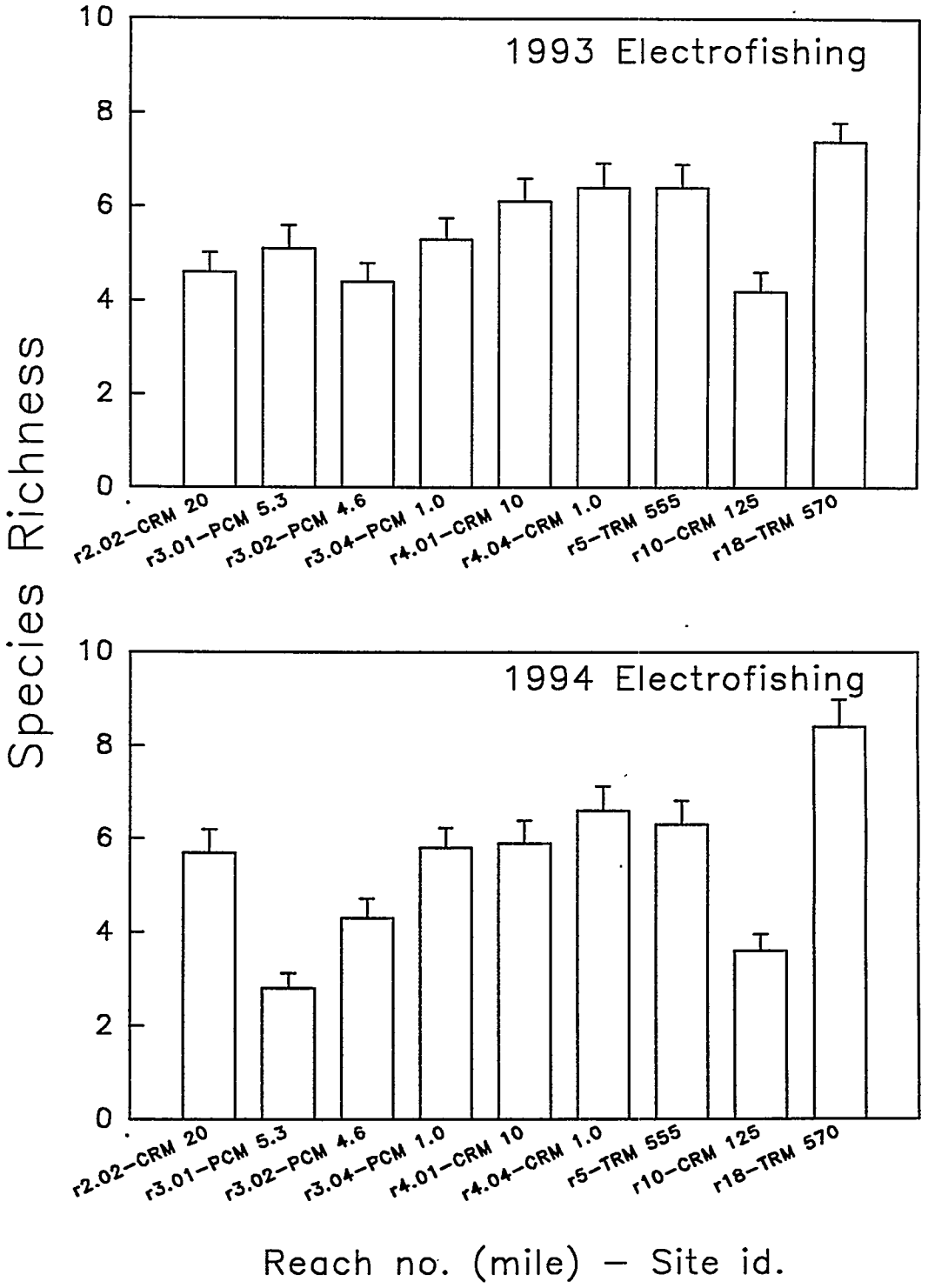


Fig. 6.11. Species richness of fish sampled by electrofishing in reaches of the Clinch River/Poplar Creek system and reference reaches in 1993–1994.

following considerations: the effects of contaminants on the fish community are not severe (i.e., there is not as much as a 50% reduction in community properties), they are not entirely attributable to DOE (Poplar Creek upstream is contaminated and somewhat toxic), and there is a significant chance that effects may be less than the endpoint level. 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, then 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 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. Because none of the reference sites are similar embayments, it is not possible to definitively say that the low species richness and abundance is due to habitat, thereby eliminating toxic effects as a cause. Hence, the survey results are consistent with a significant toxic effect, but habitat quality also contributes to an unknown extent.

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.

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, relative 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.

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).

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 relative to other Clinch R. sites.
Bioindicators	±	Reproductive indicators were modified in largemouth bass below White Oak Creek (Reach 2.02) relative to reference fish.
Toxicity Tests		No toxicity tests were performed with water from the portion of this reach downstream of DOE sources (subreaches 2.02-2.04). Reach 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.

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. 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 (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. Reach 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.

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 6.5. 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 4.01 relative 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	+	Cu and Zn 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.

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.

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 Ag and Ni.
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.

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 greater than seven 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.

The high observed mortality and abnormalities in fish embryos 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-d tests.

The relationship of the histopathologies and reproductive indicators to effects on populations and ecosystems is unknown.

The fish community surveys in Poplar Creek are compared to reference communities in the main channel of reservoirs and not to other embayments.

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

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 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 EPA Contract Laboratory Program Practical Quantification Limit (CLP PQL) and the Effects Value which is the lower of the ER-L and the Florida NOEL.
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 NAWQC 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.

al., 1995). An extensive marine data set was presented in MacDonald (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.

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 endpoints 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

Table 6.8. Results of screening of chemicals that exceed benchmarks in whole sediments or filtered pore water for COPECs

Chemical	COPEC	Reason for Inclusion or Rejection
Aluminum	YES	Exceeds chronic NAWQC in pore water in Reaches 0, 3.04, 4.01, 4.02, 4.04, 7.02, and 18. Exceeds Daphnid LTV and the LCV of all tested aquatic organisms (plants) at Reach 4.01. A sediment benchmark is not available.
Arsenic	YES	Exceeds the ER_M in whole sediment in Reach 7.01 and the ER-L, LEL_MOE, and/or REG_IV in Reaches 1, 2.04, 7, 8, 10, 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, 2.01, 2.04, 13, 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 Reach 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, 4.04, 5, 7.02, and 22. 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 Reaches 3.02, 3.03, 3.04, 4.03, 7.02, and 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.

Table 6.8 (continued)

Chemical	COPEC	Reason for Inclusion or Rejection
Cobalt	YES	Exceeds the SCV and the Daphnid LTV and LCV in pore water in Reaches 3.01, 3.03, 4.03, 4.04, and 13; the SCV in Reaches 0 and 3.02; no other benchmarks in any reach. A sediment benchmark is not available.
Copper	YES	Exceeds the REG_IV and ER-L in Reaches 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 Reach 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 Reaches 3.03, 4.04, and 7. Exceeds the chronic NAWQC and the Fish LCV in pore water in Reaches 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.
Lead	YES	Exceeds the REG_IV and/or ER-L in Reaches 0, 1, 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 Reaches 3.01, 3.02, and 4.04, but no other benchmarks in any reach.

Table 6.8 (continued)

Chemical	COPEC	Reason for Inclusion or Rejection
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 Reaches 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, and the ER_M in Reaches 3.02, 3.03, and 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, 2.04, 3.02, 3.03, and 3.04, 4, 7.02, and 13; 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 and ER_M 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
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.
Acenaphthene	YES	Exceeds the ER_L in Reaches 0, 3.02, 3.03, 3.04, 4.01, and 13; 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 Reaches 3.03, 3.04, and 4.01; 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 ER_L and/or the REG_IV (CLP PQL) in 3.02, 3.03, and 18, but not the ER_M or the EQ_PART in any reach. Not detected in pore water in any reach.
Benzo(a)pyrene	YES	Exceeds 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.

Table 6.8 (continued)

Chemical	COPEC	Reason for Inclusion or Rejection
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 ER_M in 13. Not detected in pore water in any reach.
PAHs, Total	YES	Exceeds the REG_IV in Reaches 0, 2.04, 3, 4, 7, 8, 13, and 18; 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.

Table 6.8 (continued)

Chemical	COPEC	Reason for Inclusion or Rejection
PCB-1248	YES	Exceeds the ER_L in Reaches 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.
PCB-1254	YES	Exceeds the ER_L, but not 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.
PCB-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.

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. (1994) 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)

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 2.03. Manganese is the principle contributor to toxicity in most reaches. Only Reach 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 greater than 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 Reaches 2.02 and 4.04 and significant contributors in Poplar Creek (Reaches 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

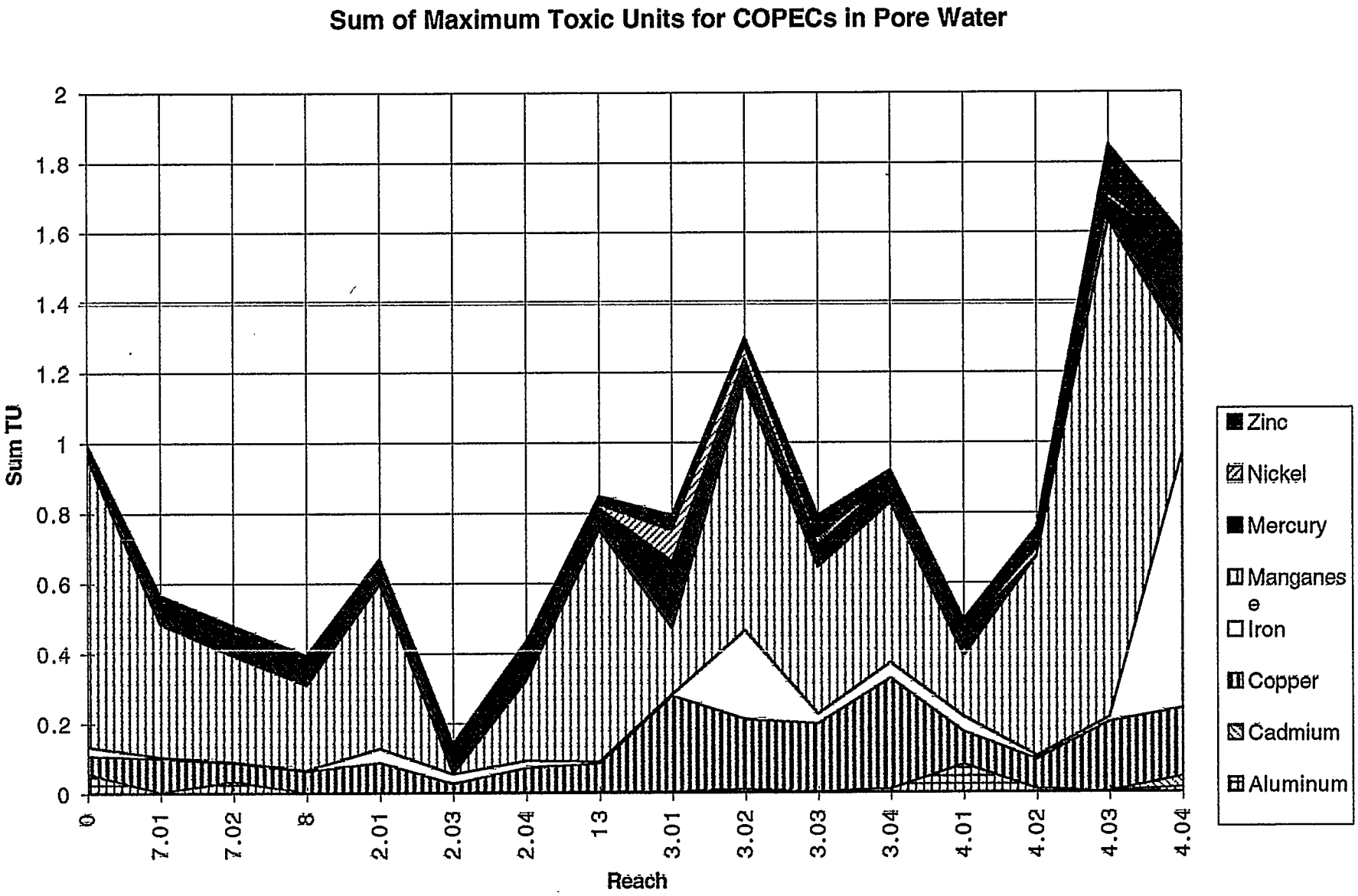


Fig. 6.12. Plot of maximum concentration of pore water COPECs normalized to toxic units (TU). The Upper contour of the graph is the sum of TU for the reaches. Includes COPECs with Maximum TUs greater than 0.01.

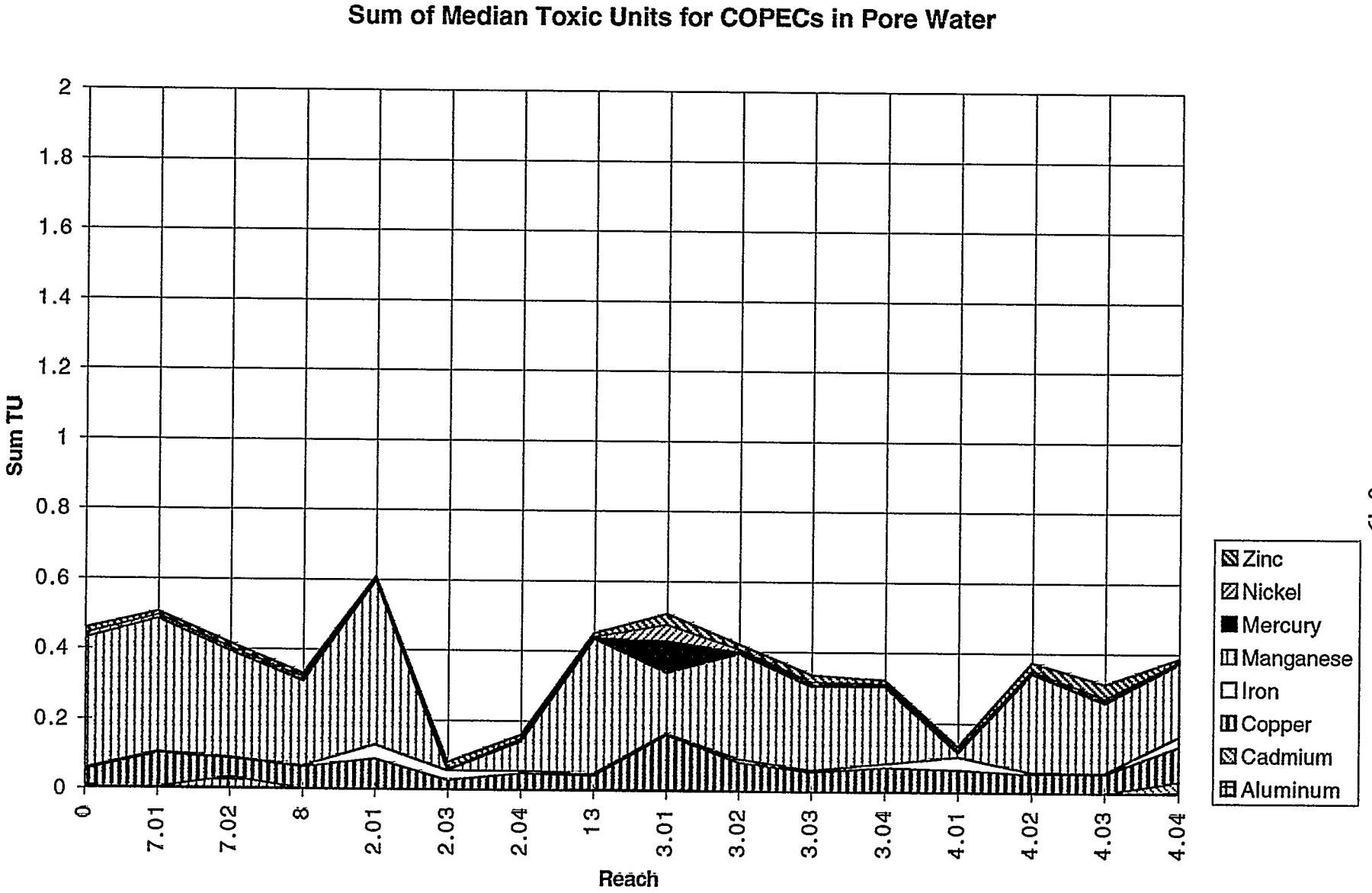


Fig. 6.13. Plot of median concentration of pore water COPECs normalized to toxic units (TU). The Upper contour of the graph is the sum of TU for the reaches. Includes COPECs with Maximum TUs greater than 0.01.

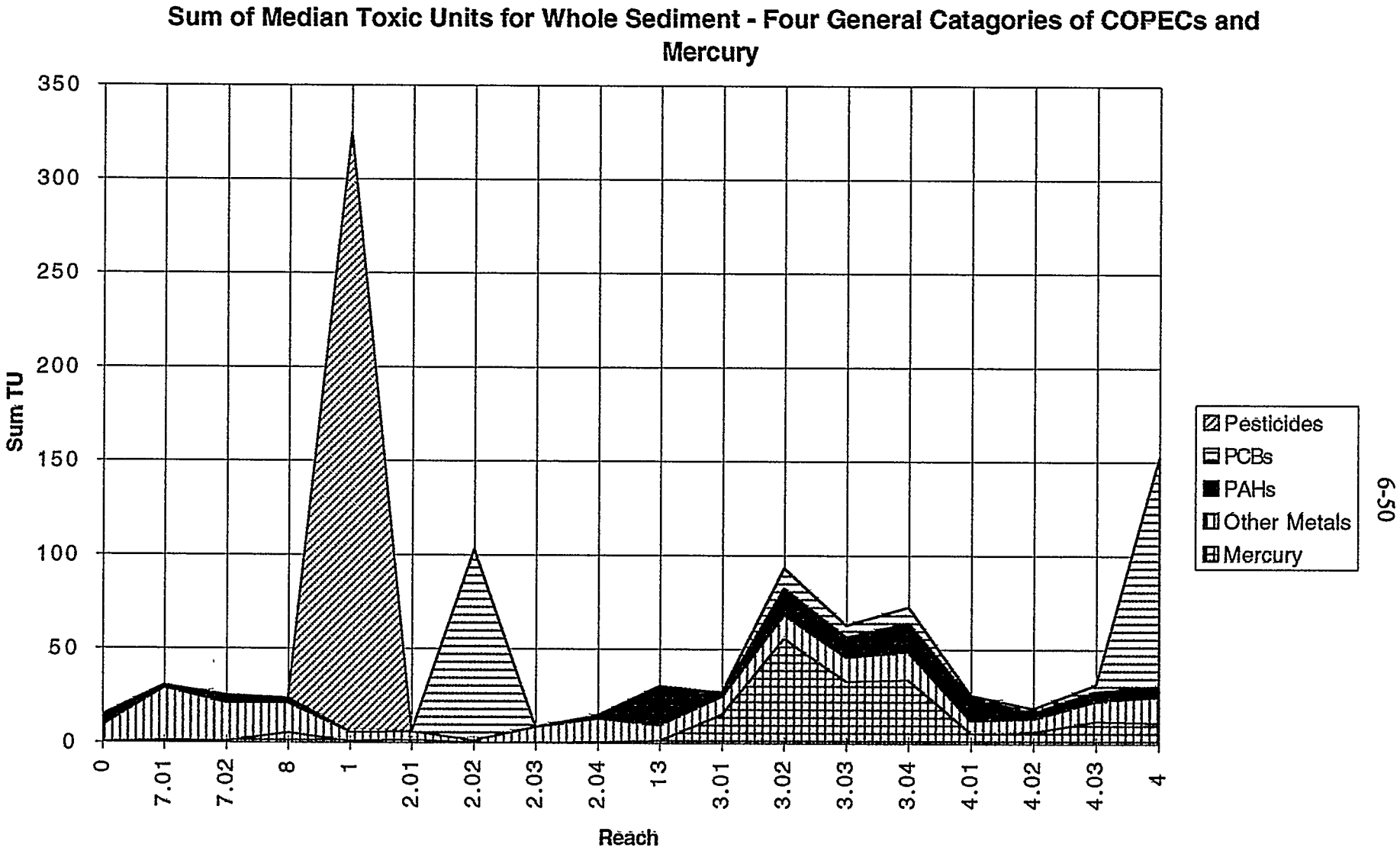


Fig. 6.14a. Plot of median concentration of whole sediment COPECs normalized to toxic units (TU). The Upper contour of the graph is the sum of TU for the reaches. Includes COPECs with Maximum TUs greater than 0.01. This graph (a) is four General Categories of COPECs and mercury.

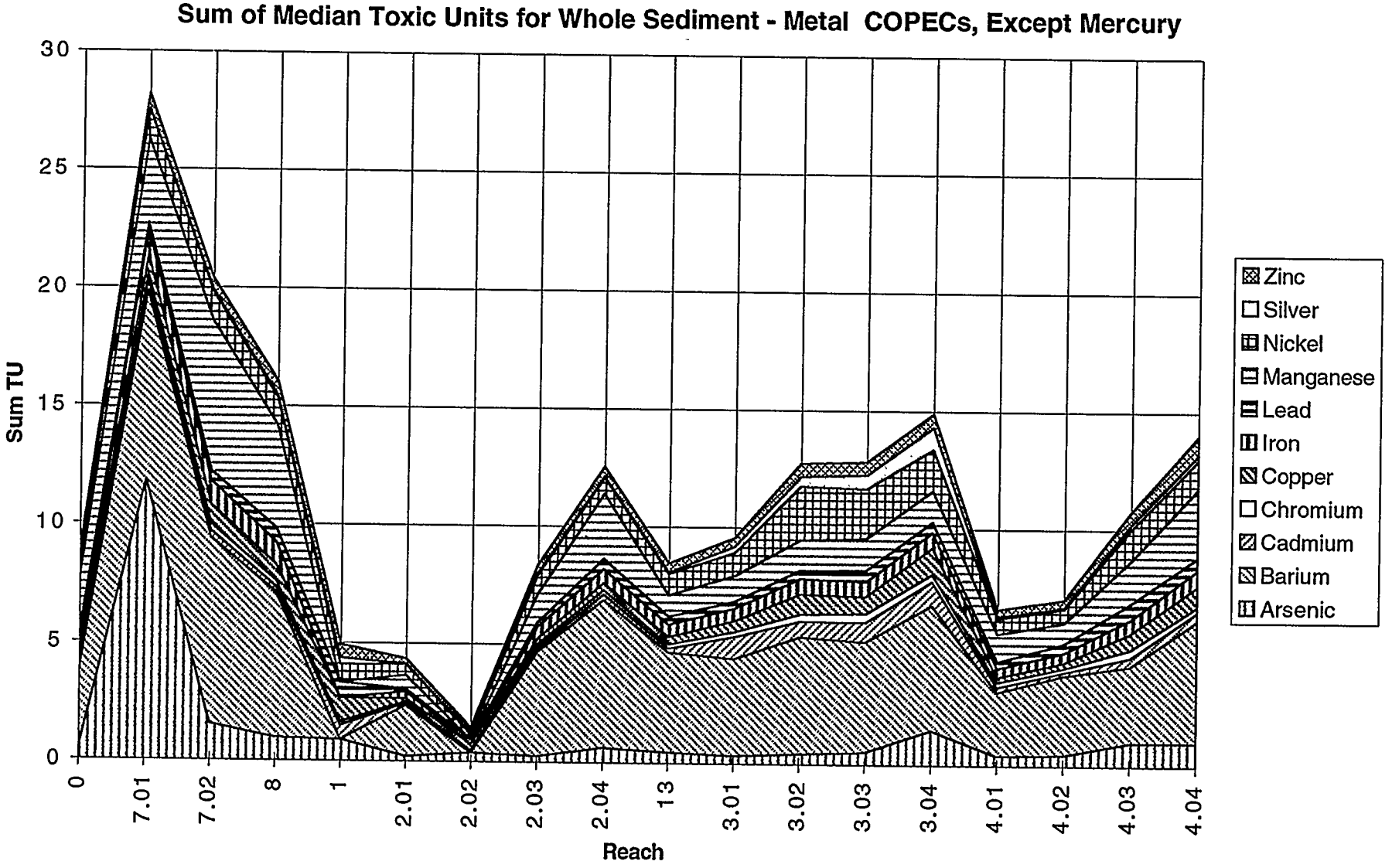


Fig. 6.14b. Plot of median concentration of whole sediment COPECs normalized to toxic units (TU). The Upper contour of the graph is the sum of TU for the reaches. Includes COPECs with Maximum TUs greater than 0.01. This graph (b) is Metal COPECs, Except Mercury.

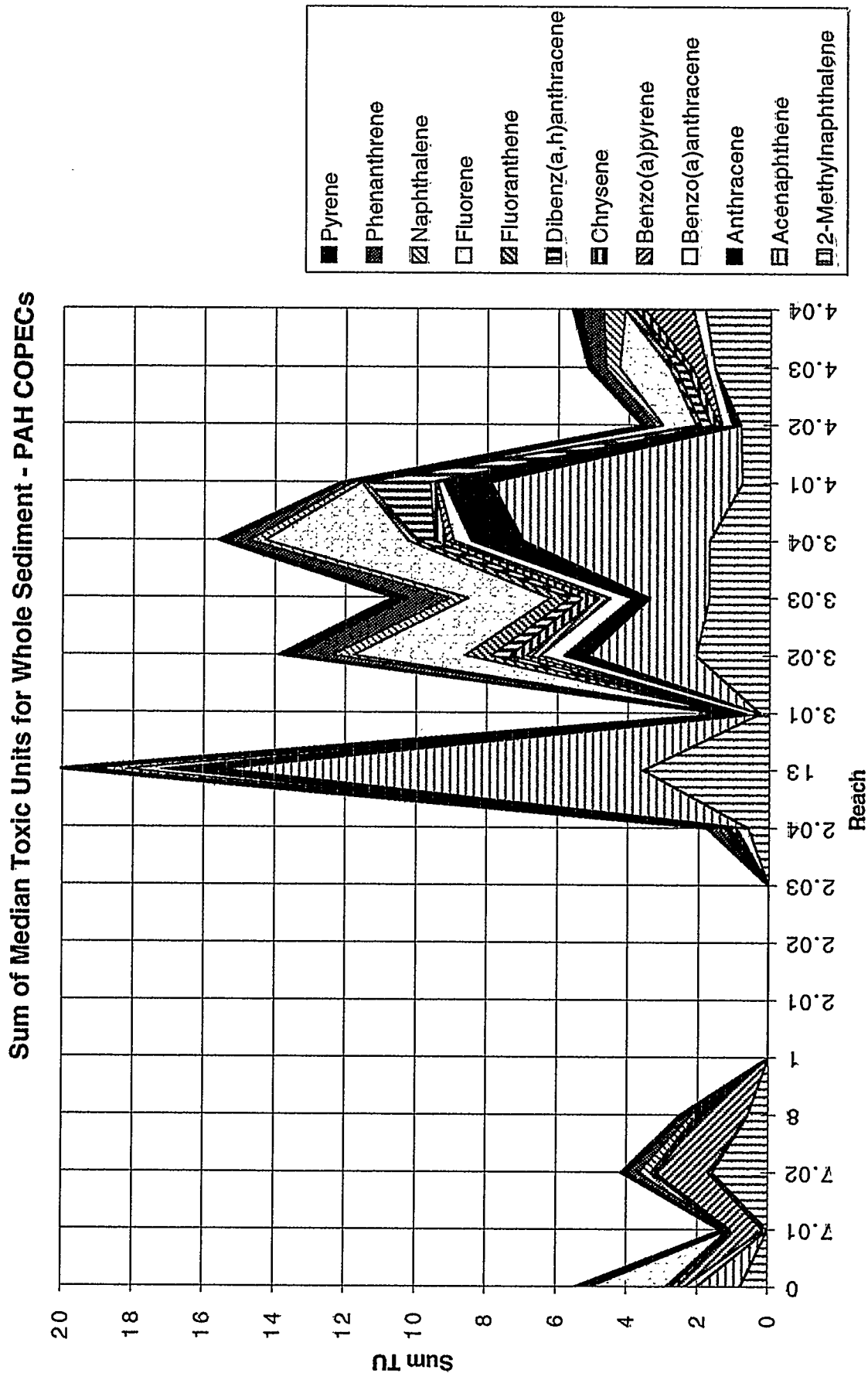


Fig. 6.14c. Plot of median concentration of whole sediment COPECs normalized to toxic units (TU). The Upper contour of the graph is the sum of TU for the reaches. Includes COPECs with Maximum TUs greater than 0.01. This graph (c) is PAH COPECs.

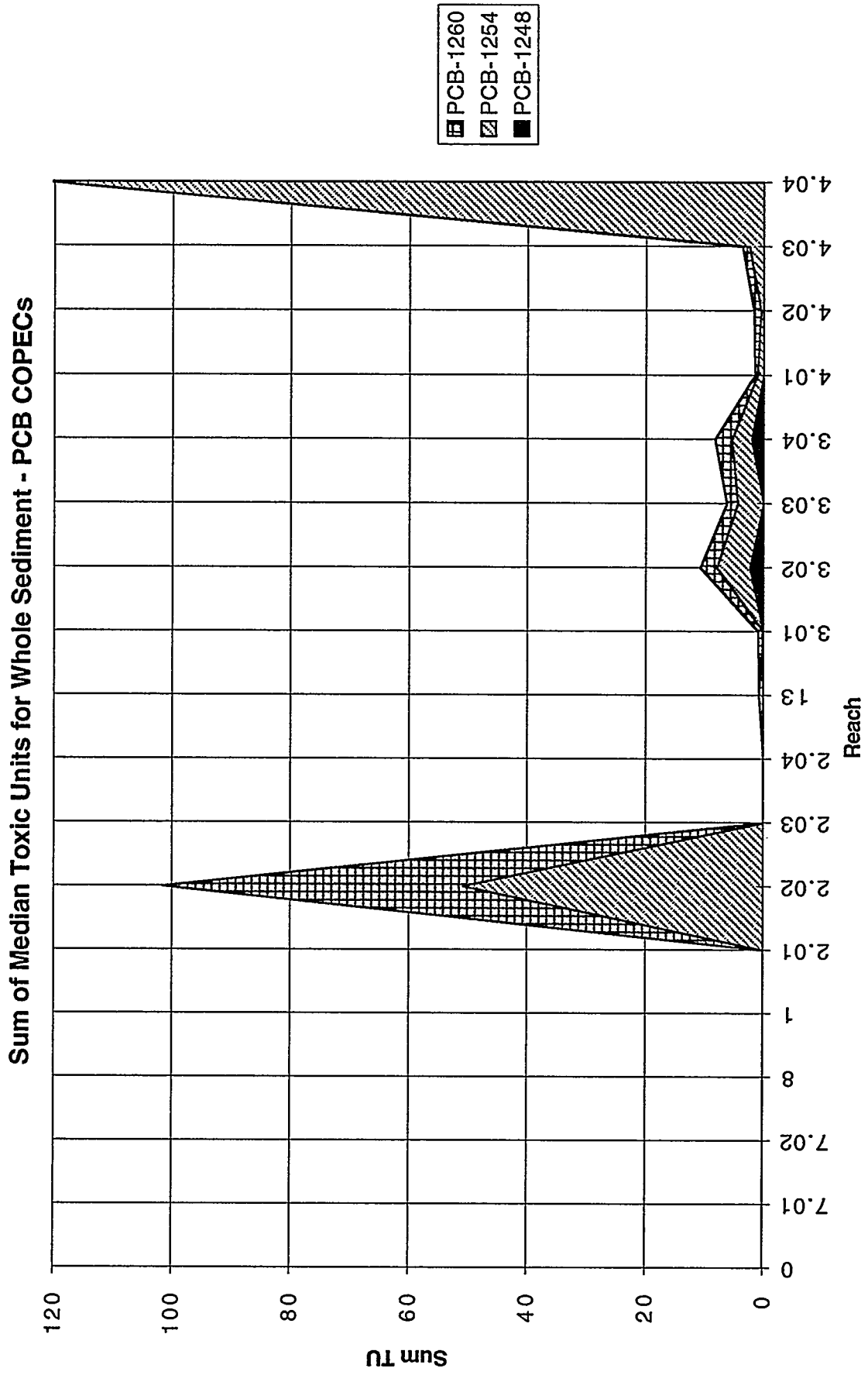


Fig. 6.14d. Plot of median concentration of whole sediment COPECs normalized to toxic units (TU). The Upper contour of the graph is the sum of TU for the reaches. Includes COPECs with Maximum TUs greater than 0.01. This graph (d) is PCB COPECs.

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.). Endpoints included taxa richness, diversity, density, mortality, growth, respiration, behavior (e.g., avoidance, emergence, reburial, etc.) and suborganismal effects (e.g., Mixed-Function 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 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 greater than 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 greater than 20%, but less than 50%, likelihood of toxic effects. Negligible risks are generally indicated if the likelihood of toxic effects is less than 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 (Reach 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 Reaches 0, 3.04, 4.01, 4.02, 4.04, and 7.02. Only Reach 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 (Reaches 7.01, 7.02, and 3.04), intermediate in areas where ash and ash-associated arsenic are diluted (Reaches 1, 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 Reach 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 Reaches 7 and 3.04. The intermediate sites (Reaches 1, 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 Reach 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 reach (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 Reach 3.04.

Cadmium. Cadmium appears to present a marginal risk in Poplar Creek below East Fork and at very localized hot spots in Reach 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 Reach 4.04. This concentration and the aqueous test endpoints 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 Reach 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 Reach 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 Reach 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 (3.01 and 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 Reaches 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 less than 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 endpoints for Cu are shown in Fig. F3.2.

Cu concentrations in Poplar Creek sediment appear to be more toxic below Mitchell Branch than in Reaches 13 and 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 (Reaches 4.03 and 4.04) and adjacent to the ORR (Reach 1) than in adjoining reaches. Cu toxicity is possible but unlikely in Reaches 1, 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 Reaches 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 Reaches 3.03, 4.04, and 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, Reach 2.04, and Reach 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 Reaches 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 endpoints 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 1994 and Barnett and Turner 1995). In general, these studies indicated that the dominant form was mercuric sulfide, that mercuric chloride constituted less than 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.

Hg concentrations in Poplar Creek sediment appear to be more toxic below Mitchell Branch than in Reaches 13 and 3.01 (Fig. F3.11b). However, there is only limited data from Reach 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 Reach 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 Reaches 3.02 and 3.03 and suggests that the ash disposal area may contribute to risks from exposure to Hg in some sediments of Reach 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 (Reaches 4.03 and 4.04) and at most sites in Reaches 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 endpoints 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 Reaches 13 and 3.01 (Fig. F3.12b). Ni concentrations exceeded the 50th percentile of the community level effects distribution at most of the sites in Reaches 3.02, 3.03, and 3.04, whereas concentrations in Reaches 13 and 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.

Ni 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 (Reaches 4.03 and 4.04) than in Reaches 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 Reach 4.04; a marginal risk in most of Reaches 1, 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 Reaches 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 Reaches 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 Reaches 1, 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 Reach 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 endpoints for Zn are shown in Fig. F3.15.

Approximately 30% of the sediment samples from Reach 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 (Reaches 2.04 and 4.01) and Poplar Creek (Reaches 3.02 and 3.03). Maximum concentrations in Reaches 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 Reach 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 Reaches 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 Reach 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 Reaches 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 Reach 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 Reaches 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 Reach 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: PCB-1248, PCB-1254, and PCB-1260. PCB-1248 was detected in two reaches (3.02 and 3.04) of Poplar Creek only, whereas PCB-1254 was present in each reach where PCBs were detected, except Reach 3.01. PCBs were not detected in pore water samples in these reaches and the concentrations estimated using the equilibrium partitioning model were less than the highly conservative SCV. This suggests that PCB toxicity is 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 (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 Reaches 13 and 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

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 less than 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 less than 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 Reaches 3.01 and 3.03 had significant mortality in subsequent tests. Furthermore, even with the first test included, the mean proportional reductions in survival were not biologically significant (i.e., reduced less than 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; Reach 2.03 and CRM 9.0; Reach 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 Reach 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 less than 20%).

Mortality of *H. azteca* in sediments from McCoy Branch Embayment (MBM 0.2; Reach 7.02 and MBM 0.4; Reach 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

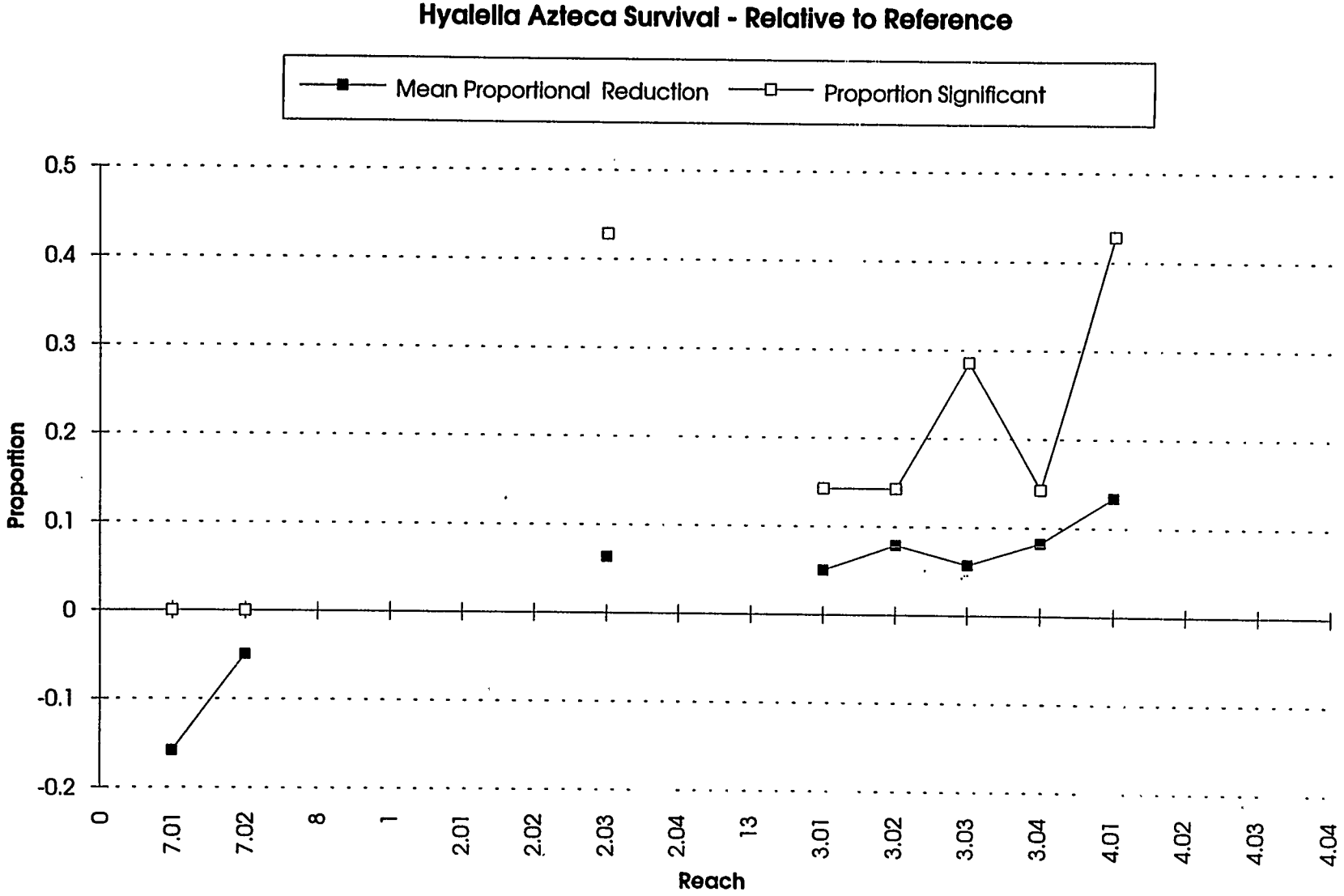


Fig. 6.15. Proportional reduction in *Hyalella azteca* survival relative to references and proportion of tests in which survival was statistically significantly decreased relative to controls or was decreased by at least 20% relative to controls (i.e., significantly decreased).

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 (Reach 4.04) and declines with increasing distance from Watts Bar Reservoir. The only exception to this pattern is Reach 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 Reach 4.01. Taxonomic richness of invertebrates is higher in Reach 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 Reaches 4.01 and above; chironomids predominate in Reaches 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 Reach 3.01, mile 5.1) are high in both abundance and richness. Reaches 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 (Reach 3.04).

These trends do not in and of themselves demonstrate a contaminant impact. Measured in terms of abundance and richness, Clinch River Reach 4.01, immediately below Poplar Creek is significantly different from Reach 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 Reaches 13, 3.02, 3.03, and 3.04 may be attributed to local conditions, including the presence of contaminants.

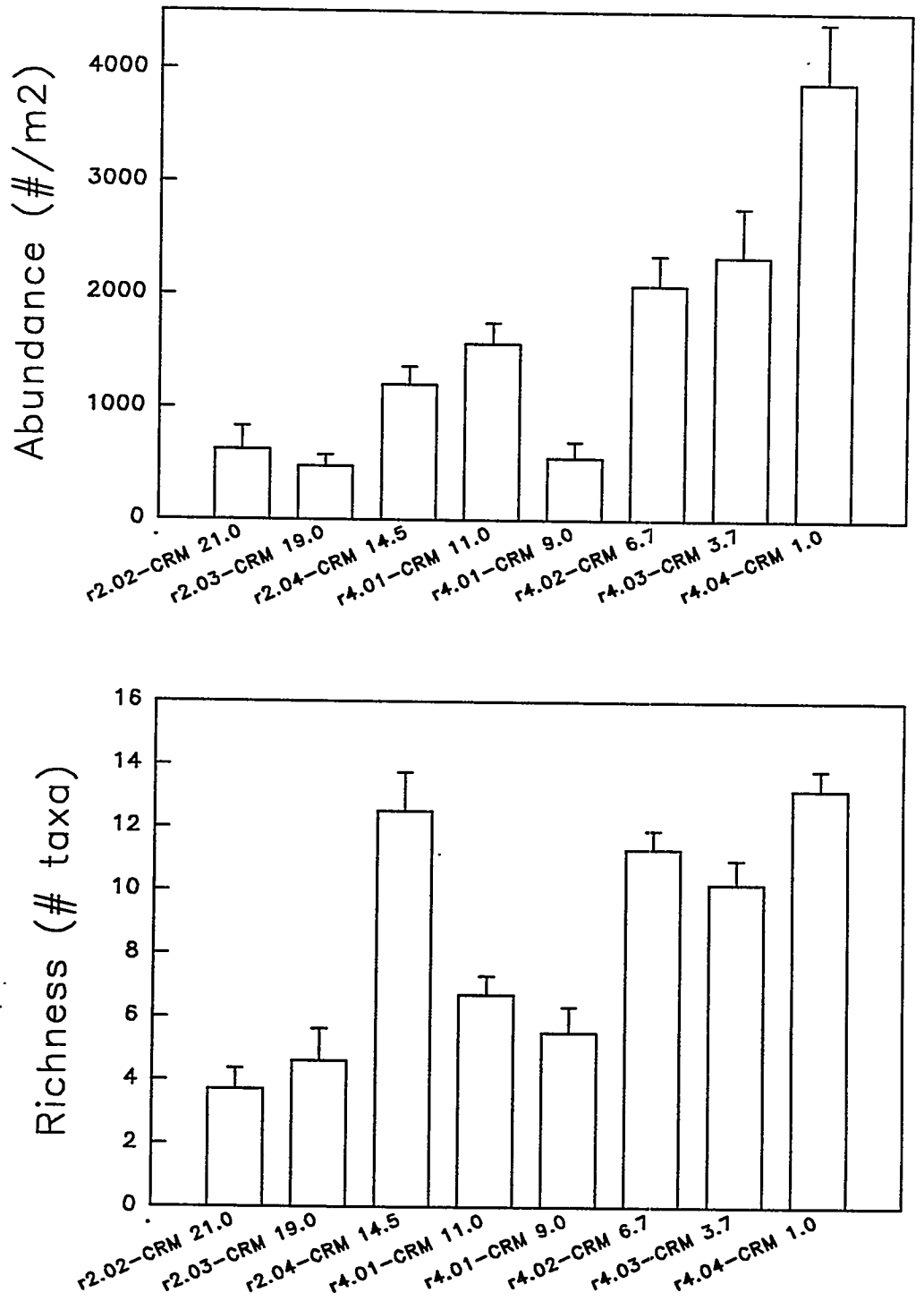


Fig. 6.16. Taxonomic richness and abundance (Mean ± SE) of benthic macroinvertebrates in reaches of the Clinch River in 1994.

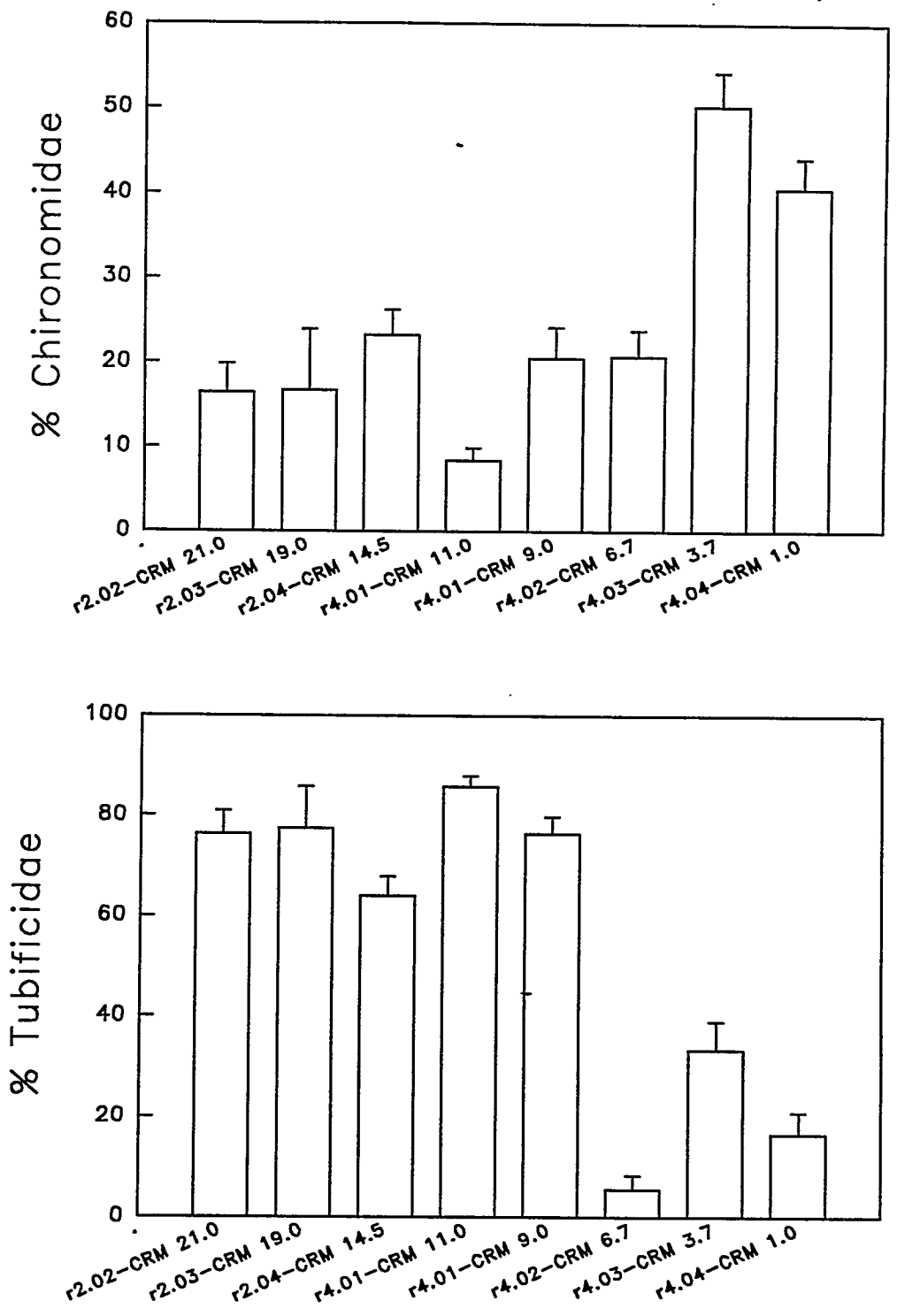


Fig. 6.17. Percentage of benthic macroinvertebrate community (Mean ± SE) represented by *Chironomidae* and *Tubificidae* in reaches of the Clinch River in 1994.

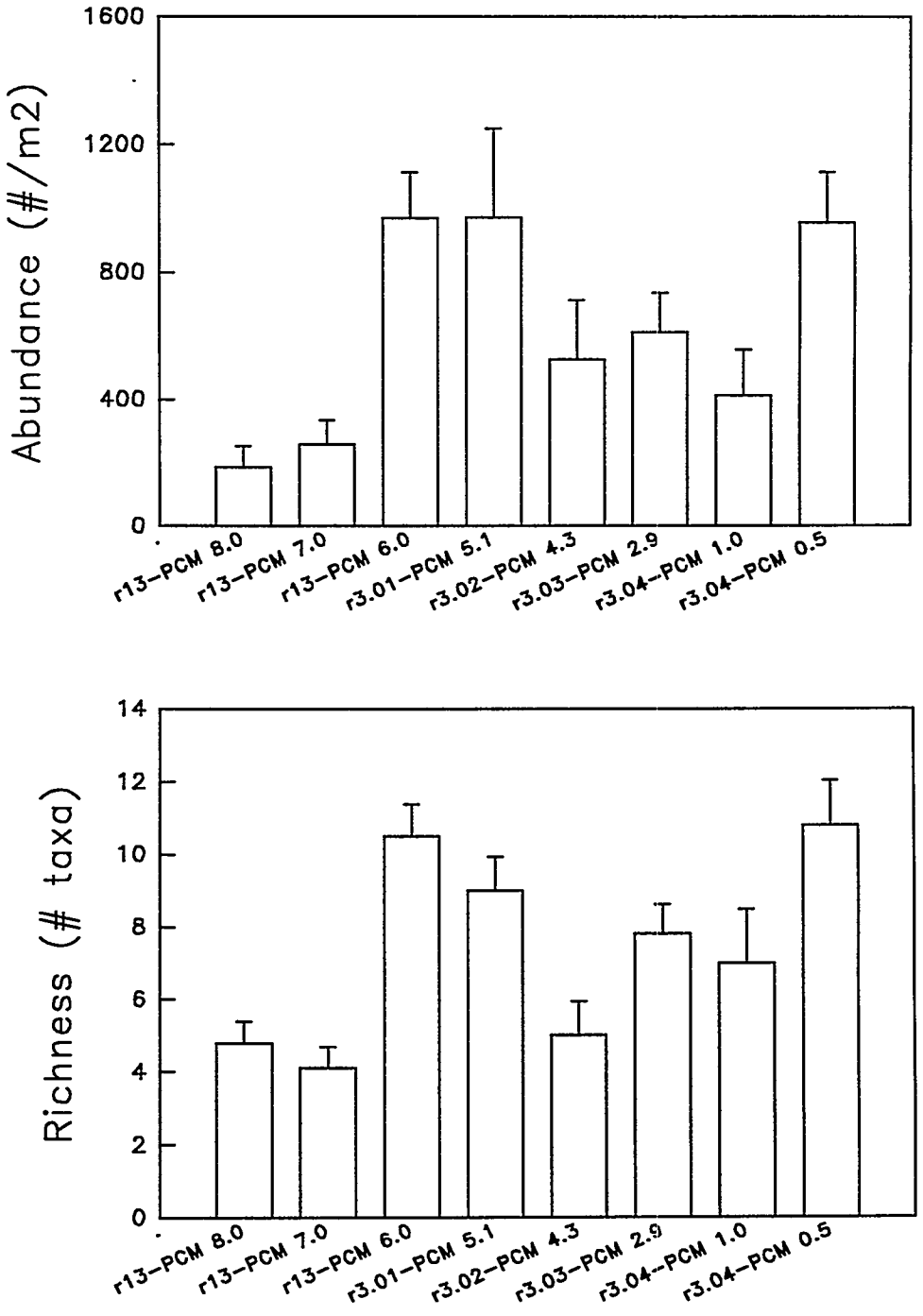


Fig. 6.18. Taxonomic richness and abundance (Mean ± SE) of benthic macroinvertebrates in reaches of Poplar Creek in 1994.

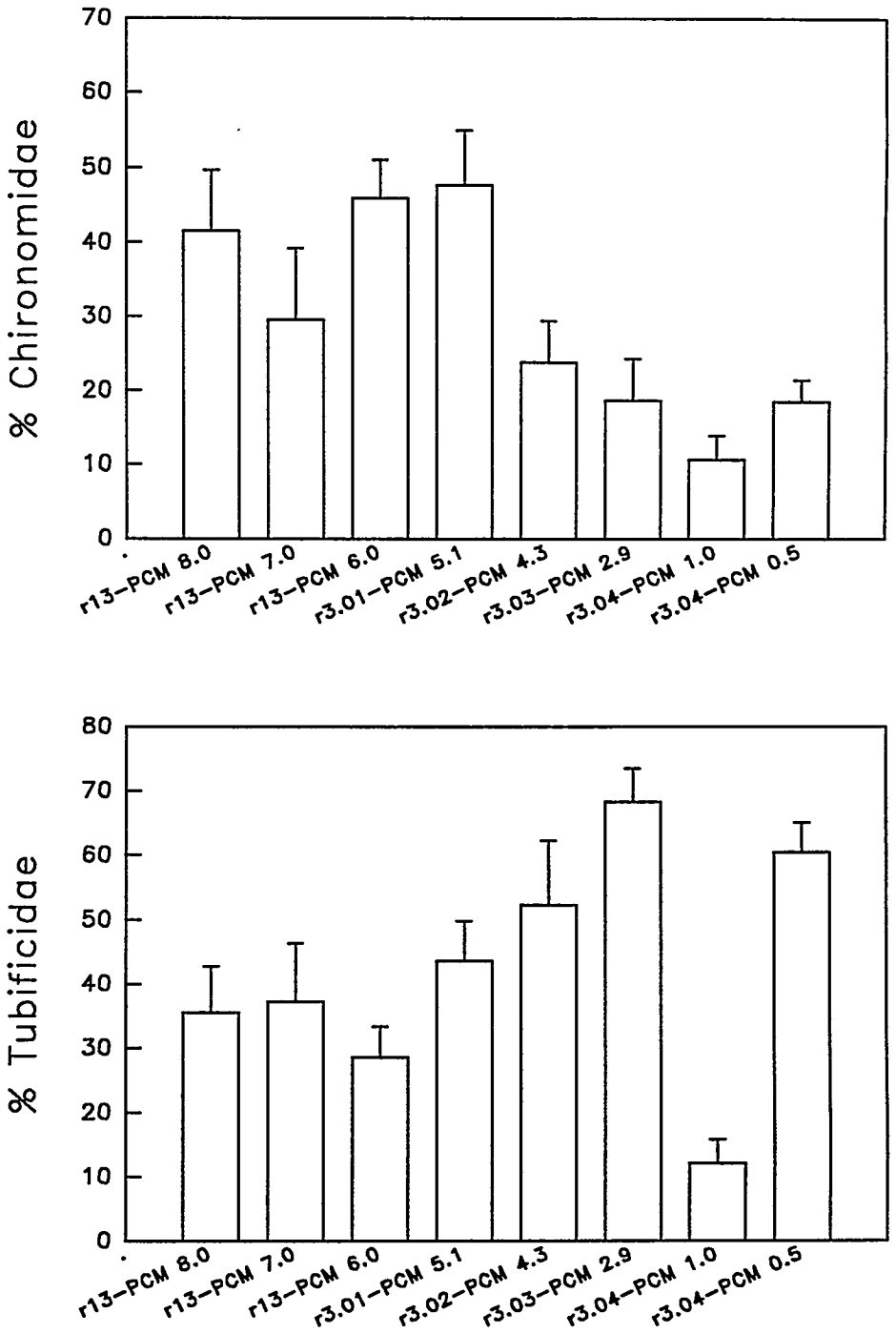


Fig. 6.19. Percentage of benthic macroinvertebrate community (Mean ± SE. represented by *Chironomidae* and *Tubificidae* in reaches of Poplar Creek in 1994.

The TVA (TVA 1994) qualitatively characterized the sediment in the CR/PC benthic invertebrates survey samples. The sites in Reaches 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 Reaches 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 Reach 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 Reach 3.04 was almost entirely composed of fine silt (TVA 1994). Poplar Creek sediments above Reach 3.04 were a mixture of fine and coarse sediments with varying amounts of silt present in nearly every survey sample (TVA 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 less than 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.

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[1]	Physical & Chemical Variables[2]	Chemical Variables Given Physicl Variables[3]	
		Clinch River		
Taxa Richness	52%	69%	17%	<0.0001
Dominance	57%	67%	10%	0.0018
Chironomidae[4]	64%	78%	14%	0.0001
Diptera (other)[4]	21%	44%	23%	0.0002
Hexagenia limbata[4]	83%	90%	7%	<0.0001
Tubificidae[4]	39%	53%	14%	0.0032
Veneroida[4]	55%	64%	9%	0.0140
		Poplar Creek		
Taxa Richness	49%	60%	11%	0.0157
Dominance	23%	28%	5%	0.5014
Chironomidae[4]	25%	44%	19%	0.0057
Diptera (other)[4]	33%	39%	5%	0.4412
Hexagenia limbata[4]	61%	70%	9%	0.0122
Tubificidae[4]	47%	53%	6%	0.2481
Veneroida[4]	45%	55%	11%	0.0336

[1] Physical variables included depth, substrate characteristics (% sand, % silt, % clay, % gravel), and total organic carbon.
[2] Chemical Variables included total PCBs, total PAH, 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.
[3] Variance explained by chemicals was derived by subtracting the R^2 for physical variables only from the R^2 for physical plus chemical variables.
[4] Abundance

Table 6.10. Signs of significant correlations between chemical variables and benthic invertebrate variables and the concsistency[1] 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[2]	2	2	3	3	0	4
PCBs (total)	3	2	2	3	1	3
PAHs (total)	2	3	2	1	3	3
Ammonia[2]	1	1	5	1	2	4
Aluminum[2]	2	2	3	1	3	3

[1] 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.

[2] 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.

[3] Concentrations in sediment pore water.

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?

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 greater than 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, reaches 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 endpoint 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	+	Reaches 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. Arsenic and Cr are significant at localized hot-spots below Mitchell Branch and East Fork. PAHs 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 endpoint 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.

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 PAHs, but no COPEC 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 Reach 4.01 was the only site tested.
Media Analyses	+	Risks from PAHs and Hg are significant at most sites in at least one reach, 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 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. Reach 4.01 had reduced richness and abundance relative to Reach 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 endpoint 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 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 As 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 endpoints 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 1985-NH4). 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 subreaches 0, 1, 2.01, 2.02, 2.04, 13, 3.01, 3.02, 3.03, 3.04, 4.01, 6, 4.04, 18, 5, and 15. 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 is the sum of the exposures attributable to each source and may be described as:

$$E_{total} \approx E_{food} + E_{water} + E_{soil}$$

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 endpoints 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 may then be described as:

$$E_{total} \approx E_{food} + E_{water}$$

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:

$$E_j = \sum_{i=1}^{m} \left(\frac{IR_i \times C_{ij}}{BW} \right) \qquad (1)$$

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 endpoint species (kg)

For the purposes of this assessment, the generalized model was modified to reflect the foraging characteristics of the endpoint 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 for Clinch River piscivorous wildlife is as follows:

$$E_j = \left(\frac{IR_w \times C_{w\text{-}j}}{BW} \right) + \left(P_a \times \sum_{i=1}^{n} \frac{(P_{f\text{-}i} \times IR_f) \times C_{ij}}{BW} \right)$$

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 95% UCL is used in exposure estimates. To prevent bias that may result from calculating 95% UCLs using data that contains values below the detection limit, product limit estimator was used to calculate the 95% UCLs 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 95% UCL 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 Endpoint 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 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 95% UCLs [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 endpoints 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 endpoint-subreach-contaminant combinations where comparison of point exposure estimates to NOAELs produced HQs≥1 (NOAELs are presented in Sect 6.4.2.1.; Screening of exposure estimates against NOAELs is presented in Sect. 6.4.3.1.). Endpoint-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: endpoint 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

Endpoint	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

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 endpoint 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 less than 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 C13 and C15). Hg concentrations in feathers and liver (Table C14) and PCB concentrations in fat (Table C16), liver (Table C17), and muscle (Table C18) 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 endpoint, 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). 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:

$$D = \frac{d \; x \; bw}{bw^{2/3}} = d \; x \; bw^{1/3}$$

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:

$$NOAEL_w = NOAEL_t \left(\frac{bw_t}{bw_w} \right)^{1/3}$$

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 endpoints, 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 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:

$$C_w = \frac{NOAEL_w \; x \; bw_w}{W} \qquad (19)$$

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 C24).

Ten mink (8 females and 2 males) were fed each diet for approximately 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 C22), and Aroclor 1260 (and other PCB congeners) in liver and fat (Tables C25 and C26) 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 C27 and C28), 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 C19) 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 endpoint. Single chemical toxicity data are available for all four endpoints. 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 greater than 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 endpoints (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 endpoints; 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 endpoint 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 95% UCL 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 subreach 1, no contaminant produced an HQ>1 for any endpoint in any Clinch River reach. In subreach 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 subreach.

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 endpoint species expected to be present in each watershed.

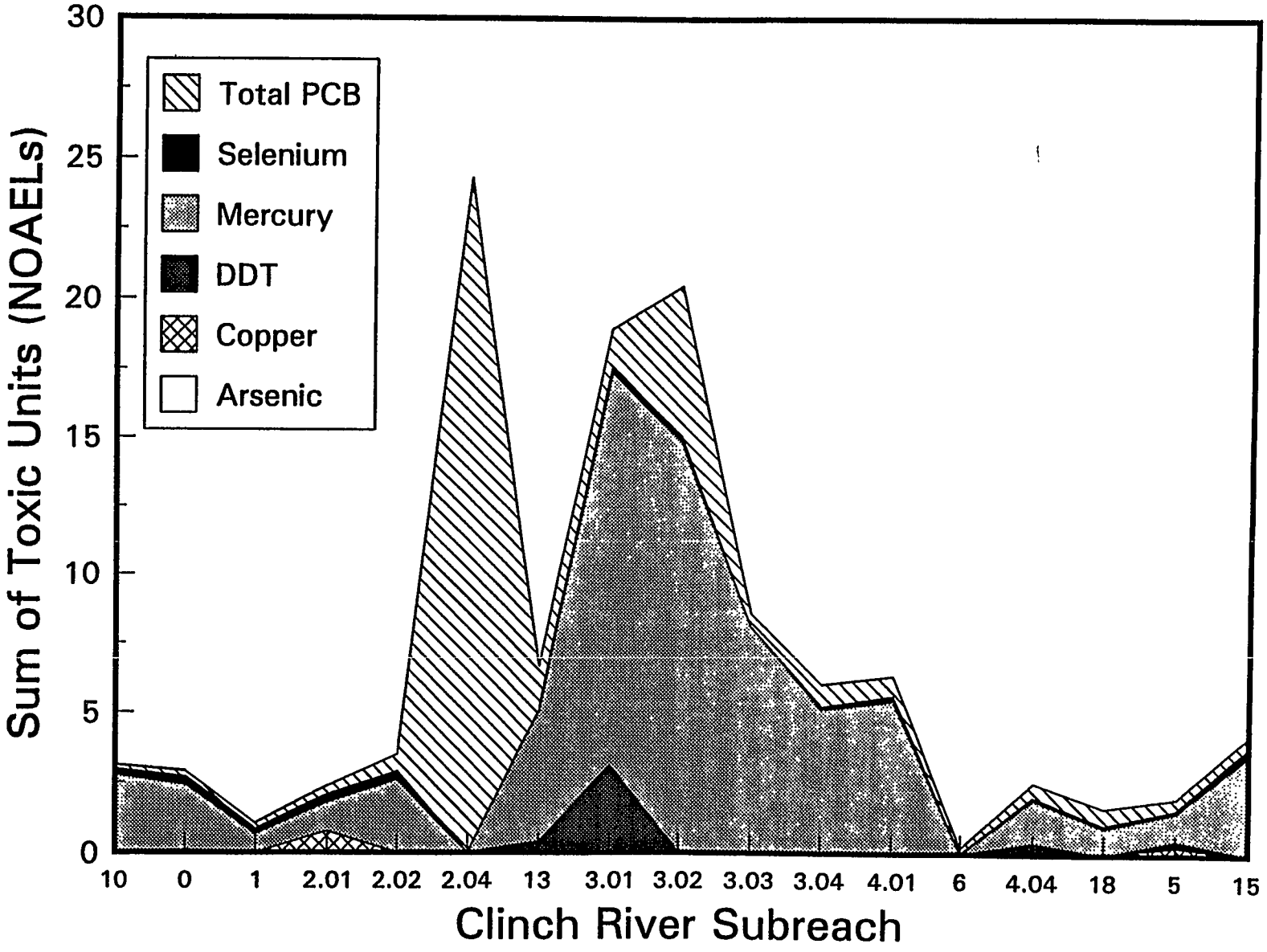


Fig. 6.20. Sum of NOAEL-based toxic units for osprey in the Clinch River/Poplar Creek system.

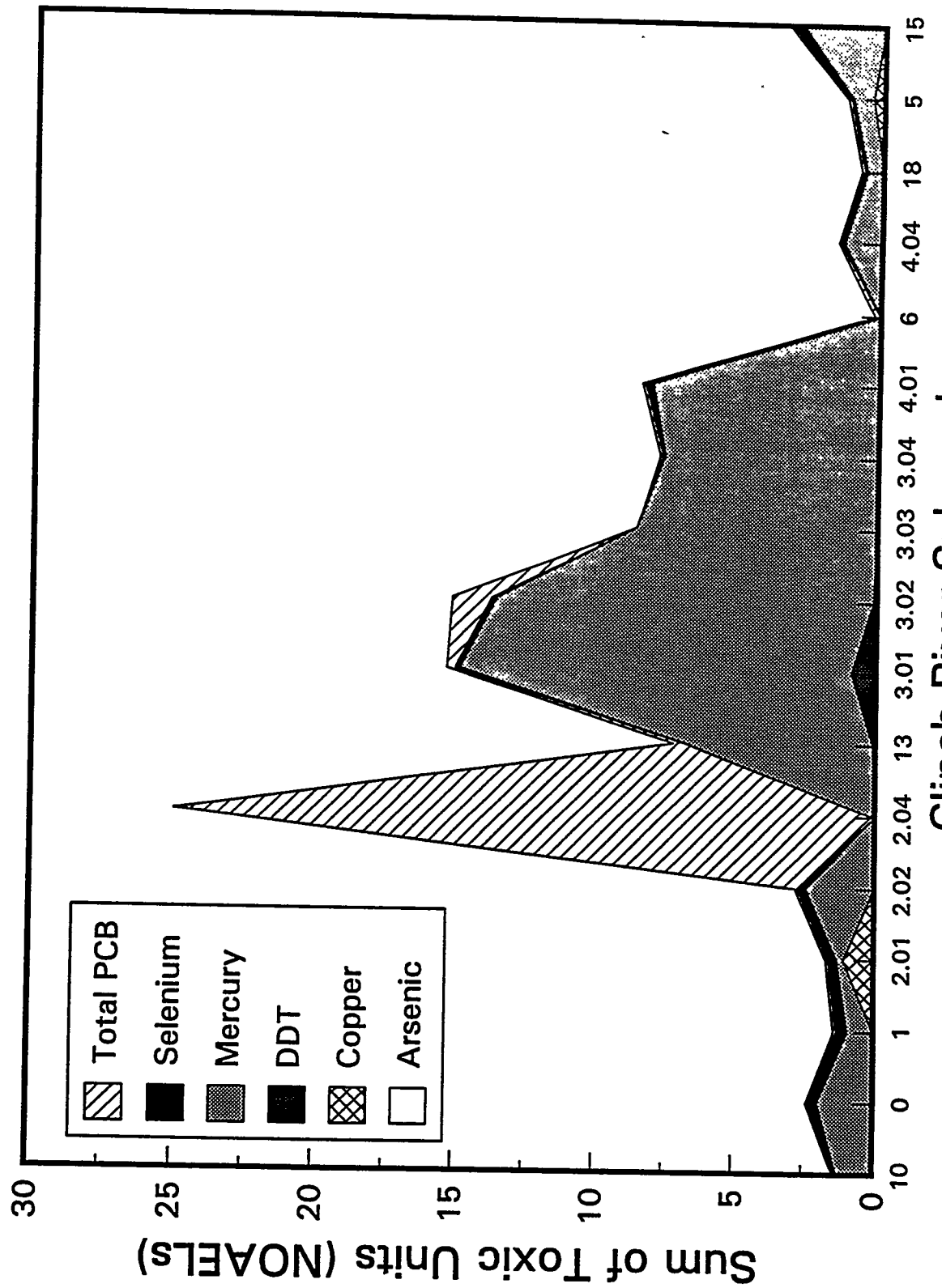


Fig. 6.21. Sum of NOAEL-based toxic units for great blue heron in the Clinch River/Poplar Creek system.

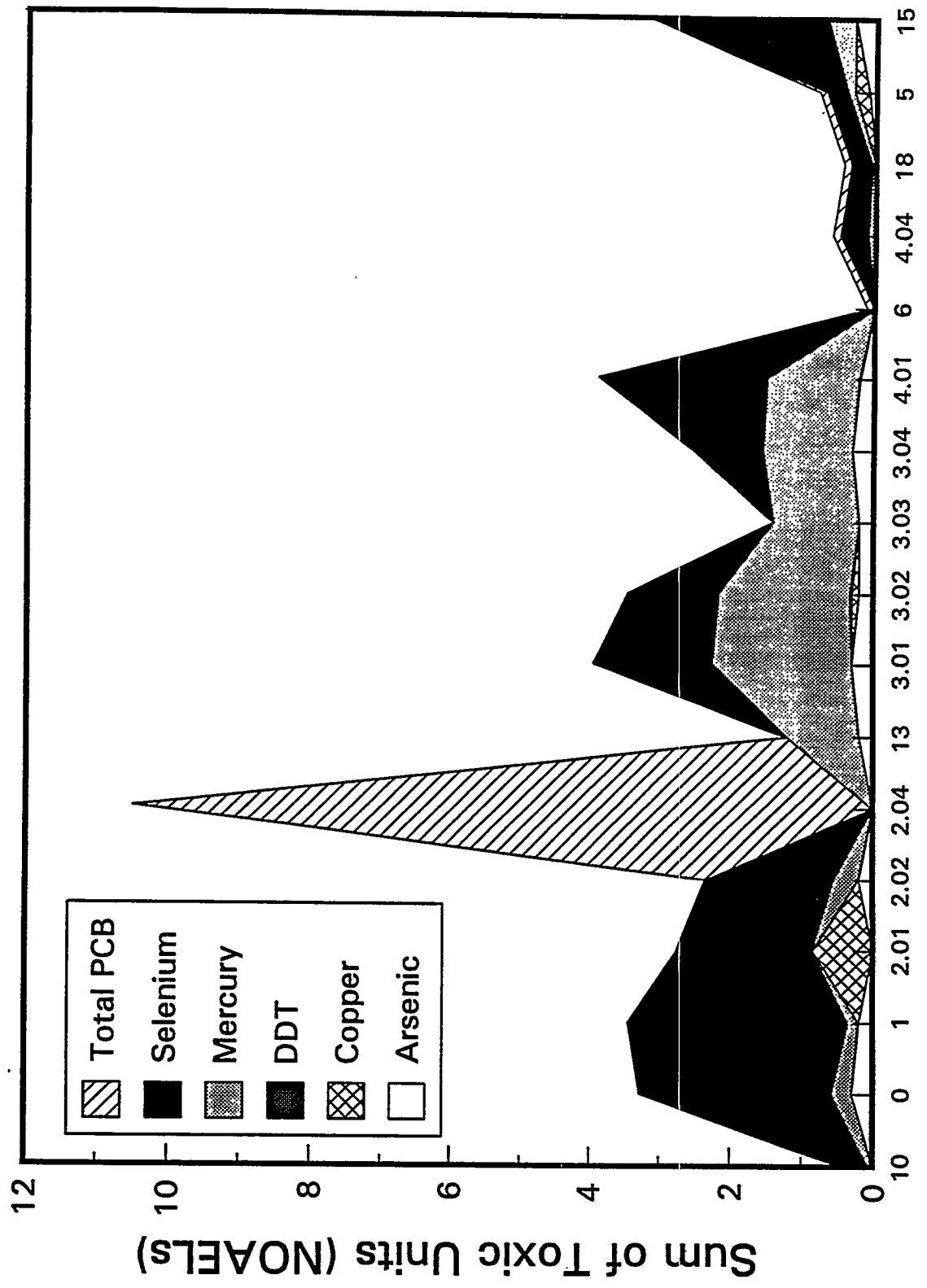


Fig. 6.22. Sum of NOAEL-based toxic units for mink in the Clinch River/Poplar Creek system.

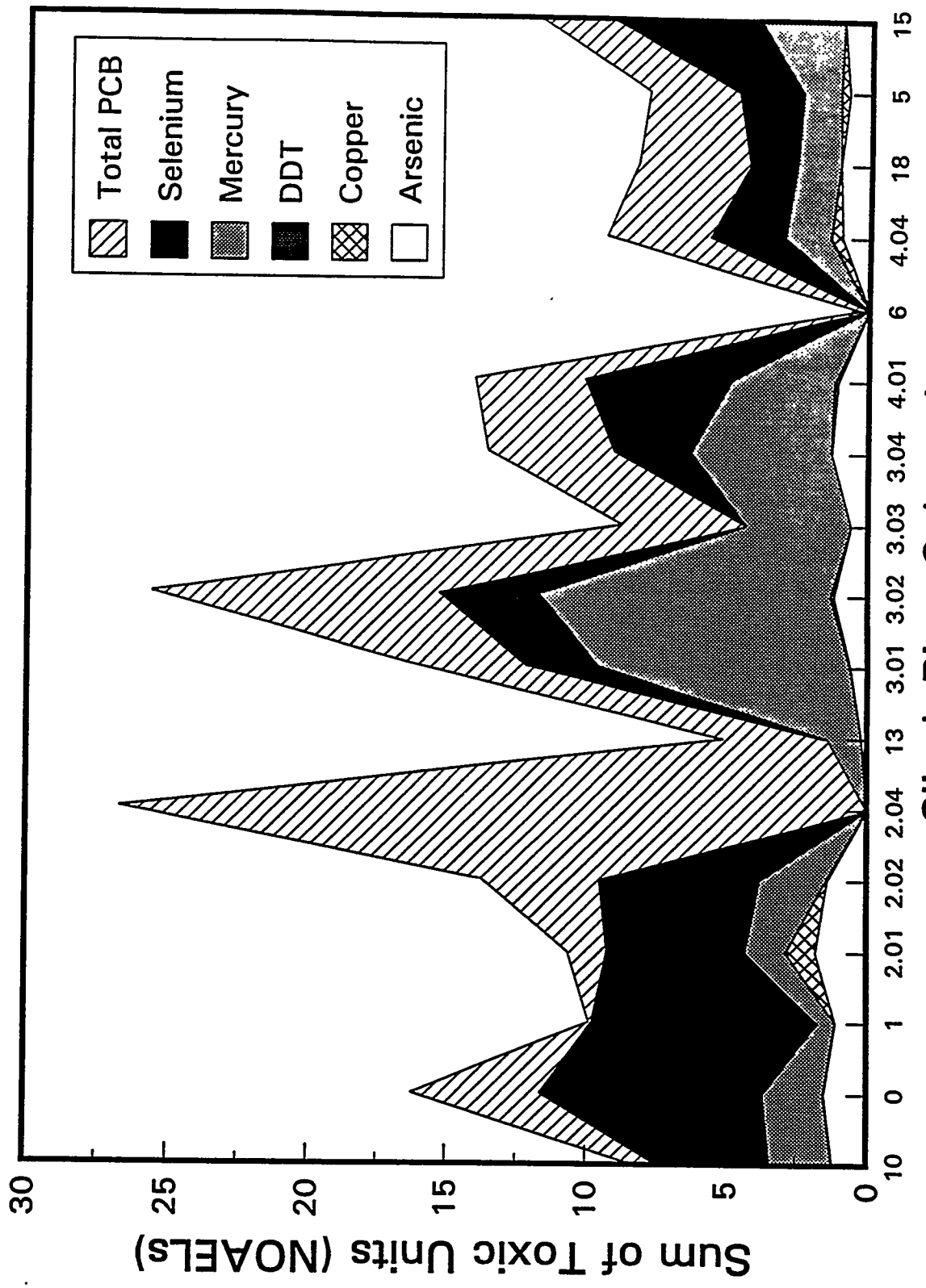


Fig. 6.23. Sum of NOAEL-based toxic units for river otter in the Clinch River/Poplar Creek system.

Table 6.17. Comparison of exposure distributions to NOAELs and LOAELs

Comparison	Meaning	Risk-based Interpretation
NOAEL>80th percentile of exposure distribution	less than 20% of exposures> NOAEL	Individual- and population-level adverse effects are highly unlikely
NOAEL<80th percentile<LOAEL	More than 20% of exposures > NOAEL, but less than 20% of exposures> 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 ORR population.
LOAEL<80th percentile of exposure distribution	More than 20% of exposures> LOAEL	Effects on some individuals are likely and they may contribute significantly to effects on the ORR population.

Literature-derived population densities used for each endpoint 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 endpoint 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:

$$b(y;n;p) = \binom{n}{y} p^{y} (1-p)^{n-y}$$

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.

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

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.

Binomial probability distributions were generated only for contaminant-endpoint-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 less than 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 greater than 80%, it was assumed that all individuals within that subreach were experiencing adverse effects. The total numbers of individuals of each endpoint 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 (1993) 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 (Subreach 1) and one on Poplar Creek approximately at the border between subreaches 3.03 and 3.04 (Fig. 6.30). While the Subreach 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 approximately 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.

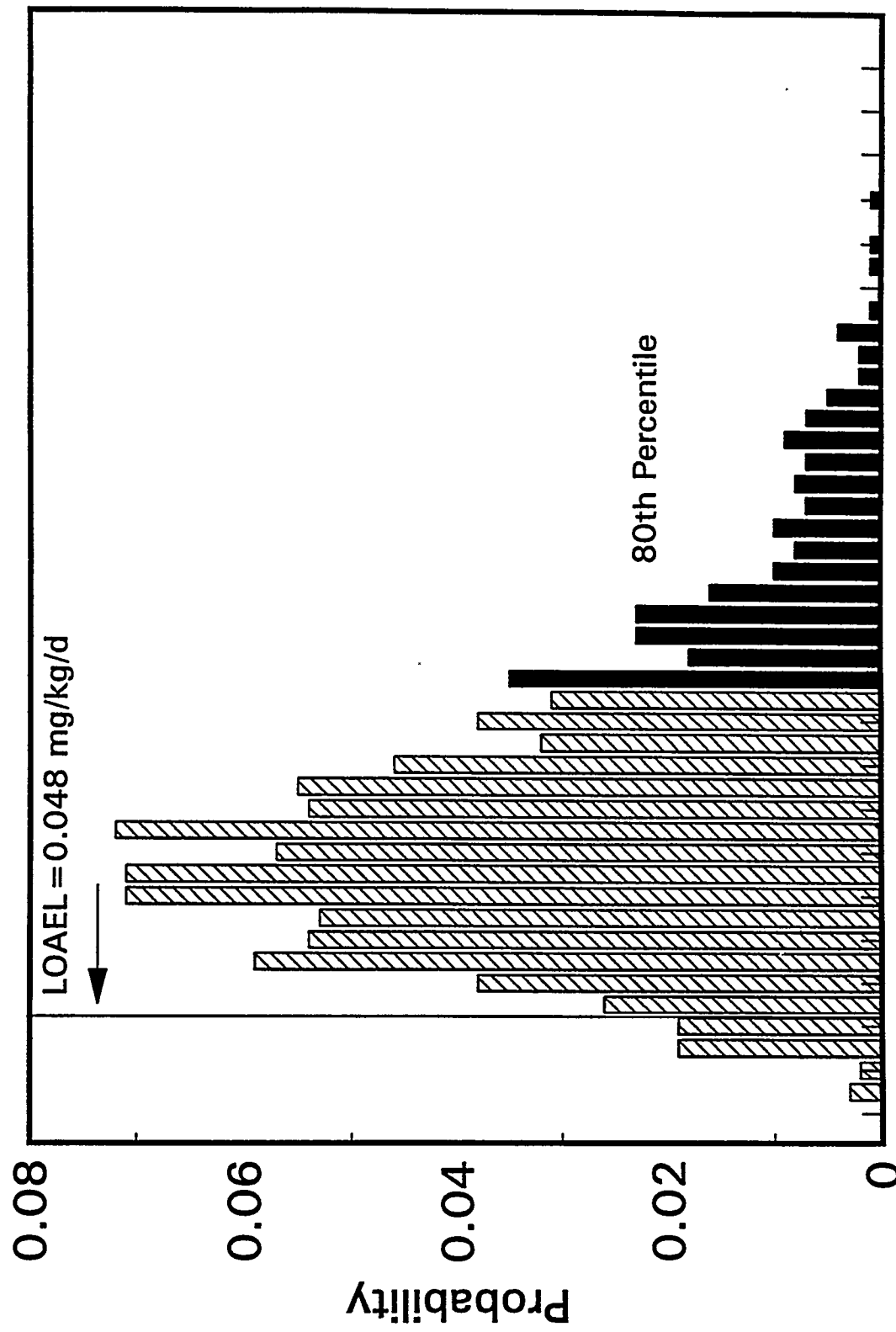


Fig. 6.24. Estimated distribution of mercury exposure for great blue heron in subreach 3.01. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution > LOAEL, adverse effects to individuals resulting from contaminant exposure are likely.

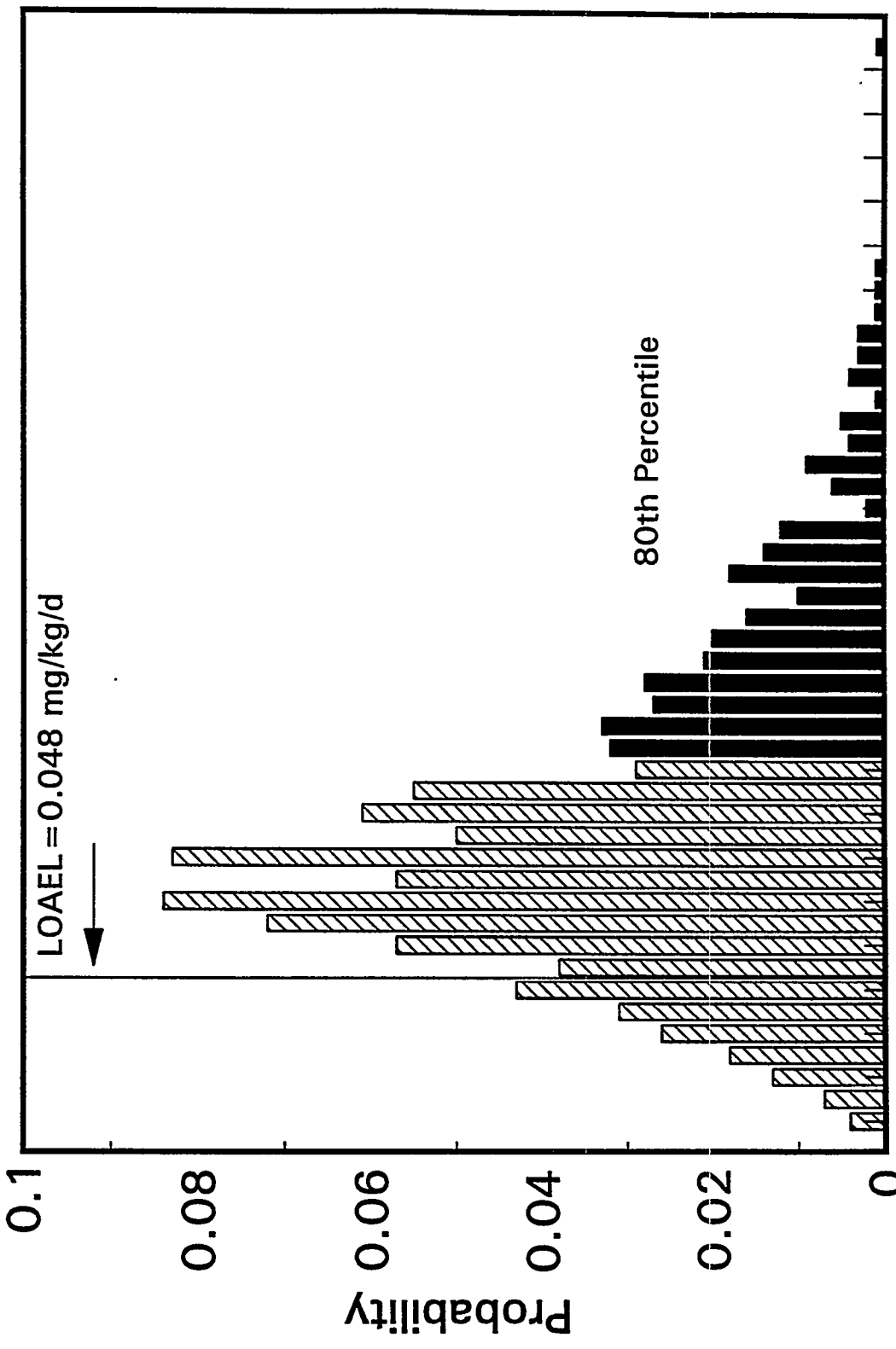


Fig. 6.25. Estimated distribution of mercury exposure for great blue heron in subreach 3.02. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution > LOAEL, adverse effects to individuals resulting from contaminant exposure are likely.

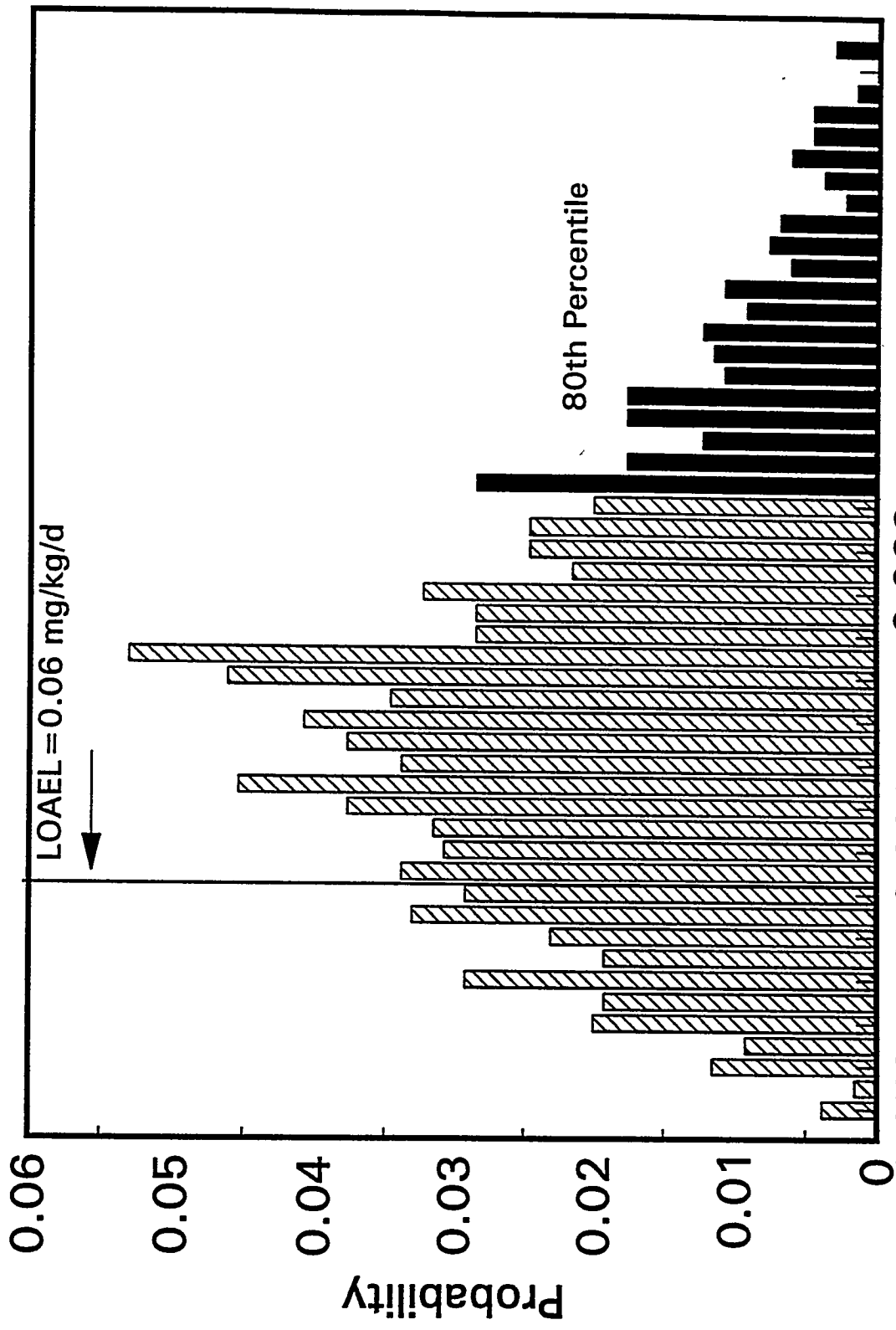


Fig. 6.26. Estimated distribution of mercury exposure for river otter in subreach 3.01. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution > LOAEL, adverse effects to individuals resulting from contaminant exposure are likely.

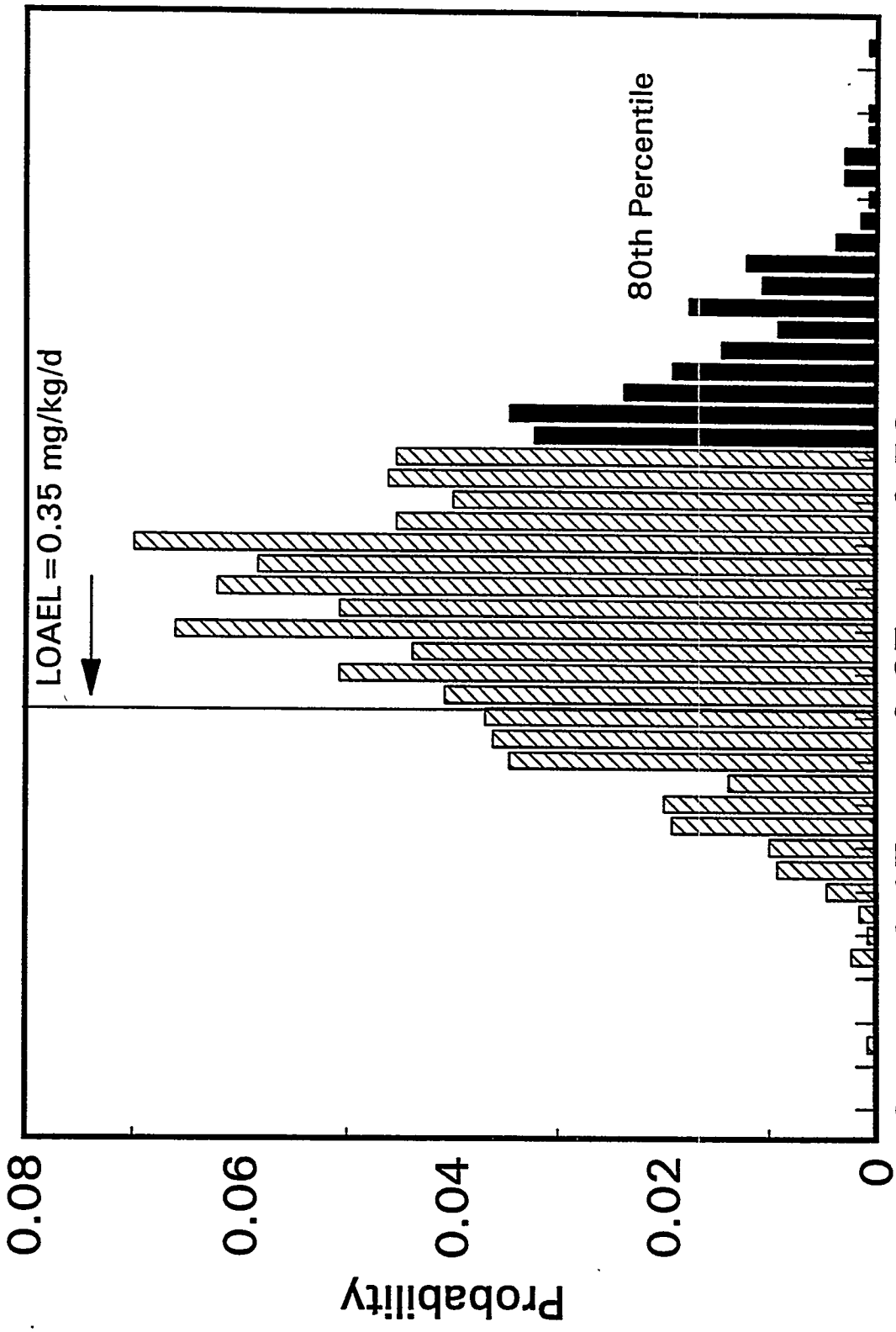


Fig. 6.27. Estimated distribution of PCB exposure for river otter in subreach 2.04. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution > LOAEL, adverse effects to individuals resulting from contaminant exposure are likely.

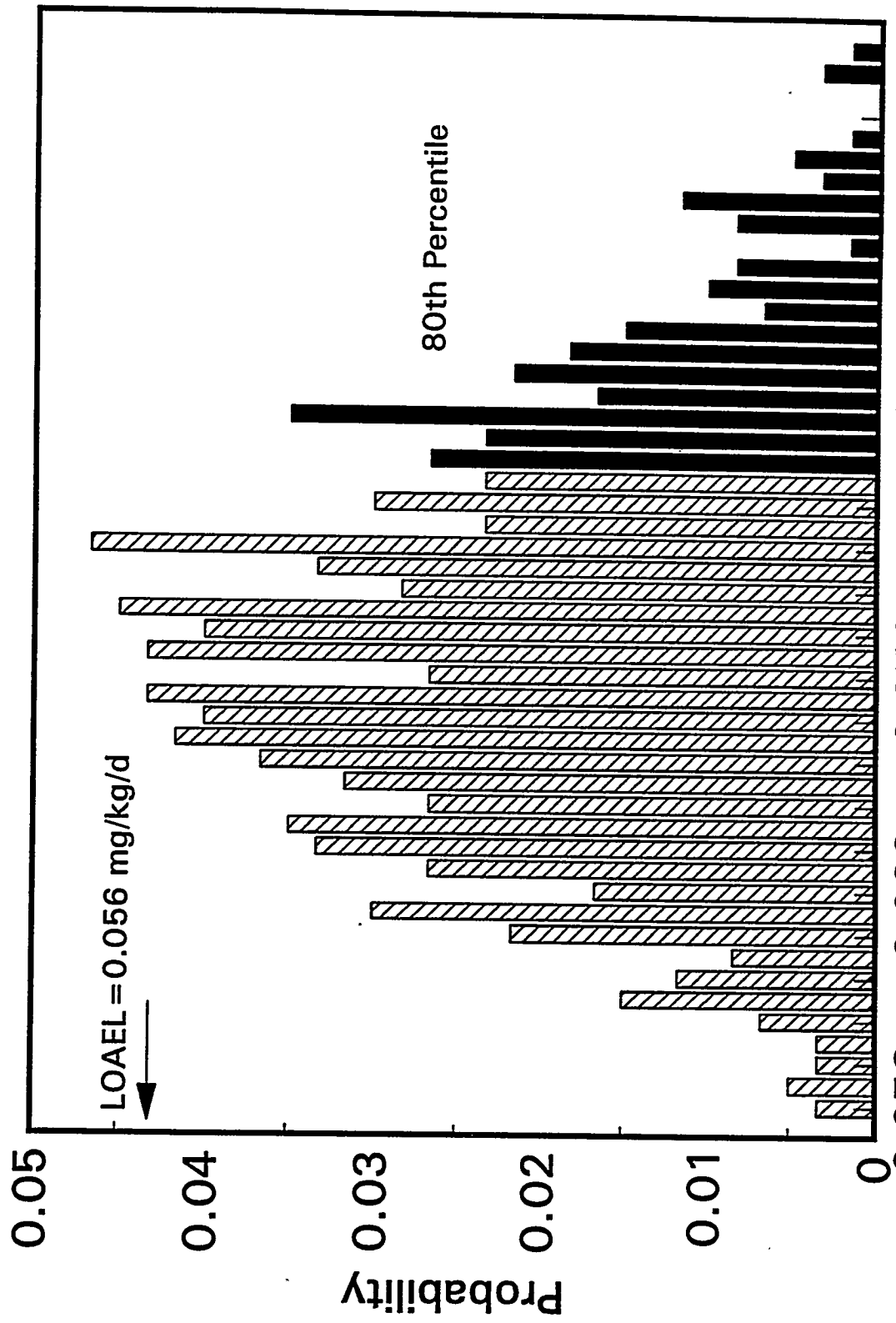


Fig. 6.28. Estimated distribution of mercury exposure for osprey in subreach 3.01. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution > LOAEL, adverse effects to individuals resulting from contaminant exposure are likely.

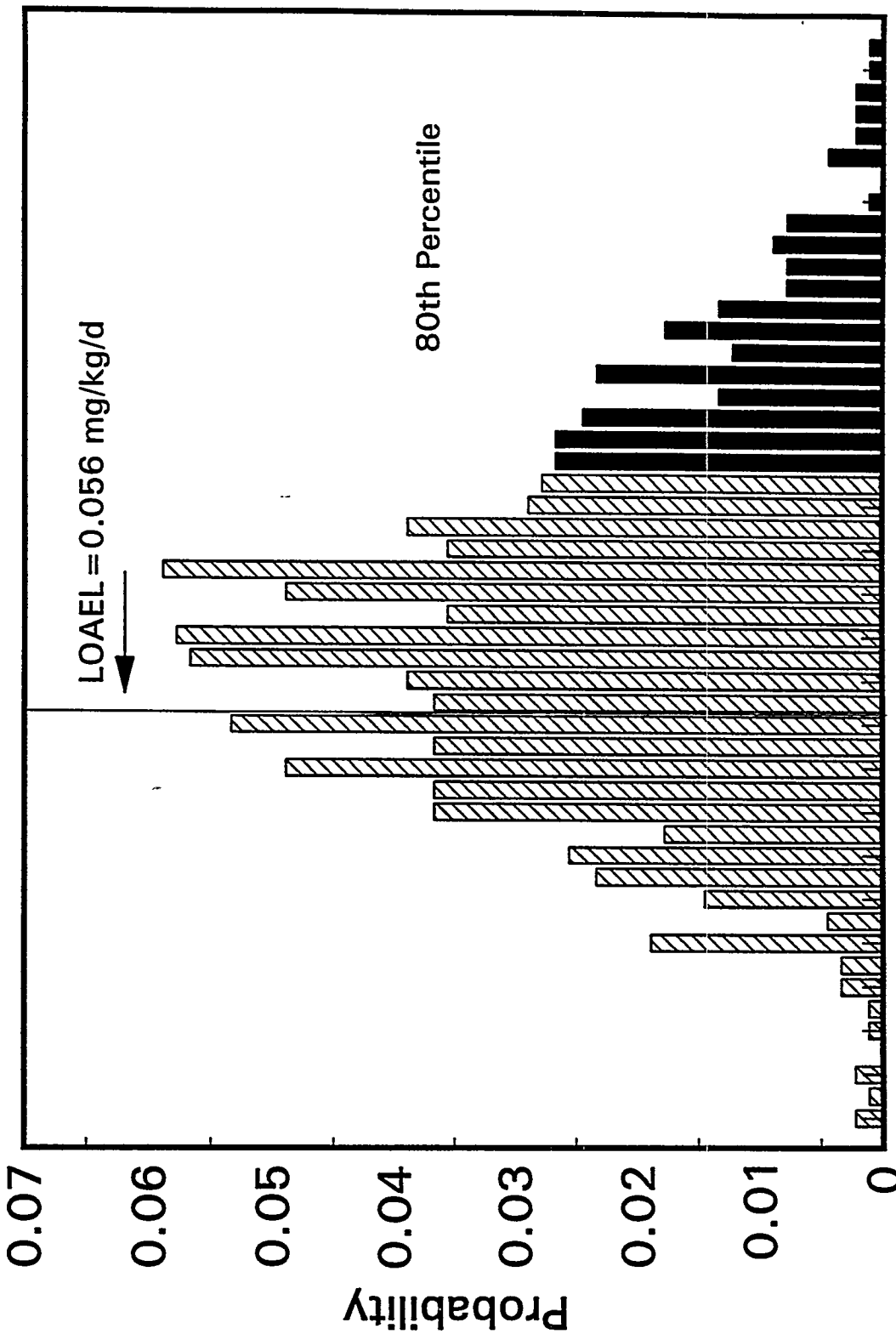


Fig. 6.29. Estimated distribution of mercury exposure for osprey in subreach 3.02. Distribution generated from Monte Carlo simulation of exposure model. If 80th percentile of distribution > LOAEL, adverse effects to individuals resulting from contaminant exposure are likely.

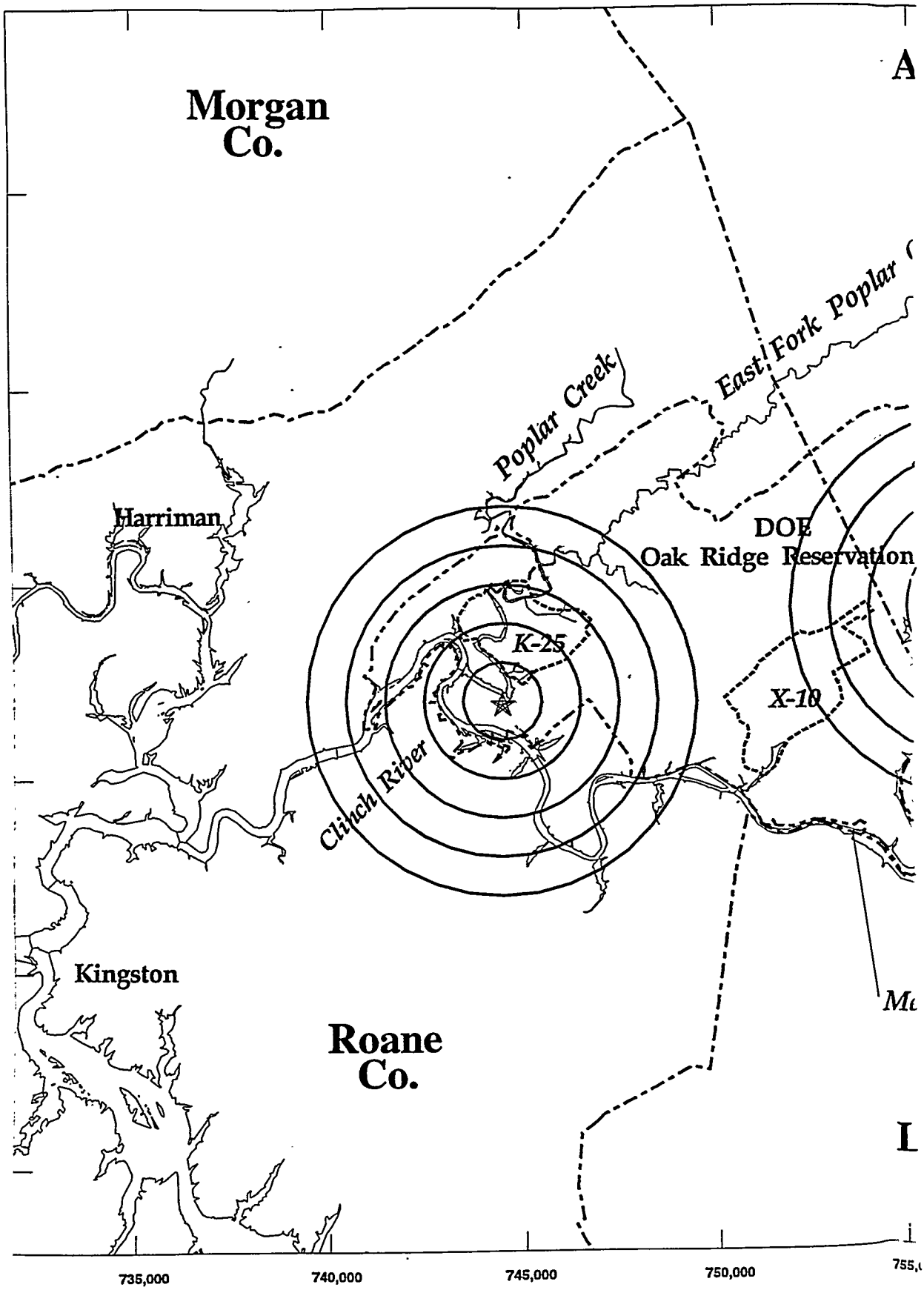
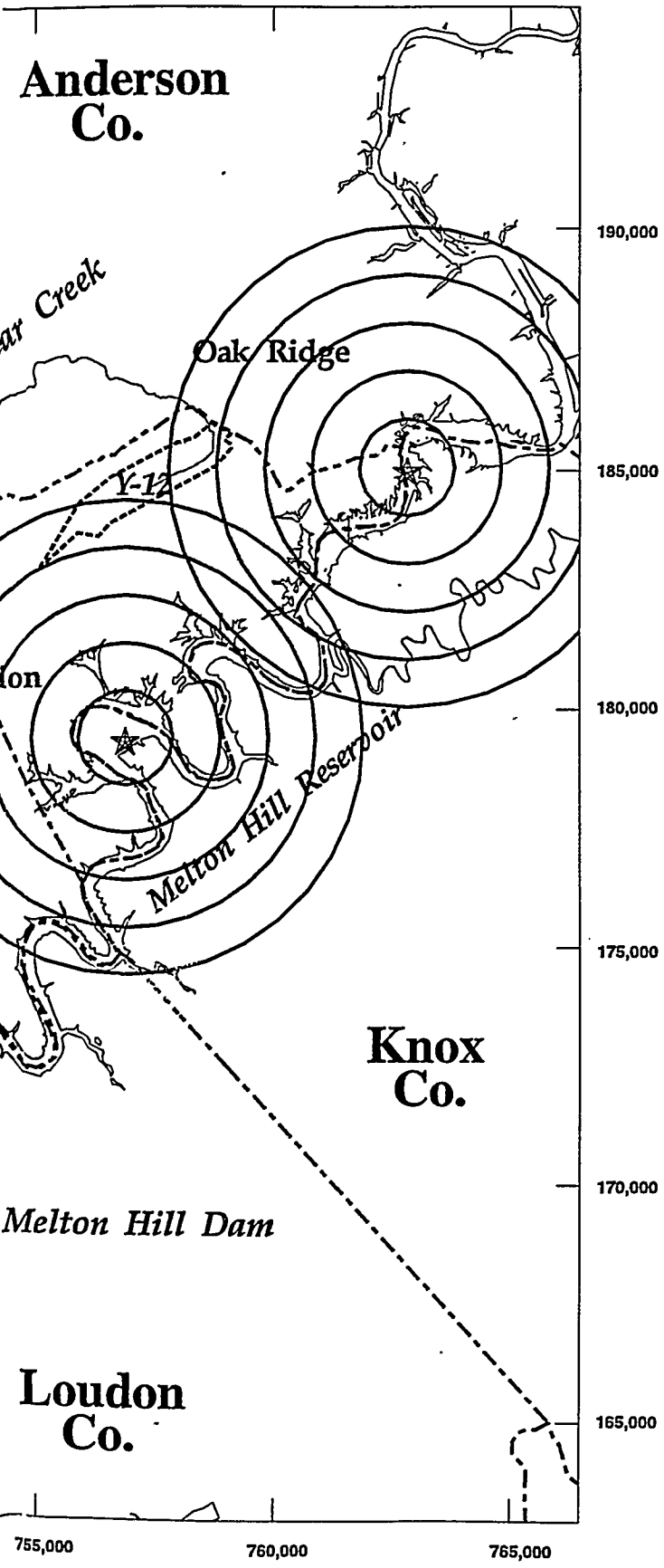


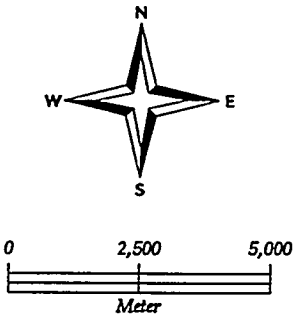
Fig. 6.30. Location of active osprey nesting locations adjacent to the Oak Ridge Reservat



ervation. Circles represent 1 km radii from nest.

Clinch River Environmental Restoration Program

Osprey Nests and Approximate Ranges



LEGEND	
★	Nest Location
▭	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 Clinch River Environmental Restoration Program

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Table 6.19. Subreaches, approximate distance from nest and estimated contribution to total diet for the nest

Subreach	Distance from Nest	Percent Contribution to Diet
4.01	~1-4	5%
2.04	No Data	
2.03	No Data	
2.02	~5	5%
13	4-5 km	5%
3.01	~3-4 km	15%
3.02	~3 km	20%
3.03	0-2km	25%
3.04	0-1 km	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

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 subreaches 13, 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 d (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.

PCBs. The 80th percentile for PCB exposure experienced by osprey exceeded the NOAEL but not the LOAEL at subreaches 2.04, 13, 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 approximately 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 C20 and C24). 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 PCB 1254 or PCB 1260 because they have ~54% or 60% Cl. The congener makeup of PCB 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 bue 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 C13 and C15), Hg in feathers and liver of chicks (Table C14) and PCBs in fat (Table C16), liver (Table C17), and muscle (Table C18) 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 C19). 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 (1993) (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 subreach 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. ORR-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

[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 less than 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

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.

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.

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.

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.

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.

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.

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.

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.

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 endpoint 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.

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 et al. 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 endpoints, 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 subreaches for which data were available (Subreaches 0, 1, 2.01, 2.02, 2.04, 3.01, 3.02, 3.03, 3.04, 4.01, 4.04, 5, 6, 10, 13, 15, and 18).

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 endpoint 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 endpoints 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:

$$E_j = \left(\frac{IR_w \times C_{w\text{-}j}}{BW}\right) + \left(\frac{(C_{i\text{-}j})(P_i) \times IR_i}{BW}\right)$$

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 95% UCL is a conservative estimate of exposure for wildlife. The PLE was used to calculate the 95% UCLs 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 endpoint 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 95% UCL 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, 4.04 and 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 subreach 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 endpoints 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 endpoint-subreach-contaminant combinations where comparison of point exposure estimates to NOAELs produced HQs $\geq$ 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) endpoint 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

Endpoint	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

Monte Carlo Simulation could not be performed for exposure to mercury (both rough-winged swallows and bats) and PCBs (bats only) at subreach 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 endpoint 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 (1993). 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 endpoint, 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 endpoints 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 endpoint 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 endpoint 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 greater than one 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 PCB-1254. The importance of each contaminant, relative to an endpoint, 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 endpoints was observed at subreach 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, 3.04, and 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 endpoint in Clinch River subreaches 0, 1, 7.01, 7.02, 8, 2.01, 2.02, 2.03, 2.04, 13, 3.03, 4.01, 4.02, 4.03, 4.04, 18, or 5.

Table 6.22. Reaches where HQs > 1 were observed

		Contaminants Where HQ > 1	
Subreach	**Subreach Name**	**Rough-winged Swallow**	**Little Brown or Gray Bats**
3.01	Poplar Creek: East Fork Poplar Creek	Hg	
3.02	Poplar Creek: Mitchell Branch	Hg	
3.04	Poplar Creek Mouth	Hg	Hg
22	White Oak Creek Embayment	Hg	Hg, PCBs

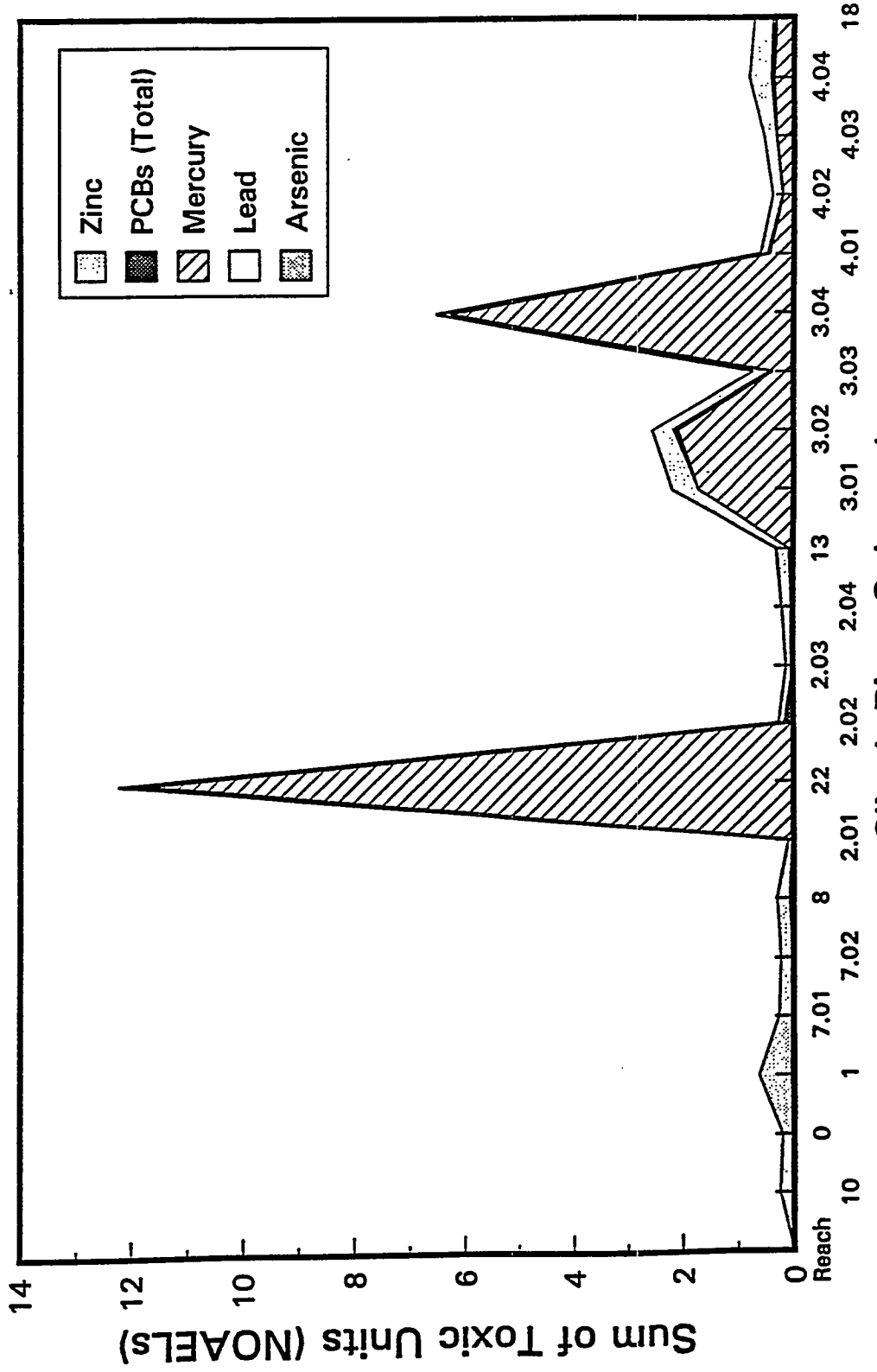

Fig. 6.31. Sum of NOAEL-based toxic units for rough-winged swallows.

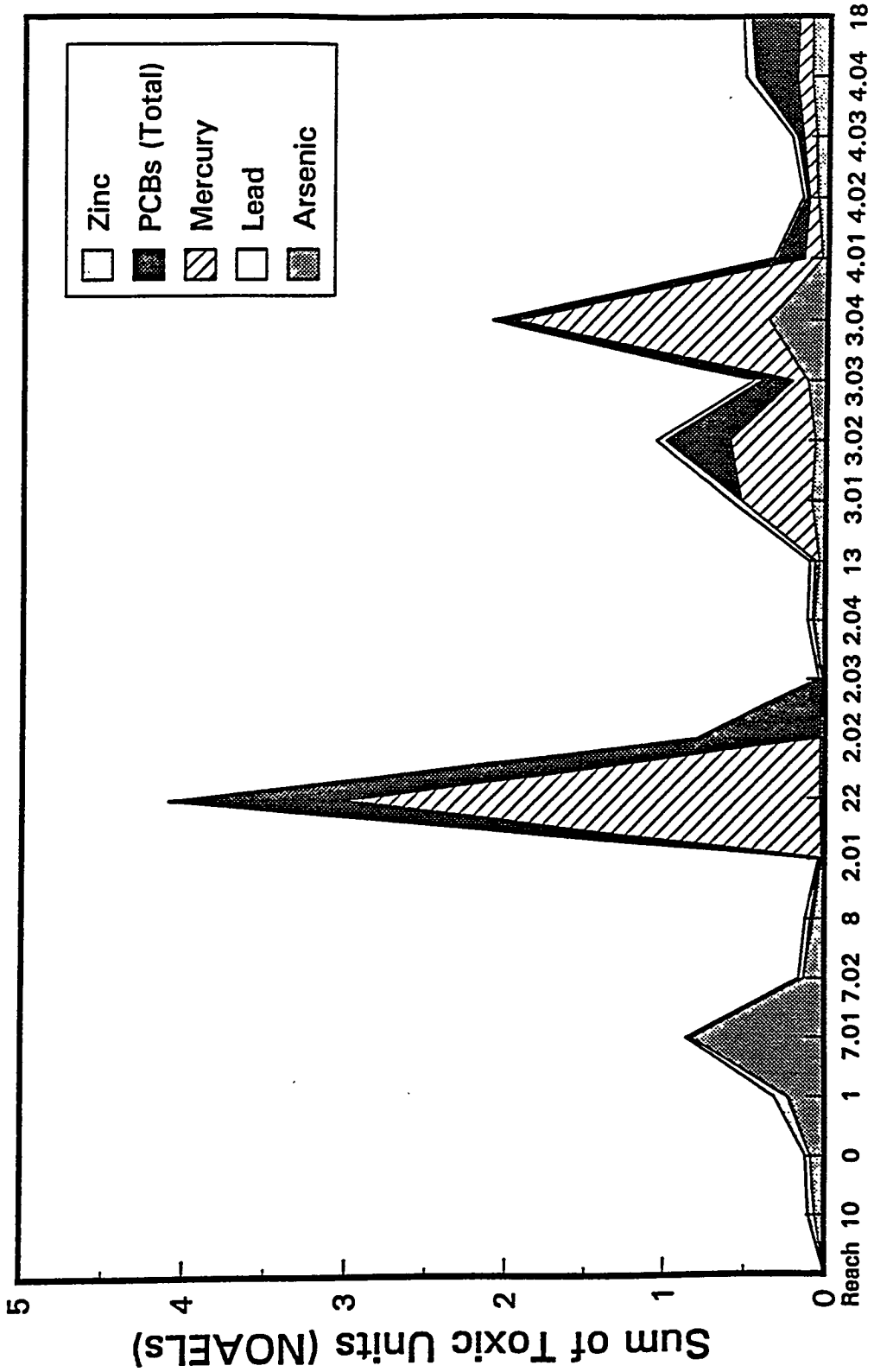

Fig. 6.32. Sum of NOAEL-based toxic units for little brown bats and gray bats.

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 95% UCL 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 endpoint 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 endpoint-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.

Table 6.23. Results of Monte Carlo simulation of insectivore contaminant exposure estimates in the Clinch River

			Simulation Results (mg/kg/d)		
Analyte	**Subreach**	**Endpoint**	**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

Hg LOAEL for rough-winged swallows = 0.25 mg/kg/d
Hg LOAEL for little brown/gray bats = 0.57 mg/kg/d

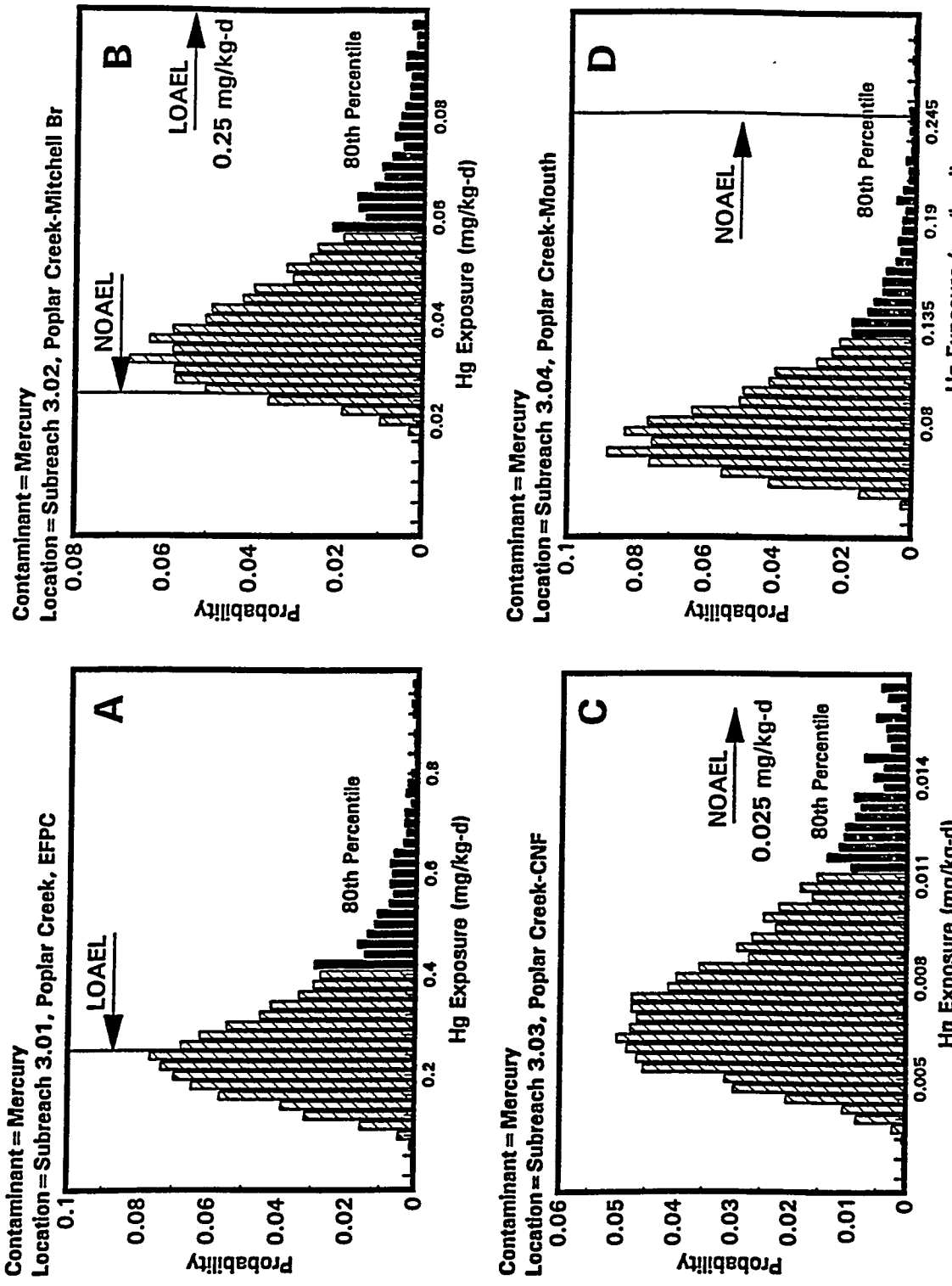

Fig. 6.33. Distribution of estimated mercury exposures for rough-winged swallows.

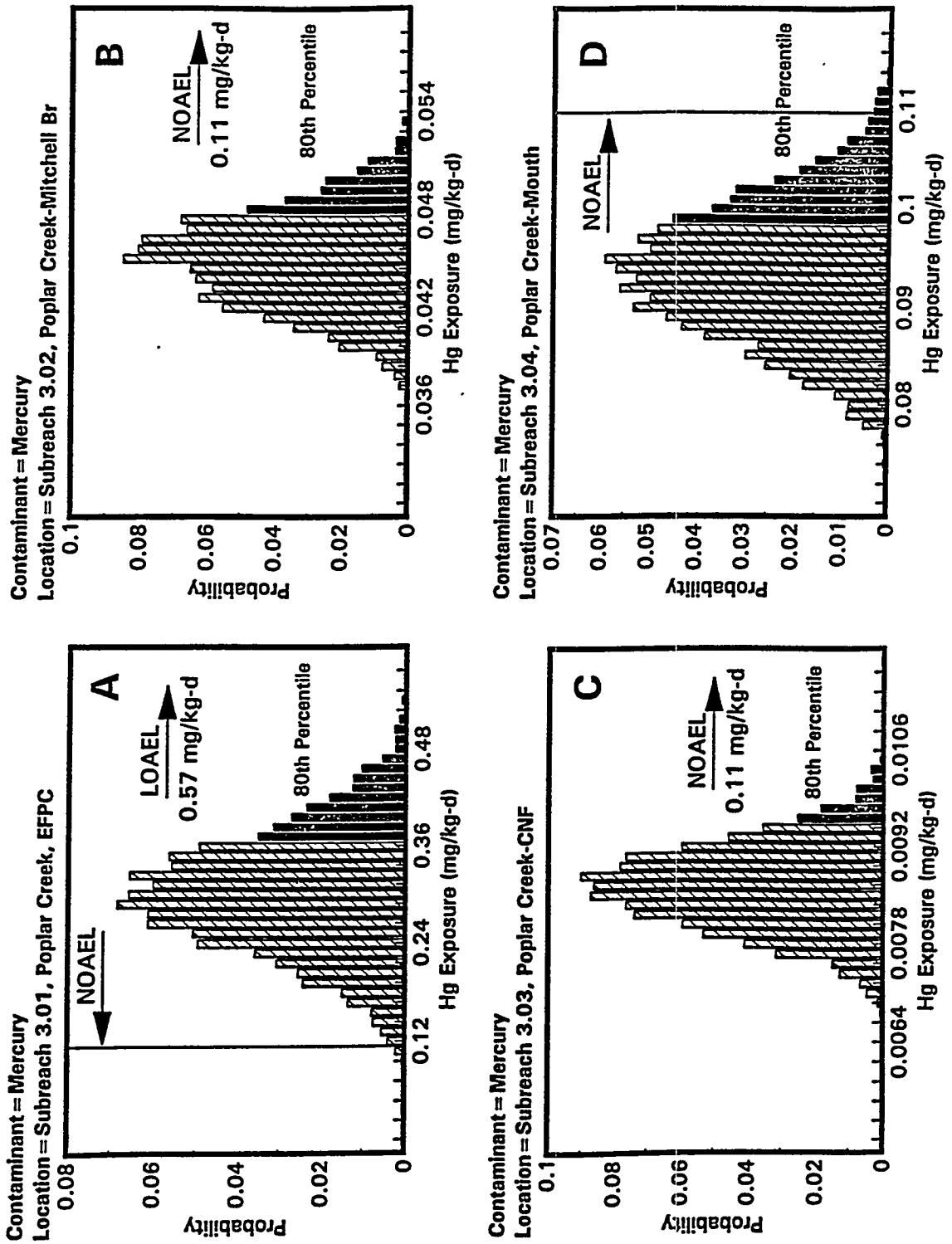

Fig. 6.34. Distribution of estimated mercury exposures for little brown and gray bats.

Table 6.24. Comparison of exposure distributions to NOAELs and LOAELs

Comparison	Meaning	Risk-based Interpretation
NOAEL>80th percentile of exposure distribution	less than 20% of population is experiencing exposures> NOAEL	Individual- and population-level adverse effects are highly unlikely
NOAEL<80th percentile<LOAEL	More than 20% of population is experiencing exposures> NOAEL, but less than 20% is experiencing exposures> 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	More than 20% of population is experiencing exposures > LOAEL	Individual- and population-level adverse effects are likely.

The interpretations described above apply to exposure estimates for populations within a given subreach. In order to determine population-level effects for each endpoint 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 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 endpoint is considered in this assessment, population-level effects can be evaluated.

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 subreach 22 (White Oak Creek Embayment) exceeded the estimated LOAEL by approximately 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 > NOAEL	Percentile > LOAEL
Rough-winged Swallows			
3.01	Hg	>99%	~60%
3.02	Hg	~90%	<1%
3.03	Hg	<1%	<1%
3.04	Hg	<1%	<1%
Little Brown/Gray Bats			
3.01	Hg	>99%	<1%
3.02	Hg	<1%	<1%
3.03	Hg	<1%	<1%
3.04	Hg	<5%	<1%

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, greater than 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 subreach 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 subreach 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 subreaches 3.01 in Poplar Creek (80th percentile=0.418 mg/kg/d; Fig. 6.33a), and 22 in White Oak Creek Embayment (mean=0.293 mg/kg/d). However, the average mercury exposure experienced by swallows feeding within subreach 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 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 subreaches 3.01 in Poplar Creek (80th percentile=0.368 mg/kg/d: Fig. 6.34a) 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 subreach 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 subreach 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 subreach 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 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 subreaches 3.01 and 22. Individuals may be at risk only if foraging takes place exclusively within those subreaches. This is highly unlikely since the foraging range of bats is approximately 8.5 km from the roost (LaVal et al. 1977).

6.5.4 Uncertainties Concerning Risks to Insectivorous Wildlife

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.

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.

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.

Single 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.

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.

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.

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.

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 endpoint are nationwide values which may overestimate or underestimate the body weight of species found at the site.

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.

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:

$$E_j = \left(\frac{IR_v \times C_{v\text{-}j}}{BW}\right) + \left(\frac{IR_s \times C_{s\text{-}j}}{BW}\right)$$

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 endpoint. Species-specific parameters necessary to estimate exposure using the above equation are listed in Table F6.1.

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 95% UCL or maximum concentration (whichever was lower) from all cores within each subreach (1, 7, 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 95% UCL 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):

$$\text{Log } B_v = 1.588 - 0.578 \log K_{ow}$$

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 (SAIC 1995).

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 endpoints 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 $\geq$ 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 95% UCLs 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

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, PCB-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

Endpoint	Distribution	Mean (kg)	Standard Deviation	Range (kg)	Source
Eastern Cottontail	triangular	1.244		0.842-1.533 based on ♀	Chapman et al. 1980

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 endpoint 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 endpoints (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 endpoint, 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 greater than 1. Specific details on development of the NOAELs and LOAELs for all wildlife endpoints 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, PCB-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 endpoint 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 greater than 1 suggest that the chemical may be hazardous to the plants that may grow on the spoil. HQs less than 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.

Table 6.28. COPECs which may pose a risk to plants within each subreach

Subreach	COPEC	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

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 endpoint 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 greater than one 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.

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 ΣTUs) for the most important contaminants. Arsenic, cadmium, and mercury are the primary contaminants contributing to risk. However, barium, manganese, PCB-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 greater than 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 greater than 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 subreaches 1, 7, 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 (subreaches 1, 7, 3.02, 3.03, 3.04, 4.03, and 4.04) in excess of the estimated LOAELs (Figures 6.36–6.44). Conversely, less than 1% of the population of cottontails feeding at the spoil within reach 7 may experience thallium exposures in excess of the NOAEL. Additionally, less than 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 subreaches 7 and 3.04, respectively, are highly unlikely. The exposures for all other contaminants within subreaches 1, 7, 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.

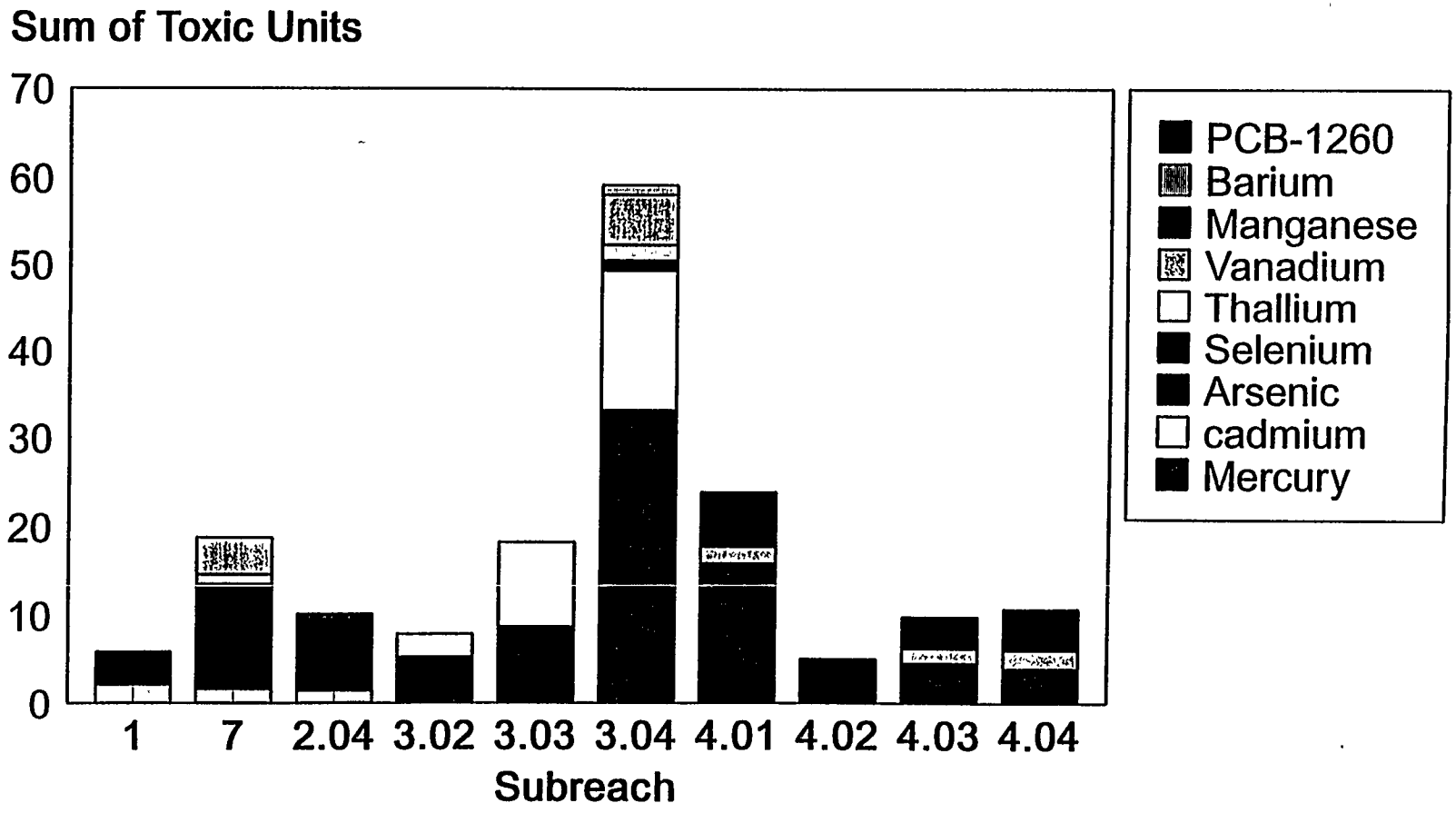

Fig. 6.35. Sum of NOAEL-based toxic units for evaluation of risks to Eastern Cottontails foraging on dredge spoils within each subreach.

Table 6.29. Summary of subreaches where HQs >1 were observed for each contaminant

SUBREACH	COPEC	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
	PCB-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

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

Table 6.30 (continued)

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	
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
	PCB-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

Table 6.31. Comparison of exposure distribution to NOAELs and LOAELs

Comparison	Meaning	Risk-based Interpretation
NOAEL>80th percentile of exposure distribution	less than 20% of population is experiencing exposures> NOAEL	Individual- and population-level adverse effects are highly unlikely
NOAEL<80th percentile<LOAEL	More than 20% of population is experiencing exposures> NOAEL, but less than 20% is experiencing exposures> 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	More than 20% of population is experiencing exposures< LOAEL	Individual- and population-level adverse effects are likely.

Table 6.32. Results from the comparison of exposure distributions with NOAELs and LOAELS

Subreach	Analyte	Percentile> NOAEL	Percentile > 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%

Table 6.32 (continued)

Subreach	Analyte	Percentile> NOAEL	Percentile > LOAEL
	Cd	>99%	>99%
	Hg	>99%	>99%
	Se	>30%	<1%
	Tl	>50%	<1%
	V	>99%	<1%
4.01	As	>85%	<1%
	Cd	>65%	>25%
	Hg	>99%	>99%
	PCB-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%

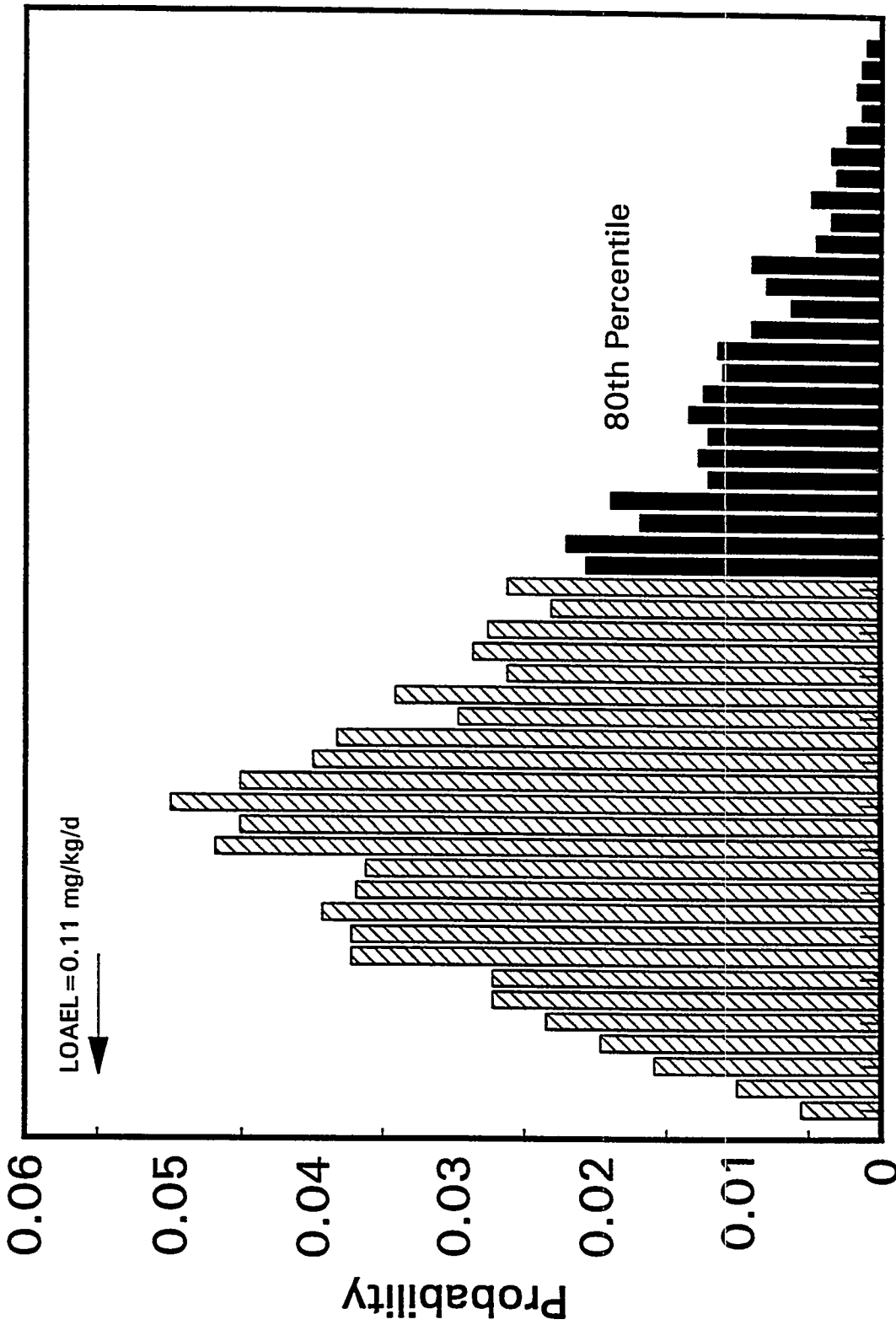

Fig. 6.36. Mercury exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.03 using Monte Carlo simulation.

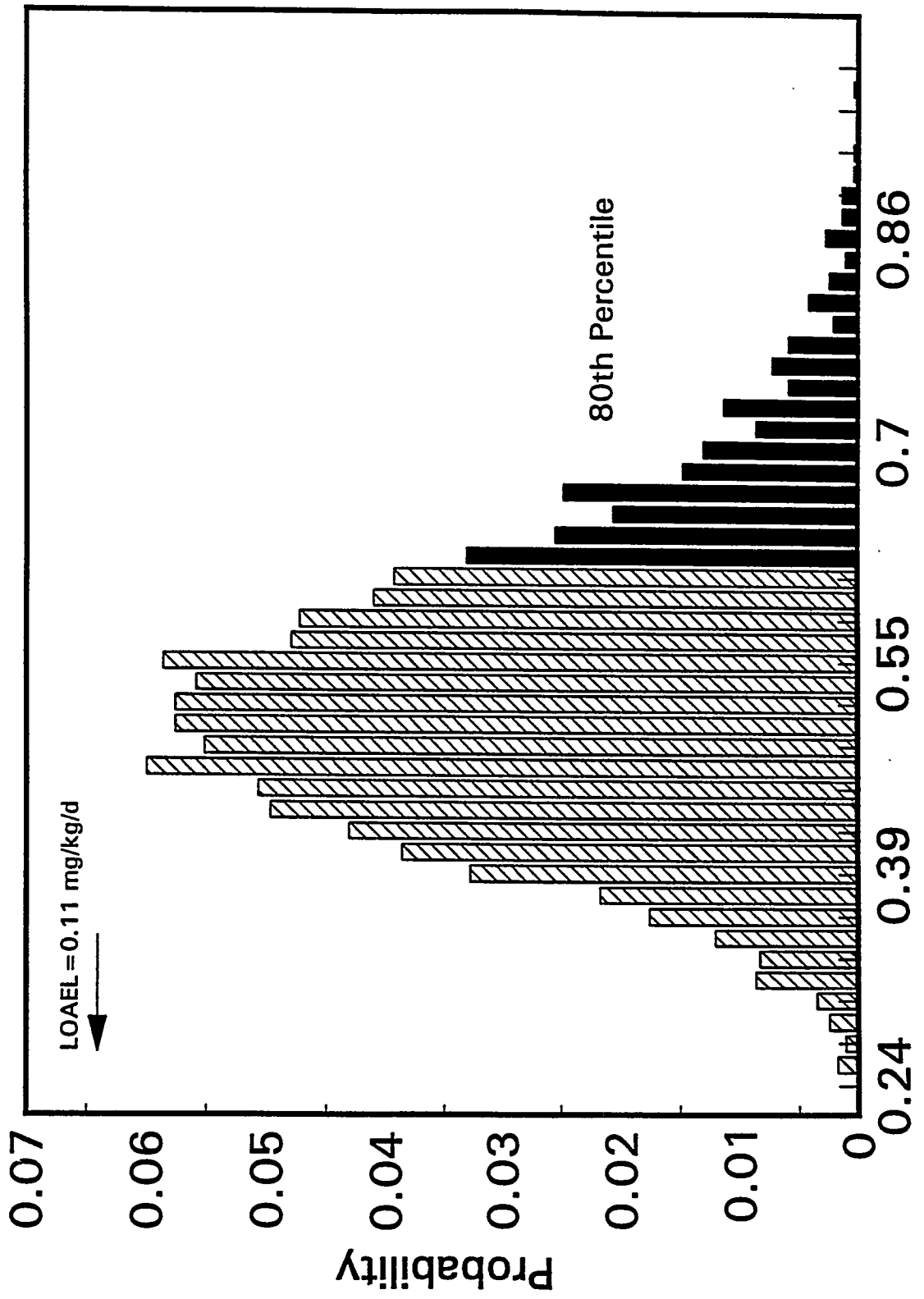

Fig. 6.37. Mercury exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.04 using Monte Carlo simulation.

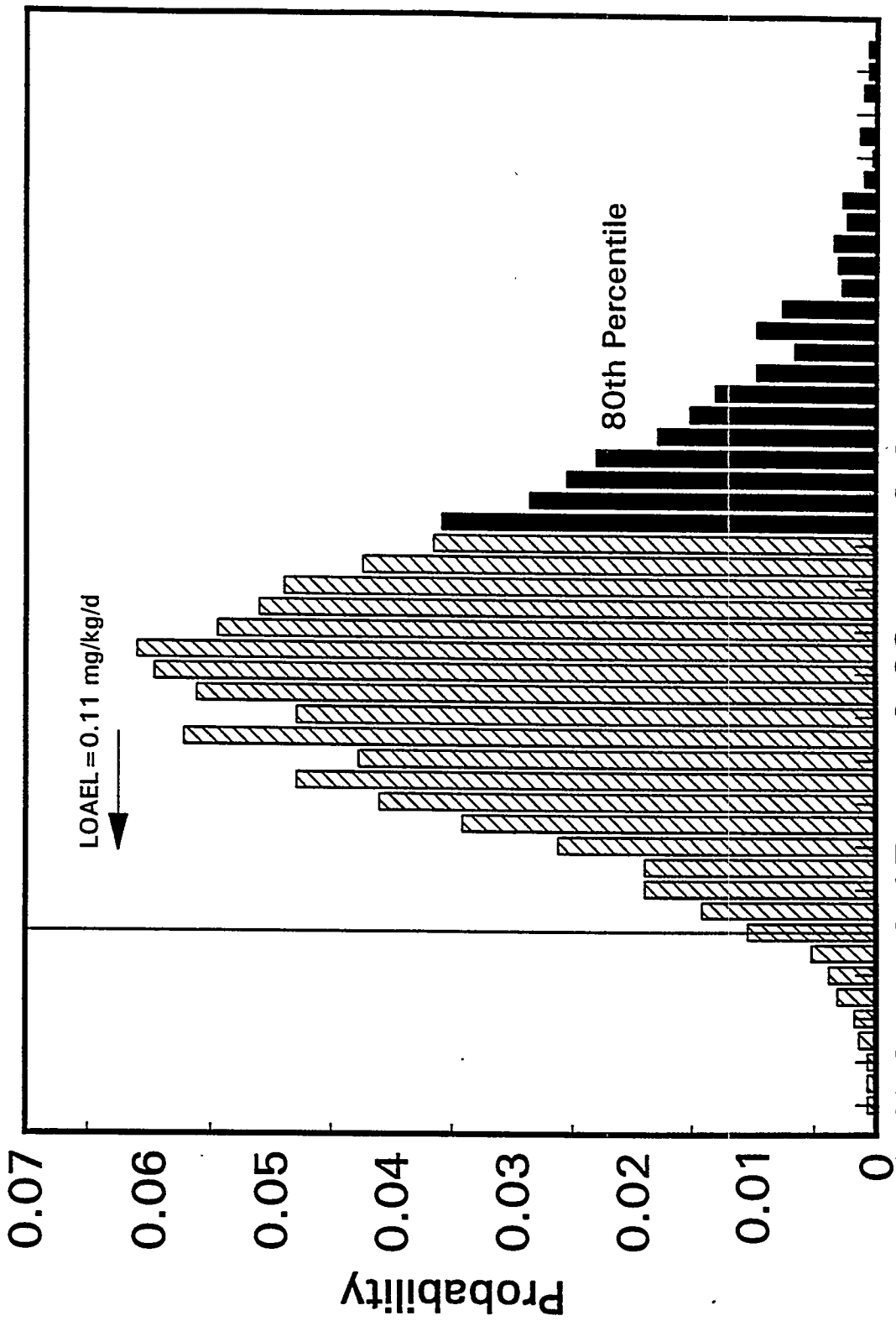

Fig. 6.38. Mercury exposure distribution for eastern cottontails foraging on a future spoil within subreach 4.01 using Monte Carlo simulation.

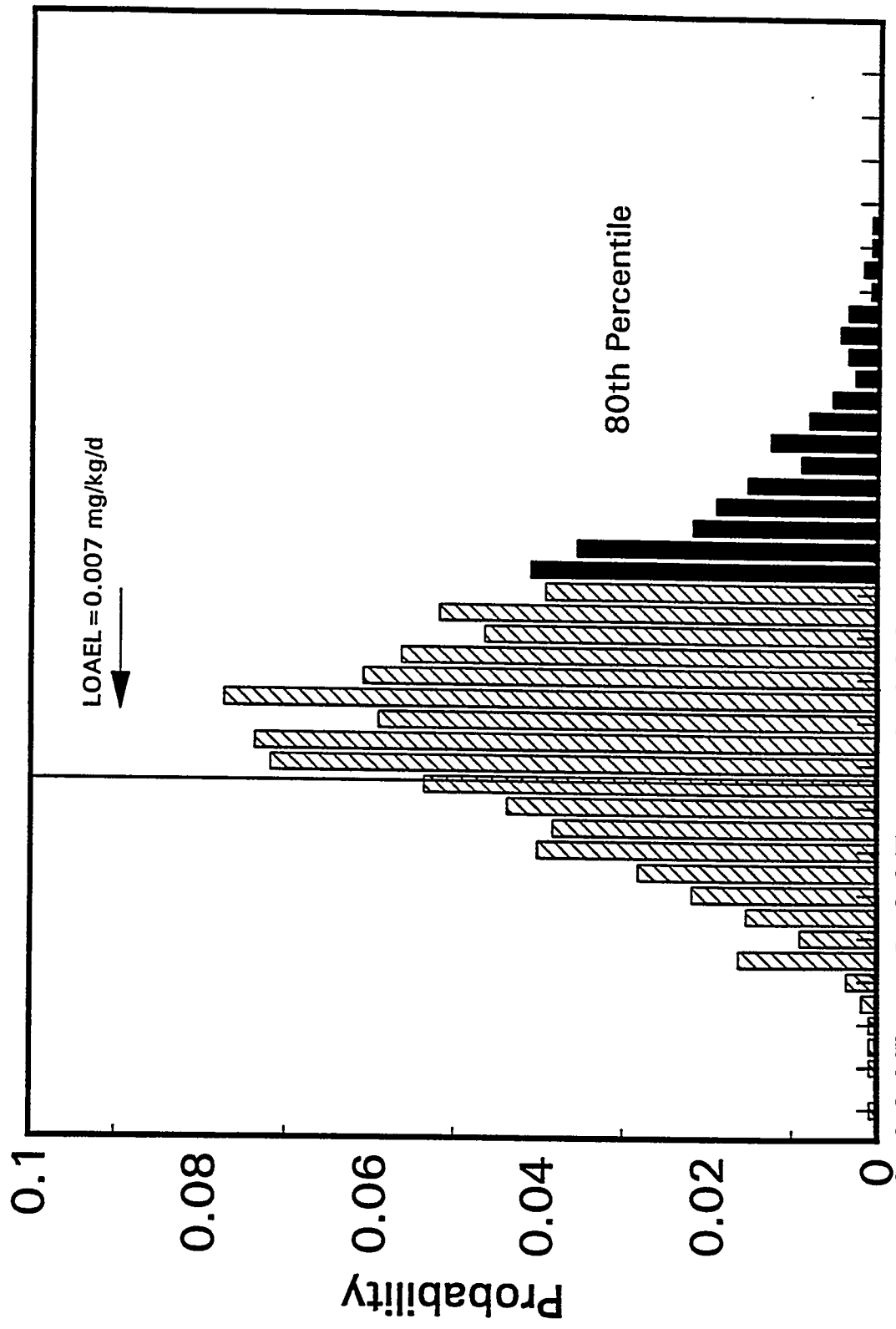

Fig. 6.39. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within reach 1 using Monte Carlo simulation.

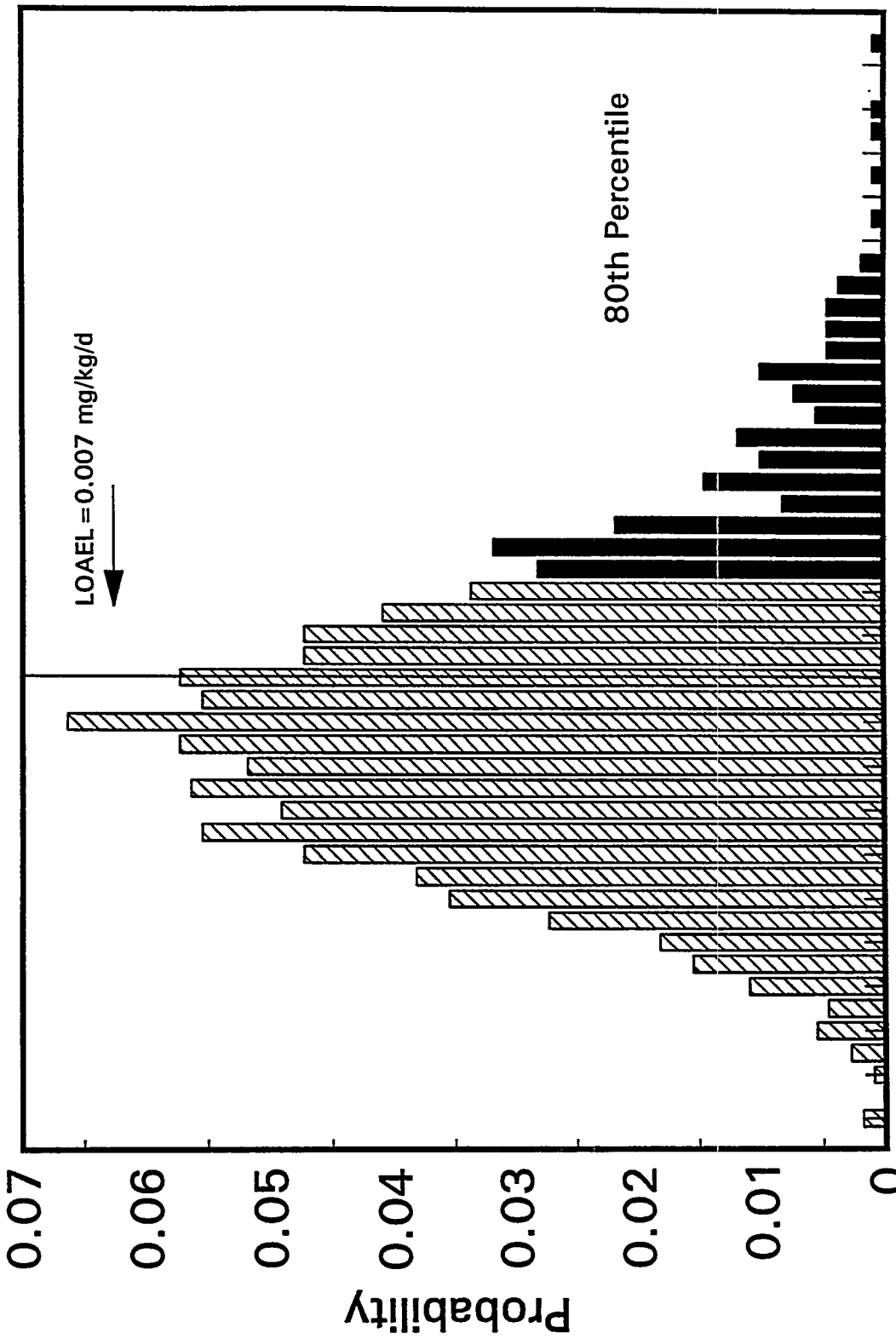

Fig. 6.40. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within reach 7 using Monte Carlo simulation.

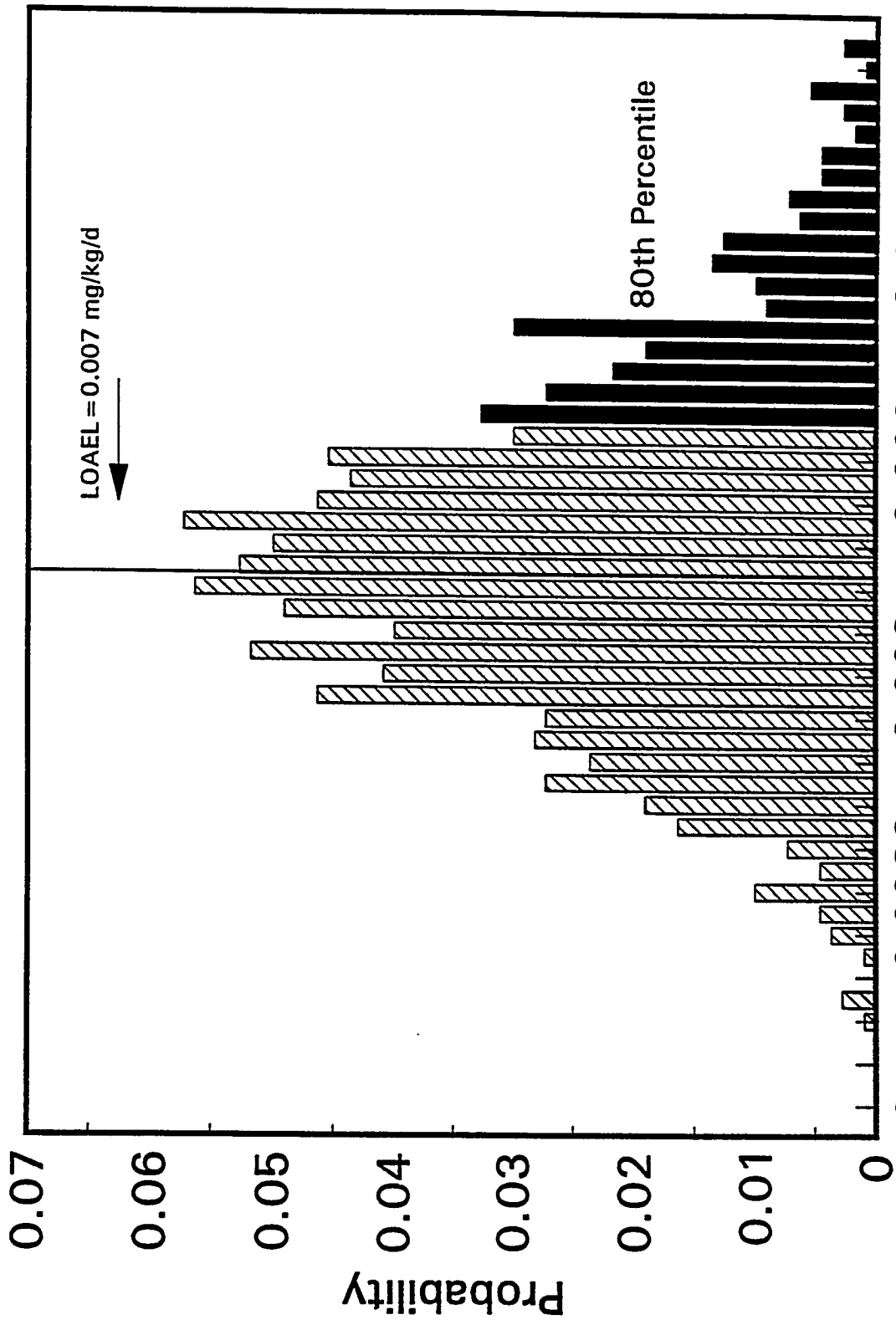

Fig. 6.41. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.02 using Monte Carlo simulation.

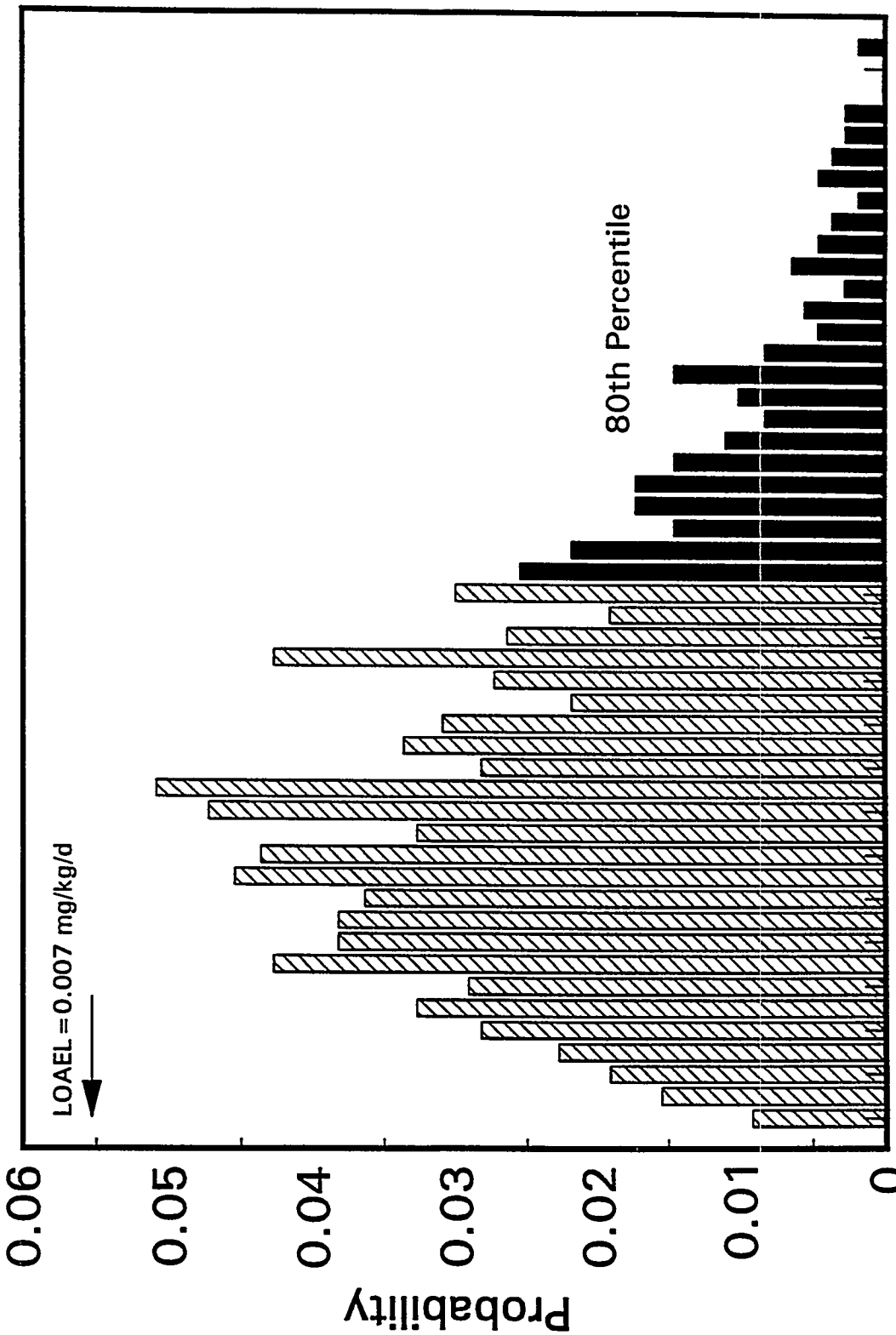

Fig. 6.42. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 3.03 using Monte Carlo simulation.

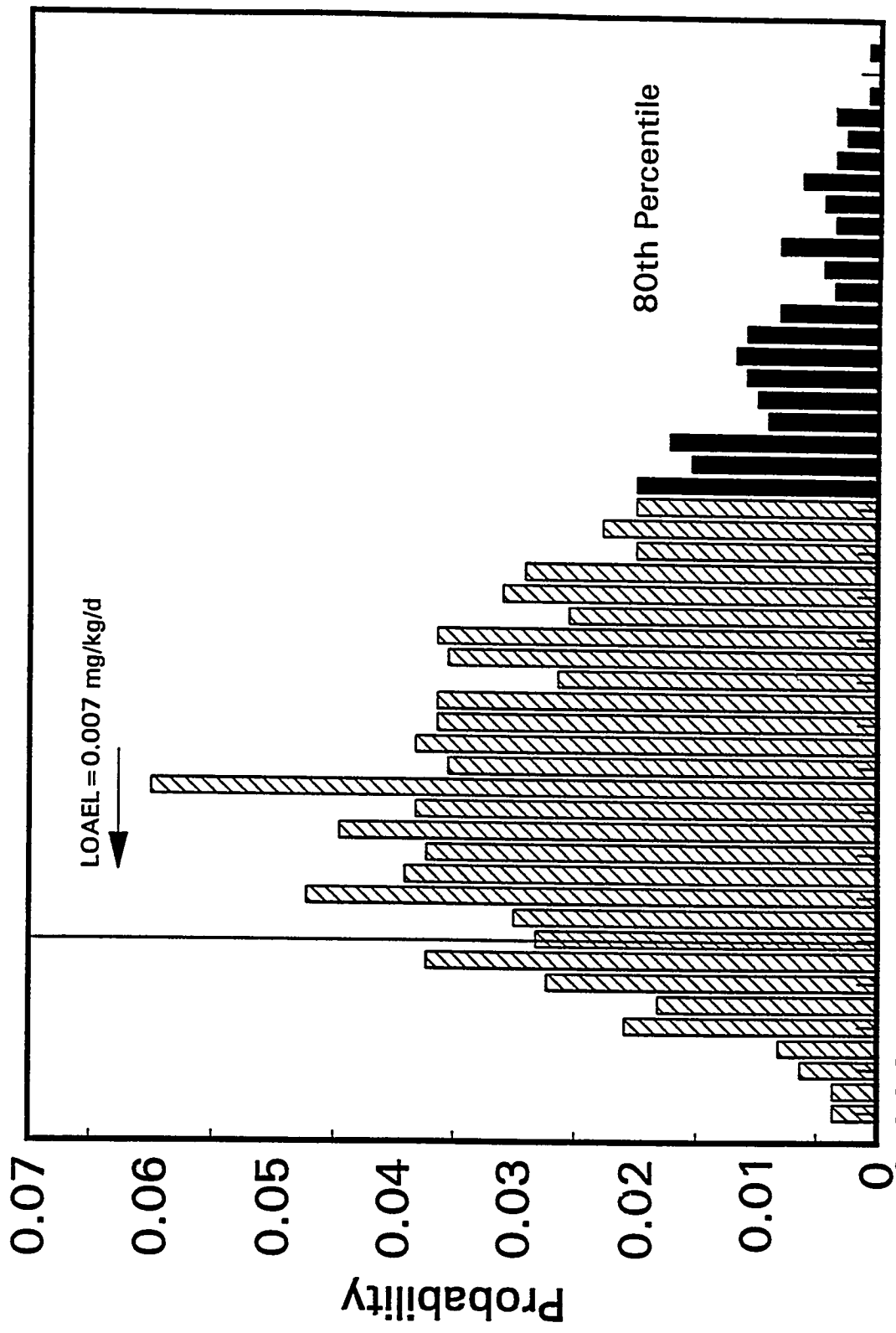

Fig. 6.43. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 4.03 using Monte Carlo simulation.

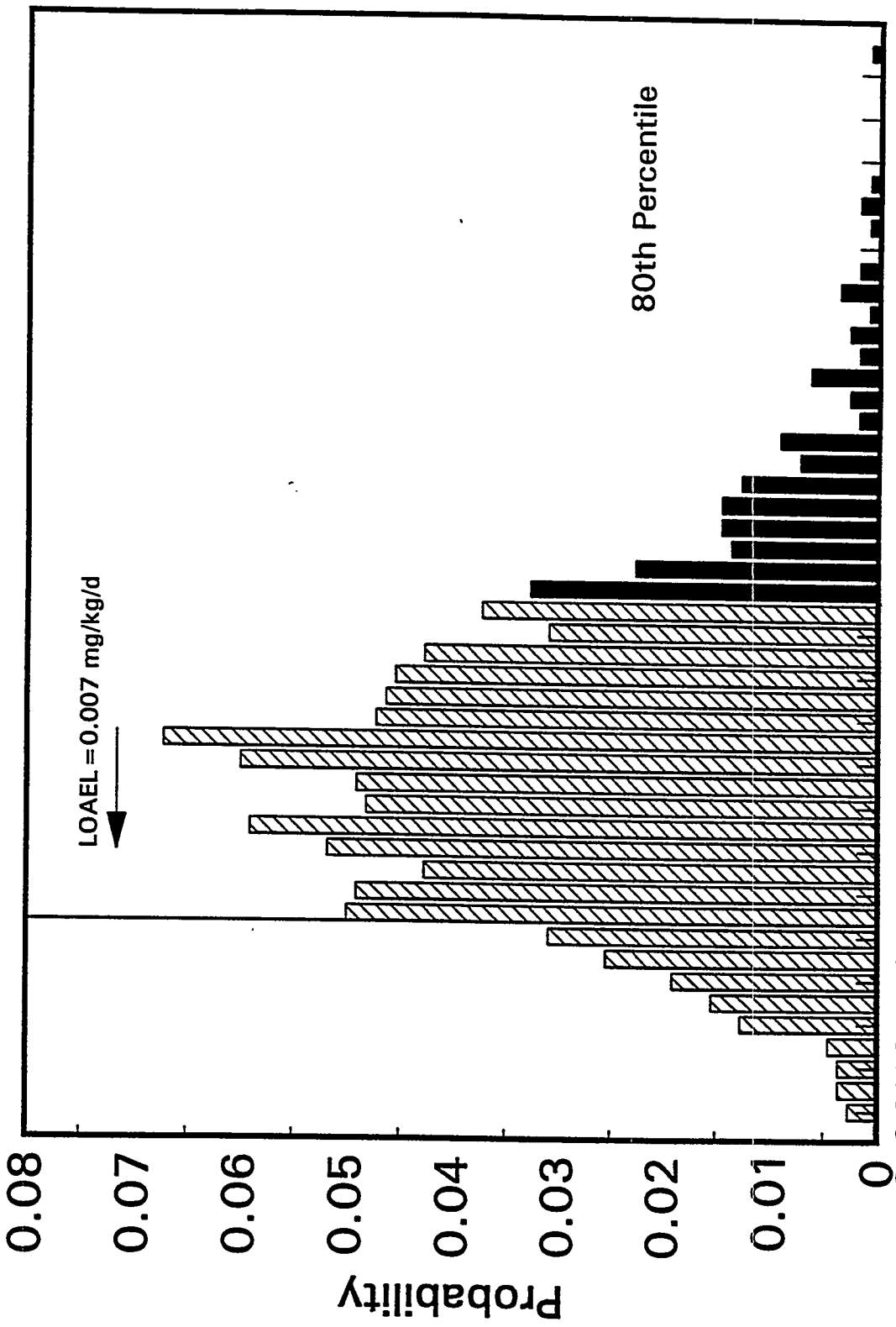

Fig. 6.44. Cadmium exposure distribution for eastern cottontails foraging on a future spoil within subreach 4.04 using Monte Carlo simulation.

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 subreaches 1, 7, 2.04, 4.01, 4.02, 4.03 and 4.04.

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 subreach 1, 7, 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 subreaches 7, 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 subreaches 7 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 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 subreaches (1, 7, 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 subreaches 1, 7, 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 subreaches 1 (As), 7 (As, Se, Tl, and V), 2.04 (as, Mn, and Se), 3.02 (Hg), 3.04 (Ba, Se, Tl, and V), 4.01 (As and PCB-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.

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.

Bioavailability of contaminants. It was assumed that 100% of the contaminant concentration reported in sediments and modeled vegetation was bioavailable. The double acid extraction method used to determine sediments concentrations reflect the total potential pool of contaminants. 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. Therefore, exposure estimates based upon the contaminant concentrations in media are highly conservative and are likely to overestimate the actual contaminant exposure experienced.

Uncontaminated surface water consumption for herbivorous wildlife. This assessment assumed that herbivorous wildlife will not be consuming surface water from the Clinch River or Poplar Creek. Since this is a future scenario, surface water concentrations found in the Clinch River can not be accurately predicted. Moreover, rabbits are likely to consume water from smaller bodies of water such as tributaries, ponds, or small pools. The high banks of the CR/PC also limit access to the water source in many areas. Additionally, the primary purpose of this scenario is to evaluate risks from dredged spoils.

Extrapolation from published toxicity data. There are no published NOAELs for eastern cottontails. To estimate toxicity of contaminants at the site, it was necessary to extrapolate from NOAELs observed for test species (i.e., rats and mice). 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, eastern cottontails may be more or less sensitive than rats or mice.

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.

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, 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.

Single contaminant tests vs exposure to multiple contaminants in the field. While plants and eastern cottontails are exposed to multiple contaminants concurrently, published toxicological values only consider effects experienced by exposures to single contaminants. Because some contaminants 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.

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 modeled vegetation or sediment from the Clinch River or Poplar Creek, potential toxicity at the sites may be overestimated.

Food source availability. This assessment assumed that the spoil deposited on land would revegetate at a level sufficient to maintain foraging area for a population of herbivores (i.e., eastern cottontails).

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 the body weight of the eastern cottontail are nationwide values which may overestimate or underestimate the body weight of species found at the site.

6.7 RISKS FROM EXPOSURE TO RADIONUCLIDES

6.7.1 Exposure Assessment

6.7.1.1 Sources of exposure

Sources of radioactivity in the environment include naturally occurring radionuclides, fallout from atmospheric testing, runoff from watersheds that have received atmospheric deposition, and radioactive effluents from medical, industrial, and nuclear facilities discharged either routinely or accidentally. The discharge of radioactive effluents to the environment has the potential for contaminating terrestrial and aquatic environments, which results in long-term, low dose rate exposures to an ecosystem and its components. Sources of radionuclides in the CR/PC system are discussed in Sect. 3.2. Native biota receive external radiation exposure from radionuclides in air, water, soil, sediment, and other biota, such as vegetation. Organisms also receive internal radiation exposure from radionuclides ingested via food and water and from radionuclides absorbed through the skin and respiratory organs (Templeton et al. 1971, IAEA 1976, Blaylock and Trabalka 1978, Woodhead 1984). Evaluation of the resulting radiation doses received by biota requires quantitative information on the radionuclides to which they are exposed. In all cases the radiation source must be known in terms of the quantity of each specific radionuclide and the corresponding energy released per disintegration. Special factors to correct for size, biological concentration, elimination, etc., are applied when appropriate.

Radiation doses were calculated for representative species of biota in the CR/PC. Estimates of radiation dose to aquatic biota were made using the methodology reported by Blaylock et al. (1993), which is based upon the "Point Source Dose Distribution" methodology recommended by the 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 endpoint 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 (percent)	Dose Rate (mrad/year)
1	2	60Co (92%), 137Cs/137mBa (8%)	200
2	14	137Cs/137mBa (93%), 90Sr/90Y (5%)	490
3	13	137Cs/137mBa (89%), 235U (5%)	180
4	13	137Cs/137mBa (91%), 60Co (5%)	280
7	3	90Sr/90Y (79%), 137Cs/137mBa (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 less than 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 (Percent)	Dose Rate (mrad/year)
1	6	60Co (91%), 90Sr/90Y (7%)	32
2	16	90Sr/90Y (52%), 137Cs/137mBa (36%)	33
3	14	137Cs/137mBa (84%), 90Sr/90Y (10%)	56
4	15	137Cs/137mBa (76%), 60Co (13%)	22
7	3	90Sr/90Y (61%), 137Cs/137mBa (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 (Percent)	Dose Rate (mrad/year)
1	6	137Cs/137mBa (~100%)	23
2	15	137Cs/137mBa (~100%)	5400
3	14	90Sr/90Y (79%), 137Cs/137mBa (21%)	350
4	15	90Sr/90Y (77%), 137Cs/137mBa (23%)	320
7	3	137Cs/137mBa (60%), 60Co (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 (USEPA 1993). 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 (Percent)	Dose Rate (mrad/year)
1	2	137Cs/137mBa (96%), 60Co (4%)	11
2	5	137Cs/137mBa (90%), 90Sr/90Y (8%)	1500
3	7	137Cs/137mBa (78%), 90Sr/90Y (17%)	530
4	6	137Cs/137mBa (98%), 90Sr/90Y (2%)	1400
7	2	90Sr/90Y (80%), 137Cs/137mBa (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 332), 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 ^{37m} 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.

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.

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.

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 (USEPA 1993) 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.

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.

Although risks to fish in Poplar Creek is not high, 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 fish in Poplar Creek is 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 ENDPOINTS

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. A toxicity identification and evaluation (TIE) should be performed 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.

Table 6.37. RGOs for reducing risks to fish from contaminants in water (mg/L)

Chemical	Acute RGO	Chronic RGO
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 NAWQC
[b] Chronic NAWQC
[c] CV for fathead minnow
[d] CV for *C. dubia*

6.9.2 Remedial Goal Options for the Benthic Invertebrate Community

Benthic invertebrates are exposed to contaminants primarily by 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.

Whole sediment and pore water concentrations of chemicals were observed in Poplar Creek that exceeded ecotoxicological benchmarks. Actions should be taken to ensure that chemicals in sediment do not occur above concentrations at which effects are expected. Whole sediment concentrations should not exceed the lesser of the ER-M and the 50th percentile of the effects distributions presented in Appendix F3. Pore water concentrations should not exceed the chronic NAWQC, or lowest chronic value for Daphnids if NAWQC are not established. These concentrations are presented in Table 6.38 for those chemicals that were characterized as presenting a significant risk in Reach 3. RGOs are not presented

for chemicals that present a greater risk above the OU (Reach 13) than in Reach 3 (e.g., 2-methylnaphthalene, acenaphthene, total PAHs). Although the 50th percentile of the community level effects distribution was the lowest RGOs available for arsenic, it was not used because it was based on only two data points. Rather, the next lowest available RGO, the 50th percentile of the lethal effects distribution, was used for arsenic.

Table 6.38. RGOs for reducing risks to benthic invertebrates from contaminants in sediment (mg/kg) and pore water (mg/L)

Chemical	Sediment RGO	Pore Water RGO
Arsenic	6.2[a]	N/A
Chromium	90[a]	N/A
Mercury	0.150[a]	N/A
Nickel	22[a]	N/A
Silver	0.8[a]	N/A
PCBs (Total)	0.180[b]	N/A

N/A Not applicable, because this chemical is not a significant risk in this medium or the observed concentrations are below the RGOs.

[a] 50th Percentile from community level or lethal effects distribution, whichever is lower.

[b] ER-M

7. PURPOSE AND ORGANIZATION OF THE FEASIBILITY STUDY

This feasibility study (FS) identifies Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) remedial actions that eliminate, reduce, or control risks to human health and the environment. The FS process, as defined in the National Contingency Plan (NCP) (42 *United States Code* 9601 et seq.), 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. Following is a review of the reaches and subreaches in the Clinch River/Poplar Creek OU.

- Reach 1: Lower Melton Hill [Clinch River mile (CRM) 23 - 51.3]
- Reach 2: Clinch River from Melton Hill Dam to Poplar Creek (CRM 12.1 - 22)

- Reach 3: Poplar Creek from East Fork Poplar Creek (EFPC) to Clinch River [Poplar Creek mile (PCM) 0 - 5.5]

  — Subreach 3.1: Poplar Creek below EFPC (PCM 4.8 - 5.5)

  — Subreach 3.2: Poplar Creek below Mitchell Branch (PCM 3.1 - 4.6)

  — Subreach 3.3: Poplar Creek below the Y-12 Plant Central Neutralization Facility (PCM 1.3 - 2.9)

  — Subreach 3.4: Poplar Creek below the ash disposal area (PCM 0 - 1)

- Reach 4: Clinch River from Poplar Creek to Lower Watts Bar Reservoir (CRM - 0 - 11.8)

- Reach 7: McCoy Branch Embayment [McCoy Branch Embayment mile (MBM) 0.2 - 0.4]

  — Subreach 7.1: Upper McCoy Branch (MBM 0.4)
  — Subreach 7.2: Lower McCoy Branch (MBM 0.2)

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, contaminants of concern (COCs), and remedial goal options (RGOs). General response actions, potential remedial technologies, and process options are developed and screened for technical implementability. Those general response actions, technologies, and process options that do not satisfy implementability are described briefly and not discussed any further. Those that meet the criteria are carried forward into an evaluation of effectiveness, implementability, and cost.

Chapter 9 combines the retained technologies and process options to develop a range of remedial alternatives consistent with RAOs. Each assembled alternative is evaluated for its implementability, effectiveness, and cost. Alternatives that best satisfy these criteria are carried forward for detailed description and analysis. Alternatives that do not satisfy these criteria will be screened out and justification offered for not carrying them forward into detailed analysis.

Chapter 10 provides a detailed analysis of each alternative carried forward from Chapter 9. This will 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 are 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 Clinch River 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 1988) 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 Clinch River OU. These assessments are based on the operational history of ORR, results of previous investigations, and results of the RI. Risk quantification is based on the defensible laboratory data generated during the RI.

The primary driver for conducting an FS for the Clinch River OU in terms of human health depends on whether the sediment largely contained beneath the river and additional layers of deposited sediment is considered a medium of potential human exposure. As noted in the human health baseline risk assessment, two types of sediment access where evaluated for several areas (subreaches) along the OU river system (1) dredging of 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 exposures pathway because dredging for agricultural land application is not generally practiced. Dredging in the TVA waterways system is tightly controlled by TVA and the U.S. Army Corps of Engineers (COE). Unauthorized dredging is prohibited. While the baseline human health risk assessment evaluated exposures, assuming a residential land use (i.e., contact is considered to occur throughout the entire period of low pool exposing near-shore sediments), all the subreaches evaluated in the RI are within the ORR controlled by DOE. In addition, no private properties have been identified with near-shore sediment contamination in excess of risk thresholds; thus, no current unacceptable residential exposures occur.

Ingestion of fish caught within the Clinch River 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 conservatively assumed that the residential exposure scenario evaluated in the baseline human health risk assessment is viable. Based on the results of that assessment, Subreach 3.02 exceeds EPA criteria for identifying chemicals of concern (COCs). The total excess lifetime cancer risk (ELCR) calculated under this scenario just exceeds the criterion of 1×10^{-4} (cumulative ELCR is 2×10^{-4}). The applicable COCs include: Cr, As, Be, ^{238}U, ^{234}U, ^{235}U, ^{137}Cs, and ^{60}Co. While Subreaches 3.04 and 4.02 are associated with a cumulative ELCR of 1×10^{-4}, this value does not exceed the EPA criterion. The fish ingestion scenario is also considered viable; although fish advisories are currently in place. The applicable COCs include: As, Hg, Be, Se, PCBs, ^{137}Cs, ^{90}Sr, aldrin, chlordane, and 4,4'-DDT. Therefore, if the near-shore sediments are considered a likely exposure medium under a residential scenario and the fish consumption scenario is considered without the current fish advisories, the primary purpose of remedial actions would be to limit exposure to the sediments and fish.

Results of the baseline ecological risk assessment suggest that significant ecological risks occur in Reach 3, but not the Clinch River. This is based primarily on weight-of-evidence indicating a 20 percent reduction in fish species richness or abundance. However, it is recognized that community characteristics can be highly variable; therefore, the results may be uncertain.

The remedial action alternatives developed and analyzed in Chapters 9 and 10 are designed to meet the following RAOs.

- If near-shore sediments are a likely exposure medium under a residential scenario, prevent direct access to contaminated sediments to eliminate ingestion, inhalation,

external exposure, and dermal contact exposure routes. This would avoid the potential need for active remediation because sediment removal may cause resuspension and transport of contaminated sediment that could result in contaminant concentrations that exceed surface water quality criteria and destruction of benthic invertebrates.

- If deep sediments are a likely exposure medium under a 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. This would avoid the potential need for active remediation because sediment removal may cause resuspension and transport of contaminated sediment that could result in contaminant concentrations that exceed surface water quality criteria and destruction of benthic invertebrates.

- Prevent exposure to fish potentially contaminated with COCs.

- Continue monitoring ecological communities for potential impacts in conjunction with actions taken in upstream OUs to prevent ongoing releases to the Poplar Creek and Clinch River OUs.

To meet these RAOs, technologies and process options described in the following sections, as well as alternatives which follow in later chapters, will not attempt to actively remediate deep sediments, but will address controls that will enable these sediments to remain in place and undisturbed.

8.2 IDENTIFICATION, SCREENING, AND EVALUATION OF TECHNOLOGIES AND PROCESS OPTIONS

Technology groups and process options can be identified from several sources. The 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* were tools used to develop a list of technologies and process options that apply to the media of interest.

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 previously listed 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 1988), 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. Site conditions, against which each process option is assessed in this screening step and the following screening step, are shown in Table 8.1. 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 discussed in this document. Each technology that passes the first screening step will have one or more process options. Eliminated process options are indicated in Figure 8.1 with cross-hatched boxes. A brief description is given to indicate why the option is not considered technically applicable to the site.

Table 8.1. Site physical and contaminant characteristics for the Clinch River/Poplar Creek Operable Unit

Parameter	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)	790 / 795
(below Melton Hill Dam)	741 / 735
Flow rate	Controlled reservoir, 0–1,215 m^3/second 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
Parameter	**Contaminant characteristics**
Contaminants of concern	Cr, As, Mn, ^{137}Cs, ^{234}U, ^{238}U, ^{40}K, 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 Rads, half-life dependent
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
Natural degradation	Good for organics, poor for PCBs, and not applicable to metals
Radioactivity	Low at surface sediments, fish, and surface water; moderate in deep sediments

As = arsenic
CERCLA = Comprehensive Environmental Response, Compensation, and Liability Act
Cr = chromium
Cs = cesium
DDT = dichlorodiphenyltrichloroethane
ft = foot
K = potassium
m = meter
Mn = manganese
MSL = mean sea level
PCB = polychlorinated biphenyl
Rad = radioactive
Tc = technetium
U = uranium

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 options that appears to offer the best blend of effectiveness, implementability, and cost is carried forward for development into remedial alternatives in Chapter 9. These process options are indicated in Figure 8.1 with double-outlined boxes.

Process options or technology groups not carried forward to development of alternatives remain technically site applicable, but under current conditions do not 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. These process options are indicated in single-outlined boxes.

A comprehensive discussion of all process options remaining after the first screening step for technical applicability is provided in Appendix I. 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.

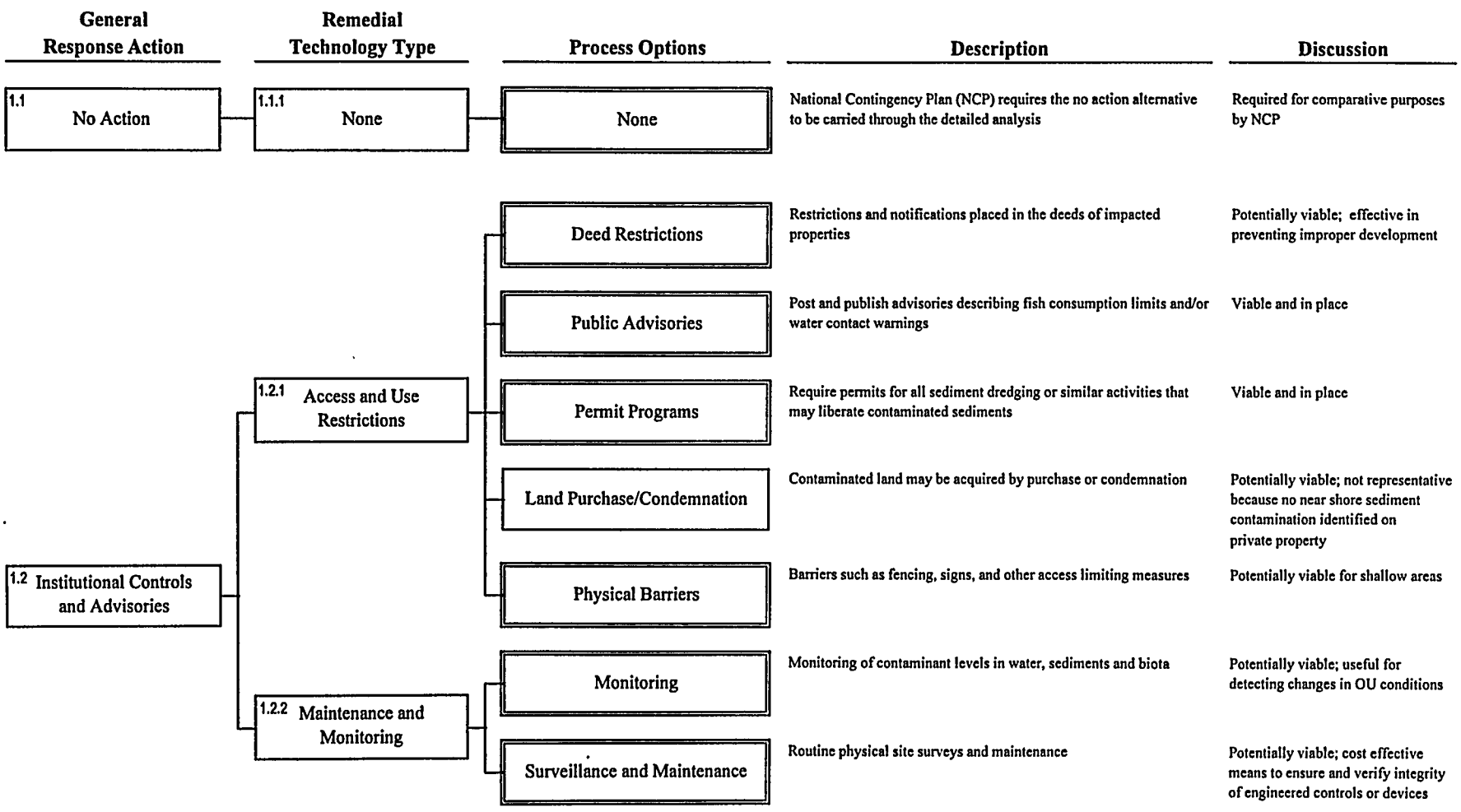

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Fig. 8.1

Page 1 of 6

Results of process option screening

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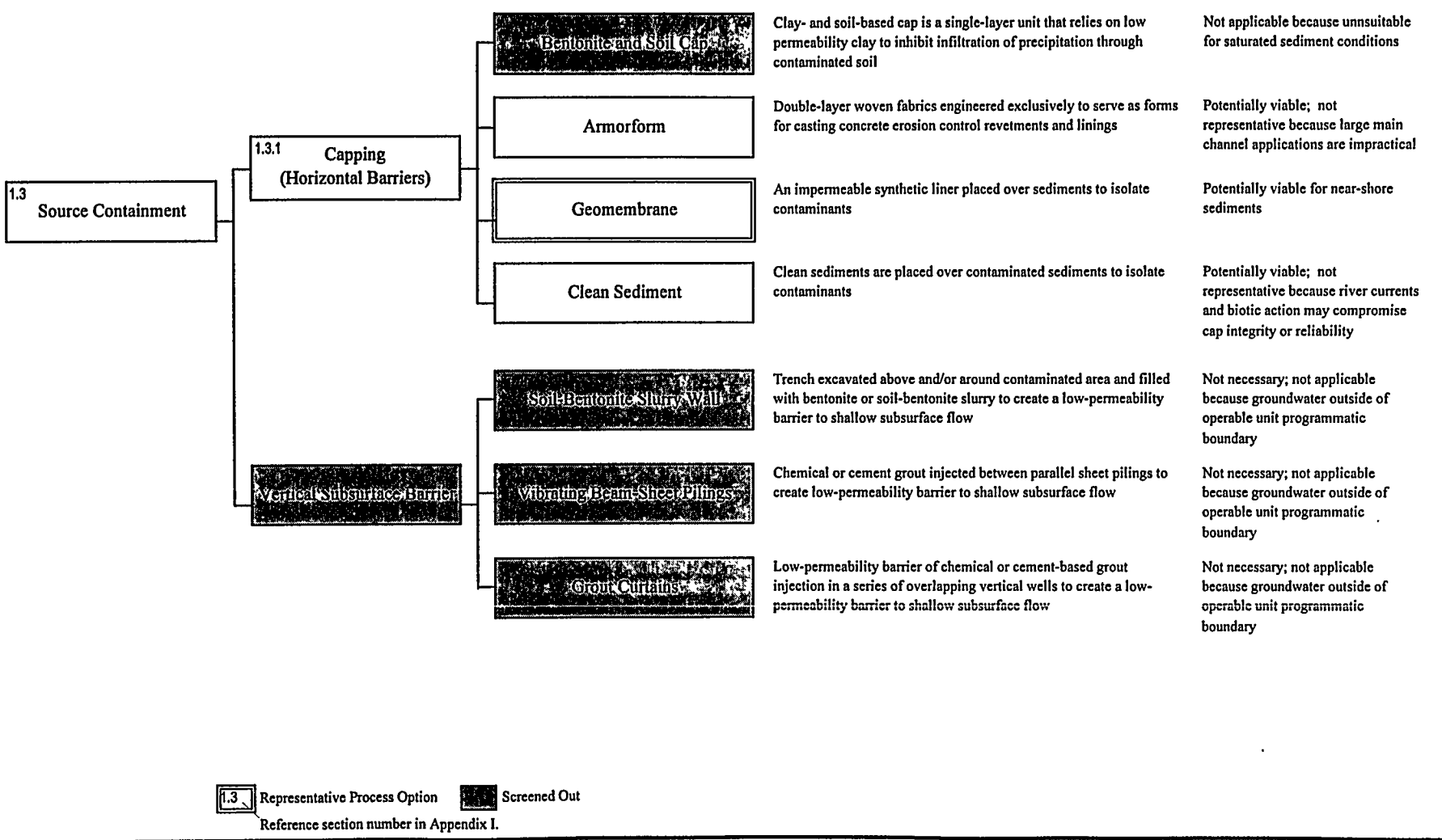

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Fig. 8.1

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Results of process option screening

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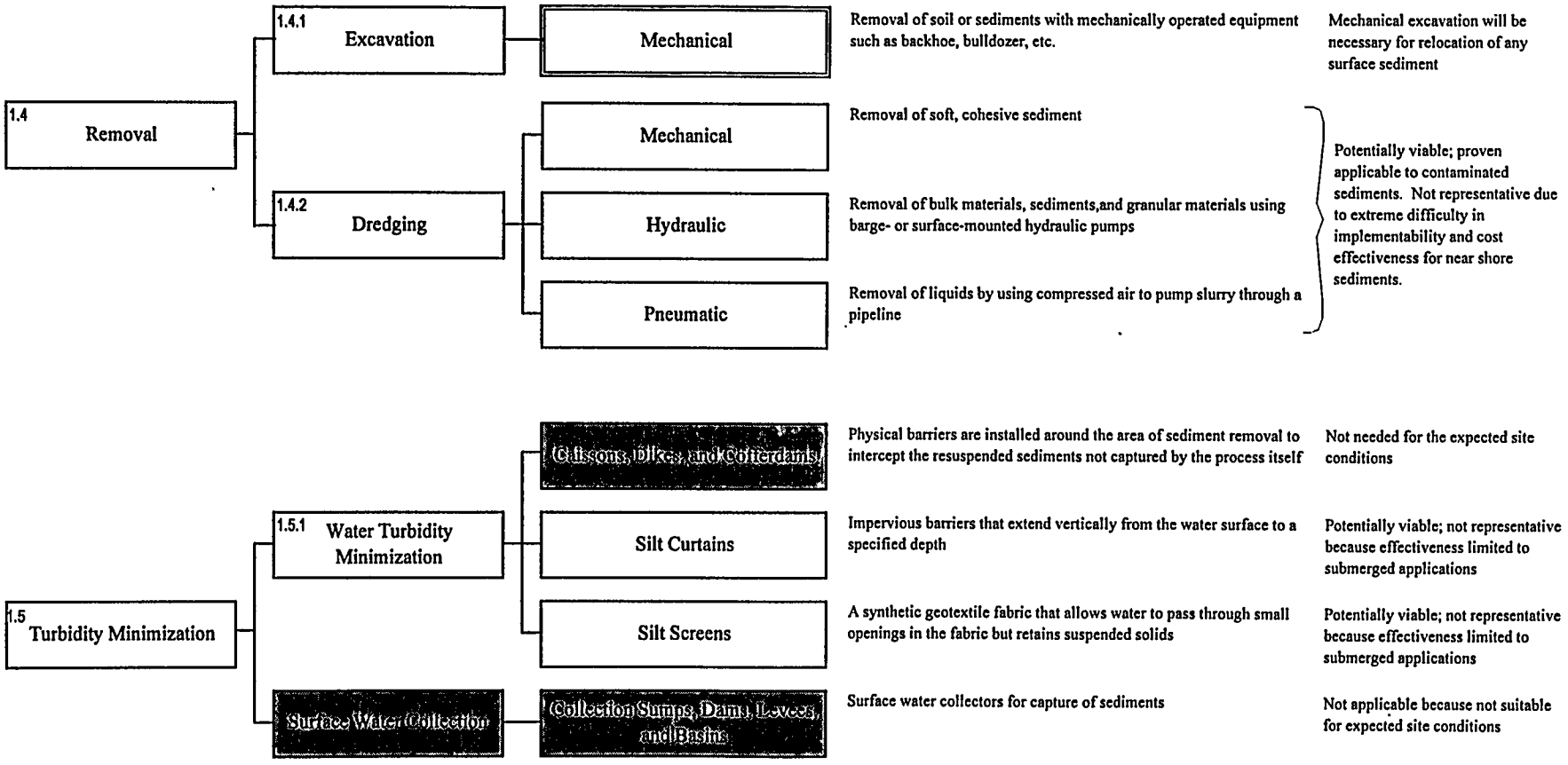

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Fig. 8.1

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Results of process option Screening

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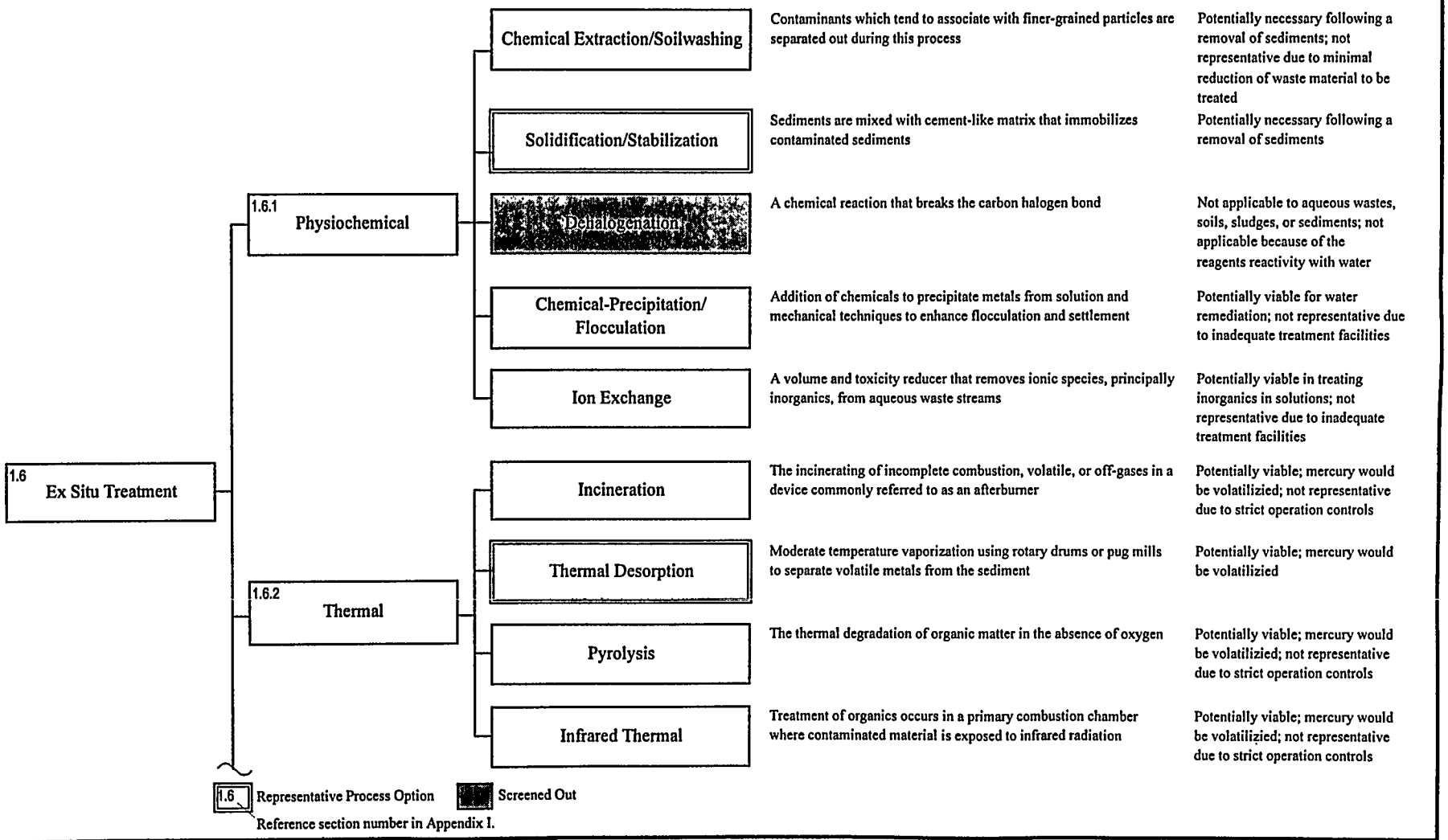

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Fig. 8.1

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Results of process option screening

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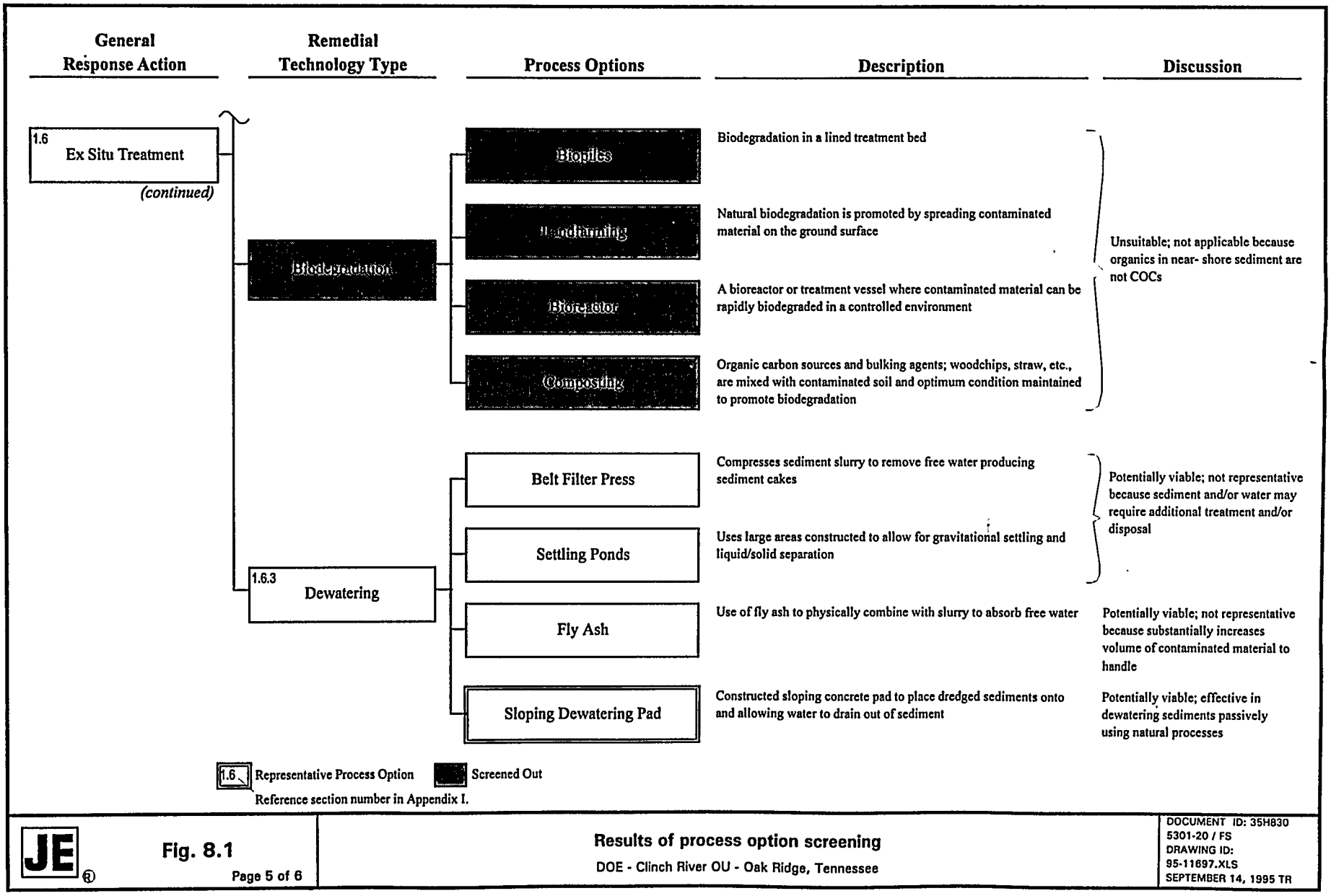

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Fig. 8.1

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Results of process option screening

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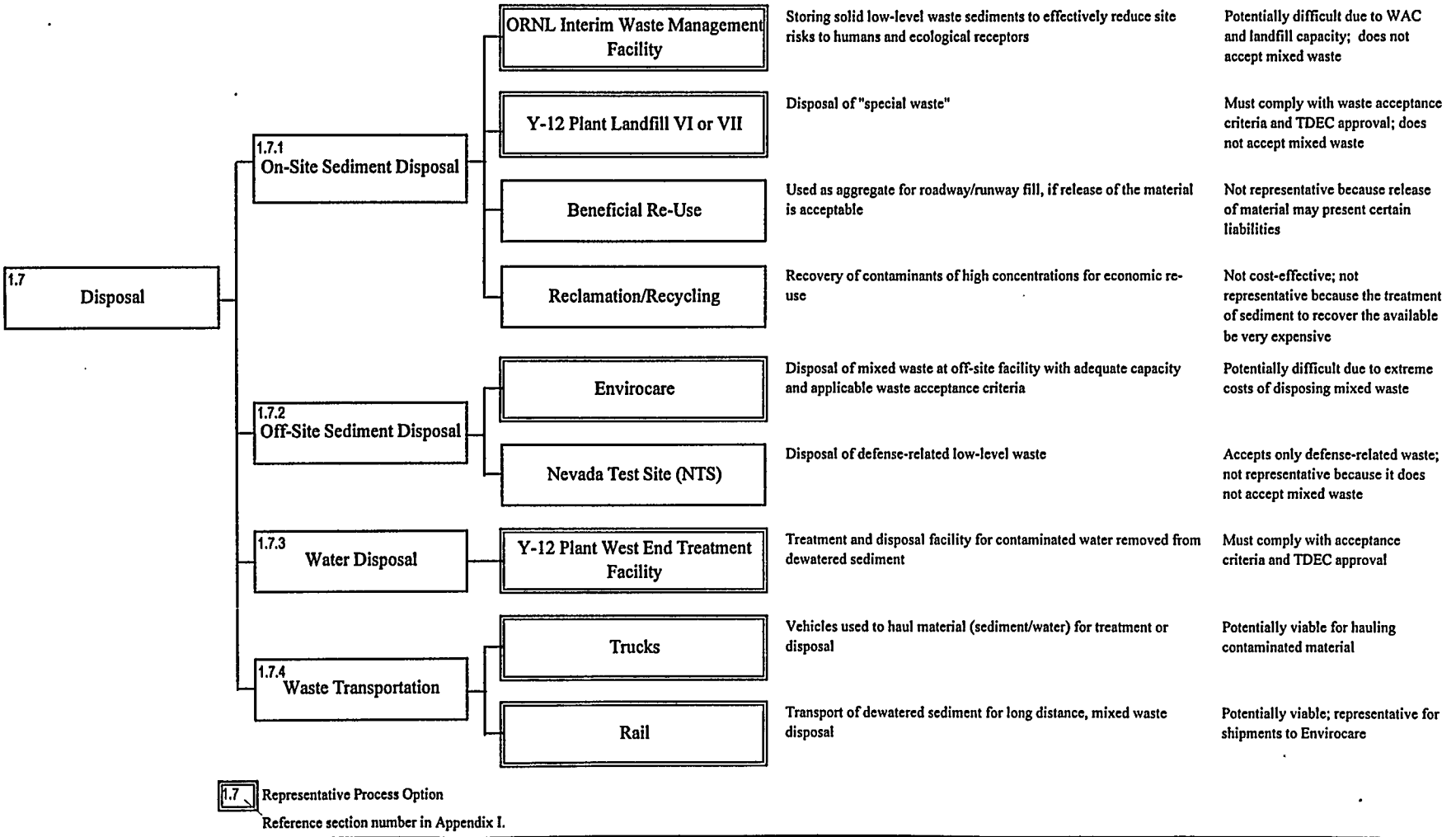

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Fig. 8.1

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Results of process option screening

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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. Section 9.3 screens alternatives based on effectiveness, implementability, and cost. The alternatives that remain following the Section 9.3 screening are described in detail in Section 9.4, including process options, areas by reach or subreach where an alternative is proposed, and descriptions of process options, as applied.

Alternatives 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 1988) establishes an approach to develop appropriate remedial action alternatives. Using this approach, the scope, characteristics, and complexity of the specific conditions at the Clinch River/Poplar Creek OU site 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 for treatment of residuals and untreated waste);

- 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" 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 web. This process will continue to isolate and further reduce availability of the contaminants. In spite of the benefit natural sedimentation gives with respect to 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, if contributing sources continue to add contaminants to the system, then 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 Subreach 3.02 at the mouth of Mitchell Branch 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 remediation of contaminated biota was based on the following:

- Time, preventing contaminant migration, and natural degradation of contaminants are the only reasonable solutions to an overall biotic remediation in this OU. 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 fish pathway of concern (to humans) is based on the consumption of contaminated fish. If release of contaminants to the reservoir and its tributaries cease, then contaminant levels in the fish population will decline. For the near term, interruption of this pathway is the primary focus that each remedial alternative will have with regard to human health risk reduction.

Alternative assembly for the remediation of contaminated shallow sediments was based on the following:

- Near shore sediments are submerged at summer pool under as much as 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 down stream 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 immobile under current daily river conditions and fluctuations.

- Deep sediments <u>do not</u> pose unacceptable risk to human populations or biota when left in place during either summer or winter pool.

- Deep sediments could pose unacceptable risk to humans and biota when dredged and used untreated, exclusive of surrounding clean sediments/soils.

9.2 DESCRIPTION AND SCREENING OF ALTERNATIVES

The following sections present abbreviated descriptions of all alternatives developed for Clinch River and Poplar Creek. 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.

The NCP requires that alternatives be screened primarily on the basis of effectiveness and implementability, and to a lesser extent on cost. Following the screening of alternatives, each of the retained remedial alternatives is summarized, including contingencies.

9.2.1 Alternative 1—No Action

As required by the NCP, the 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 Clinch River and Poplar Creek 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.

Implementability of this alternative is technically feasible. No construction or monitoring activities would be performed. There are no associated capital or operation and maintenance (O&M) costs.

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 both OUs. Components of Alternative 2 include:

- deed restrictions,
- public advisories,
- permit programs,
- monitoring, and
- surveillance and maintenance.

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. 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.

This alternative is effective. The low-level threat contaminants within the OU can be managed with relative ease. Covenants and permit programs will allow shallow and deep sediments to remain undisturbed with little or no unacceptable risk. Approved sediment-disturbing activities would require additional sediment sampling to adequately define the vertical and horizontal extent of contaminated material in the area planned for disturbance. Thus, selective dredging or other sediment-disturbing activities can occur, leaving ORR peak release layers with minimal or no disturbance. Casual or recreational users would not be at risk because they do not receive the same amount of exposure as an on-site resident would receive. Monitoring of fish and sediment is an effective tool to detect changes in the system, such as erosion of contaminated sediment and increased bioavailability, or changes to the system, such as increased input of contaminants. Early detection (best detected with surface water monitoring near the contaminant sources) of these changes will allow for proper response time with appropriate measures.

This alternative is technically implementable. Materials for monitoring and trained technicians to sample fish and sediment are available locally. Administrative implementability is considered to be moderately difficult. Meetings and information exchange among DOE, COE, and TVA regarding sediment-disturbance permits, monitoring data, and implementation of deed restrictions all require an organized effort to remain effective at long-term management of in situ contaminants.

Cost for this alternative is considered moderate. Direct capital costs include administrative costs for implementation of covenants, public advisories, and permit programs. Indirect capital costs include costs from remedial action and remedial design work plans and report preparations. Long-term costs include monitoring supplies and equipment, sampling effort labor, and laboratory costs.

9.2.3 Alternative 3—Source Containment

Alternative 3 is proposed as a means to provide an additional component of source containment for shallow sediments where contaminants exceed risk thresholds. Alternative 3 consists of institutional controls and advisories and geomembrane capping.

Institutional controls and advisories would be the same as described in Alternative 2. Geomembrane capping (described below) would be used in lieu of institutional controls when technically feasible. Access restrictions such as fencing and signs would be used only where site conditions such as steepness of shore slopes are too great to implement source containment measures.

Geomembrane capping would place an impermeable barrier over contaminated shallow sediment to impede availability of these sediments to biological and human receptors. Potential wind and water erosion would be minimized to prevent redistribution of contaminants. Geomembrane capping within Poplar Creek would consist of approximately 44,000 ft^2 of geotextile, membrane, and rip-rap.

Protection of the public from contaminants is further enhanced by adding a horizontal barrier which isolates contaminants from potential receptors, human (now and future on-site resident) and ecological receptors. Shallow sediments would be protected from erosion and thus potential redistribution of contaminants would be minimized. Should a larger area of contamination be identified, advisories, controls, and containment options (at potentially significant additional cost) would be expanded to address the new threat.

This alternative is considered effective. All of the institutional control options in Alternative 2 are applied again for this alternative. Containment of shallow sediments will offer a greater degree of protection from those sediments currently exposed at winter pool. Contaminated deep sediments will continue to be protected from disturbance with permit programs. Fish consumption advisories will remain in effect. Covenants will continue to protect property from improper development, and monitoring will continue to track system conditions. If the property were developed into a residential area the capped portion of the shoreline would protect humans from contact with contaminants thereby keeping risks well below unacceptable levels. Aquatic plants would be displaced during the construction phases. Fish, benthos, and species that feed on these would initially be displaced too, but may ultimately recover, recolonize the area, and benefit by no longer residing in contaminated sediment. Long-term effectiveness of this alternative should be very good because contaminants exceeding risk thresholds would be isolated, leaving no realistic pathway of concern.

Technical implementability for Alternative 3 is moderately difficult. Capping construction requires considerable amounts of vegetative clearing for site access and area preparation. Materials and labor for the cap option are readily available.

Costs associated with this alternative are medium. Direct capital costs for the cap include materials and associated labor, administrative costs for implementation of covenants, public advisories, and permit programs. Indirect capital costs will include costs from remedial action and remedial design work plans and report preparations. Long-term costs will include monitoring supplies and equipment, sampling effort labor, and laboratory costs. Costs for cap maintenance include annual surveillance to ensure cap integrity. Costs for cap repairs are not included in this estimate because of the historical durability of the cap design and the unlikelihood of storms or floating debris causing damage to the cap. Some cap maintenance costs may be expected after that period. Adequate monitoring will detect early deterioration in the cap and allow for less costly repairs, if they become necessary.

9.2.4 Alternative 4—Removal, Treatment, and Disposal

Alternative 4 is proposed as a means to permanently remove approximately 14,000 yd^3 of source contaminants in shallow sediments within Poplar Creek. The main components of this alternative are:

- institutional controls and advisories, as described in Alternative 2;
- mechanical excavation;
- a sloping dewatering pad;
- thermal desorption;

- solidification/stabilization; and
- disposal.

Mechanical excavation of shallow sediments would be performed while the reservoir is at winter pool to minimize the amount of water removed with the sediment. Mechanical excavation would have a very high rate of sediment resuspension if conducted while the sediment is covered with water. Precipitation and run-on could remobilize sediments in uncontrolled erosion, depending on the intensity of the precipitation. A daily cover, such as a tarp, would divert direct precipitation from entering the excavation area. Silt fencing would be tied to staked straw bales installed at the top and bottom of the slope to control run-on and runoff, respectively.

A sloping dewatering pad would be used to dewater sediment. The pad would be bermed on three sides with the open side having the highest elevation. Trucks will bring in the excavated sediment, accessing the open side, and dump it. A collection trench on the lowest side of the pad would allow for the collection of the water draining from the sediment and any seepage through the pad. Only small amounts of pore water are expected, but a means to collect the water is needed. Uncontrolled discharge of this water would be unacceptable.

Thermal desorption is proposed for some of sediments removed from Poplar Creek. Mercury concentrations exceed acceptable risk levels and potentially exceed allowable Toxicity Characteristic Leaching Procedure (TCLP) concentrations.

Solidification/stabilization is proposed as a contingent action for arsenic in McCoy Branch Embayment sediments. Existing data are inconclusive with regard to the extent of arsenic contamination that would fail TCLP tests. Confirmatory stockpile sampling of the sediment may reveal which areas, if any, need solidification/stabilization treatment before disposal.

Disposal options remain uncertain. Suitable disposal areas hinge mainly on the contaminant profile of the sediment and the ability of the facilities to accept, or have the capacity to accept, contaminated material. Although radionuclides are not a serious human or ecological risk concern, they severely limit the options when disposal of sediments is considered. A combination of waste facilities may be necessary to handle different segments of waste streams to accommodate all waste materials. Each of the following disposal options can handle certain contaminant types. All disposal options require that sediment be dewatered (i.e., no free liquid). Additional treatment, such as those proposed previously, may be necessary for the waste to meet individual disposal site waste acceptance criteria (WAC).

Oak Ridge National Laboratory Interim Waste Management Facility (IWMF) may be used to store sediment as a solid low-level radioactive waste only. If removed sediment has radioactivity above background values, but metal contaminants are below TCLP concentration maximums for the facility WAC or the metals component is properly treated and meets these criteria, then storage of the material here is a potential disposal option. This facility cannot accept mixed waste.

Y-12 Plant Landfill VI or VII may be used to dispose of material as an untreated "special waste" if the radioactivity of the material is at or below background values, concentrations of the metals are relatively low, and regulatory officials will grant special waste permit status. Normally Landfill VI or VII is a construction waste landfill, but special waste status will allow the material to be disposed there. This facility cannot accept mixed waste.

Envirocare, Inc. (Envirocare) of Utah is permitted to accept Class A mixed waste. No treatment, other than dewatering, would be necessary before transport to Envirocare. Envirocare would treat the sediments as necessary for legal disposal.

The Y-12 Plant West End Treatment Facility is proposed to treat and dispose of any water drained from removed sediments. The water would be treated, as necessary, for legal discharge.

Trucks are proposed to be the principal on-site means to move contaminated sediment or water to appropriate treatment and/or disposal areas. Spills and leaks from wet sediment must be avoided. Liners or suitable substitutes will be used to protect potential receptors from contaminants dripping onto roadways. Tanker trucks will be used to collect water drained from sediment, accumulate treatable quantities, and deliver water for treatment.

Rail shipments are only proposed for off-site shipments of sediments to Envirocare. Shipments will be packaged in "7 ANV" B-25 boxes and loaded onto rail cars at a suitable railroad spur. If other subreaches are found to be contaminated, or the known contamination is deeper than originally thought, then the removal area or depth can be expanded (at potentially significant additional cost) to address the new threat. If previously unknown contaminants are identified at significant levels, then treatment (or additional treatment) may be necessary to meet acceptance criteria for disposal of the waste sediments.

Effectiveness of this alternative is expected to be good. Removal of the contaminated shallow sediments will ultimately eliminate risks to human and ecological receptors associated with those contaminants. Deep sediments pose no unacceptable risk because they are protected from sediment-disturbing activities through use of a permit program. Fish consumption

advisories would remain in effect. Covenants would remain to protect against improper development although reliance on them would sharply diminish as contaminated shallow sediments are gone. Monitoring of system conditions would remain effective. Long-term protection to humans and the environment would be very good because shallow sediment contaminants would be removed.

Implementing Alternative 4 will be difficult. Implementation around a limited window of opportunity, the winter pool water level, will require proper planning and materials staging so that execution of the task can be accomplished quickly and efficiently. The level of effort to get in to excavate and get out before the reservoir begins filling will create an explosion of activity focused on a few small areas during a relatively short time period. Administrative implementability too would be difficult. Meetings and information exchange among DOE, COE, and TVA regarding sediment-disturbance permits (CWA Section 404 and the Rivers and Harbors Act), tracking of personnel, equipment, and materials will require much effort to initiate and sustain the level of effort needed.

The cost for this alternative will be high. Treatment of Poplar Creek sediments is expected. All sediments will, at a minimum, require dewatering before any additional treatment (such as thermal desorption) to meet WAC. Each step from removal of sediment to dewatering pad to final disposition will involve repeated waste-handling activities. Direct capital costs include heavy equipment to excavate and haul the material, erosion control devices, dewatering pad materials, construction equipment, and labor. Large amounts of indirect capital costs would be associated with remedial design and work plan reports because of the technical complexity of the alternative.

9.3 SUMMARY OF REMEDIAL ALTERNATIVES

Alternatives 1, 2, and 3 are retained for detailed evaluation for various parts in the Clinch River/Poplar Creek OU previously described. Alternative 1 is retained to meet NCP requirements, and will be carried forward for a detailed description and analysis. Alternatives 2 and 3 are retained for possible implementation in the OU. Each will be carried forward for a detailed description and analysis. Alternative 4 will not be carried forward for a detailed analysis at this time for the following reasons:

- Property Ownership—No private properties have been identified with shallow surface sediment contamination in excess of risk thresholds; therefore, no current on-site residents are at risk.

- Short-Term Impacts—Extensive impacts on the ecosystem will result if this alternative is implemented. Aquatic and terrestrial habitat will be destroyed by the level of invasion necessary for excavation and site preparation. Risks to biota will increase because fugitive sediment within contained areas will further contaminate benthic organisms and the fish which feed on them.

- Implementability—The engineering and planning necessary to implement this alternative would be immense relative to the other alternatives. The reservoir, although controlled when water is released and stored, remains a dynamic system. Excavation is difficult due in part to the measures needed to contain fugitive sediment in uncontrolled run-off. Administrative implementability will also be difficult because meetings and information exchange among DOE, COE, and TVA regarding sediment-disturbance permits (as per CWA Section 404) may be required to implement actions in state or federal waterways.

- Risks From Transport—Transport of approximately 58,300 yd^3 of contaminated sediments from the site to the dewatering pad and then from the dewatering pad for approximately 10 miles along public roads to the disposal site may cause traffic hazards because 11-yd^3-capacity dumptrucks would likely transport this material. Potential traffic accidents may be a concern due to spillage of contaminated material.

- Limited Landfill Space Availability—The expected quantities of sediments to be removed from the OU is approximately 58,300 yd^3. Landfill space at all on-site facilities are limited. Additional disposal space would possibly need to be constructed to accommodate the amount of excavated sediment.

- Cost—The approximate minimum direct capital cost for removal, treatment, and placing the treated soil was estimated, using liberal assumptions, at $18,000,000. When transportation, monitoring, personal protective equipment (PPE), and other potential direct cost and indirect activities such as engineering, project management, and quality assurance are included, the estimated cost for Alternative 4 is three or four times greater than any of the other proposed alternatives.

9.4 DETAILED DESCRIPTION OF ALTERNATIVES

Following is a description of each alternative and the manner in which it is intended to be implemented for the OUs.

9.4.1 Alternative 1—No Action

As required by the NCP, the no action alternative provides a comparative baseline against which other alternatives can be evaluated. Under this alternative, the contaminated sediment in Clinch River and Poplar Creek would be left as is, without implementing any containment, removal, treatment, or other mitigating actions. This alternative does not provide for the monitoring of sediment, surface water, or groundwater, and does not provide any support for existing institutional controls to reduce the potential for exposure. Existing controls could not be removed, but they would not be replaced or maintained by DOE.

9.4.2 Alternative 2—Institutional Controls and Advisories

Alternative 2 would manage contaminants in both OUs in situ and without treatment. Long-term management of these contaminants must continue because they would remain in place and only minor reduction of risk levels would occur due to natural sedimentation and natural degradation of the contaminants. Because the contaminants in the OUs are low-level wastes they could be managed fairly easily because some of the proposed options are already in place (public advisories for fish consumption and permit programs that regulate sediment-disturbing activities). The remaining options in this alternative strengthen management of the contaminants, protection of the public, and monitoring of the ecosystem with minimal activities in the sediment (sampling) and only limited action on the shore (fencing).

9.4.2.1 Deed restrictions

The limited contaminant data obtained from shallow sediment sampling locations along Clinch River will not allow for discrete areas TBC independently. For this reason, deed restrictions are recommended for all DOE properties along Clinch River Reaches 1, 2, 3, and 4 to provide blanket coverage for properties that may be released in the future. Covenant-specific activities should include, but are not limited to, restrictions on drinking water well installations, excavations, or other activities that may elevate risk in association with other DOE activities that presently or formerly took place.

The liability for contamination remaining in place (sediments or soils) or in a transient condition (groundwater) will continue to be held by DOE as long as the next owner abides by the conditions set forth in the covenant. Failure to abide by the conditions may absolve DOE from either full or partial responsibility for any damages incurred.

9.4.2.2 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. Administration of these advisories is currently managed by the TDEC. This FS assumes that TDEC management will continue.

9.4.2.3 Permit program

Restrictions on sediment-disturbing activities are jointly managed by TVA and COE. For this FS, it applies 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 the OUs.

9.4.2.4 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 Clinch River/Poplar Creek site would be performed annually to observe system conditions.

Fish monitoring would be conducted semi-annually. 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.

9.4.3 Alternative 3—Source Containment

Options provided in Alternative 2 cannot physically isolate contaminated sediments from potential receptors. The options rely on the voluntary compliance to warnings, signs, and staying out of fenced areas.

Infrequent visitors probably are not at risk because risk scenarios developed in the RI are built on the daily ingestion and/or inhalation of sediment contaminants. Only under the residential scenario does exposure to sediments pose unacceptable risk. However, the sediments are under as much as 6 ft of water for a portion of the year and daily exposures are not likely to occur. In this case, containment of the sediments may be necessary to lower the risk potential by isolating the contaminants and removing the pathway for the future on-site resident.

The geomembrane cap would place a physical barrier between the contaminated sediment and the receptor and remove the pathway of concern. 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. Deep sediments remain in place and protected from disturbance with the use of institutional controls. Public advisories are also planned to advise and protect the public by supplying fish consumption advisories.

Deed restrictions, public advisories, permit programs, and monitoring are the same as in Alternative 2.

Physical barriers, such as fencing, may be used for areas where capping is determined to be technically infeasible (e.g., shore slopes too steep to install the 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.

The geomembrane cap combines the low permeability of a synthetic geomembrane layer and the erosion resistance of riprap to cover and isolate contaminants in place and reduce risk of either direct exposure or redistribution of contaminants by erosion). The area 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 any 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 shall be mulched and seeded to minimize erosion.

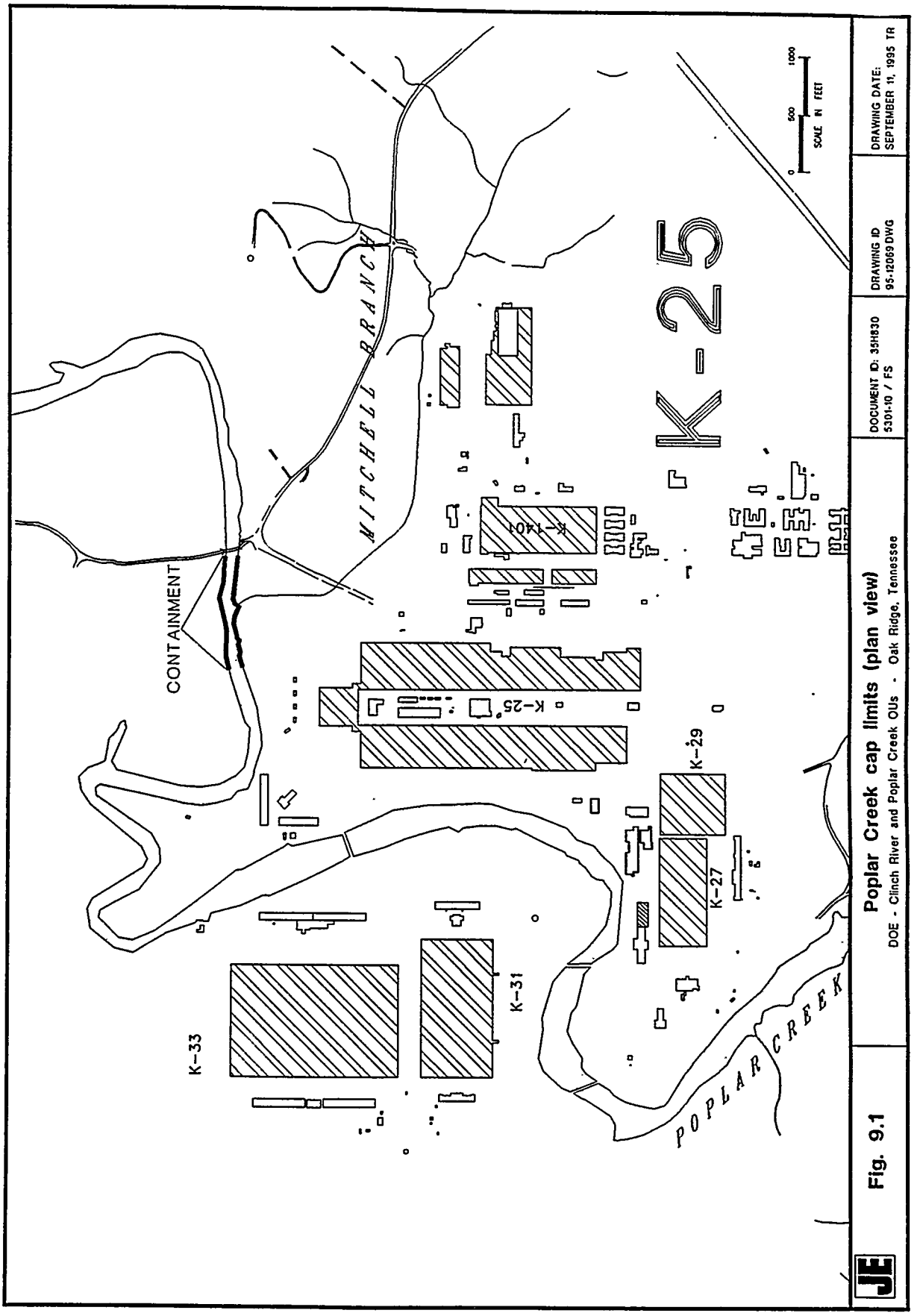
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Fig. 9.1
Poplar Creek cap limits (plan view)
DOE - Clinch River and Poplar Creek OUs - Oak Ridge, Tennessee
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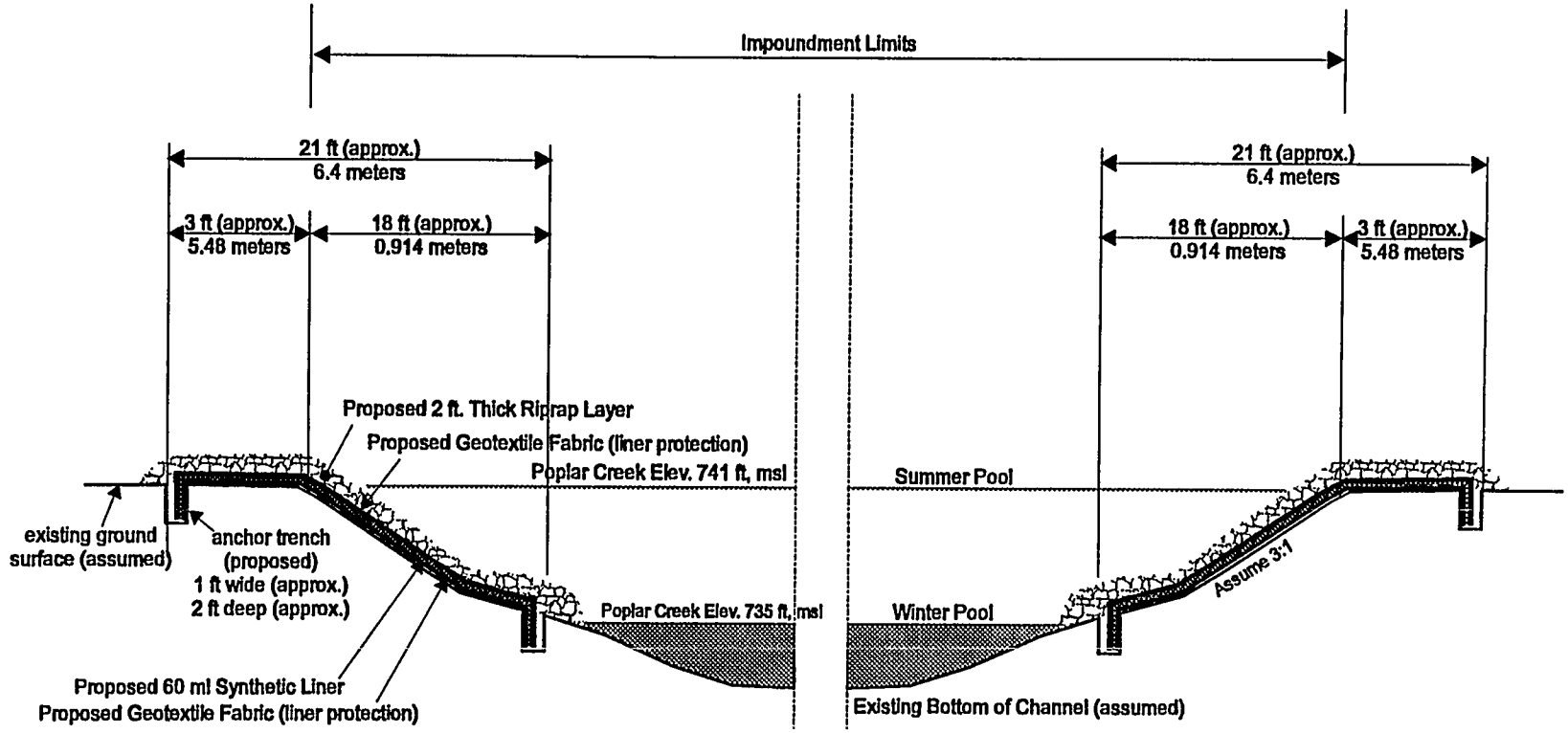

JE	Fig. 9.2	Poplar Creek - Alternative 3 (containment detail) DOE - Clinch River OU - Oak Ridge, Tennessee	DOCUMENT ID: 35H630 5301-10 / FS	DRAWING ID: 95-11623.CDR-1	DRAWING DATE: SEPTEMBER 5, 1995 Ba

The conceptual sequence of construction activities for Alternative 3 is presented in the following list.

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 would 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 a upper-protective geotextile layer over the geomembrane.

9. Place 0.6 m (2 ft) of riprap over the geotextile.

10. Seed and mulch any construction disturbed areas that extend beyond capping limits.

Postclosure monitoring would be conducted annually to ensure the cap remains reliable and performing as designed. Following detection, repairs to the cap should be made as soon as possible. It is anticipated that this cap design 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.

The containment portion of Alternative 3 would be implemented for Subreach 3.02 of Poplar Creek. The Alternative 3 controls and advisories are for the entire area of both OUs.

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 tradeoffs 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.

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 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.

10.1.5 Reduction of Toxicity, Mobility, or Volume

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. As described in Chapter 9, no treatment options are planned at this time.

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 implementing any removal, treatment, containment, or mitigating technologies. 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 provide controls necessary to protect human and ecological receptors from fish consumption or future contaminant disturbance/migration. Alternative 1 is not protective to humans who may catch and eat unacceptable amounts of fish from the OU. Exposure to humans and piscivorous wildlife of 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 deep sediments be disturbed and redistributed in the OU or perhaps elsewhere

downstream. Exposure for contaminated sediment and biotic populations may elevate 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.

There are no short-term risks associated with implementing this alternative since no actions would occur.

Investigations have shown current potential adverse effects to ecological receptors. Ambient toxicity tests on medaka embryos suggest a significant risk to sensitive fish populations in Reach 3 of Poplar Creek. Maximum concentrations of COPECs in pore water appear to be acutely toxic in only four subreaches and chronically toxic in all reaches except Subreach 2.03. Maximum and median concentrations of COPECs in whole sediment appear to be toxic in every reach. Fish from Poplar Creek appear to adversely affect piscivorous wildlife. Mink fed fish from Poplar Creek experienced weight reduction, reduction in litter size, and increase in liver ethoxyresorufin-o-deethylase activity. Insectivorous birds, such as the rough-winged swallow, may be adversely effected by mercury concentrations in aquatic insects in Subreaches 4.01, 4.02, and 4.03 in the Clinch River.

10.2.1.2 Compliance with ARARs

CERCLA Section 121 cleanup standards, including compliance with ARARs, apply only to remedial actions EPA decides should be taken under CERCLA authority; therefore, there are no ARARs for this alternative.

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 risk assessment indicates that contaminant levels in certain fish species exceed the cancer risk threshold of 1.0×10^{-4}. No risk reduction would occur under the no action alternative for either fish or sediment.

The no action alternative would not provide controls for existing contamination, and would not reduce existing site risks. Future risks to the environment may increase if the deep sediment is disturbed, is redistributed without notice, or if existing public advisories are discontinued (assumed to be possible under a no action alternative).

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. There would be no increase in short-term risks to workers or the community and no short-term environmental impacts. There would be no short-term unavoidable adverse impacts to humans, the environment, socioeconomics, or cultural resources as a result of construction activities. Cumulative impacts from construction would not exist because there would be no construction. Risk to humans and the environment would remain at current baseline risk assessment and BERA 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 deed restrictions, 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 RAOs and RGOs listed in Chapter 8 and protects human health. Alternative 2 maintains the existing institutional controls that provide this protection.

Institutional control would include the adoption of deed covenants for all areas with sediment contamination in the event of sale or other release of DOE properties. Continued federal ownership of the site would restrict future on-site residential and agricultural land uses that could result in direct exposure to the 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 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 preventing 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 any ARARs or TBCs. 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 *Code of Federal Regulations* (CFR) 300.430(e)(3), respectively. Identified TBCs (Appendix C) would be met by this alternative.

10.2.2.3 Long-term effectiveness and permanence

Alternative 2 meets the RAOs of preventing future residential exposure to sediments. Long-term protection of human health would be provided by deed restrictions, 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 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 became bioavailable.

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 the 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 EPA, 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.

This alternative does not involve handling contaminated sediments; therefore, there is little risk of potential exposure to workers. Risk of occupational injury is considered negligible. It is anticipated that workers conducting monitoring would use standard monitoring procedures and wear PPE to provide worker safety. There would be no change in the worker population or traffic of the Oak Ridge area during implementation because the existing work force could be used.

This alternative protects the environment 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 duration of sign posting activities will vary according to the extent of the area selected for implementation.

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 and deed restrictions, would be easily implementable. Fencing is implementable although certain areas are heavily vegetated and may require brush clearing during construction.

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 present worth cost of Alternative 2 is approximately $6.1 million. Table 10.1 provides a detailed breakdown of the escalated project costs. More detailed information is provided in Appendix H.

Direct Capital Cost. The escalated cost for the direct capital cost is estimated at $778,000. Direct capital cost includes deed restrictions, public advisories, and permit programs.

Indirect Capital Cost. Escalated indirect capital cost, including contingency, is estimated to be $409,000. Indirect capital cost includes remedial action integration, remedial design work plan, remedial design report, and remedial action work plan.

O&M Cost. Escalated cost, including contingency, for O&M is estimated to be $12.5 million. O&M cost includes monitoring of fish, water, and sediment for approximately 30 years. Monitoring would support the required CERCLA 5-year reviews conducted by the regulators.

10.2.3 Analysis of Alternative 3—Source Containment

The institutional controls and monitoring to be implemented under Alternative 3 are the same as those previously discussed for Alternative 2 with modifications, as necessary, and the augmentation of capping with physical barriers when conditions dictate. Capping would be for the area shown in Figure 9.1.

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.

Table 10.1. Alternative 2—cost estimate

Project Cost Item	Cost ($Thousands)[a]
CAPITAL COSTS	
Direct Cost:	
Deed Restrictions	17
Public Advisories	727
Permit Programs	34
Direct Cost Total (rounded)	778
Indirect Cost:	
Remedial Action Integration	81
Remedial Design Work Plan	16
Remedial Design Report	58
Remedial Action Work Plan	18
Contingency—25 percent	236
Indirect Cost Total (rounded)	409
TOTAL CAPITAL COST	1,187
MONITORING AND MAINTENANCE COSTS	
Monitoring	10,030
Contingency—25 percent	2,510
TOTAL O&M COST	12,540
TOTAL PROJECT ESCALATED COST	13,727
TOTAL PROJECT PRESENT WORTH[b]	**6,076**

[a]Escalated.
[b]Present Value Cost based on 30-year present value, 7 percent discount rate.

Note: Costs presented in table are rounded.

O&M = operation and maintenance

10.2.3.1 Overall protection of human health and the environment

Alternative 3 meets the RAOs developed for Clinch River/Poplar Creek by ultimately minimizing direct contact of humans with the contaminants in shallow sediments by physically isolating contaminated sediments from the surface, rendering them unavailable for ingestion.

Alternative 3 also prevents migration of contaminants through sediment erosion and contaminant redistribution caused by physical disturbance of deep sediments. The level of environmental protection offered by this alternative is increased by the isolation of shallow sediments. These sediments would no longer be in a pathway for biota to contact thus reducing long-term ecological risks.

10.2.3.2 Compliance with ARARs

Although DOE Orders are not ARARs, the radiation exposure limits of 100 mrem/year for exposure of the public, environment, and workers defined in DOE Order 5400.5 would be met. This Order is the primary contaminant-specific TBC for this alternative.

Under the COE authority Nationwide Permit 38, "Cleanup of Hazardous and Toxic Waste," requires wetlands delineation as part of notification for discharges into special aquatic sites. The substantive requirements would be applicable to all activities related to capping. If adverse effects to the wetlands are determined, consultation with the COE and mitigation would be required (e.g., banking) under 33 CFR 330, Appendix A (C)(13). Nationwide Permits 13 (Bank Stabilization) and 18 (Minor Discharge) could also apply.

All remedial activities for Alternative 3 would comply with the TDEC Aquatic Resource Alteration regulations by being as conservative as possible in disturbing soil, sediment, and vegetation.

Before construction activities begin, specific occurrences in soil and sediment disturbance areas of the tall larkspur (*Delphinium exaltatum*) must be identified. The tall larkspur is a state-listed endangered species and a federal candidate for listing as threatened and endangered. If found, consultation with the state would help determine potential mitigation. Probable mitigation would be to transfer the plant(s) to another area of suitable habitat in which no disturbance would occur.

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 activities will allow for at least a 50-ft buffer around the cemeteries. If additional archaeological resources are discovered during installation of fences or access roads or during capping, 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.

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 soil or sediment excavated from the anchor trench areas is determined to be a hazardous, mixed, and/or PCB-contaminated waste, storage, treatment, and/or disposal regulations under the Resource Conservation and Recovery Act and/or the Toxic Substances Control Act would be triggered.

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. Residual risk from Alternative 3, source containment, is the same as from Alternative 2, institutional controls and advisories, for deep sediments. 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 Alternative 3 is very good so long as repairs are made and cap integrity remains intact.

10.2.3.4 Reduction of toxicity, mobility, or volume through treatment

Alternative 3 would not reduce toxicity, mobility, or volume of the waste through treatment, because no treatment is applied to the waste materials.

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. As with any construction activities the

potential for disturbing site sediments exists. The contaminated sediments would not be overly disturbed; however, engineering controls for run-on, direct precipitation, and run-off would be used to minimize sediments escaping; therefore, minimal potential for off-site migration of waterborne contaminants during remedial activities would occur. There is the potential for dust migration from earth-moving activities that would occur during placement of the cap. The baseline risk assessment indicates that fugitive dust emission is not currently a pathway of concern. 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.

There is expected to be an increase in traffic during implementation as access road construction materials are brought in. 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 minimal. Occupational injury and dust inhalation pose the most significant risks to remediation workers. Remediation workers placing the cap would use standard construction and monitoring equipment and PPE to provide worker safety as specified in the construction 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. On-site monitoring during construction would assess the potential effects on workers from construction and determine if additional protection would be needed.

Environmental Impacts. Short-term impacts to ecological receptors would be higher under Alternative 3 than for Alternative 2 during cap installation. The environmental effects would be disruption of habitat and potential sediment resuspension during construction activities.

Disruption of habitat would be permanent for aquatic plants and most benthic organisms in shallow sediments. Deep water plants and animals should not be affected because construction will occur on exposed winter pool sediments and not extend into deep water. 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 to a suitable habitat elsewhere. Replacement of diverse vegetation will

be likely in order to restore areas on which capping is implemented following the close of cap construction. However, while these remedial activities may affect individuals of a species, they are not anticipated to have any effects at the community level.

The effects of containment on the floodplain would be minimal. Some grading and slope alteration will be inevitable; however, existing shoreline slopes would remain as 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 may necessitate a substitution.

Wetland areas could also be encountered in the area intended for implementation of capping. 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 assessment 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 would be required.

Because the contaminated sediments would be disturbed to some extent, there is potential for contaminants to migrate from the site during implementation due to 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.

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 1 year. Administrative duration is the same as in 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.

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, COE, and TDEC.

Materials required for implementation of Alternative 3 include geomembrane, geotextile, and riprap; earth-moving and compaction equipment; silt fences; and sampling equipment. All of these are readily available materials.

10.2.3.7 Cost

The project present worth cost of Alternative 3 is approximately $7.5 million. Table 10.2 provides a detailed breakdown of the escalated project costs. More detailed information is provided in Appendix H.

The escalated direct capital cost is estimated at $1.7 million. Direct capital cost includes deed restrictions, public advisories, permit programs, physical barriers, and capping.

The escalated indirect capital cost, including contingency, is estimated to be $1 million. Indirect capital cost includes remedial action integration, remedial design work plan, remedial design report, and remedial action work plan.

Total escalated cost, including contingency, for O&M is estimated to be $12.4 million. O&M cost includes fence maintenance and monitoring of fish, water, and sediment for approximately 30 years. Monitoring would support the required CERCLA 5-year reviews conducted by the regulators.

Table 10.2. Alternative 3—cost estimate

Project Cost Item		Cost ($ Thousands)[a]
CAPITAL COSTS		
Direct Cost:		
Deed Restrictions		16
Public Advisories		722
Permit Programs		34
Resource Surveys		322
Capping		621
	Direct Cost Total (rounded)	1,715
Indirect Cost:		
Remedial Action Integration		118
Remedial Design Work Plan		64
Remedial Design Report		227
Remedial Action Work Plan		72
Contingency—25 percent		548
	Indirect Cost Total (rounded)	1,029
	TOTAL CAPITAL COST	2,744
O&M COSTS		
Monitoring		9,943
Contingency—25 percent		2,486
	TOTAL O&M COST	12,429
TOTAL PROJECT ESCALATED COST		15,173
	TOTAL PROJECT PRESENT WORTH[b]	**7,487**

[a]Escalated.
[b]Present Value Cost based on 30-year present value, 7 percent discount rate.

Note: Costs presented in table are rounded.

O&M = operation and maintenance

10.3 COMPARATIVE ANALYSIS

A summary of the comparative analysis of alternatives for Clinch River/Poplar Creek is presented in Table 10.3.

10.3.1 Overall Protection of Human Health and the Environment

Alternative 1, no action, would least protect human health or the environment because no action would be taken. The risk assessment indicates unacceptable risk from exposure to contaminants in certain shallow areas with contaminated surface sediments, when deep sediments

Table 10.3. Detailed analysis summary and assessment

Assessment criteria	Alternative 1: No Action	Alternative 2: Institutional Controls and Advisories	Alternative 3: Containment
Overall protection			
Human health	No	Yes	Yes
Environment	No	Yes	Yes
Compliance with ARARs			
Chemical-specific	Not applicable	None	Complies
Location-specific	Not applicable	None	Complies; however, complicated or extensive efforts may be required to avoid adverse effects to wetlands, threatened and endangered species, cultural resources
Action-specific	Not applicable	Complies	Complies
Long-term effectiveness and permanence			
Residual risk	Risks may become unacceptable	Risks remain at current level to humans and biota	Low; contaminant pathway eliminated by isolating contaminated sediment
Adequacy of controls	Not applicable; no controls provided	Adequate	Adequate with proper maintenance
Reliability of controls	Not applicable	Reliable	Reliable, but requires periodic inspection and repairs
Short-term effectiveness			
Worker protection	No risk	Low risk	Moderate risk
Community protection	No protection	Low risk	Low risk
Environmental protection	No protection	No increase	High risk
Time between implementation and completion	Not applicable	6 months	1 year
Reduction of toxicity, mobility, and volume through treatment			
Toxicity, mobility, and volume	None	None	None

Table 10.3. (continued)

Assessment criteria	Alternative 1: No Action	Alternative 2: Institutional Controls and Advisories	Alternative 3: Containment
Implementability			
Technical	Easy	Easy	Moderate
Administrative	Easy	Moderate	Difficult
Material availability	Not applicable	Readily available	Readily available
Cost ($)			
Capital	0	1.5 million	16.9 million
Present worth	0	6.1 million	17.1 million

ARAR = applicable or relevant and appropriate requirement

are dredged and used for agricultural purposes, and if excess fish consumption occurs. If alternative risk assumptions are used, then there would be no unacceptable risks to humans from shallow sediments. 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 than Alternative 1. If properties are released by DOE, the restrictions placed on deeds would prohibit land usages that could elevate risk levels. Alternative 2 would be more protective of the environment than Alternative 1 due to 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 risks to the environment or humans increase.

Alternative 3 would be most protective to human health and the environment by containing of shallow sediment contaminants with a geomembrane cap in addition to 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 would destroy the benthos and their habitat on the capped area. Benthic habitat similar to the existing habitat may never return to the capped area.

During remediation, Alternatives 2 and 3 would protect the community and workers through the use of engineered and institutional controls. Short-term risks to the community (not including transportation) and to nonremediation workers would be approximately equal and within acceptable limits for all action alternatives.

10.3.2 Compliance with ARARs

No ARARs are triggered for the no action alternative. 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 which may be used in Alternative 2. All ARARs for Alternative 3 would be met without waivers. TBCs for Alternative 3 institutional controls are the same as for Alternative 2. Consultations will be held with TVA, COE, and

TDEC to ensure substantive requirements are met for the TVA Act, the COE Nationwide Permit Program, and the TDEC Aquatic Resource Alteration Permit Program. If Alternative 3 is chosen as the preferred alternative, a wetland and/or floodplain analyses would be required before the initiating of actions for containment activities as well as potential mitigation for the tall larkspur (*Delphinium exaltatum*), wetland mitigation (if required), and a Phase I archaeological resource survey.

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 and 3 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 and institutional controls to reduce the residual risks to potential future human receptors and current ecological receptors. Alternative 3 would use containment to prevent access and exposure to the contaminated shallow sediments; however, the waste would remain in place. 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.

10.3.4 Reduction of Toxicity, Mobility, or Volume

None of the proposed alternatives reduce toxicity, mobility, or volume because no treatment options are used.

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.

Alternatives 2 and 3 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 is not expected to have an effect off DOE property.

The risks to workers are expected to be similar in implementing Alternatives 2 and the institutional controls portion of Alternative 3. 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 due to contaminant exposure would be controlled to acceptable levels during construction of a cap.

The short-term environmental effects would vary for the alternatives. Alternative 2 would have little or no short-term impacts. Alternative 3 would likely increase some sediment disturbance, although the use of erosion controls during capping activities would minimize the impact. 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 areas, but full recovery of most species is anticipated, except aquatic plants and shallow sediment benthic organisms.

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. Alternative 3 could be completed within 1–2 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. The materials for caps and the technical experience are available locally. Alternative 3 would be more difficult to implement because of the 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, COE, TDEC, and DOE concerning the substantive requirements of permit programs, monitoring results (and reports), and record keeping.

10.3.7 Cost

Cost estimates are used in the CERCLA FS process to eliminate remediation alternatives that are significantly more expensive than competing alternatives unless they offer commensurate increases in performance or overall protection of human health and the environment. The cost estimates developed are order-of-magnitude estimates with an intended accuracy range of −30

to +50 percent. Final costs would depend on actual labor and material costs, actual site conditions, productivity, competitive market conditions, final scope, final schedule, final engineering design, and other variables.

Costs developed were direct capital cost (including fixed-price remedial construction), indirect capital cost (engineering and project management during remediation), and long-term maintenance and monitoring (post-remediation). Overall, 30-year present value costs were developed for each alternative assuming the following: a discount rate of 7 percent, variable escalation per the ORER0294.VAL standard file for the Automated Estimating System cost estimating program, and a 25–40 percent contingency. Additional cost estimate detail and backup is provided in Appendix H. In summary, Alternative 3 would be noticeably more expensive than Alternatives 1 or 2, due to greater numbers of needed laborers and an extended construction calendar, heavy equipment, and materials.

11. REFERENCES

Adams, S.M., and Hackney, C.T. 1992. *Ecological processes in southeastern United States aquatic ecosystems*. In: C.T. Hackney, S.M. Adams, and W.H. Martin, (eds), Biodiversity of the southeastern United States. pgs. 3-17. John Wiley, New York.

Adams, S.M. and R.B. McLean. 1985. Estimation of largemouth bass, (Micropterus salmoides) Lacepede, growth using condition indices and physiological variables. J. Fish Biology 26:111-126.

Adams, S.M., K.D. Ham, and J.J. Beauchamp. 1994. Application of canonical variate analysis in the evaluation and presentation of multivariate biological response data. Environmental Toxicology and Chemistry 13:1673-1683.

Adams, S.M. 1990. Status and use of bioindicators for evaluating effects of chronic stress on fish. American Fisheries Society Symposium 8:1-8.

Adams, S. M. and M.G. Ryon. 1994. Comparison of health assessment approaches for evaluating the effects of contaminant-related stress on fish populations. J. Aquatic Ecosystem Health 3:15-25.

Adams, S.M., L.R. Shugart, G.R. Southworth, and D.E. Hinton. 1990. Application of bioindicators in assessing the health of fish populations experiencing contaminant stress. Pgs. 333-353, In: J.F. McCarthy and L.R. Shugart, (eds.), Biomarkers of environmental contamination. Lewis Pub., Boca Raton, FL.

Adams, S.M., W.D. Crumby, M.S. Greeley, M.G. Ryon, and E.M. Schilling. 1992a. Relationships between physiological and fish population responses in a contaminated stream. Environmental Toxicology and Chemistry 11:1549-1557.

Adams, S.M., W.D. Crumby, M.S. Greeley, and L.R. Shugart. 1992b. Responses of fish populations and communities to pulp mill effluents: a holistic assessment. Ecotoxicology and Environmental Safety 24:347-360.

Adams, S.M., A.M. Brown, and R.W. Goede. 1993. A quantitative health assessment index for rapid evaluation of fish condition in the field. Transactions of the American Fisheries Society. 122:63-73.

Adams, S.M., W.D. Crumby, M.S. Greeley, Jr., and L.R. Shugart. 1992a. Responses of fish populations and communities to pulp mill effluents: a holistic assesssment. Ecotoxicol. Environ. Safety 24:347-360.

Ankley, G.T., V.S. Blazer, R.E. Reinert, and M. Agosin. 1986. Effects of aroclor 1254 on

cytochrome P-450-dependent monooxygenase, glutathione S-transferase, and UDP-glucuronosylransferase activities in channel catfish liver. Aquatic Toxicol. 9:91-103.

Addison, R.F. 1984. Hepatic mixed function oxidase (MFO) induction in fish as a possible biological monitoring system. pp. 51-60. IN: V.W. Cairns, P.V. Hodson, and J.O. Nriagu, (eds.), Contaminant Effects on Fisheries. John Wiley & Sons, New York.

Alexander, C. M. 1987. *Fishery Status Assessment of Watts Bar Reservoir with Managment Recommendations.* Tennessee Valley Authority Office of Natural Resources and Economic Development, Knoxville, Tennessee. TVA/ONRED/AWR-88/5.

Ashwood, T. L., 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, Tennessee.

Auerlich, R. J. and R. K. Ringer. 1977. "Current status of PCB toxicity including reproduction in mink." Arch. Environ. Contam. Toxicol. 6:279-292.

Ahokas, J. T., Kärki N. T., Oikari, A., and Soivio A. 1976. Mixed function monooxygenase of fish as an indicator of pollution of Aquatic Environment by industrial effluent. Bull. Environ. Contam. Toxicol. 16: 270-274.

ATSDR (Agency for Toxic Substances and Disease Registry). 1989b. *Toxicological Profile for Plutonium.* U.S. Department of Health and Human Services, Public Health Service. Atlanta, GA.

ATSDR (Agency for Toxic Substances and Disease Registry). 1990a. *Toxicological Profile for Dibenz(a,h)anthracene.* U.S. Department of Health and Human Services, Public Health Service, Atlanta, GA.

ATSDR (Agency for Toxic Substances and Disease Registry). 1994. *Toxicological Profile for 4,4'DDT, 4,4'-DDE, 4,4'-DDD.* U.S. Department of Health and Human Services, Public Health Service. Atlanta, GA.

ATSDR (Agency for Toxic Substances and Disease Registry). 1993a. *Toxicological Profile for Aldrin.* U.S. Department of Health and Human Services, Public Health Service. Atlanta, GA.

ATSDR (Agency for Toxic Substances and Disease Registry). 1993b. *Toxicological Profile for Polychlorinated Biphenyls.* U.S. Department of Health and Human Services, Public Health Service. Atlanta, GA.

ATSDR (Agency for Toxic Substances and Disease Registry). 1992. *Toxicological Profile for Boron.* U.S. Department of Health and Human Services, Public Health Service. Atlanta, GA.

ATSDR (Agency for Toxic Substances and Disease Registry). 1990b. *Toxicological Profile for Cresols: o-cresol, p-cresol, m-cresol.* U.S. Department of Health and Human Services, Public Health Service, Atlanta, GA.

Aulerich, R. J. and R. K. Ringer, 1979. Toxic effects of dietary polybrominated biphenyls on mink. Arch. Environ. Contam. Toxicol. 8:487-498.

Aulerich, R. J., R. K. Ringer, and S. Iwamoto, 1973. Reproduetive failure and mortality in mink fed on Great Lakes fish. J. Reprod, Fertil. Suppl. 19:365-376.

Aulerich, R. J., R. K. Ringer, and S. Iwamoto, 1974. Effects of dietary mercury on mink. Arch. Environ. Contam. Toxicol. 2:43-51.

Aulerich, R. J., R. K. Ringer, H. L. Seagran, and W. G. Youatt, 1971. Effects of feeding Coho salmon and other Great Lakes fish on mink reproduction. Canad. J. Zool. 49:611-616.

ACGIH (American Conference of Governmental Industrial Hygienists). 1986. Copper. In: Documentation of the Threshold Limit Values and Biological Exposure Indices, 5th ed. ACGIH, Cincinnati, OH, p. 146.

Adema D. M. M. and L. Henzen. 1989. "A comparison of plant toxicities of some industrial chemicals in soil culture and soilless culture." *Ecotoxicol. Environ. Saf.* 18:219-29.

Aery, N.C., and S. Saker. 1991. "Studies on the effect of heavy metal stress on growth parameters of soybean." *J. Environ. Biol.* 12(1):15-24.

Alexander, G.R. 1977. Food of vertebrate predators on trout waters in north central lower Michigan. The Michigan Academician. 10: 181-195.

Allen, J.G., H.G. Masters, R.L Peet, K.R. Mullins, R.D. Lewis, S.Z. Skirrow, and J. Fry. 1983. Zinc toxicity in ruminants. J. Comp. Pathol. 93:363-377. (Cited in ATSDR 1989, Eisler 1993).

Aller, A.J., J.L. Bernal, M.J. del Nozal, and L. Deban. 1990. "Effects of selected trace elements on plant growth." *J. Sci. Food Agric.* 51:447-479.

Althoff, D.P. and G.L. Storm. 1989. Daytime spatial characteristics of cottontail rabbits in central Pennsylvania. J. Mamm. 70: 820-823.

Amur, M., J. McCarthy, M. Gill. 1982. Respiratory response of guinea pigs to zinc oxide fume. Am. Ind. Hyg. Assoc. J. 43:887-889. (Cited in ATSDR 1989).

Anderson, D.W., R.W. Risebrough, L.A. Woods, Jr., L.R. DeWeese, and W.G. Edgecomb. 1975. "Brown pelicans: improved reproduction off the southern California coast." Science 190: 806-808.

Anthony, E.L.P. and T.H. Kunz. 1977. Feeding strategies of the little brown bat, *Myotis lucifugus*, in Southern New Hampshire. Ecology. 58: 775-786.

APHA. 1989. Standard Methods for the examination of water and watsewater. 17th edition. American Public Health Association. Washington, D.C.

Arnold, T.W. and E.K. Fritzell. 1987. Food habits of prairie mink during the waterfowl breeding season. Can. J. Zool. 65: 2322-2234.

Arthur, E., I. Motzok, and H.D. Branion. 1958. Interaction of dietary copper and molybdenum in rations fed to poultry. Poult. Sci. 37:1181. (Cited in Mehring et al. 1960)

Arthur, J. W., and E. N. Leonard. 1970. Effects of copper on *Gammarus pseudolimnaeus, Physa integra,* and *Campeloma decisum* in soft water. J. Fish. Res. Bd. Canada 27:1277-1283.

Arthur, W.J., III and R.J. Gates. 1988. Trace elements intake via soil ingestion in pronghorns and in black-tailed jackrabbits. J. Range Manage. 41: 162-166.

Asher, C.J., and P.F. Reay. 1979. "Arsenic uptake by barley seedlings." *Aust. J. Plant Physiol.* 6:459-466.

ATSDR (Agency for Toxic Substances and Disease Registry). 1989. Toxicological profile for selected PCBs (Aroclor-1260, -1254, -1248, -1242, -1232, -1221, and -1016). Syracuse Research Corporation for ATSDR, U.S. Public Health Serv. ATSDR/TP-88/21. 135pp.

ATSDR (Agency for Toxic Substances and Disease Registry). 1989. Toxicological Profile for Zinc. Agency for Toxic Substances and Disease Registry, U.S. Public Health Service, Atlanta, GA. 121 pp. ATSDR/TP-89-25.

ATSDR (Agency for Toxic Substances and Disease Registry). 1990. Toxicological Profile for Copper. Prepared by Syracuse Research Corporation for ATSDR, U.S. Public Health Service under Contract 88-0608-2. ATSDR/TP-90-08.

Aughey, E., L. Grant, B.L. Furman, and W.F. Dryden. 1977. The effects of oral zinc supplementation in the mouse. J. Comp. Pathol. 87:1-14.

Aulerich, R.J., R.K. Ringer, M.R. Bleavins, et al. 1982. "Effects of supplemental dietary copper on growth, reproductive performance and kit survival of standard dark mink and the acute toxicity of copper to mink." J. Animal Sci. 55: 337-343.

Azar, A., H.J. Trochimowicz, and M.E. Maxwell. 1973. "Review of lead studies in animals carried out at Haskell Laboratory: two-year feeding study and response to hemorrhage study." In: *Environmental Health Aspects of Lead: Proceedings, International Symposium*, D. Barth et al., eds. Commission of European Communities. pp. 199-210.

R., R. I, Aulerich, and R. K. Ringer, 1984. Effects of chronic dietary hexachlorobenzene exposure on the reproductive performance and survivability of mink and European ferrets. Arch. Environ. Contam. Toxicol. 13:357-365.

Baes C. F. III, R. D. Sharp, A. L. Sjoreen, and R. W. Shor. 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture,* ORNL-5786, Oak Ridge National Laboratory, Martin Marietta Energy Systems, Oak Ridge, Tennessee. September 1984.

Bates, L. D. 1983. *Radioactive Solid Waste Storage and Disposal at Oak Ridge National Laboratory, Description and Safety Analysis.* ORNL/TM-8201. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Bates, J. A., D. L. Dycus, and G. E. Hall. 1992. *Reservoir Monitoring - 1991: Fish Tissue Studies in the Tennessee Valley in 1990.* Tennessee Valley Authority, Water Resources, Chattanooga, Tennessee. TVA/WR-92/7.

BEIR (Biological Effects of Ionizing Radiations). 1990. *Health Effects of Exposure to Low Levels of Ionizing Radiation.* BEIR V. National Academy Press. Washington, DC.

Blaylock, B. G., C. C. Brandt, R. B. Cook, M. L. Frank, S. K. Holladay, L. A. Hook, E. R. Sain, J. L. Keith, and G. W. Suter. 1994. *The Utility of Existing Data Sets for Conducting a CERCLA Baseline Risk Assessment for Lower Watts Bar Reservoir and an Analysis of Quantitation Limits.* Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Blaylock, B. G., 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, Tennessee.

Bogan, A. E., and P. W. Parmalee. 1983. *Tennessee's Rare Wildlife, Volume II: The Mollusks.* Tennessee Wildlife Resources Agency, Nashville, Tennessee.

Brenkert, A. L., C. C. Brandt, K. A. Rose, R. B. Cook, and M. A. Wood. 1992. "A

comparison of two methods for estimating spatial patterns of sediment accumulation in the Clinch River-Watts Bar Reservoir system." pp 78-81 in *Fifth Tennessee Water Resources Symposium,* F. Quinones and K. L. Hoadley (eds.), American Water Resources Association, Nashville, Tennessee.

Brown, A. M., G. D. Jenkins, and G. D. Hickman. 1993. *Reservoir monitoring - 1992, Summary of fish community results.* Tennessee Valley Authority, Norris, Tennessee.

Bruce, G. M, J. E. Buddenbaum, J. K. Lamb, and T. E. Widner 1993. *Dose Reconstruction Feasibility Study, Volume II - Part A.* Tennessee Deprtment of Health Division of Environmental Epidemiology, Nashville, Tennessee.

Baker, D. A. and J. K. Soldat. 1992. *Methods for Estimating Doses to Organisms from Radioactive Materials Released into the Aquatic Environment.* PNL-8150, Battelle Pacific Northwest Laboratories, Richland, Washington.

Barnett, M. O. and R. R. Turner. 1995. *Bioavailability of Mercury in East Fork Poplar Creek Soils.* Oak Ridge National Laboratory, Oak Ridge, Tennessee. Y/ER-215

Bartlett, R.J., and B. James. 1979. "Behavior of chromium in soils: III. Oxidation." *J. Environ. Qual.* 8:31-35.

Beal, F.E.L. 1918. Food Habits of the Swallows, A family of Valuable Native Birds. United States Department of Agriculture Bulletin No. 619: 1-28.

Belwood, J.J. and M.B. Fenton. 1976. Variation in the diet of *Myotis lucifugas (Chiroptera: Vespertilionidae).* Can. J. Zool. 54: 1674-1678.

Benoit, D. A. 1975. Chronic effects of copper on survival, growth, and reproduction of the bluegill (Lepomis macrochirus). Trans. Am. Fish. Soc. 104:353-358.

Bensen, W. H. et al. 1994. Synopsis of discussion section on the bioavailability of inorganic contaminants. pp. 63-71. In J. L. Hamelink, P. F. Landrum, H. L. Bergman, and W. H. Benson. Bioavailability: Physical, Chemical and Biological Interactions. Lewis Pub., Boca Raton, FL.

Berlin, M. 1979. Mercury. pp. 503-530. in L. Friberg (ed). Handbook on the toxicology of metals. Elsevier Press, NY. 709 pp.

Berlin, W. H., R. J. Hesselberg, and M. J. Mac. 1981. Growth and mortality of fry of Lake Michigan lake trout during chronic exposure to PCB's and DDE. Techn. Pap. U. S. Fish Wildl. Ser. 105:11-22.

Bevelhimer, M.S. and Adams, S.M. In press. Assessing contaminant distribution and

effects in a reservoir fishery. Third national reservoir fisheries symposium. Chattanooga, TN.

Bicknell, B.R. J.C. Imhoff, J.L. Kittle, Jr., A. S. Donigan, Jr., and R.C. Johansen. 1993. Hydrologis Simulation Program - Fortran, User's Manual for Release 10. EPA /600/R-93/174. Environmental Research Office, U.S. Environmental Protection Agency, Athens, Georgia.

Biesinger, K. E. and G. M. Christensen. 1972. "Effects of various metals on survival, growth, reproduction and metabolism of the *Daphnia magna*." *J. Fish. Res. Bd. Canada.* 29:1691-1700.

Blamberg, D.L., U.B. Blackwood, W.C. Supplee, and G.F. Combs. 1960. Effect of zinc deficiency in hens on hatchability and embryonic development. Proc. Soc. Exp. Biol. Med. 104:217-220. (Cited in Eisler 1993)

Blaylock, B. G., M. L. Frank and B. R. O'Neal. 1993. *Methodology for Estimating Radiation Dose Rates to Freshwater Biota Exposed to Radionuclides in the Environment.* ES/ER/TM-78. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Blaylock, B. G. and J. R. Trabalka. 1978. "Evaluating the effects of ionizing on aquatic organisms." 7: 103-152. In: J. T. Lett and H. Alder (eds.), *Advances in Radiation Biology.* Academic Press, New York.

Bleavins, M. R., R. J. Aulerich, and R. K. Ringer. 1980. Polychlorinated biphenyls (Aroclors 1016 and 1242): effects on survival and reproduction in mink and ferrets. Arch. Environ. Contam. Toxicol. 9(5):627-635.

Bleavins, M.R. and R.J. Aulerich. 1981. Feed consumption and food passage time in mink (Mustela vison) and European ferrets (Mustela putorius furo). Lab. Anim. Sci. 31: 268-269.

Boyden, R., V.R. Potter, and C.A. Elvehjem. 1938. Effect of feeding high levels of copper to albino rats. J. Nutr. 15(4):397-402.

Brink, M.F., D.E. Becker, S.W. Terril., and A.H. Jensen. 1959. Zinc toxicity in the weanling pig. J. Anim. Sci. 18:836-842.

Brown D. A., R. W. Gossett, G. P. Hershelman, C. F. Ward, A. M. Westcott, and J. N. Cross, 1986. Municipal wastewater contamination in the Southern California Bight: Part I-Metal and organic contaminants in sediments and organisms. Mar. Environ. Res. 18:291-310.

Brown , A. M., G. D. Jenkins, and G. D. Hickman. 1993. Reservoir monitoring - 1992.

Summary of fish community results. Tennessee Valley Authority, Norris, Tennessee.

Broyer, T.C., C.M. Johnson, and R.P. Huston. 1972. "Selenium and nutrition of *Astragalus*. I. Effects of selenite or selenate supply on growth and selenium content." *Plant Soil* 36:635-649.

Burgess, S.A. and J.R. Bider. 1980. Effects of stream habitat improvements on invertebrates, trout populations, and mink activity. J. Wildl. Manage. 44: 871-880.

Burt, W.H. and R.P. Grossenheider. 1976. A field guide to the mammals of America north of Mexico. Third Edition. Houghton Mifflin Co., Boston

Biomedical and Environmental 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, TN.

Biomedical and Environmental Analysis Section. 1994. *Toxicity Values for Use in Hazardous Waste Risk Assessment and Remediation.* Vol. I. ES/ER/TM-76. Martin Marietta Energy Systems. Health Sciences Research Division. Oak Ridge National Laboratory. Oak Ridge, TN.

Cook, R. B., et al. 1993. *Phase 2 Sampling and Analysis Plan, Quality Assurance Project Plan, and Health and Safety Plan for the Clinch River Remeidal Investigation: An Addendum to the Clinch River RCRA Facility Investigation Plan*

Cook, R. B., et al. 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, Tennessee

Calabrese, E. J., R J Aulerich, and G. A. Padgett. Mink as a predictive model in toxicology. Drug Metab. Rev (in press).

Cain, B.W., L. Sileo, J. C. Franson, and J. Moore. 1983. Effects of dietary cadmium on mallard ducklings. Environ. Res. 32: 286-297.

Calder, W.A. and E.J. Braun. 1983. Scaling of osmotic regulation in mammals and birds. Am. J. Physiol. 224: Rr601-R606.

Callazo, J.A. 1985. Food habits of nesting Great Blue herons at Heyburn State Park, Idaho. Northwest Science. 59: 144-146.

Camardese, M.B., D.J. Hoffman, L.J. LeCaptain, and G.W. Pendleton. 1990. Effects of arsenate on growth and physiology in mallard ducklings. Environ. Toxicol. Chem. 9: 785-795.

Campbell, P.G.C., and P.M. Stokes. 1985. Acidification and toxicity of metals to aquatic biota. Can. J. Fish. Aquatic Sci. 42:2034-2049.

Carlson, R.W., D.S. Adriano, K.S. Sajwan, S.L. Abels, D.P. Thoma, and J.T. Driver. 1991. Effects of selected trace metals on germinating seeds of six plant species. Water Air Soil Pollut. 59:231-240.

Chapman, H. D. 1966. Zinc. In H.D. Chapman (ed) *Diagnostic Criteria for Plants and Soils.*

Chapman, J.A., J.G. Hockman, and M.M. Ojeda C. 1980. Sylvilagus floridanus. Mamm. Species. No. 136, pp. 1-8.

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. EPA, Corvallis Environmental Research Laboratory, Corvallis, OR.

Clark, D.R., Jr., P.A. Ogasawara, G.J. Smith, and H.M. Ohlendorf. 1989. Selenium accululation by raccoons exposed to irrigation drainwater at Kesterson National Wildlife Refuge, California, 1986. Arch. Environ. Contam. Toxicol. 18: 787-794.

Collins, J.C. 1981. "Zinc." In N.W. Lepp (ed) *Effects of Heavy Metal Pollution on Plants, Vol. 1. Effects of Trace Metals on Plant Function.* pp. 145-170.

Cunningham, G.N., and M.B. Wise, E.R. Barrick. 1966. Effect of high dietary levels of manganese on the performance and blood constituents of calves. J. Anim. Sci. 25: 532-538

Custer, T.W. and G.H. Heinz. 1980. Reproductive success and nest attentiveness of mallard ducks fed Aroclor 1254. Environmental Pollution (Series A) 21: 313-318.

Cameron, I. L., W. C. Lawrence, and J. B. Lum. 1985. Medaka eggs as a model system for screening potential teratogens. pp. 239-243. IN Prevention of Physical and Mental Congenital Defects, Part C: Basic and Medical Science, Education and Future Strategies. Alan R. Liss, Inc.

Cloud, J. G. 1981. Rapid Communication: Deoxycorticosterone-induced precocious hatching of teleost embryos. J. Exp. Zool. 216:197-199.

Cooper, K. R., H. Liu, P-A. Bergqvist, and C. Rappe. 1991. Evaluation of Baltic herring

and Icelandic cod liver oil for embryo toxicity, using the Japanese medaka (*Oryzias latipes*) embryo larval assay. Environ. Toxicol. and Chem. 10: 707-714.

Carrigan, P. H., Jr., R. J. Pickering, T. Tamura, and R. Forbes 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part A, Investigations of Radionuclides in Upper Portion of Sediment.* ORNL-3721 Suppl. 2A. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Carrigan, P. H. Jr., and R. J. Pickering 1967. *Radioactive Materials in Bottom Sediment of Clinch River: Part B, Inventory of Radionuclides in Undisturbed Cores.* ORNL-3721 Suppl. 2B. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Churchill, M. A., J. S. Cragwall, Jr., R. W. Andrew, 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, Tennessee.

Cook, R. B., 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, Tennessee.

Cottrell, W. D. 1960. *Radioactivity in Silt of the Clinch and Tennessee Rivers.* ORNL-2847. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Cowser, K. E. and W. S. Snyder. 1966. *Safety Analysis of Radionuclide Release to the Clinch River.* ORNL-3721 Suppl. 3. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Daniel, W. W. 1987. Biostatistics: A Foundation for Analysis in the Health Sciences, 4th ed. John Wiley and Sons, New York. 733 pp.

Dey, S., and S. Bhattacharya. 1989. Ovarian damage to *Channa punctatus* after chronic exposure to low concentrations of elsan, mercury, and ammonia. Ecotoxicology and Environmental Safety. 17: 247-257.

Dahlgren, R.B., R.L. Linder, and C.W. Carlson. 1972. "Polychlorinated biphenyls: their effects on penned pheasants." Environ. Health Perspect. 1: 89-101.

DeBuchananne, G.D. and R.M. Richardson, 1956. *Ground-Water Resources of East Tennessee.* Tennessee Department of Conservation, Division of Geology Bulletin 58, Part I. Nashville, Tennessee.

Decisioneering, Inc. 1994. *Crystal Ball: User's Guide.* Denver, Colorado.

Dial, N. 1978a. Methylmercury: some effects on embryogenesis in the Japanese medaka, *Oryzias latipes*. Teratology 17: 83-92.

Dial, N. 1978b. Some effects of methylmercury on development of the eye in medaka fish. Growth. 42: 309-318.

DOE (U.S. Department of Energy). 1994. *Oak Ridge Reservation Site Management Plan for the Environmental Restoration Program.* DOE/OR-1001/R3.

Dunstone, N. 1993. The mink. T&AD Poyser, London. 232pp.

Dycus, D.L. and Meinert, D.L. 1990. reservoir monitoring-1990: Summary of vital statistics and use impairment monitoring on Tennessee Valley Reservoirs, Tennessee Valley Authority, Chattanooga, TN.

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

DOE (U.S. Department of Energy). 1994. *Addendum to the Phase 2 Sampling and Analysis Plan for the Clinch River Remedial Investigation*, DOE/OR/01-1254&D1.

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

DOE (U.S. Department of Energy). 1987. *Remedial Actions Priority Systems (RAPS) Mathematical Formulations.* DOE/RL/87-06.

Dunne, L. J., 1990. *Nutrition Almanac. Third Edition.* Nutrition Search, Inc. John D. Kirschmann, Director. McGraw Hill Publishing Company.

DiToro, D. M. 1985. "A Particle Interaction Model of Reversible Organic Chemical Sorption." Chemosphere 14:1503-1538.

DOE (U.S. Department of Energy). 1979. *Environmental Assessement of the Oak Ridge Gaseous Diffusion Plant Site.* DOE/EA-0106.

DOE (U.S. Department of Energy). 1988. *Historical Releases form Current DOE Oak Ridge Operations Office Facilities.* Oak Ridge operations Office, Oak Ridge, Tennessee. OR-890.

DOE (U.S. Department of Energy). 1987. *Remedial Actions Priority Systems (RAPS) Mathematical Formulations.* DOE/RL/87-06.

DOE (U.S. Department of Energy). 1994. *Oak Ridge Reservation Site Management Plan for the Environmental Restoration Program.* DOE/OR-1001/R3.

Dycus, D. L. 1989. *PCB Studies on Fish from Watts Bar, Ft. Loudoun, Tellico and Chilhowee Reservoirs, 1987.* Tennessee Valley Authority, Water Resources, Chattanooga, Tennessee. TVA/AWR-89/10.

Dycus, D. L. 1990. *PCB Studies on Fish from Watts Bar, Ft. Loudoun, Tellico and Chilhowee Reservoirs, 1988.* Tennessee Valley Authority, Water Resources, Chattanooga, Tennessee. TVA/WR/AB-90/11.

Dycus, D. L. and G. D. Hickman. 1988. *PCB Levels in Fish from Fort Loudoun Reservoir, Fort Loudoun Tailrace, Tellico Reservoir, and Chilhowee Reservoir, Autumn 1986 to Winter 1987.* Tennessee Valley Authority, Water Resources, Knoxville, Tennessee. TVA/ONRED/AWR-88/19.

Dalke,P.D. and P.R. Sime. 1941. Food habits of the eastern and New England cottontails. J. Wildl. Manage. 5: 216-228.

Dave, G. 1984. Effects of waterborne iron on growth, reproduction, survival and haemoglobin in *Daphnia magna.* Comp. Biochem. Physiol. 78C:433-438.

de Boer. 1988. Chlorobiphenyls in bound and non-bound lipids of fishes; comparison of different extraction methods. Chemosphere 17:1803-1810.

DeFoe, D. L., G. D. Veith, and R. W. Carlson. 1978. Effects of Aroclor 1248 and 1260 on the fathead minnow (*Pimephales promelas*). J. Fish. Res. Board Can. 35:997-1002.

DeGraaf, R.M., G.M. Witman, J.W. Lanier, B.J. Hill, and J.M. Keniston. 1981. Forest habitat for birds of the northeast. U.S.D.A. Forest Service. Northeast Forest Experiment Station and Eastern Region. 598 pp.

Deuel, L. E. and A. R. Swoboda. 1972. "Arsenic toxicity to cotton and soybeans." *J. Environ. Qual.* 1:317-20.

Dixon, R. K. 1988. "Response of ectomycorrhizal *Quercus rubra* to soil cadmium, nickel and lead." *Soil Biol. Biochem.* 20(4):555-59.

DOE (United States Department of Energy). 1994. *Addendum to the East Fork poplar Creek - Sewer Line Beltway Remedial investigation Report.* DOE/OR/02-1119&D2/A1/R1

Dowdy, S, and S. Wearden. 1983. Statistics for research. John Wiley and Sons. New York. 537 pp.

Drinker, K., and P. Drinker. 1928. Metal fume fever: V. Results of the inhalation by animals of zinc and magnesium oxide fumes. J. Ind. Hyg. 10:56-70. (Cited in ATSDR 1989).

Drinker, K.R., P.K. Thompson, and M. Marsh. 1927. An investigation of the effect of long continued ingestion of zinc, in the form of zinc oxide, by cats and dogs, together with observations upon the excretion and storage of zinc. Am. J. Physiol. 80:31-64.

Dunning, J.B. 1984. Body weights of 686 species of North American birds. West. Bird Banding Assoc. Monogr. No. 1. Eldon Publ. Co. Cave Crk, AZ. 38 pp.

Dycus, D.L. and Meinert, D.L. 1991. *Reservoir monitoring-1990: Summary of vital statistics and use impairment monitoring on Tennessee Valley Reservoirs.* Tennessee Valley Authority, Chattanooga, TN.

Eberhardt, L., T.J. Peterle, and R. Schofield. 1963. Problems in a rabbit population study. Wildl. Monog. No. 10: 6-7.

Eckerman, K. F. and J. C. Ryman. 1993. External exposure to radionuclides in air, water, and soil. Federal Guidance Report No. 12. Oak Ridge National Laboratory. U. S. Environmental Protection Agency. EPA 402-R-93-081.

Edens, F.W., E. Benton, S.J. Bursian, and G.W. Morgan. 1976. "Effect of Dietary Lead on Reproductive Performance in Japanese Quail, Coturnix coturnix japonica." Toxicol. Appl. Pharmacol. 38: 307-314.

Edmonds, G. F., Jr., S. L. Jensen, and L. Bernes. 1976. Mayflies of North America. U. Minnesota Press, Minneapolis.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1988. The Birder's Handbook: A firld guide to the natural history of North American Birds. Simon and Schuster, Inc. New York. 785 pp.

Eisler, R. 1985. "Cadmium hazards to fish, wildlife, and invertebrates: a synoptic review." *U.S. Fish Wildl. Serv. Biol. Rep.* 85(1.2). 46pp.

Eisler, R. 1985. Selenium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.5). 57pp.

Eisler, R. 1986. Polychlorinated biphenyl hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.7). 72 pp.

Eisler, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.10). 90pp.

Eisler, R. 1988. "Lead hazards to fish, wildlife, and invertebrates: a synoptic review." *U.S. Fish Wildl. Serv. Biol. Rep.* 14. 134pp.

Eisler, R. 1988. Arsenic hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.12). 92pp.

Eisler, R. 1993. Zinc hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.26). 106pp.

Elser, J.J. and B.L. Kimmel 1985. *Nutrient Availability for Phytoplankton Production in a Multiple-Impounded Series.* Canadian Journal of Fisheries and Aquatic Sciences 42(8):1359-1370.

EPA (U. S. Environmental Protection Agency). 1980. *Ambient water quality criteria for polychlorinated biphenyls.* EPA 440/5-80-068. U. S. Environmental Protection Agency, Washington, D. C. 211pp.

EPA (U. S. Environmental Protection Agency). 1980a. "Guidelines and methodology used in the preparation of health effects assessment chapters of the consent decree water quality criteria documents." Fed. Regist. 45(231): 79347-79356.

EPA (U.S. Environmental Protection Agency). 1985. Ambient Water Quality Criteria for Ammonia - 1984. EPA/440/5-85-001. U.S. Environmental Protection Agency, Washington, D.C.

EPA (U. S. Environmental Protection Agency). 1988. *Interim Sediment Criteria Values for Nonpolar Hydophobic Organic Contaminants.* SDC#17. Office of Water Regulations and Standards. Washington, D.C.

EPA (U. S. Environmental Protection Agency). 1993-Acen. *Sediment Quality Criteria for the protection of Benthic Organisms: Acenaphthene.* EPA-822-X 93-013. Office of Science and Technology. Washington D.C.

EPA (U. S. Environmental Protection Agency). 1993-Fluor. *Sediment Quality Criteria for the Protection of Benthic Organisms: Fluoranthene.* EPA-822-R-93-012. Office of Water & Office of Research and Development, Office of Science and Technology, Washington, D.C.

EPA (U. S. Environmental Protection Agency). 1993-Phen. *Sediment Quality Criteria for the Protection of Benthic Organisms: Phenanthrene.* EPA-822-R-93-014. Office of Water & Office of Research and Development, Office of Science and Technology, Washington, D.C.

EPA (U. S. Environmental Protection Agency). 1993b. "Wildlife criteria portions of the proposed water quality guidance for the Great Lakes system." EPA/822/R-

93/006. Office of Science and Technology, Washington, D.C.

EPA (United States Environmental Protection Agency) Region IV. 1994. *Draft Region IV Waste Management Division Sediment Screening Values for Hazardous Waste Sites.* Region IV, United States Environmental Protection Agency, Atlanta, Georgia. 2/16/94 Version.

EPRI (Electric Power Research Institute). 1987. *Measurement of Bioavailable Mercury Species in Fresh Water and Sediments.* Palo Alto, California. EPRI EA-5197.

Eagar, D. C., and R. M. Hatcher. 1980. *Tennessee's Rare Wildlife, Volume I: The Vertebrates.* Tennessee Wildlife Resources Agency, Nashville, Tennessee.

Eckerman and Young. 1980. *A methodology for calculating residual radioactivity levels following decommissioning.* NUREG-0707. U.S. Nuclear Regulatory Commission. Washington, D.C.

Egli, D., D. Conner, R. Erikson, C, Garcia, M. Zamorski, C. Richmond, and R. Collier. 1985. *The Report of the Joint Task Force on Uranium Recycle Materials Processing.* DOE/OR-859, DE88 012848. Cited In: Bruce, G.M, J.E. Buddenbaum, J.K. Lamb, and T.E. Widner 1993. *Dose Reconstruction Feasibility Study, Volume II - Part A.* Tennessee Deprtment of Health Division of Environmental Epidemiology, Nashville, Tennessee.

Eichholz, G. G. 1983. Source terms for nuclear facilities and medical and industrial sites, In: *Radiological Assessment* (J.E. Till and H. R Meyer eds.). U.S. Nuclear Regulatory Commission, Washington, D. C. NUREG/CR-3332, ORNL-5968.

Energy Systems (Martin Marietta Energy Systems). 1990. *Clinch River RCRA Facility Investigation Plan.* ES/ER-1/D1. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Energy Systems (Martin Marietta Energy Systems). 1992. *Oak Ridge Reservation Environmental Monitoring Report for 1991.* EH/ESH-22/(2 vols). Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Energy Systems (Martin Marietta Energy Systems). 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31/(2 vols). Oak Ridge National Laboratory, Oak Ridge, Tennessee.

EPA (U.S. Environmental Protection Agency). 1993a. *Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning.* EPA 822-R-93-011. Office of Water Regulations and Standards, Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1993b. *Integrated Risk Information System (IRIS).* EPA/600/8-86/032A. EPA Office of Health and Environmental Assessment, Cincinnati, Ohio.

EPA (U.S. Environmental Protection Agency). 1992. "Criteria Charts," December update, Region IV Water Management Division.

EPA (U.S. Environmental Protection Agency). 1991a. *Human Health Evaluation Manual, Supplemental Guidance; Standard DeGault Exposure Factors.* OSWER Directive 9285.6-03. U.S. Environmental Protection Agency.

EPA (U.S. Environmental Protection Agency). 1991b. *Uptake/biokinetic model for lead. Version 0.50.* Environmental Criteria and Assessment Office, Cincinnati. January.

EPA (U.S. Environmental Protection Agency). 1990a. *Guidance on Remedial Actions for Superfund Sites with PCB Contamination,* EPA/540/G-90/007, Office of Emergency and Remedial Response, Washington, D.C.

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

EPA (U.S. Environmental Protection Agency). 1989b. *Briefing Report to the EPA Science Advisory Board on the Equilibrium Partitioning Approach to Generating Sediment Quality Criteria.* EPA 440/5-89-002. Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1989c. Risk Assessment Guidance for Superfund Volume I Human Health Evaluation Manual (Part A) Interim Final. EPA/540/1-89/002. Office of Emergency and Remedial Response, Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1988a. *Methodology for evaluating potential carcinogenicity in support of reportable quantity adjustments pursuant to CERCLA Sec. 102.* OHEA-C-073. Office of Health and Environmental Assessment. Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1988. *Laboratory Data Validation Functional Guidelines for Evaluating Organics Analyses.* Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1988. *Laboratory Data Validation Functional Guidelines for Evaluating Inorganics Analyses.* Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1986. *Quality criteria for water.* EPA440/5-86-001. Office of Water Regulations and Standards, Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1986a. *Test Methods for Evaluating Solid Waste.* Office of Solid Waste and Emergency Response, Washington, D.C.

EPA. (U.S. Environmental Protection Agency). 1985a. *Reference Values for Risk Assessment.* Environmental Criteria and Assessment Office, Cincinnati, Ohio.

EPA. (U.S. Environmental Protection Agency). 1982. *Pesticide assessment guidelines, Subdivision E, Hazard evaluation: Wildlife and Aquatic Organisms.* EPA-540/9-82-024. Office of Pesticide Programs. U.S. Environmental Protection Agency, Washington, D.C.

Ewing, L. K. 1993. *Suspended-Sediment Inflows to Watts Bar Reservoir.* Tennessee Valley Authority, Hydraulic Engineering, Norris Tennessee. Report No. WR28-3-14-105.

Eckerman and Young. 1980. *A methodology for calculating residual radioactivity levels following decommissioning.* NUREG-0707. U.S. Nuclear Regulatory Commission. Washington, D.C.

Eicholz, G.G. 1983. "Source Terms for Nuclear Facilities and Medical and Industrial Sites," in *Radiological Assessment.* NUREG/CR-3332. ORNL-5968. J.E. Till and H.R. Meyer, (eds). U.S. Nuclear Regulatory Commission. Washington, DC.

Energy Systems (Martin Marietta Energy Systems). 1992. *Oak Ridge Reservation Environmental Monitorin Report for 1991.* EH/ESH-22/(2 vols). Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Energy Systems (Martin Marietta Energy Systems). 1993. *Oak Ridge Reservation Environmental Monitoring Report for 1992.* EH/ESH-31/(2 vols.). Oak Ridge National Laboratory, Oak Ridge, Tennessee.

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, DC.

EPA (United States Environmental Protection Agency). 1994a. *Integrated Risk Information System.* IRIS database. Office of Research and Development. Washington, DC.

EPA (U.S. Environmental Protection Agency). 1991a. *Human Health Evaluation Manual, Supplemental Guidance; Standard Default Exposure Factors.* OSWER Directive 9285.6-03. U.S. Environmental Protection Agency.

EPA (U.S. Environmental Protection Agency). 1991b, Memorandum from the USEPA, Region IV Dated March 20, 1991 to Hazardous Waste Contractor from Elmer W.

Akin, Health Assessment Officer Subject: Region IV Risk Assessment Guidance.

EPA (U.S. Environmental Protection Agency). 1991c. *Uptake/biokinetic model for lead. Version 0.50.* Environmental Criteria and Assessment Office, Cincinnati, Ohio, January 1991.

EPA (United States Environmental Protection Agency). 1991e. *Test Methods for Evaluating Solid Waste.* SW-846, 3rd Edition. Washington, DC.

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

EPA (U.S. Environmental Protection Agency). 1989c. *Risk Assessment Guidance for Superfund Volume I Human Health Evaluation Manual (Part A) Interim Final.* EPA/540/1-89/002. Office of Emergency and Remedial Response, Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1985a. *Reference Values for Risk Assessment.* Environmental Criteria and Assessment Office, Cincinnati, Ohio.

Energy Systems (Martin Marietta Energy Systems). 1990. *Clinch River RCRA Facility Investigation Plan.* ES/ER-1/D1. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

EPA (U. S. Environmental Protection Agency). 1993a. "Wildlife exposure factors handbook." Volume I. Office of Research and Development, Washington, DC. EPA/600/R-93/187a.

Fenton, M.B. and R.M.R. Barclay. 1980. *Myotis lucifugus.* Mammal. Species. 142: 1-8.

Feldman, I., J. Jones, and R. Cross. 1967. "Chelation of uranyl ions adenine nucleotides." *J. Am. Chem. Soc.* 89:49-55.

Fenton, M.B. and G.P. Bell. 1979. Ecolocation and feeding behavior in four species of *Myotis (Chiroptera).* Can. J. Zool. 57: 1271-1277.

Ferm, V.H. and D.P. Hanlon. 1974. Toxicity of copper salts in hamster embryonic development. Biol. Reprod. 11: 97-101. (Cited in U.S. AF 1990)

Ferm, V.H., and W.M. Layton, Jr. 1981. Teratogenic and mutagenic effects of cadmium. Pp 743-756 in J.O. Nriagu (ed) Cadmium in the environment. Part 2. Healthe Effects. John Wiley, New York.

Fimreite, N. 1970. Effects of methyl mercury treated feed on the mortality and growth of leghorn cockerels. Can. J. Anim. Sci. 50: 387-389.

Fimreite, N. 1971. Effects of dietary methylmercury on ringnecked pheasants. Can. Wildl. Serv. Occ. Paper No. 9. 37 pp.

Fitzhugh, O.G. 1948. "Use of DDT insecticides on food products." Ind. Eng. Chem. 40: 704-705.

Fox, M.R.S., and R.M. Jacobs. 1986. Zinc: essentiality, function, effects of deficiency, and requirements. In: Metal Ions in Biological Systems, H. Sigel, ed. Marcel Dekker, Inc., New York. pp. 214-248.

Foy, C.D., R.L. Chaney, and M.C. White. 1978. "The physiology of metal toxicity in plants." *Ann. Review Plant Physiol.* 29:511-566.

Francis, P.C., W.J. Birge and J.A. Black. 1984. "Effects of Cadmium-Enriched Sediment on Fish and Amphibian Embryo-Larval Stages." *Ecotoxicology and Environmental Safety* 8:378-387.

Fletcher, G.L, M.J. King, J.W. Kiceniuk, and R.F. Addison. 1982. Liver hypertrophy in winter flounder following exposure to experimentally oiled sediments. Comp. Biochem. Physiol. 73C:457-462.

Fehring, J. P. and D. L. Meinert 1993. *Reservoir Vital Signs Monitoring - 1992: Physical and Chemical Characteristics of Water and Sediment.* Tennessee Valley Authority, Water Resources, Chattannoga, Tennessee.

Ford, C. J., J. T. Byrd, J. M. Grebmeier, R. A. Harris, R. C. Moore, S. E. Madix, C. D. Rash, and M. A. Smith. 1993. *Final Project Report on Arsenic Biogeochemistry in the Clinch River and Watts Bar Reservoir.* ORNL/ER-206. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Goede, R. W., and B. A. Barton. 1990. Organismic indices and an autopsy-based assessment as indicators of health and condition of fish. American Fisheries Society Symposium 8:93-108.

Greeley, M. S., Jr., D. R. Calder, and R. A. Wallace. 1987. Oocyte growth and development in the striped mullet, *Mugil cephalus*, during seasonal ovarian recrudescence: Relationship to fecundity and size at maturity. Fish. Bull. 85: 187-200.

Greeley, M. S., Jr., R. MacGregor III, and K. R. Marion. 1988a. Changes in the ovary of the Gulf killifish, *Fundulus grandis* (Baird and Girard), during seasonal and semilunar spawning cycles. J. Fish Biol. 33: 97-107.

Greeley, M.S., S.M. Adams, W.D. Crumby, and R. Epler. 1994. Bioindicator assessment of fish health and reproductive success in Lake hartwell and Twelve Mile Creek. U. S. Corps of Engineers, Tech. Memorandum No. 8, Hartwell, S.C.

Greeley, M. S., Jr., R. MacGregor III, and K. R. Marion. 1988b. Variation in plasma oestrogens and androgens during the seasonal and semilunar spawning cycles of female gulf killifish, *Fundulus grandis* (Baird and Girard). 33: 419-429.

Gatz, A. J., and S.M. Adams. 1994. Patterns of movement of centrachids in two warmwater streams in eastern Tennessee. J. Freshwater Fish Ecology 3:35-48.

Gardner, R.H., W.W. Hargrove, D.A. Levine, S.M. Pearson, and K.A. Rose. In press. *Spatial analysis of cesium in sediments of Watts Bar Reservoir.*

Garner, J. M. and O. W. Kochitzky 1956. "Radioactive Sediments in the Tennessee River System." *Journal of the Sanitary Engineering Division of the American Society of Civil Engineers,* Volume 82, No. SA2, August 1956.

Gasaway, W. C. and I. O. Buss. 1972. "Zinc Toxicity in the Mallard Duck." J. Wildl. Mgmt. 36:1107-1117.

Goddard, P. L., A. J. Legeay, D. S. Pesce, and A. M. Stanley. 1991. *Site Description of Environmental Restoration Units at the Oak Ridge K-25 Site, Oak Ridge, Tennessee.* K/ER-47. Martin Marietta Energy Systems. Oak Ridge, Tennessee.

Griffith, W. L. 1957. "Uranium Losses from the Y-12 Plant to the Environment." Y-B92-13. Cited In: Bruce, G.M, J.E. Buddenbaum, J.K. Lamb, and T.E. Widner 1993. *Dose Reconstruction Feasibility Study, Volume II - Part A.* Tennessee Department of Health Division of Environmental Epidemiology, Nashville, Tennessee.

Gallup, W.D., and L.C. Norris. 1939. The amount of manganese required to prevent perosis in the chick. Poult. Sci. 18: 76-82.

Gasaway, W.C., and I.O. Buss. 1972. Zinc toxicity in the mallard duck. J. Wildl. Manage. 36: 1107-1117.

Gay, D.D. 1975. "Biotransformation and chemical form of mercury in plants." In *International Conference on Heavy Metals in the Environment Symposium Proceedings, Vol. II,* pt. 1. pp. 87-95.

Geertz-hansen, P., and E. Mortensen. 1983. The effect of dissolved and precipitated iron on the reproduction of brown trout *(Salmo trutta).* Vatten 39:55-62.

Gerell, R. 1970. Home ranges and movements of the mink Mustela vison Schreber in southern Sweden. Oikos 21: 160-173.

Gerisch, F. M. 1984. Evaluation of static renewal chronic toxicity test method for *Daphnia magna* Straus using boric acid. Environ. Toxicol. Chem. 3:89-94.

Gopinath, C., G.A. Hall and J.McC. Howell. 1974. Title not given. Res. Vet. Sci. 16: 57-69. (Cited in Aaseth and Norseth 1986)

Goyer, R.A. 1991. Toxic effects of metals. In: M.O. Amdur, J. Doull and C.D. Klaasen, Eds., Casarett and Doull's Toxicology, 4th ed. Pergamon Press, New York, NY, p. 653-655.

Gould, Ed. 1955. The feeding efficiency of insectivorous bats. J. Mammal. 36: 399-407.

Grummer, R.H., O.G. Bentley, P.H. Phillips, and G. Bohstedt. 1955. The role of manganese in growth, reproduction, and and lactation of swine. J. Anim. Sci. 19: 170-175.

Gupta, U.C. 1984. "Boron nutrition of alfalfa, red clover, and timothy grown on podzol soils of eastern Canada." *Soil Sci.* 137(1):16-22.

Halstead, R.L., B.J. Finn, and A.J. MacLean. 1969. "Extractability of nickel added to soils and its concentration in plants." *Can. J. Soil Sci.* 49:335-342.

Hamilton, W.J.,Jr. 1940. The summer food of minks and raccoons on the Montezuma marsh, New York. J. Wildl. Manage. 4: 80-84.

Hammonds, J.S., F.O. Hoffman, and S.M. Bartell. 1994. An introductory guide to uncertainty analysis in environmental and health risk assessment. ES/ER/TM-35/R1. Oak Ridge National Laboratory, Oak Ridge, TN.

Hansen, L. G., W. B. Wiekhorst, and J. Simon. 1976. Effects of dietary Aroclor 1242 on channel catfish (*Ictalurus punctatus*) and the selective accumulation of PCB components. J. Fish. Res. Board Can. 33:1343-1352.

Harr, J.R. 1978. Biological effects of selenium. pp. 393-426. in F.W. Oehme (ed.) Toxicity of heavy metals in the environment. Part I. Marcel Dekker, Inc. New

Harris, H.J., T.C. Erdman, G.T. Ankley, and K.B. Lodgel. 1993. Measures of Reproductive Success and Poly-chlorinate Biphenyl Residues in Eggs and Chicks of Forster's Terns on Green Bay, Lake Michigan, Wisconsin--1988. Arch. Environ. Contam. Toxicol. 25: 304- 314.

Harvey, M.J. 1992. BATS of the Eastern United States. Arkansas Game & Fish Commission/ U.S. Fish & Wildlife Service.

Haywood, S. 1985. Copper toxicosis and tolerance in the rat. I. Changes in copper content of the liver and kidney. J. Pathol. 145: 149-158.

Heinz, G.H. 1979. "Methyl mercury: reproductive and behavioral effects on three generations of mallard ducks." J. Wildl. Mgmt. 43: 394-401.

Heinz, G.H., and S.D. Haseltine. 1983. Altered avoidance behavior of young black ducks fed cadmium. Environ. Toxicol. Chem. 2: 419-421.

Heinz, G.H., D.J. Hoffman, A.J. Krynitsky, and D.M.G. Weller. 1987. "Reproduction in mallards fed selenium." Environ. Toxicol. Chem. 6: 423-433.

Heinz, G.H., D.J. Hoffman, and L.G. Gold. 1989. Impaired reproduction of mallards fed an organic form of selenium. J. Wildl. Mgmt. 53: 418-428.

Hill, E.F., and M.B. Camardese. 1986. Lethal dietary toxicities of environmental contaminants and pesticides to Coturnix. U.S. Fish Wildl. Serv. Tech. Rep. 2. 147pp.

Hoffman, D.J., and G.H. Heinz. 1988. Embryotixic and teratogenic effects of selenium in the diet of mallards. J. Toxicol. Environ. Health. 24: 477-490.

Hoffman, R.D. 1978. The diets of herons and egrets in southwestern Lake Erie. Natl. Audubon Soc. Res. Ref. 7: 365-369.

Holly Jr., F.M., J.C. Yang, P. Schwartz, J. Schaefer, S.H. Hsu and R. Einhellig. 1990. Charima: Numerical Simulation of Unsteady Water and Sediment Movement in Multiple Connected Networks of Mobile-be Channels. IIHR Report No. 343, Iowa Institute of Hydraulic Research, The University of Iowa, Iowa City, Iowa 52242.

Hornshaw, T.C., R.J. Aulerich and H.E. Johnson. 1983. Feeding Great Lakes fish to mink: effects on mink and accumulation and elimination of PCBs by mink. J. Toxicology and Environ. Health 11: 933-946.

Huckabee, J.W., and B.G. Blaylock. 1973. "Transfer of mercury and cadmium from terrestrial to aquatic ecosystems." *Adv. Exp. Med. Biol.* 40:125-160.

Huckins, J. N., T. R. Schwartz, J. D. Petty, and L. M. Smith. 1988. Determination, fate, and potential significance of PCBs in fish and sediment samples with emphasis on selected AHH-inducing congeners. Chemosphere 17:1995-2016.

Hudson, R. H., R. K. Tucker, and M. A. Haegele. 1984. Handbook of toxicity of pesticides to wildlife. 2nd ed. U.S. Fish Wildl. Serv. Resour. Publ. 153. 90pp.

Hussein, A.S., A.H. Cantor, and T.H. Johnson. 1988. Use of high levels of dietary aluminum and zinc for inducing pauses in egg production of Japanese quail. Poultry Sci. 67:1157-1165.

Hull, R.N. and Suter, G.W. II. 1994. *Toxicological benchmarks for screening of potential contaminants of concern for effects on sediment-associated biota: 1994 Revision.* Oak Ridge National Laboratory, Oak Ridge, TN. ES/ER/TM-95/R1.

Hutchinson, T.C. 1981. "Nickel". In N.W. Lepp (ed) *Effects of Heavy Metal Pollution on Plants, Vol. 1.* Effects of Trace Metals on Plant Function. pp. 171-212.

Hall, G. E. and D. L. Dycus. 1991. *Fish Tissue Studies in the Tennessee Valley in 1989.* Tennessee Valley Authority, Water resources, Chattanooga, Tennessee. TVA/WR/AB-91/12.

Harr, J. R. 1978. "Biological effects of selenium." pp. 393-426. In F.W. Oehme (ed.) *Toxicity of heavy metals in the environment.* Part I. Marcel Dekker, Inc. New York. 515 pp.

Heinz, G. H. 1979. "Methyl mercury: reproduction and behavioral effects on three generations of mallard ducks." J. Wildl. Mgmt. 43: 394-401.

Heitman, J. F. 1979. *Impact of a Commercial Net Fishery on Striped Bass Populations in Two Tennessee Reservoirs.* M.S. thesis. University of Tennessee, Knoxville, Tennessee.

Hickman, G. D., E. M. Scott Jr., and A. M. Brown. 1991. *Fish community results - 1990.* Tennessee Valley Authority, Norris, Tennessee.

Hoffman, F.O., B.G. Blaylock, C.C. Travis, K.L. Daniels, E.L. Etnier, K.E. Cowser, and C. W. Weber. 1984. *Preliminary Screening of Contaminants in Sediments.* ORNL/TM-9370. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Hoffman, F.O., 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, Tennessee.

Holladay, S. K., 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, Tennessee.

Hull, R. N. and G. W. Suter II. 1994. *Toxicological benchmarks for screening potential contaminants of concern for effects on sediment-associated biota.* ORNL/ER-177.

Hacking, M.A., J. Budd, and K. Hodson. 1978. The ultrastructure of the liver of the rainbow trout: normal structure and modifications after chronic administration of a polychlorinated biphenyl Aroclor 1254. Canadian Journal Zoology 56:477-491.

Hinton, D.E., and D.J. Lauren. 1990. Liver structural alterations accompanying chronic toxicity in fishes:potential biomarkers of exposure.

Heisinger, J. F., and W. Green. 1975. Mercuric chloride uptake by eggs of the ricefish and resulting teratogenic effects. Bull. Environ. Contam. Toxicol. 14: 655-673.

Hinzman, R. L. (ed.). 1993. Second Report on the Y-12 Plant Biological Monitoring and Abatement Program for East Fork Poplar Creek. Preliminary Draft ORNL/TM Report. Oak Ridge, Tennessee.

Heaton S. N., 1992. Effects on reproduction of ranch mink fed carp from Saginaw Bay, Michigan . M. S. Thesis, Department of Animal Science, Michigan State University, East Lansing, MI. 152 pp.

Hiraoka, Y. 1985. A re-examination of the toxicity test for pollutants. Hiroshima J. of Med.Sci. 34: 323-326.

Hiraoka, Y., S. Ishizawa, T. Kamada, and H. Okuda. 1985. Acute toxicity of 14 different kinds of metals affecting medaka fry. Hiroshima J. of Med. Sci. 34: 327-330.

Hiraoka, Y. and H. Okuda. 1983. Characteristics of vertebral abnormalities of medaka as a water pollution indicator. Hiroshima J. of Med. Sci. 32: 261-266.

Hiraoka, Y. and H. Okuda. 1984. Time course study of occurrence of anomalies in medaka's centrum by cadmium or fenitrothion emulsion. Bull. Environ. Contam. Toxicol. 32: 693-697.

Hiraoka, Y., and H. Okuda. 1983. Characteristics of vertabral abnormalities of medaka as a water pollution indicator. Hiroshima Journal of Medical Sciences 32(3): 261-266.

Helder, T. 1980. Effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) on early life stages of the pike (*Esox lucius* L.). The Science of the Total Environment 14: 255-264.

Heath, A.G. 1987. Use of physiological and biochemical measures in pollution biology. Pgs. 221-234, In A.G. Heath , editor. Water pollution and fish physiology. CRC

Press, Boca Raton, Florida.

Helder, T. 1981. Effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) on early life stages of rainbow trout (*Salmo Gairderi*, Richardson). Toxicology 19: 101-112.

Hinton, D.E., and J.A. Couch. 1984. Pathological measures of marine pollution effects. pp.7-32. IN: H.H. Harris (ed.), Concepts in Marine Pollution Measurements. Maryland Sea Grant College, University of Maryland, College Park, Maryland.

Hinton, D.E., and D.J. Lauren. 1990. Integrative histopathological approaches to detecting effects of environmental stressors on fishes. Amer. Fish. Soc. Sym. 8:51-66.

Hornshaw, T. C., R J, Aulerich, and H. E. Johnson, 1983. Feeding Great Lakes fish to mink: Effects on mink and acculmulation and elimination of PCBs by mink. J. Toxicol. Environ. Health 11:933-946.

Hammonds, J.A. and Hoffman. 1992. *Toxicity Profiles for Radionuclides*. SENES Oak Ridge, Inc. Center for Risk Analysis. Oak Ridge, TN.

Hoffman, F. O., et al. 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, Tennessee.

Hickman, G.D., Scott, E.M., and Brown, A.M. 1991. *Reservoir vital signs monitoring-1990. Fish community results*. TVA, River Basins Operation. Norris, Tennessee.

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

Holladay, S. K., et al. 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, Tennessee.

Hickman, G.D., Scott, E.M., and Brown, A.M. 1991. Reservoir vital signs monitoring-1990. Fish community results. TVA, River Basins Operation. Norris, Tennessee.

Ishida, J. 1951. Effects of 2,4-dinitrophenol on oxygen consumption during early development of the teleost, *Oryzias latipes*. Annot. Zool. Jpn. 24: 181-186.

Ishihara, K. 1956. Effect of 2,4-dinitrophenol and sodium azide on the ammonia production in the embryos of *Oryzias latipes*. J. Fac. Sci. Univ. Tokyo. Section IV: Zoology. 7: 525-533.

IAEA (International Atomic Energy Agency). 1989. *Evaluating the Reliability of Predictions Made Using Environmental Transfer Models.* IAEA Safety Series 100. Vienna, Austria.

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

IAEA (International Atomic Energy Agency). 1976. *Effects of Ionizing Radiation on Aquatic Organisms and Ecosystems.* IAEA Technical Report Series 172. Vienna, Austria.

IAEA (International Atomic Energy Agency). 1979. *Methodology for Assessing Impacts of Radioactivity on Aquatic Ecosystems.* IAEA Technical Report Series 190. Vienna, Austria.

IAEA (International Atomic Energy Agency). 1992. *Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards. Technical Report Series* . No. 332, International Atomic Energy Agency, Vienna Austria

ICRP (International Commission on Radiological Protection). 1991. *1990 Recommendations of the International Commission on Radiological Protection.* ICRP Publication No. 60. Pergamon Press, Inc. New York.

ICRP. (International Commission on Radiological Protection). 1990. *Age-dependent Doses to Members of the Public from Intake of Radionuclides: Part 1.* ICRP Publication No. 56. Pergamon Press, Inc. New York.

Iman, R. L. and Helton, J. C. 1991. *The Repeatability of Uncertainty and Sensitivity Analyses for Complex Probabilistic Risk Assessments.* Risk Analysis, 11(4):591-606.

Iman, R. L., Helton, J. C., and Campbell, J. E. 1981a. *An Approach to Sensitivity Analysis of Computer Models, Part 1.* Introduction, Input Variable Selection and Preliminary Variable Assessment. Journal of Quality Technology, 13(3):174-183.

Iman, R. L., Helton, J. C., and Campbell, J. E. 1981b. *An Approach to Sensitivity Analysis of Computer Models, Part 1.* Ranking of Input Variables, Response Surface Validation, Distribution Effect and Techniques Synopsis. Journal of Quality Technology, 13(4):232-240.

IAEA (International Atomic Energy Agency). 1989. *Evaluating the Reliability of Predictions Made Using Environmental Transfer Models.* IAEA Safety Series 100. Vienna, Austria.

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

ICRP (International Commission on Radiological Protection). 1991. *1990 Recommendations of the International Commission on Radiological Protection.* ICRP Publication No. 60. Pergamon Press, Inc. New York.

ICRP (International Commission on Radiological Protection). 1990. *Age-dependent Doses to Members of the Public from Intake of Radionuclides: Part 1.* ICRP Publication No. 56. Pergamon Press, Inc. New York.

Iman, R. L. and Helton, J. C. 1991. *The Repeatability of Uncertainty and Sensitivity Analyses for Complex Probabilistic Risk Assessments.* Risk Analysis, 11(4):591-606.

Johnson, L.L., E. Casillas, T.K. Collier, B.B. McCain, and U. Varanasi. 1988. Contaminant effects on ovarian development in english sole (*Parophrys vetulus*) from Puget Sound, Washington. Canadian Journal of Fisheries and the Aquatic Environment 45: 2133-2146.

Johnson, L.L., J.E. Stein, T.K. Collier, E. Casillas, and U. Varanasi. 1994. Indicators of reproductive development in prespawning female winter flounder (*Pheuronectes americanas*) from urban and non-urban estuaries in the northeast United States. The Science of the Total Environment 141: 241-260.

Johnson, L.L., E. Casillas, T.K. Collier, B.B. McCain, and U. Varanasi. 1988. Contaminant effects on ovarian development in English Sole (*Parophrys vetulus*) from Puget Sound, Washington. Can. J. Fish. Aquat. Sci. 45: 2133-2146.

John, M. K., H. H. Chuah, and C. J. Van Laerhoven. 1977. "Boron response and toxicity as affected by soil properties and rates of boron." *Soil Sci.* 124:34-39.

Jenkinson, J. J. 1991. *Reservoir vital signs monitoring - 1990, benthic macroinvertebrate community results.* Tennessee Valley Authority, Chattanooga, Tennessee.

Kocher, D. C. 1981. Radioactive decay data tables. Technical Information Center, U.S. Department of Energy. DOE/TIC-11026.

Kabata-Pendias, A., and H. Pendias. 1984. *Trace Elements in Soils and Plants.* CRC Press, Inc. Boca Raton, Florida.

Keen, R. and H.B. Hitchcock. 1980. Survival and longevity of the little brown bat (*Myotis lucifugus*) in southeastern Ontario. J. Mamm. 61: 1-7.

Kirubagaran, R., and K.P. Joy. 1992. Toxic effects of mercury on testicular activity in the freshwater teleost, *Clarias batrachus* (L.). Journal of Fisheries Biology 41: 305-315.

Kaplan, E. L., and Meier, P. (1958), "Nonparametric estimation from incomplete observations," Journal of the American Statistical Association, 53, 457-81.

Killough, G.G. and K.F. Eckerman. 1983. "Internal Dosimetry," pp. 7-1 to 7-98 in *Radiological Assessment.* NUREG/CR-3332. ORNL-5966. J.E. Till and H.R. Meyer, (eds). U. S. Nuclear Regulatory Commission. Washington, DC.

Kocher, D.C. 1981. *Radioactive Decay Data Tables.* DOE/TIC-11026. Technical Information Center. U.S. Department of Energy.

Kszos, L. A., A. J. Stewart, and P. A. Taylor. 1992. An evaluation of nickel toxicity to *Ceriodaphnia dubia* and *Daphnia magna* in a contaminated stream and laboratory tests. Environ. Toxicol. Chem. 11:1001-1012.

Kushlan, J.A. 1978. Feeding ecology of wading birds. Wading Birds. National Audobon Society. p. 249-297.

Ketcheson, M.R., G.P. Barron, and D.H. Cox. 1969. Relationship of maternal dietary zinc during gestation and lactation to development and zinc, iron, and copper content of the postnatal rat. J. Nutr. 98:303-311.

Kimball, G. n.d. The effects of lesser known metals and one organic to fathead minnows [*Pimephales promelas*] and *Daphnia magna.* U.S. Environmental Protection Agency, Duluth, MN.

King, K. A., T.W. Custer, and J.S. Quinn. 1991. Effects of Mercury, Selenium, and Organochlorine Contaminants on Reproduction of Forster's Terns and Black Skimmers Nesting in a Contaminated Texas Bay. Arch. Environ. Contam. Toxicol. 20:32-40.

Kinnamon, K.E. 1963. Some independent and combined effects of copper, molybdenum, and zinc on the placetal transfer of zinc-65 in the rat. J. Nutrition 81: 312-320.

Klecka, W. R. 1980. *Discriminant analysis.* Sage University Paper series Quantitative Applications in the Social Sciences, series no. 07-019. Beverly Hills and London: Sage Publications.

Kline, R.D., V.W. Hays and G.L. Cromwell. 1971. Effects of copper, molybdenum and sulfate on performance, hematology and copper stores of pigs and lambs. J. Animal Sci. 33: 771-779. (Cited in ATSDR 1990)

Kloke, A. 1979. "Content of arsenic, cadmium, chromium, fluorine, lead, mercury, and nickel in plants grown on contaminated soil." Paper presented at United Nations-ECE Symp. Effects of Air-borne Pollution on Vegetation. Warsaw. Cited in A. Kabata-Pendias and H. Pendias (eds.). 1984. T*race Elements in Soils and Plants*. CRC Press, Inc. Boca Raton, Florida.

Kirubagaran, R., and K.P. Joy. 1992. Toxic effects of mercury on testicular activity in the freshwater teleost, *Clarias batrachus* (L.). Journal of Fisheries Biology 41: 305-315.

Krarup, T. 1969. Oocyte destruction and ovarian tumorigenesis after direct application of a chemical carcinogen (9:10-dimethyl-1:2-benzanthracene) to the mouse ovary. Int. J. Cancer 4: 61-75.

Loar, J. M., S.M. Adams, H. Amano, J.B. Berry, B.G. Blaylock, H.L. Boston, M.L. Frank, C.T. Garten, M.A. Huston, B.D. Jimenez, B.L. Kimmel, J.T. Kitchings, L. Meyers-Schone, D.A. Mohrbacher, C.R. Olsen, M.G. Ryon, J.G. Smith, G.R. Southworth, A.J. Stewart, S.S. Talmage, B.T. Walton. 1992. *First Annual Report on the Biological Monitoring and Abatement Program at Oak Ridge National Laboratory.* ORNL/TM-10399. Oak Ridge National Laboratory. Oak Ridge, Tennessee.

Loar, J.M. 1981. *Ecological studies of the biota communities in the vicinity of the Oak Ridge gaseous diffusion plant.* ORNL/TM-6714. Oak Ridge National laboratory, Oak Ridge, TN.

Larsen, I. L., and N. H. Cutshall. 1981. "Direct determination of ^{7}Be in sediments." Earth Planet. Sci. Lett. 54:379-384.

Long, E. R. and L. G. Morgan. 1991. *The potential for biological effects of sediment-sorbed contaminants tested in the national status and trends program.* NOAA Technical Memo. NOS OMA 52. Seattle, WA.

Leung, T., S-Y, and R. V. Bulkley. 1979. Effects of petroleum hydrocarbons on length of incubation and hatching success in the Japanese medaka. Bull. Environ. Contam. 23: 236-243.

Lin, Y-W. P., M. S. Greeley, Jr., and Robin A. Wallace. 1989. *Fundulus heteroclitus* gonadotropin(s) 2. Year-round husbandry of animals with active pituitaries and responsive follicles. Fish Physiol. Biochem. 6:139-148.

Loar, J.M. (ed.) 1992. Sixth annual report on the ORNL Biological Monitoring and Abatement Program. Draft ORNL/TM report. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Levine, et al. 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, Tennessee.

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

Leonards, P.E.G., M. D. Smit, A. W. J. J. de Jongh, B. van Hattum. 1994. Evaluation of dose-response relationships for the effects of PCBs on the reproduction of mink (Mustela vison). Institute for Environmental Studies, Amsterdam. 50pp.

Loar, J. M. 1990. Fourth annual report on the ORNL biological monitonng and abatement program. Environmental SciencesDivision, Oak Ridge National Laboratory, ORNL/TM-Draft.

Loar, J.M., Soloman, J.A., and Cada, G.F. Technical background information for the ORNL environmental and safety report. Vol. 2. ORNL/TM-7509/V2. Oak Ridge National Laboratory, Oak Ridge, TN.

Lockhart, W.G., and D.A. Metner. 1984. Fish serum chemistry as a pathology tool, Pgs. 73-85, In: V.W. Cairns, P.V. Hodson, and J.O. Nriagu (eds.), Contaminant Effects on Fisheries. John Wiley & Sons, N.Y.

Labanauskas, C.K. 1966. "Manganese." In *Diagnostic Criteria for Plants and Soils.* Chapman, H.D. (ed.). Univ. of California, Div. Agric. Sci., Riverside. pp. 264-285.

Lam, H.F., M.W. Conner, A.E. Rogers, S. Fitzgerald, and M.O. Amdur. 1985. Functional and morphological changes in the lungs of guinea pigs exposed to freshly generated ultrafine zinc oxide. Toxicol. Appl. Pharmacol. 78:29-38. (Cited in ATSDR 1989, Eisler 1993).

Lamb, J.C., IV, R.E. Chapin, J. Teague, A.D. Lawton, and J.R. Reel. 1987. "Reproductive effects of four phthalic acid esters in the mouse." Toxicol. Appl. Pharmacol. 88: 255-269.

Laskey, J.W., G.L. Rehnberg, J.F. Hein, and S.D. Carter. 1982. Effects of chronic manganese (Mn_3O_4) exposure on selected reproductive parameters in rats. J. Toxicol. Environ. Health. 9: 677-687.

Lata, K. and B. Veer. 1990. "Phytotoxicity of Zn amended soil to *Spinacia* and *Coriandrum.*" *Acta Bot. Indica.* 18:194-198.

LaVal, R.K., R.L. Clawson, M.L. LaVal, and W. Caire. 1977. Foraging behavior and

nocturnal activity patterns of Missouri bats with emphasis on the endangered species *Myotis gresescens* and *Myotis sodalis*. J. Mammol. 58(4):592-599.

Lecyk, M. 1980. Toxicity of cupric sulfate in mice embryonic development. Zool. Pol. 28: 101-105. (Cited in U.S. AF 1990)

Lewis, M. A. and L. C. Valentine. 1981. "Acute and chronic toxicities of boric acid to *Daphnia magna* Staus." *Bull. Environ. Contam. Toxicol.* 27:309-315.

Lieb, A. J., D. D. Bills, R. O. Sinnhuber. 1974. Accumulation of dietary polychlorinated biphenyls (Aroclor 1254) by rainbow trout (*Salmo gairdneri*). J. Agr. Food Chem. 22:638-642.

Linzey, A.V. 1987. Effects of chronic polychlorinated biphenyls exposure on reproductive success of white-footed mice (*Peromyscus leucopus*). Arch. Environ. Contamin. Toxicol. 16: 455-460.

Linzon, S.N. 1978. Phytotoxically excessive levels for contaminants in soil and vegetation. Report of Ministry of the Environment. Ontario, Canada. Cited in A. Kabata-Pendias and H. Pendias (eds.). 1984. Trace Elements in Soils and Plants. CRC Press, Boca Raton, Florida.

Lloyd, T.B. 1984. Zinc compounds. In: Kirk-Othmer Encyclopedia of Chemical Technology, 3rd ed., H.F. Mark, D.F. Othmer, C.G. Overberger, G.T. Seaborg, eds. John Wiley & Sons, New York. pp. 851-863.

Lo, M.T., and E. Sandi. 1980. Selenium: occurance in foods and its toxicological significance. A review. J. Environ. Pathol. Toxicol. 4: 193-218.

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.

Long, E.R. and L.G. Morgan. 1991. "Potential for biological status effects of sediment-sorbed contaminants tested in the National Status and Trends Program." NOAA Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration. Second Printing, August 1991. Seattle, Washington.

Lunk, W.A. 1962. The Rough-Winged Swallow *Stelgidopteryx ruficollis* (Vieillot), A Study Based On Its Breeding Biology in Michigan. Oscar Root (editor), Publications of the Nuttal Ornitholgical Club, No. 4.

Mattison, D. R., K. Shiromizu, and M.S. Nightingale. 1983. Oocyte destruction by polycyclic aromatic hydrocarbons. Am. J. Ind. Med. 4: 191-202.

McCormick, J.H., K.M. Jensen, R.L. Leino, and G.N. Stokes. 1987. Fish blood osmolality, gill histology, and oocyte atresia as early warning acid stress indicators. Annls. Soc. R. Zool. Belg. 117 (suppl. 1): 309 - 319.

McMaster, M.E., G.J. Van Der Kraak, C.B. Portt, K.R. Munkittrick, P.K. Sibley, I.R. Smith, and D.G. Dixon. 1991. Changes in hepatic mixed-function oxygenase (MFO) activity, plasma steroid levels and age at maturity of a white sucker (*Catostomus commersoni*) population exposed to bleached kraft pulp mill effluent. Aquat. Toxicol. 21: 199-218.

MacDonald, D. D., B. L. Charlish, M. L. Haines, and K. Brydges. 1994. *Approach to the Assessment of Sediment Qaulity Criteria in Florida Coastal Waters. Volume 3 - Supporting Documentation: Biological Effects Database for Sediments.* Florida Department of Environmental Protection, Tallahassee, Florida.

MacDonald, J. M., J. D. Shields, and R. K. Zimmer-Faust. 1988. Acute toxicities of eleven metals to early life-history stages of the yellow crab (*Cancer anthonyi*) . Mar. Biol. 98:201-207.

Mackay, A.D., J.R. Caradus, and M.W. Pritchard. 1990. "Variation for aluminum tolerance in white clover." *Plant Soil* 123:101-105.

Maita, K., M. Hirano, K. Mitsumori, K. Takahashi, and Y. Shirasu. 1981. Subacute toxicity with zinc sulfate in mice and rats. J. Pest. Sci. 6:327-336. (Cited in ATSDR 1989; Eisler 1993).

Maleug, K.W., G.S. Schuytema, J.H. Gakstatter and D.F. Krawczyk. 1984. "Toxicity of Sediments from Three Metal-Contaminated Areas." *Environ. Toxicol. Chem.* 3: 279-291.

Masters, A. E. 1993. Reservoir monitoring - 1992. Benthic macroinvertebrate community results. Tennessee Valley Authority, Chatannoga, Tennessee.

Massie, H.R. and V.R. Aiello. 1984. Excessive intake of copper: Influence on longevity and cadmium accumulation in mice. Mech. Ageing Dev. 26: 195-203.

Mauck, W. L., P. M. Mehrle, and F. L. Mayer. 1978. Effects of the polychlorinated biphenyl Aroclor 1254 on growth, survival, and bone development in brook trout (*Salvelinus fontinalis*). J. Fish. Res. Board Can. 35:1084-1088.

Mayer, F. L., P. M. Mehrle, and H. O. Sanders. 1977. Residue dynamics and biological effects of ppolychlorinated biphenyls in aquatic organisms. Arch. Environ. Contam. Toxicol. 5:501-511.

Mayer, F. L., Jr., and H. O. Sanders. 1973. Toxicology ofphthalic acid esters in aquatic

organisms. Environmental Health Perspectives 5:153-157.

Mayo, R.H., S.M. Hauge, H.E. Parker, F.N. Andrews, and C.W. Carrick. 1956. Copper tolerance of young chickens. Poult. Sci. 35:1156-1157. (Cited in Mehring et al. 1960)

McCarty, L. S. and D. Mackay. 1993. Enhancing ecotoxicological modeling and assessment, body residues and modes of toxic action. ES&T 27:1719-1728.

McGrath, S.P. 1982. "The uptake and translocation of tri- and hexa-valent chromium and effects on the growth of oat in flowing nutrient solution." *New Phytol.* 92:381-390.

McKim, J. M., and D. A. Benoit. 1971. Effects of long-term exposures to copper on survival, growth, and reproduction of brook trout (*Salvelinus fontinalis*). J. Fish. Res. Board Can. 28:655-662.

McKim, J. M., G. F. Olson, G. W. Holcombe, and E. P. Hunt. 1976. Long-term effects of methylmercuric chloride on three generations of brook trout (*Salvelinus fontinalis*): toxicity, accumulation, distribution, and elimination. J. Fish. Res. Board Can. 33:2726-2739.

McKim, J. M., J. G. Eaton, and G. W. Holcombe. 1978. Metal toxicity to embryos and larvae of eight freshwater fish - II: Copper. Bull. Environ. Contam. Toxicol. 19:608-616.

McLane, M.A.R. and D.L. Hughes. 1980. Reproductive success of screech owls fed Aroclor 1248. Arch. Environm. Contam. Toxicol. 9: 661-665.

Mehring, A.L. Jr., J.H. Brumbaugh, A.J. Sutherland, and H.W. Titus. 1960. "The tolerance of growing chickens for dietary copper." Poult. Sci. 39: 713-719.

Mehrle, P. M., and F. L. Mayer. 1976. Di-2-ethylhexyl phthalate:residue dynamics and biological effects in rainbow trout and fathead minnows. Trace Subst. Env. Health 10:519-524.

Mendenhall, V.M., E.E. Klaas, and M.A.R. McLane. 1983. "Breeding success of barn owls (*Tyto alba*) fed low levels of DDE and dieldrin." Arch. Environ. Contam. Toxicol. 12: 235-240.

Miller, W.J., H.E. Amos, R.P. Gentry, D.M. Blackman, R.M. Durrance, C.T. Crowe, A.S. Fielding, and M.W. Neathery. 1989. Long-term feeding of high zinc sulfate diets to lactating and gestating dairy cows. J. Dairy Sci. 72:1499-1508.

Mitchell, J.L. 1961. Mink movements and population on a Montana river. J. Wildl. Manage. 25: 48-54.

Moore, J. W. 1991. Inorganic Contaminants of Surface Waters, Research and Monitoring Priorities. Springer-Verlag, NY.

Mount, D. I. 1968. Chronic toxicity of copper to fathead minnows (Pimephales promelas, Rafinesque). Water Res. 2:215-223.

Mount, D. I., and C. E. Stephan. 1969. Chronic toxicity of copper to the fathead minnow (Pimephales promelas) in soft water. J. Fish. Res. Board Can. 26:2449-2457.

Muramoto, S., H. Nishizaki, and I. Aoyama. 1990. "The critical levels and the maximum metal uptake for wheat and rice plants when applying metal oxides to soil." *J. Environ. Sci. Health, Part B* 25(2):273-80.

Marty, G. D., J. M. Núñez, D. J. Lauren, and D. E. Hinton. 1990. Age-dependent changes in toxicity of N-nitros compounds to Japanese medaka (*Oryzias latipes*) embryos. Aquat. Toxicol. 17: 45-62.

1983 Mercury Task Force. 1983. Mercury at Y-12: A Study of Mercury Use at the Y-12 Plant, Accountability, and Impacts on Y-12 Workers and the Environment - 1950 to 1983. Y/EX-34. Union Carbide Corporation- Nuclear Division, Oak Ridge Y-12 Plant, Oak Ridge, TN. 418 pages.

Meinert, D.L. 1991. *Reservoir Vital Signs Monitoring - 1990: Physical and Chemical Characteristics of Water and Sediment.* Tennessee Valley Authority, Water resources, Chattannoga, TN. TVA/WR/WQ-91/10.

Michibata, H. 1981a. Effect of water hardness on the toxicity of cadmium to the egg of teleost, *Oryzias latipes*. Bull. Environ. Contam. Toxicol. 27: 187-192.

Michibata, H. 1981b. Uptake and distribution of cadmium in the egg of the teleost, *Oryzias latipes*. J. Fish Biol. 19: 691-696.

Michibata, H., Y. Nojima, and M. K. Kojima. 1987. Stage sensitivity of eggs of the teleost *Oryzias latipes* to cadmium exposure. Environ. Res. 42: 321-327.

Michibata, H., S. Sahara, and M. K. Kojima. 1986. Effects of calcium and magnesium ions on the toxicity of cadmium to the egg of the teleost, *Oryzias latipes*. Environ. Res. 40:110-114.

Melancon, M.J., and J.J. Leach. Dose-effect relationship for induction of hepatic monooxygenase activity in rainbow trout and carp by arochlor 1254. Aquatic Toxicol. 4:51-61.

Monosson, E., W. J. Fleming, C. V. Sullivan. 1994. Effects of the planer PCB 3,3',4,4'-tetrachlorobiphenyl (TCB) on ovarian development, plasma levels of sex steroid hormones and vitellogenin, and progeny survival in the white perch (*Morone americana*). Aquatic Toxicology 29: 1-19.

McMaster, C.B. Portt, G.J. Van Der Kraak, I.R. Smith, and D.G. Dixon. 1992. Changes in maturity, plasma sex steroid levels, hepatic mixed-function oxygenase activity, and the presence of external lesions in lake whitefish (*Coregonus clupeaformis*) exposed to bleached kraft mill effluent. Can. J. Fish. Aquat. Sci. 49: 1560-1569.

Mancuso, T.F. 1975. *International Conference on Heavy Metals in the Environment.* Toronto, Ontario, Canada.

Mancuso, T.F. and W.C. hueper. 1951. "Occupational cancer and other health hazards on a chromate plant: a medical appraisal." Ind. Med. Surg. 20:358–363.

Masters, A. E. 1992. *Reservoir vital signs monitoring - 1991, benthic macroinvertebrate community results.* Tennessee Valley Authority, Chattanooga, Tennessee.

Masters, A. E. 1993. *Reservoir vital signs monitoring - 1992, benthic macroinvertebrate community results.* Tennessee Valley Authority, Chattanooga, Tennessee.

McNutt, Charles and F. W. Fisher. 1960. *Archaeological Investigations in the Upper Melton Hill Reservoir, Anderson County, Tennessee.* The University of Tennessee, Department of Anthropology, Knoxville.

McNutt, Charles and J. Bennett Graham. 1961. *Archaeological Investigations in the Lower Melton Hill Reservoir, Anderson, Knox, Loudon, and Roane Counties, Tennessee.* The University of Tennessee, Department of Anthropology, Knoxville.

Meinert, D.L. and D.L. Dycus, 1993. *Reservoir Monitoring - 1992: Summary of Vital Signs and Use Suitability Monitoring on Tennessee Valley Reservoirs.* Tennessee Valley Authority, Water Resources. Chattanooga Tennessee.

Meinert, D.L. and J.P. Fehring 1992. *Reservoir Vital Signs Monitoring - 1991: Physical and Chemical Characteristics of Water and Sediment.* Tennessee Valley Authority, Water Resources, Chattanooga, Tennessee. TVA/WR-91/1.

Moore, G.K and L.E. Toran. 1992. *Supplement to a Hydrologic Framework for the Oak Ridge Reservation.* ORNL/TM-12191. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Morton, R.J. 1965. (Ed.) *Status Report No. 5 on Clinch River Study. Clinch River Study Steering Committee,* ORNL-3721. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Moses, J. and D. C. Wade. 1991. *Acute toxicity screening of reservoir water and sediment.* Tennessee Valley Authority, Muscle Shoals, AL.

Moses, J. and D. C. Wade. 1992. *Acute toxicity screening of reservoir water and sedimentusing rotifers (ROTOTOX®) and light emitting bacteria (MICROTOX®), reservoir monitoring, summer 1991.* Tennessee Valley Authority, Muscle Shoals, Alabama.

Moses, J., T. C. Sesler, D. C. Wade, and D. L. Meinert. 1993. *Reservoir monitoring - 1992; Acute toxicity screening of reservoir water and sediment using rotifers (ROTOTOX®) and light emitting bacteria (MICROTOX®).* Tennessee Valley Authority, Muscle Shoals, AL.

NCRP (National Council on Radiation Protection and Measurements). 1991. *Effects of Ionizing Radiation on Aquatic Organisms.* Report Number 109.

NCRP (National Council on Radiation Protection and Measurements). 1984. *Radiological Assessments: Predicting the Transport, Bioaccumulation, and Uptake by Men of Radionuclides Released to the Environment.* Report Number 76.

Neff, J. M., B. W. Cornaby, R. M. Vaga, T. C. Gulbransen, J. A. Scanlon and D. J. Bean. 1988. "An evaluation of the screening level concentration approach for validation of sediment quality criteria for freshwater and saltwater ecosystems" in *Aquatic Toxicology and Hazard Assessment: 10th Volume,* ASTM STP 971. Adams, W. J., G. A. Chapman, and W. G. Landis (eds). American Society for Testing and Materials, Philadelphia. pp. 115-127.

Nelson, D.J. 1967. "Ecological Behavior of radionuclides in the Clinch and Tennessee Rivers." pp.169-187 In *Reservoir Fisheries Resources Symposium.* Southern Division, American Fisheries Society, Athens Georgia.

Nelson, D.J. 1962. *Strontium, strontium-90, and calcium analyses of Clinch and Tennessee River clams.* ORNL/TM-270. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

NRC (National Research Council). 1983. *Risk assessment in the federal government: Managing the process.* National Academy Press, Washington, D.C.

Nasci, C., G. Campesan, V.U. Fossato, L. Tallandini, and M. Turchetto, 1991. Induction of cytochrome P-450 and mixed function oxygenase activity by low concentrations of polychlorinated biphenyls in marine fish Zosterisessor ophiocephalus (Pall.). Aquatic Toxicol. 19:281-290.

Nagy, K.A. 1987. Field metabolic rate and food requirement scaling in mammals and birds. Ecol. Monogr. 57: 111-128.

National Academy of Sciences (NAS). 1979. Zinc. U.S. National Acad. Sci., National Res. Counc., Subcommittee on Zinc. Univ. Park Press, Baltimore, Md. 471pp.

National Academy of Sciences (NAS). 1980. Mineral Tolerance of Domestic Animals. National Academy Press, Washington, DC.

NCRP (National Council on Radiation Protection and Measurements). 1991. *Effects of Ionizing Radiation on Aquatic Organisms.* NCRP Report No 109, National Council on Radiation Protection and Measurements, Bethesda, Maryland.

National Geographic Society. Field Guide to the Birds of North America. 2nd Edition. 1987.

Neary, D.G., G. Schneider, and D.P. White. 1975. "Boron toxicity in red pine following municipal waste water irrigation." *Soil Sci. Soc. Am. Proc.* 39:981-982.

Nebeker, A. V., F. A. Puglisi, and D. L. DeFoe. 1974. Effect of polychlorinated biphenyl compounds on survival and reproduction of the fathead minnow and flagfish. Trans. Am. Fish. Soc. 3:562-568.

Nebeker, A. V., C. Savonen, R. J. Baker, and J. K. McCrady. 1984. Effects of copper, nickel and zinc on the life cycle of the caddisfly *Clistoronia magnifica* (Limnephilidae). Envrion. Toxicol. Chem. 3:645-649.

Niimi, A. J., and B. G. Oliver. 1983. Biological half-lives of polychlorinated biphenyl (PCB) congeners in whole fish and muscle of rainbow trout (*Salmo gasirdneri*). Can. J. Fish. Aquat. Sci. 40:1388-1394.

NRCC. 1978. Effects of arsenic in the Canadian environment. National Res. Counc. Canada Publ. No. NRCC 15391. 349pp.

NCRP (National Council on Radiation Protection and Measurements). 1984. *Radiological Assessments: Predicting the Transport, Bioaccumulation, and Uptake b Men of Radionuclides Released to the Environment.* National Council on Radiation Protection and Measurement, Bethesda, MD.

National Council on Radiation Protection and Measurements (NCRP). 1977. *Environmental Radiation Measurements.* NCRP Report No. 50. Washington, DC.

Neff, J. M. 1978. Polycyclic Aromatic Hydrocarbons. Applied Science Publisher, Ltd., London, 262.

Nutrient Requirements of Mink and Foxes, No. 7. National Academy of Sciences, Washington, D.C. 72 pp.

Olsen, C.R., et. al. 1992. *Transport and Accumulation of Cesium-137 and Mercury in the Clinch River and Watts Bar Reservoir System.* ORNL/ER-7, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Oakes, T.W, 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, Tennessee.

Ohnesorge, W.F. 1986. *Historical Releases of Radioactivity to the Environment from ORNL.* ORNL/M-135. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Olsen, C.R., I.L. Larsen, P.D. Lowry, C.R. Moriones, C.J. Ford, K.C. Dearstone, R.R. Turner, B.L. Kimmel, and C.C. Brandt, 1992. *Transport and Accumulation of Cesium-137 and Mercury in the Clinch River and Watts Bar Reservoir System.* ORNL/ER-7, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Opresko, D. M., B. E. Sample, and G. W. Suter II. 1994. *Toxicological benchmarks for wildlife.* ORNL/ER/TM-86.

ORNL (Oak Ridge National Laboratory). 1961. *Applied Health Physics Annual Report for 1960.* ORNL-3159. Health Physics Division, Oak Ridge, Tennessee.

ORNL (Oak Ridge National Laboratory). 1962. *Applied Health Physics Annual Report for 1961.* ORNL-3284. Health Physics Division, Oak Ridge, Tennessee.

ORNL (Oak Ridge National Laboratory). 1963. *Applied Health Physics Annual Report for 1962.* ORNL-3490. Health Physics Division, Oak Ridge, Tennessee.

ORNL (Oak Ridge National Laboratory). 1964. *Applied Health Physics Annual Report for 1963.* ORNL-3665. Health Physics Division, Oak Ridge, Tennessee.

ORNL (Oak Ridge National Laboratory). 1965. *Applied Health Physics Annual Report for 1964.* ORNL-3820. Health Physics Division, Oak Ridge, Tennessee.

ORNL (Oak Ridge National Laboratory). 1966. *Applied Health Physics and Safety Annual Report for 1965.* ORNL-3969. Health Physics Division, Oak Ridge, Tennessee.

ORNL (Oak Ridge National Laboratory). 1967. *Applied Health Physics and Safety Annual Report for 1966.* ORNL-4146. Health Physics Division, Oak Ridge, Tennessee.

Oh, S.H., H. Nakaue, J.T. Deagen, P.D. Whanger, and G.H. Arscott. 1979. Accumulation and depletion of zinc in chick tissue metallothioneins. J. Nutr. 109:1720-1729. (Cited in Eisler 1993)

Ohlendorf, H.M., R.L. Hothem, C.M. Bunck, T.W. Aldrich, and J.F. Moore. 1986. Relationships between selenium concentrations and avian reproduction. Trans. N. Am. Wildl. Nat. Res. Conf. 51: 330-342.

Opresko, D.M., B.E. Sample, and G.W. Suter, II. 1994. Toxicological benchmarks for wildlife: 1994 Revision. ES/ER/TM-6/R1. Oak Ridge National Laboratory, Oak Ridge, TN.

Palafox, A.L., and E. Ho-A. 1980. Effect of zinc toxicity in laying white leghorn pullets and hens. Poultry Sci. 59:2024-2028. (Cited in Eisler 1993)

Pattee, O.H. 1984. Eggshell thickness and reproduction in American kestrels exposed to chronic dietary lead. Arch Environ. Contam. Toxicol. 13: 29-34.

Peakall, D.B. 1974. "Effects of di-N-buylphthalate and di-2-ethylhexylphthalate on the eggs of ring doves." Bull. Environ. Contam. Toxicol. 12: 698-702.

Persaud, D., R. Jaagumagi, and A. Hayton. October 1990. *The Provincial Sediment Quality Guidelines*. Ontario Ministry of the Environment.

Peterson, P.J., L.M. Benson, and R. Zieve. 1981. "Metalloids." In N.W. Lepp (ed) *Effects of Heavy Metal Pollution on Plants, Vol. 1. Effects of Trace Metals on Plant Function.* pp. 279-342.

Peterson, P.J., and C.A. Girling. 1981. "Other trace metals." In N.W. Lepp (ed) *Effect of Heavy Metal Pollution on Plants. Vol 1. Effects of Trace Metals on Plant Function.* 279-342. Applied Science Publishers, New Jersey. pp. 213-278.

Platonow, N.S. and L.H. Karstad. 1973. Dietary effects of polychlorinated biphenyls on mink. Can. J. Comp. Med. 37:391-400.

Platonow, N.S. and B.S. Reinhart. 1973. The effects of polychlorinated biphenyls (Aroclor 1254) on chicken egg production, fertility and hatchability. Can. J. Comp. Med. 37: 341-346.

Prasad, A.S. 1979. Clinical, biochemical, and pharmacological role of zinc. Ann. Rev. Pharmacol. Toxicol. 20:393-426. (Cited in Eisler 1993)

Prasad, P.V.D. 1984. Effect of magnesium, strontium, and barium on the calcification of the freshwater green alga *Goeotaenium*. PHYKOS 23:202-206.

Pratt, P.F. 1966. "Vanadium." In *Diagnostic Criteria for Plants and Soils*. Chapman, H.D. (ed.). Univ. of California, Div. Agric. Sci., Riverside. pp. 480-483.

Pullar, E.M. 1940. The toxicity of various copper compounds and mixtures for domesticated birds. 2. Australian Vet. J. 16:203-213.

Pearson, S.M. and K.A. Rose. In press. *Estimating the magnitude and distribution of a contaminant in the bottom sediments of a large reservoir: effects of sampling design and spatial heperogeneity.* Submitted to Water Resources Research.

Persaud, D., R. Jaagumagi and A. Hayton. October 1990. *The Provincial Sediment Quality Guidelines.* Ontario Ministry of the Environment. 21 pages.

Prothro, M. G. 1993. Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria. Memorandum to Water Management Division Directors and Environmental Services Division Directors, Regions I-X, October 1, 1993.

Payne J. F. and W. R. Penrose. 1975. Induction of aryl hydrocarbon benzo(a)pyrene hydroxylase in fish by petroleum. Bull. Environ. Contam. Toxicol. 14: 112-116.

Pullin, B. P. 1990. Size and trends of wading bird populations in Tennessee during 1977-1988. Migrant 61:95-104.

Peters, N., A. Kohler, and H. Kranz. 1987. Liver pathology in fishes from the lower Elbe as a consequence of pollution. Dis. Aquat. Org. 2:87-97.

Rhodes, L., E. Casillas, B. McKnight, W. Gronlund, M. Myers, O. Olson, and B. McClain. 1985. Interactive effects of cadmium, polychlorinated biphenyls, and fuel oil on experimentally exposed english sole (Parophrys vetulus). Can. J. fish. Aquat. Sci. 42:1870-1880.

Ruby, S.M., D.R. Idler, and Y.P. So. 1993. Plasma vitellogenin, 17β-estradiol, T_3 and T_4 levels in sexually maturing rainbow trout *Oncorhynchus mykiss* following sublethal HCN exposure. Aquat. Toxicol. 26: 91-102.

Scientifur, 1987. Scientifur Index. I. G. Joergensen (ed.). Scientifur, Hillerod Denmark, 196 pp.

Sundqvist, C., 1989. Mink Encyclopedia Release 1.0 Parts I and II. Mink Encyclopedia Project, Depart. Biology, Abo Akademi, Porthansgatan 3, Turku, Finland. Part I, 391 pp; Part II, 387 pp.

Suter, G. W. 1990. Screening level risk assessment for off-site ecological effects in surface waters downstream from the U. S. Department of Energy Oak Ridge

Reservation. Enviromental Sciences Division Publication 3483. ORNL/ER-8.

Saidapur, S. K. 1978. Follicular atresia in the ovaries of nonmammalian vertebrates. Intl. Rev. Cytology 54: 225 -244.

Sakaizumi, M. 1980. Effect of inorganic salts on mercury-compound toxicity to the embryo of the medaka, *Oryzias latipes*. J. Fac. Sci. Univ. Tokyo. 14: 369-384.

Schreiweis, D. O. and G. J. Murray. 1976. Cardiovascular malformations in *Oryzias latipes* embryos treated with 2,4,5-trichlorophenoxyacetic acid (2,4,5-T). Teratology. 14: 287-290.

Shi, M. and E. M. Faustman. 1989. Development and characterization of a morphological scoring system for medaka (*Oryzias latipes*) embryo culture. Aquat. Toxicol. 15: 127-140.

Shul'man, G.E. 1974. Life cycles of fish. Physiology and biochemistry. John Wiley and Sons, New York. 258 pp.

Singh, S., and T.P. Singh. 1987. Evaluation of toxicity limit and sex hormone production in response to cythion and BHC in the vitellogenic catfish *Clarias batrachus*. Environ. Res. 42: 482-488.

Sivarajah, K., C.S. Franklin and P. Williams. 1978. The effects of polychlorinated biphenyls on plasma steroid levels and hepatic microsomal enzymes in fish. J. Fish Biol. 13: 401-409.

Slooff, W., and J. H. Canton. 1983. Comparison of the susceptibility of 11 freshwater species to 8 chemical compounds. II: (Semi)chronic toxicity tests. Aquat. Toxicol. 4:271-282.

Slooff, W., J. H. Canton, and J. L. M. Hermens. 1983. Comparison of the susceptibility of 22 freshwater species to 15 chemical compounds. I: (sub) acute toxicity tests. Aquat. Toxicol. 4:113-128.

Solomon, H. M. 1977a. Teratogenic effects of carbaryl, malathion and parathion on developing medaka eggs. Abstract Only. Bull. N.J. Acad. Sci. 22(2):46-47.

Solomon, H. M. 1977b. Teratogenic effects of carbaryl, malathion and parathion on developing eggs of medaka (*Oryzias latipes*). Abstract only. Animals as Monitors of Environmental Pollutants: Symposium on Pathobiology of Environmental Pollutants. University of Conn., 1977. Natl. Academy of Sci., Washington, D.C. 393 pp.

Solomon, F. P. and E. M. Faustman. 1987. Developmental toxicity of four model

alkylating agents on Japanese medaka fish (*Oryzias latipes*) embryos. Environ. Toxicol. and Chem. 6: 747-754.

Stoss, F. W. and T. A. Haines. 1979. The effects of toluene on embryos and fry of the Japanese medaka *Oryzias latipes* with a proposal for rapid determination of maximum acceptable concentrations. Environ. Pollut. 20: 139-148.

Sokal, R. R., and F. J. Rohlf. 1981. Biometry, The Principles and Practice of Statistics in Biological Research. W. H. Freeman and Company, San Francisco, California. 859 pp.

Taylor, J.K. 1987. Quality assurance of chemical measurements. Lewis Publ., Inc., Chelsea, MI. 328 pp.

Takizawa, K., H. Yagi, D.M. Jerina and D.R. Mattison. 1984. Murine starin differences in ovotoxicity following intraovarian injection with benzo(α)pyrene, (+)-(7R, 8S)-oxide,(-)-(7R,8R)-dihydodiol, or (+)-(7R,8S)-diol-(9S,10R)-epoxide-2. Canc. Res. 44: 2571-2576.

Thomas, P. 1990. Teleost model for studying the effects of chemicals on female reproductive endocrine function. J. Exp. Zool. Suppl. 4: 126-128.

Truscott, B., J.M. Walsh, M.P. Barton, J.F. Payne, and D.R. Idler. 1983. Effect of acute exposure to crude petroleum on some reproductive hormones in salmon and flounder. Comp. Biochem. Physiol. 75C: 121-130.

TVA (Tennessee Valley Authority). 1990. Tennessee River and reservoir system operation and planning review, Final Environmental Impact Statement, TVA/RDG/EQS-91/1.

Van Winkle, W., R. W. Counts, J. G. Dorsey, J. W. Elwood, V. W. Lowe, Jr., R. McElhaney, S. D. Schlotzhauer, F. G. Taylor, Jr., and R. R. Turner. 1984. Mercury Contamination in East Fork Poplar Creek and Bear Creek. ORNL/TM-8894, Environmental Sciences Division Publication No. 2051. 72 pages.

Waterman, A. J. 1939. Effects of 2,4-dinitrophenol on the early development of the teleost, *Oryzias latipes*. Biol. Bull. 76: 162-170.

Yamagami, K. 1972. Isolation of a choriolytic enzyme (hatching enzyme) of the teleost, *Oryzias latipes*. Dev. Biol. 29: 343-348.

Zar, J. H. 1984. Biostatistical Analysis. Prentice-Hall, Inc., Englewood Cliffs, New Jersey. 718 pp.

Spitsbergen, J. M., M.K. Walker, J.R. Olson, and R.E. Peterson. 1991. Pathological alterations in early life stages of lake trout, *Salvelinus namaycush*, exposed to

2,3,7,8-tetraclorodibenzo-*p*-dioxin as fertilized eggs. Aquatic Toxicology 19: 41-72.

Scott, E.M. 1994. Clinch River remedial investigation. Task 9-Benthic macroinvertebrates. Tennessee Valley Authority, Resource Group.

Zar, 1984. Biostatistical analysis. Prentice-Hall, Inc., Englewood Cliffs, New Jersey.

USACE. 1993. HEC-6: Scour and Deposition in Rivers and Reservoirs, User's manual. CPD-6, Hydrologic Engineering Center. US Army Corps of Engineers, Davis, Ca.

SAS users guide:statistics. Version 5. 1985. SAS Institute Inc., Cary, N.C. 956 pp.

Tietz, N.W. 1986. Textbook of Clinical Chemistry. W.b. Saunders, Co., Philadelphia, Pennsylvania. 1919 pp.

Pgs. 17-57, In: J.F. McCarthy and L.R. Shugart, (eds.), Biomarkers of environmental contamination. Lewis Pub., Boca Raton, FL.

Schmoyer, R. L., Beauchamp, J. J., Brandt, C. C., and Hoffman, F. O. (1995),Difficulties with the lognormal model in mean estimation and testing," Journal of Environmental and Ecological Statistics, (submitted).

Travis, C. C. and A. D. Arms. 1988. *Bioconcentration of Organics in Beef, Milk, and Vegetation.* Environmental Science and Technology, Vol. 22(3):271-274.

Turner, J.E. 1986. *Atoms, Radiation, and Radiation Protection.* Pergamon Press, Inc. New York, NY.

TWRA (Tennessee Wildlife Resources Agency) 1994. *Wildlife Research Report: Tennessee Waterfowl Report 1992-93*, Tennessee Wildlife Resources Agency, Technical Report No. 93-14, March 1994.

USDA (U.S. Department of Agriculture). 1970. *Soil Conservation Service and Agricultural Research, et al. Ohio Irrigation Guide.*

Pounds, R.P., P.D. Parr, and M.G. Ryon. 1993. Resource Management Plan For the Oak Ridge Reservation. Volume 30: Oak Ridge National Environmental Research Park Natural Areas and Reference Areas- Oak Ridge Reservation Environmentally Sensitive Sites Containing Special Plants, Animals, and Communities. ORNL/NERP-8, Environmental Sciences Division Publication No. 3995.

U.S. Army Corp of Engineers 1987. Wetlands Delineation Manual. Technical Report Y-87-1. Waterways Experiment Station, Vicksburg, M.S.

Solomon, D.K., G.K. Moore, L.E. Toran, R.B. Dreier, and W. M. McMaster, 1992. *Status Report: A Hydrologic Framework for the Oak Ridge Reservation.* ORNL/TM-12026. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Moore, G.K and L.E. Toran. 1992. *Supplement to a Hydrologic Framework for the Oak Ridge Reservation.* ORNL/TM-12191. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Fehring, J.P and D.L. Meinert 1993. *Reservoir Vital Signs Monitoring - 1992: Physical and Chemical Characteristics of Water and Sediment.* Tennessee Valley Authority, Water Resources, Chattannoga, Tennessee.

Trimble, S.W. and W.P. Carey 1984. *Sediment Characteristics of Tennessee Streams and Reservoirs.* U.S. Geological Survey Open File Report 84-749.

TVA (Tennessee Valley Authority) 1990. *Tennessee River and Reservoir System Operation and Planning Review.* Final Environmental Impact Statement. Tennessee Valley Authority. TVA/RDG/EQS-91/1.

Struxness, E.G., P.H.Carrigan, Jr., M.A. Churchill, K.E. Cowser, R.J. Morton, D.J. Nelson, and F.L. Parker. 1967. *Comprehensive Report of the Clinch River Study.* ORNL-4035. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Scott, E.M. 1994. *Clinch River Remedial Investigation: Task 9-Benthic macroinvertebrates.* Unpublished Report, Tennessee Valley Authority, Resource Group.

Scott, E.M. 1994. Clinch River remedial investigation. Task 9-Benthic macroinvertebrates. Tennessee Valley Authority, Resource Group.

Wlostowski, T., W. Chetnicki, W. Gierlachowska-Baldyga, and B. Chycak. 1988. Zinc, iron, copper, manganese, calcium and magnesium supply status of free-living bank voles. Acta Theriol. 33:555-573. (Cited in Eisler 1993)

Radian Corporation. 1993. *Initial Screening of Alternatives for the Clinch River and Watts Bar Reservoir, Oak Ridge, Tennessee.* Prepared for U.S. Department of Energy, Oak Ridge, Tennessee (93-225-230-01).

Risk Assessment Forum. 1992. *Framework for Ecological Risk Assessment.* EPA/630/R-92/001. U.S. Environmental Protection Agency, Washington, D.C.

Roels, H., R. Lauwerys, J.P. Buchet. 1987. "Epidemiological survey among workers exposed to manganese: effects on lung, central nervous system, and some biological indices." Am. J. Ind. Med. 11:307–328.

Rodgers, John. 1953. Geologic map of east Tennessee with explanatory text. Tennessee Department of Conservation, Division of Geology Bulletin SP, Part II. Nashville, Tennessee.

SAIC (Science Applications International Corporation). 1991.*Laboratory Data Validation Guidelines for Evaluating Radionuclide Analyses,* Revision 3. Oak Ridge, Tennessee.

Scott, E. M. Jr. 1992. *TVA Reservoir monitoring program fish community results - 1991.* Tennessee Valley Authority, Norris, Tennessee.

Simbeck, D. and J. Moses. 1994. *Acute Toxicity Screening of Reservoir Water and Sediment Using Daphnids (Ceriodaphnia Dubia) and Rotifers (Rotox®).* Reservoir Vital Signs Monitoring, Summer 1994. TVA Resource Group, Water Managemnt, Muscle Shoals, Alabama.

Soballe, D.M., B.L. Kimmel, R.H. Kennedy, and R.F. Gangrish. 1992. Reservoirs. In: *Biodiversity of the Southeastern United States: Aquatic Communities.* John Wiley and Sons, Inc.

Solomon, D.K., G.K. Moore, L.E. Toran, R.B. Dreier, and W. M. McMaster, 1992. *Status Report: A Hydrologic Framework for the Oak Ridge Reservation.* ORNL/TM-12026. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Spalding, , B.P. and W.J. Boegley, Jr. 1985. *ORNL Radioactive Liquid Waste Disposal Pits and Trenches: History, Status, and Closure Characterization Needs.* ORNL/CF-85/70. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Struxness, E.G., P.H. Carrigan, Jr., M.A. Churchill, K.E. Cowser, R.J. Morton, D.J. Nelson, and F.L. Parker 1967. *Comprehensive Report of the Clinch River Study.* ORNL-4035. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Suter, G. W. II. 1991. *Screening level risk assessment for off-site ecological effects in surface waters downstream from the U.S. Department of Energy Oak Ridge Reservation.* ORNL/ER-8. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Suter, G.W. II, M.E. Will, and C. Evans. 1994b. *Toxicological benchmarks for screening potential contaminants of concern for effects on terrestrial plants.* ES/ER/TM-85.

TDEC (Tennessee Department of Conservation). 1992. Division of Water Pollution Fish Control. Consumption Advisories in Tennessee. Nashville, Tennessee.

Tennessee Wildlife Resources Commission. 1991. *Tennessee Fishing Regulations.* (brochure)

Travis, C.C. and A. D. Arms. 1988. Bioconcentration of Organics in Beef, Milk, and Vegetation, Environmental Science and Technology, Vol 22(3):271-274.

Trimble, S.W. and W.P. Carey 1984. Sediment Characteristics of Tennessee Streams and Reservoirs. U.S. Geological Survey Open File Report 84-749.

Turner, J. E. 1986. *Atoms, Radiation, and Radiation Protection.* Pergamon Press, Inc. New York.

Turner, R.R., C.R. Olsen, and W.J. Wilcox, Jr. 1984. "Environmental Fate of Hg and ^{137}Cs Discharged from Oak Ridge Facilities." In Hemphill, D.D., Ed. *Trace Substances in Environmental Health-XVIII, 1984,* A Symposium. Univ. of Missouri, Columbia.

TVA (Tennessee Valley Authority). 1970. *Drainage Areas for Streams in Tennessee River Basin.* Report No. 0-5829-R-2. Knoxville, Tennessee. pp 116.

TVA (Tennessee Valley Authority). 1980. *Behavior of coal ash particles in water: trace metal leaching and ash settling.* Report EDT-110. Energy Demonstration and Technology, Chattanooga, Tennessee.

TVA (Tennessee Valley Authority). 1985a. *Instream Contaminant Study - Task 2, Sediment Characterization, Volume I.* Office of Natural Resources and Economic Development. Knoxville, Tennessee.

TVA (Tennessee Valley Authority). 1985b. *Instream Contaminant Study - Task 2, Sediment Characterization, Volume II- Appendices.* Office of Natural Resources and Economic Development. Knoxville, Tennessee.

TVA (Tennessee Valley Authority). 1985c. *Quality Assurance Program for the Oak Ridge Instream Contaminant Study.* Division of Services and Field Operations, Laboratory Branch. Knoxville, Tennessee.

TVA (Tennessee Valley Authority). 1985d. *Instream Contaminant Study - Task 4, Fish Sampling and Analysis.* Office of Natural Resources and Economic Development.

TVA (Tennessee Valley Authority). 1986a. *Heavy Metals and PCB Concentrations in Sediments from Selected TVA Reservoirs - 1982.* Office of Natural Resources and Economic Development. Chattanooga, Tennessee. TVA /ONRED/AW 86/35.

TVA (Tennessee Valley Authority). 1986b. *Instream Contaminant Study- Task 5, Summary Report.* Office of Natural Resources and Economic Development. Knoxville, Tennessee.

TVA (Tennessee Valley Authority). 1987. *Watts Bar Reservoir Land Management Plan*

(Final Draft).

TVA (Tennessee Valley Authority). 1990a. *Tennessee River and Reservoir System Operation and Planning Reveiw. Final Environmental Impact Statement.* TVA/RDG/EQS-91/1.

TVA (Tennessee Valley Authority). 1990b. Tennessee River Navigation Charts. Knoxville, Tennessee.

TVA (Tennessee Valley Authority). 1991a. *Results of Sediment and Water Sampling for Inorganic, Organic, and Radionuclide Analysis at Recreation Areas and Water Intakes - Norris, Melton Hill, and Watts Bar Lakes - Data Report.* Water Resources. Chattanooga, Tennessee.

TVA (Tennessee Valley Authority). 1991b. *TVA Environmental Chemistry Quality Assurance Program.* Water Management Branch, Office of Natural Resources, Chattanooga, Tennessee.

TVA (Tennessee Valley Authority). 1992. *Annual Radiological Environmental Monitoring Report, Watts bar Nuclear Plant, 1991.* Nuclear Operations/Technical Programs. Muscle Shoals, Alabama.

TVA (Tennessee Valley Authority). 1993. *Supplemental Environmental Impact Statement—Aquatic Plant Management Program.* TVA/WM-93/8. Vols I & II. Water Resources/Aquatic Biology Department.

USACE. 1990. *HEC-6: Scour and Deposition in Rivers and Reservoirs, User's Manual.* CPD-6, Hydrological Engineering Center, United States Army Corps of Engineers, Davis, California.

USDA (U.S. Department of Agriculture). 1970. Soil Conservation Service and Agricultural Research, et al. Ohio Irrigation Guide.

U.S. Department of Commerce. 1990. *1990 Census of the Population and Housing.* Bureau of the Census.

Warren, R.B. 1974. *Soil Survey of Meigs County, Tennessee.* U. S. Department of Agriculture Soil Conservation Service in cooperation with the Tennessee Agricultural Experiment Station.

Webb, William S. 1938. *An Archaeological Survey of the Norris Basin in Eastern Tennessee.* Bulletin 118. Bureau of American Ethnology, Washington, D.C.

Wilcox, W.J. Jr. 1983. *Mercury at Y-12: A Summary of the 1983 UCC-ND Task Force Study.* Report Y/EX-23.

Williams, D. L. and D. L. Dycus. 1993. *Reservoir Monitoring - 1992: Fish Tissue Studies in the Tennessee Valley in 1991 and 1992.* Tennessee Valley Authority, Water Resources, Chattanooga, Tennessee.

Wilson, W. K. 1991. *Hydroacoustic estimates of fish abundance.* Tennessee Valley Authority, Chattanooga, Tennessee.

Will, M.E. and G.W. Suter II. 1994. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on terrestrial Plants: 1994 Revision. ES/ER/TM-85/R1.

Wren, C.D., D.B. Hunter, J.F. Leatherland and P.M. Stokes. 1987. The effects of polychlorinated biphenyls and methylmercury, singly and in combination on mink. II: Reproduction and Kit Development. Arch. Environ. Contam. Toxicol. 16: 449-454.

Woolson, E.A., J.H. Axley, and P.C. Kearney. 1971. "Correlation between available soil arsenic, estimated by six methods and response of corn (*Zea mays* L.)." *Soil Sci. Soc. Am. Proc.* 35:101-105.

Rabinowitz, A.R. 1978. Habitat Use and Prey Selection by the Endangered Gray Bat, *Myotis Gresescens* in East Tennessee. M.S. Thesis. The University of Tennessee, Knoxville.

Rana, S. V.S. and A. Kumar. 1978. Simultaneous effects of dietary molybdenum and copper on the accumulation of copper in the liver and kidney of copper poisoned rats. A histochemical study. Ind. Health 18: 9-17. (Cited in U.S. AF 1990)

Reece, R.L., D.B. Dickson, and P.J. Bunowes. 1986. Zinc toxicity (new wire disease) in aviary birds. Australian Vet. J. 63:199. (Cited in Eisler 1993)

Roberson, R.H., and P.J. Schaible. 1960. The tolerance of growing chicks for high levels of different forms of zinc. Poult. Sci. 39: 893-896.

Sample, and G.W. Suter, II. 1994. Toxicological benchmarks for wildlife. ES/ER/TM-6/R1. Oak Ridge National Laboratory, Oak Ridge, TN.

Sanders, O.T., R.L. Zepp, and R.L. Kirkpatrick. 1974. Effect of PCB ingestion on sleeping times, organ weights, food consumption, serum corticosterone and survival of albino mice. Bull. Environ. Contam. Toxicol. 12(4):394-399.

Schlicker, S.A., and D.H. Cox. 1967. Maternal dietary Zn in excess, fetal development, and Fe and Cu metabolism. (Abstract). Fed. Amer. Proc. 26:520. (Cited in ATSDR 1989).

Schlicker, S.A., and D.H. Cox. 1968. Maternal dietary zinc and development; zinc, iron, and copper content of the rat fetus. J. Nutr. 95:287-294.

Schroeder, H. A., J. J. Balassa, and W. H. Vinton, Jr. 1965. Chromium, cadmium and lead in rats: effects on life span, tumors and tissue levels. J. Nutr. 86:51-66.

Schroeder, H. A., and J. J. Balassa. 1967. Arsenic, germanium, tin and vanadium in mice: effects on growth, survival and tissue levels. J. Nutr. 92:245-251.

Schroeder, H.A and M. Mitchener. 1971. "Toxic effects of trace elements on the reproduction of mice and rats." Arch. Environ. Health. 23: 102-106.

Schroeder, H. A., and M. Mitchener. 1972. Selenium and Tellurium in mice. Arch. Environ. Health 24:66-71.

Sealander, J.A. 1943. Winter food habits of mink in southern Michigan. J. Wildl. Manage. 7: 411-417.

Sheppard, M. I., T.T. Vandergraaf, D.H. Thibault, and J.A.K. Reid. 1983. "Technetium and uranium: Sorption by and plant uptake from peat and sand." *Health Physics* 44(6):635-643.

Short, H.L. and R.J. Cooper. 1985. Habitat suitability index models: Great Blue heron. Biological report 82(10.99). Fish and Wildlife Service, U.S. Department of the Interior.

Smith, S. P.J. Peterson, and K.H.M. Kwan. 1989. "Chromium accumulation, transport and toxicity in plants." *Toxicol. Environ. Chem.* 24:241-251.

Snarski, V. M., and G. F. Olson, 1982. Chronic toxicity and bioaccumulation of mercuric chloride in the fathead minnow (*Pimephales promelas*). Aquat. Toxicol. 2:143-156.

Sokal, R.R., and F.J. Rohlf. 1981. Biometry. W.H. Freeman and Company, San Francisco.

Sutton, W.R., and V.E. Nelson. 1937. Studies on zinc. Proc. Soc. Exp. Biol. Med. 36:211-213.

Soldat, J. K., N. M. Robinson and D. A. Baker. 1974. *Models and Computer Codes for Evaluating Environmental Radiation Doses*, BNWL-1754, Battele Pacific Northwest Laboratories, Richland, Washington.

Soltanpour, P.N. and S.M. Workman. 1980. Use of NH_4HCO_3-DTPA soil test to assess availability and toxicity of selenium to alfalfa plants. Commun. Soil Sci. Plant

Anal 11(12):1147-1156.

Southworth, G. R., M. J. Peterson, S. M. Adams, and B. G. Blaylock. 1994. Estimation of appropriate background concentrations for assessming mercury contamination in fish. Bull. Environ. Contam. Toxicol. 53:211-218.

Stahl, J.L., M.E. Cook, M.L. Sunde, and J.L. Greger. 1989. Enhanced humoral immunity in progeny chicks fed practical diets supplemented with zinc. Appl. Agric. Res. 4:86-89.

Stahl, J.L., J.L. Greger, and M.E. Cook. 1990. Breeding-hen and progeny performance when hens are fed excessive dietary zinc. Poultry Sci. 69:259-263. (Cited in Eisler 1993)

Stickel, L.F., W.H. Stickel, R.A. Dyrland, and D.L. Hughes. 1983. "Oxychlordane, HCS-3260, and nonachlor in birds: lethal residues and loss rates." J. Toxicol. Environ. Health. 12: 611-622.

Stokinger, H.E. 1981. Copper. In: G.D. Clayton and E. Clayton, Eds, Patty's Industrial Hygiene and Toxicology, Vol. 2A. John Wiley & Sons, New York, NY, pp. 1620-1630.

Stone, E.L., and G. Baird. 1956. "Boron level and boron toxicity in red and white pine." *J. Forestry* 54:11-12.

Stoner, D. and L.C. Stoner. 1941. Feeding of Nestling Bank Swallows. AUK? Volume 58: 52-55.

Straube, E.F., N.H. Schuster, and A.J. Sinclair. 1980. Zinc toxicity in the ferret. J. Comparative Pathol. 90:355-361.

Suedel, B.C., J.H. Rodgers, Jr., and P.A. Clifford. 1993. "Bioavailability of fluoranthene in freshwater sediment toxicity tests." *Environ. Toxicol. and Chemistry* 12: 155-165.

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.* Oak Ridge National Laboratory, Oak Ridge, TN. ES/ER/TM-96/R1.

Templeton, W. L., R. E. Nakatani, and E. E. Held. 1971. "Radiation Effects". *Radioactivity in the Marine Environments.* pp 223-239. National Academy of Science, Washington DC.

Traynor, M. F. and B. D. Knezek. 1973. "Effects of nickel and cadmium contaminated

soils on nutrient composition of corn plants." *Proc. Annual Conf. on Trace Substances in the Environment.* 7:82-87.

Trelease, S.F., A.A. Di Somma, and A.L. Jacobs. 1960. "Seleno-amino acid found in *Astragalus bisulcatus*." *Science* 132:618.

Treon, J.F. and F.P. Cleveland. 1955. "Toxicity of certain chlorinated hydrocarbon insecticides for laboratory animals, with special reference to aldrin and dieldrin." Ag. Food Chem. 3: 402-408.

Turner, M. A. and R. H. Rust. 1971. "Effects of chromium on growth and mineral nutrition of soybeans." *Soil Sci. Soc. Am. Proc.* 35:755-58.

TVA (Tennessee Valley Authority). 1994. *Clinch River Remedial Investigation Task 9 - Benthic Macroinvertebrates.*

USDA (U.S. Department of Agriculture). 1981. *Soil Survey of Anderson County, Tennessee.* Soil Conservation Service.

U.S. AF (U.S. Air Force). 1990. Copper. In: The Installation Program Toxicology Guide, Vol. 5. Wright-Patterson Air Force Base, Ohio, pp. 77(1-43).

U.S. EPA. 1975. *Preliminary Investigation of Effects of Boron, Indium, Nickel, Selenium, Tin, Vanadium, and Their Compounds, Vol. VI, Vanadium.*

U.S. EPA. 1980a. Guidelines and methodology used in the preparation of health effects assessment chapters of the consent decree water quality criteria documents. Federal Register 45(231): 79347-79356.

U.S. EPA Report No. EPA-560/2-75-005f. Cited in EPA. 1980b. *A Screening Procedure for the Impacts of Air Pollution Sources on Plants, Soils, and Animals.* EPA 450/2-81-078. Washington, D.C.

U.S. EPA. 1980-Ag. Ambient Water Quality Criteria for Silver. United States Environmental Protection Agency, Office of Water, Washington, D.C.

U.S. EPA. 1980-bis. Ambient water quality criteria for phthalate esters, EPA 440/5-80-067. U.S. Environmental Protection Agency, Washington, D.C.

U. S. EPA. 1980-phth. Ambient water quality criteria for phthalate esters, EPA 440/5-80-067. U.S.Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1984. Health Effects Assessment for Zinc (and Compounds). U.S Environmental Protection Agency, Office of Research and Development, Washington, DC. EPA/540/1-86-048.

U.S. EPA 1985-Cd. Ambient Water Quality Criteria for Cadmium. United States Environmental Protection Agency, Office of Water. EPA 440/5-84-032. Washington D.C.

U.S. EPA. 1985-Cr. Ambient water quality criteria for chromium. EPA 440/5-80-035. U.S. Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1985-Cu. *Ambient water quality criteria for copper - 1984.* EPA 440/5-84-031. U.S. Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1985-Hg. Ambient water quality criteria for mercury-1984. EPA 440/5-84-026. Office of Water Regulations and Standards, Washington, D.C.

U.S. EPA. 1985-Pb. Ambient water quality criteria for lead - 1984, EPA 440/5-84-027. U.S. Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1986-Ni. Ambient water quality criteria for nickel - 1986, EPA 440/5-86-004. U.S. Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1987-Zn. Ambient water quality criteria for zinc, 1987, EPA 440/5-87-003. U.S. Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1988-Al. Ambient water quality criteria for aluminum. EPA 440/5-86-008. U.S. Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1993. Integrated Risk Information System (IRIS). Carcinogenicity Assessment for Zinc and Compounds, and Oral RfD Assessment for Zinc phosphide. Office of Health and Environmental Assessment, Cincinnati, OH.

U.S. EPA (U. S. Environmental Protection Agency). 1993a. "Wildlife exposure factors handbook." Volume I. Office of Research and Development, Washington, DC. EPA/600/R-93/187a.

U.S. EPA (U. S. Environmental Protection Agency). 1993b. "Wildlife criteria portions of the proposed water quality guidance for the Great Lakes system." EPA/822/R-93/006.Office of Science and Technology, Washington, D.C.

U.S. Fish and Wildlife Service. 1964. "Pesticide-wildlife studies, 1963: a review of Fish and Wildlife Service investigations during the calendar year." FWS Circular 199.

Van Daele, L.J., and H.A. Van Daele. 1982. Factors affecting the productivity of ospreys nesting in west-central Idaho. Condor. 84: 292-299.

van Goor, B.J., and D. Wiersma. 1976. "Chemical forms of Mn and Zn in phloem exudates." *Physiol. Plant.* 36:213-216.

Van Leeuwen, C. J., J. L. Maase-Diepeveen, G. Niebeek, W. H. A. Vergouw, P. S. Griffioen, M. W. Luijken. 1985. Aquatic toxicological aspects of dithiocarbamates and related compounds. I. Short-term toxicity tests. Aquatic Toxicology 7:145-164.

Venugopal, B., and T. D. Luckey. 1978. Metal toxicity in mammals: vol. 2. Chemical toxicity of metals and metalloids. Plenum press, New York. 409pp.

Verschuuren, H.G., R. Kroes, E.M. Den Tonkelaar, J.M. Berkvens, P.W. Helleman, A.G. Rauws, P.L. Schuller, G.J. Van Esch. 1976a. Toxicity of methylmercury in rats I. short-term study. Toxicology 6: 85-96.

Verschuuren, H.G., R. Kroes, E.M. Den Tonkelaar, J.M. Berkvens, P.W. Helleman, A.G. Rauws, P.L. Schuller, G.J. Van Esch. 1976b. Toxicity of methylmercury in rats II. long-term toxicity study. Toxicology 6: 107-123.

Verschuuren, H.G., R. Kroes, E.M. Den Tonkelaar, J.M. Berkvens, P.W. Helleman, A.G. Rauws, P.L. Schuller, and G.J. Van Esch. 1976c. Toxicity of methyl mercury chloride in rats. II. Reproduction study. Toxicol. 6: 97-106.

Vohra, P., and F.H. Kratzer. 1968. Zinc, copper and manganese toxicities in turkey poults and their alleviation by EDTA. Poult. Sci. 47: 699-704

Wallace, A. and E. M. Romney. 1977a. "Roots of higher plants as a barrier to translocation of some metals to shoots of plants." *In Biological Implications of Metals in the Environment.* Proceeding of the Fifteenth Annual Hanford Life Sciences Symposium, Richland, WA. Tech Info Center, ERDA, Washington, DC. pp. 370-379.

Wallace, A., R. M. Romney, J. W. Cha, S. M. Soufi, and F. M. Chaudhry. 1977b. "Nickel phytotoxicity in relationship to soil pH manipulation and chelating agents." *Commun. Soil Sci. Plant Anal.* 8(9):757-64.

Walters, M., and F.J.C. Roe. 1965. A study of the effects of zinc and tin administered orally to mice over a prolonged period. Food Cosmet. Toxicol. 3:271-276. (Cited in U.S. EPA 1993).

Wan H.F., R.L. Mikkelsen, and A.L. Page. 1988. Selenium uptake by some agricultural crops from central California soils. J. Environ. Qual. 17(2):269-272.

Wang, W. 1988. Site-specific barium toxicity to common duckweed, *Lemna minor*. Aquat. Toxicol. 12:203-212.

Warington, K. 1954. "The influence if iron supply on toxic effects of manganese, molybdenum and vanadium on soybeans, peas and flax." *Ann. Appl. Biol.*

41(1):1-22.912.

Weigel, H. J., D. Ilge, I. Elmadfa, and H. J. Jaeger. 1987. Availability and toxicological effects of low levels of biologically bound cadmium. Arch. Environ. Contam. Toxicol. 16(1):85-93.

Westmoreland, N., and W.G. Hoekstra. 1969. Pathological defects in the epiphyseal cartilage of zinc-deficient chicks. J. Nutr. 98:76-82. (Cited in Eisler 1993)

Whanger, P. D. 1973. Effect of dietary cadmium on intracellular distribution of hepatic iron in rats. Res. Commun. Chem. Pathol. Pharmacol. 5:733-740.

White, D.H., and M.T. Finley. 1978. Uptake and retention of dietary cadmium in mallard ducks. Environ. Res. 17: 53-59.

White, D.H., M.T. Finley, and J.F. Ferrell. 1978. Histopathological effects of dietary cadmium on kidneys and testes of mallard ducks. J. Toxicol. Environ. Health.

White, M. C., R. L. Chaney, and A. M. Decker. 1979. "Differential cultivar tolerance in soybean to phytotoxic levels of soil Zn. II. Range of Zn additions and the uptake and translocation of Zn, Mn, Fe, and P." *Agronomy J.* 71:126-131.

Whitehead, N.E., R.R. Brooks, and P.J. Peterson. 1971. "The nature of uranium occurrence in the leaves of *Coprosma australis* (A. Rich) Robinson." *Aust. J. Biol. Sci.* 24:67-73.

Whitworth, M.R., G.W. Pendleton, D.J. Hoffman, and M.B. Camardese. 1991. Effects of dietary boron and arsenic on the behavior of mallard ducklings. Environ. Toxicol. Chem. 10: 911-916.

Woodhead, D. S. 1984. "Contamination due to Radioactive Materials". p. 1111-1287. In: O. Kinne (ed.), *Marine Ecology*. Vol. V, Part 3: Pollution and Protection of the Seas-Radioactive Materials, Heavy Metals and Oil. John Wiley and Sons, New York, Ny.

Keen and Leach 1988 (Mn)

Khalid, B. Y. and J. Tinsley. 1980. Some effects of nickel toxicity on rye grass." *Plant Soil.*

Korschgen, L.J. 1958. December food habits of mink in Missouri. J. Mamm. 39: 521-527.

U.S. Army Corp of Engineers 1987. *Wetlands Delineation Manual.* Technical Report Y-87-1. Waterways Experiment Station, Vicksburg, MS.

NAS. 1977. Arsenic. National Acad. Sci., Washington, D.C. 332pp.

WHO (World Health Organization). 1984. "Chlordane." Environ. Health Criter. 34. 82 pp.

Williams, L. L., J. P. Giesy, N. DeGalan, D. A. Verbrugge, D. E. Tillitt, G. T. Ankley, R. L. Welch. 1992. Prediction of concentrations of 2,3,7,8-tetrachlorobenzo-*p*-dioxin equivalents from total concentrations of polychlorinated biphenyls in fish fillets. Environ. Sci. Technol. 26:1151-1159.

Wobeser, G., N. O. Nielsen, and B. Schiefer. 1976. Mercury and mink, Part 2 Experimental methyl mercury intoxication. Can. J. Comp. Med. J. 40:34-45.

Wobeser, G. and M. Swift. 1976. Mercury poisoning in a wild mink. J. Wildl. Dis. 12:335-345.

Wobeser, G., N. O. Nielsen, and B. Schiefer. 1976. Mercury and mink. II. Experimental methylmercury intoxication. Can. J. Comp. Med. 40:34-45.

Wren, C.D. 1986. A review of metal accumulation and toxicity in wild mammals: I. Mercury. Environ. Res. 40: 210-244.

Wright, L.L., J.S. Mattice, and J.J. Beauchamp. 1982. "Effect of temperature and sex on growth patterns in nymphs of the mayfly *Hexagenia bilineata* in the laboratory." Freshwater Biol. 12:535-545.

Wong, M.H. and A.D. Bradshaw. 1982. A comparison of the toxicity of heavy metals, using root elongation of rye grass, *Lolium perenne*. New Phytol. 92:255-61.

Zar, 1984. Biostatistical analysis. Prentice-Hall, Inc., Englewood Cliffs, New Jersey.

Holladay, S. K. et al. 1995. *Quality Assurance/Quality Control Summary Report for Phase 2 of the Clinch River Remedial Investigation.* ORNL/ER-245. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

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