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Energy Systems Environmental Restoration Program
Y-12 Environmental Restoration Program

# Remedial Investigation Report on Chestnut Ridge Operable Unit 2 (Filled Coal Ash Pond/Upper McCoy Branch) at the Oak Ridge Y-12 Plant Oak Ridge, Tennessee

## Volume 1. Main Text

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# ACRONYMS

| | |
|---|---|
| AET | apparent effects threshold |
| ARAR | applicable or relevant and appropriate requirement |
| ASTM | American Society for Testing and Materials |
| CCJM | C. C. Johnson & Malhotra, P.C. |
| CDI | chronic daily intake |
| CDM Federal | CDM Federal Programs Corporation |
| CERCLA | Comprehensive Environmental Response, Compensation, and Liability Act |
| CFR | Code of Federal Regulations |
| CL | confidence level |
| COC | contaminant of concern |
| COPC | contaminant of potential concern |
| CPI | Chemical Pertubation Index |
| CV | chronic value |
| DNA | deoxyribonucleic acid |
| DOC | dissolved organic carbon |
| DOE | U.S. Department of Energy |
| EA | Environmental Assessment |
| $EC_{50}$ | median effective concentration |
| Energy Systems | Martin Marietta Energy Systems, Inc. |
| EPA | U.S. Environmental Protection Agency |
| EqP | Equilibrium Partitioning |
| ER | Environmental Restoration |
| ERA | Ecological Risk Assessment |
| ER-L | Effects Range—Low |
| ER-M | Effects Range—Median |
| FCAP | Filled Coal Ash Pond |
| FFA | Federal Facility Agreement |
| FS | Feasibility Study |
| HHRA | Human Health Risk Assessment |
| HI | hazard index |
| HQ | hazard quotient |
| IQ | intelligence quotient |
| LCT | lowest concentration tested |
| LCV | lowest chronic value |
| $LD_{50}$ | dose that is lethal to 50% of test subjects |
| LET | linear energy transfer |
| LOAEL | lowest-observed-adverse-effect level |
| LOEC | lowest-observed-effect concentration |
| MCL | maximum contaminant level |
| MCLG | maximum contaminant level goal |
| MOE | Ministry of the Environment |
| msl | mean sea level |
| NA | Natural Area |
| NAWQC | National Ambient Water Quality Criteria |
| NCP | National Contingency Plan |

| | |
|---|---|
| NEPA | National Environmental Policy Act |
| NERP | National Environmental Research Park |
| NOAA | National Oceanic and Atmospheric Administration |
| NOAEL | no-observed-adverse-effect level |
| NOEC | no-observed-effect concentration |
| NPDES | National Pollutant Discharge Elimination System |
| NPDWS | National Primary Drinking Water Standard |
| NPL | National Priorities List |
| NSDWS | National Secondary Drinking Water Standard |
| OREIS | Oak Ridge Environmental Information System |
| ORNL | Oak Ridge National Laboratory |
| ORO | Oak Ridge Operations |
| ORR | Oak Ridge Reservation |
| OSHA | Occupational Safety and Health Administration |
| OSWER | Office of Solid Waste and Emergency Response |
| OU | operable unit |
| PAH | polycyclic aromatic hydrocarbon |
| PCB | polychlorinated biphenyl |
| PCP | pentachlorophenel |
| PRG | preliminary remediation goal |
| RA | Reference Area |
| RAGS | Risk Assessment Guidance for Superfund |
| RCRA | Resource Conservation and Recovery Act of 1976 |
| RfA | Reference Area |
| RfC | reference concentration |
| RfD | reference dose |
| RFI | RCRA Facility Investigation |
| RI | Remedial Investigation |
| ROD | Record of Decision |
| RPD | relative percent difference |
| RSA | Research Area |
| SARA | Superfund Amendments and Reauthorization Act |
| SCA | Sluice Channel Area |
| SDWA | Safe Drinking Water Act |
| SEN | Secretary of Energy Notice |
| SF | slope factor |
| SI | screening index |
| SLC | screening level concentration |
| SMCL | secondary maximum contaminant level |
| SNA | State Natural Area |
| SQB | sediment quality benchmark |
| TAL | Target Analyte List |
| TBC | to be considered |
| TCL | Target Compound List |
| TDEC | Tennessee Department of Environment and Conservation |
| TIC | tentatively identified compound |
| TOC | total organic carbon |
| U.S.C. | United States Code |
| UCL95 | upper 95% confidence limit |
| UMB | Upper McCoy Branch |

| | |
|---|---|
| VOC | volatile organic compound |
| WAG | waste area grouping |
| WCK | WOC kilometer |
| WI DNR | Wisconsin Department of Natural Resources |
| WQB | water quality benchmark |
| WQC | water quality criteria |

# EXECUTIVE SUMMARY

This document is a report on the remedial investigation (RI) of Chestnut Ridge Operable Unit (OU) 2 at the Oak Ridge Y-12 Plant. Chestnut Ridge OU 2 consists of Upper McCoy Branch (UMB), the Filled Coal Ash Pond (FCAP), and the area surrounding the Sluice Channel formerly associated with coal ash disposal in the FCAP.

Chestnut Ridge OU 2 is located within the U.S. Department of Energy's (DOE's) Oak Ridge Reservation in Anderson County, Tennessee, approximately 24 miles west of Knoxville. The pond is an 8.5-acre area on the southern slope of Chestnut Ridge, 0.5 mile south of the main Y-12 Plant and geographically separated from the Y-12 Plant by Chestnut Ridge. The elevation of the FCAP is ~950 ft above mean sea level (msl), and it is relatively flat and largely vegetated. Two small ponds are usually present at the northeast and northwest corners of the FCAP. The Sluice Channel Area extends ~1000 ft from the northern margin of the FCAP to the crest of Chestnut Ridge, which has an elevation of ~1100 ft above msl. The Sluice Channel Area is largely vegetated also.

McCoy Branch runs from the top of Chestnut Ridge across the FCAP into Rogers Quarry and out of the quarry where it runs a short distance into Milton Hill Lake at McCoy Embayment, termed UMB. The portion south of Rogers Quarry, within Chestnut Ridge OU 4, is termed Lower McCoy Branch.

The DOE Oak Ridge Y-12 Plant disposed of coal ash from its steam plant operations as a slurry that was discharged into an ash retention impoundment; this impoundment is the FCAP. The FCAP was built in 1955 to serve as a settling basin after coal ash slurried over Chestnut Ridge from the Y-12 Plant. The FCAP was constructed by building an earthen dam across the northern tributary of McCoy Branch. The dam was designed to hold 20 years of Y-12 steam plant ash. By July 1967, ash had filled up the impoundment storage behind the dam to within 4 ft of the top. As the sediment trap efficiency of the FCAP decreased, the slurry was released directly into UMB. In 1967 and 1968, UMB was diverted into Rogers Quarry. Ash continued to flow through the FCAP and UMB to the quarry until 1989, when a bypass pipe was constructed to carry it directly from the steam plant to Rogers Quarry.

The objectives of this RI report are first to summarize existing site information and to draw conclusions from the information gathered during the field investigations concerning the following:

- the types and quantities of waste present at Chestnut Ridge OU 2,
- the potential for migration of contamination from the site, and
- the potential for adverse effects on human health and the environment.

These conclusions are then used to develop a Human Health Risk Assessment (HHRA) and an Ecological Risk Assessment (ERA) to identify contaminant-specific and location-specific applicable or relevant and appropriate requirements and to provide information on waste characterization and preliminary remediation goals for Clinch River OU 2 for use in the feasibility study.

The Phase 2 RI field investigation was conducted in response to comments by regulators on the Phase 1 RI report and on the scope of work performed and in compliance with the requirements of the Federal Facilities Agreement between DOE, the U.S. Environmental Protection Agency, and the Tennessee Department of Environment and Conservation. The comments noted by the regulators in the Phase 1 RI report were addressed through an additional and expanded sampling program. The coal ash was more thoroughly characterized by drilling in the FCAP. Groundwater, surface water/sediment samples, and biological samples and measurements were collected/performed at several locations. The areal extent and thickness of the coal ash located upslope of the FCAP were measured by completing transects across the Sluice Channel Area leading into the FCAP. The data collected in Phases 1 and 2, summarized in the RI report, were used to support the HHRA and an ERA.

An evaluation of potential contaminant migration pathways (soil, ash, sediment, surface water, and groundwater) was conducted. The baseline HHRA addresses potential risks to human health associated with contaminants identified at the site. Risks were quantified for two land use scenarios:

- trespasser/hunter—current land use with passive land use controls and
- residential—future land use without access controls, assuming reasonable maximum exposure.

A baseline ERA also was conducted. Ecological risks were characterized for aquatic biota, sediment-associated biota, terrestrial plants, soil invertebrates, soil microbes, and wildlife.

Organic compounds were detected in the Chestnut Ridge OU 2 media at low concentrations and low frequencies, with bis(2-ethylhexyl)phthalate, diethylphthalate, di-n-butylphthalate, and phenol being the most common. Bis(2-ethylhexyl)phthalate is a common laboratory contaminant and is not believed to be associated with Chestnut Ridge OU 2 operations. However, based on comparisons with blanks and a toxicity screening technique, bis(2-ethylhexyl)phthalate could not be eliminated from the COPs.

A number of inorganic constituents were detected in the soil, ash, surface water, sediments, and groundwater above background concentrations. The most frequent detections above reference levels were for Al, As, Fe, Mg, and Zn.

As, Be, $^{228}$Th, and Mn are the most important waste-related constituents of Chestnut Ridge OU 2. Analysis of the underlying native soils indicated that inorganic contaminants have migrated from the waste into the soils. Elevated levels of the contaminants of potential concern were also measured in the surface waters, sediments, spring waters, and groundwaters.

The data suggest that contaminant migration from Chestnut Ridge OU 2 is rather limited. Surface water data indicated that most contaminants are settling out of the water column before exiting OU 2; for example, elevated metals concentrations were reported from surface water at the toe of the dam but not in successive samples downstream of the FCAP. Sediment transport is a potential pathway for contaminant migration, but the analyses of sediment data from UMB also do not indicate elevated concentrations of contaminants seeps or moves very sluggishly at depth in the bedrock. In either case, the potential for contaminant

transport off site is limited; results of analysis of groundwater samples support this conclusion. The extensive vegetative cover at OU 2 limits the potential for transport by way of surface soil; the ERA, however, did identify uptake of some metals by trees. The ERA also identified potential transport of contaminants off site by wildlife.

For the Chestnut Ridge OU 2 HHRA, the two risk scenarios listed below were evaluated. Elevated risks (carcinogenic greater than $10^{-4}$ or hazard index equal to or greater than 1.0) associated with the scenarios and their respective pathways derive from the following COCs:

- trespasser scenario
  - ash external exposure: $^{228}$Th
- residential scenario
  - ash external exposure: $^{228}$Th
  - ash incidental ingestion: As
  - soil below the ash external exposure: $^{228}$Th
  - sediment incidental ingestion: As
  - surface (pond) water ingestion: As, Mn
  - sediment external exposure: $^{228}$Th
  - groundwater ingestion: Be, Mn
  - homegrown garden vegetable/fruit ingestion: As, Cd, Mn, Hg

In summary, regulatory unacceptable risks are eliminated for all media and contaminants, except $^{228}$Th in ash, with the scenario that assumes access controls are maintained, because the access controls cut off pathways of exposure.

The ecological risks characterization indicates the potential for adverse effects from the COCs at the following endpoints:

- Fish—Al, Cr, Cu, Fe Mn, and Zn concentrations in the waters of the FCAP ponds and UMB.
- Benthic macroinvertebrates—as in FCAP sediments and Al, Cr, Cu, Fe, Pb, Mn, Zn concentrations in FCAP surface waters. Ba, Mn, and Ni in UMB sediments and Al, As, Cu, Fe, Pb, Hg, and Zn concentrations in UMB surface waters.
- Plants—Al, Sb, As, Ba, Cd, Cr, Co, Cu, Fe, Mn, Hg, Ni, Se, Ag, V, and Zn concentrations in ash soil and shallow groundwater in the FCAP, Sluice Channel Area, and UMB.
- Soil invertebrates—Cd, Cr, Cu, Hg, and Tl concentrations in the ash and soil of the FCAP and Sluice Channel Area.
- Soil microbes and microbial processes—Al, Cd, Cr, Cu, Fe, Mn, V, and Zn concentrations in ash and soil of the entire OU 2.
- Small mammals—Al, As, Ba, Cd, Se, Tl, and V concentrations in ash, food, and water in the entire OU 2.
- Reservation-wide endpoints—metals concentrations exposures for white-tailed deer (Al, As, Ba, Cd, Se, and Tl), red fox (Al, As, Ba, Cr, Hg, Se, and TL) and red-tailed hawk (As, Cr, Hg, and Se) in ash, food, and water in the entire OU 2.

The ERA for Chestnut Ridge OU 2 determined that all lines of evidence available for fish, benthic macroinvertebrates, plants, soil invertebrates, small mammals, and wide-ranging mammals indicate the potential for adverse effects from the current inorganic contaminant levels at the site. The ecological risks associated with the future scenario are identical to the current risk scenario, given that a catastrophic failure of the dam does not occur.

# 1. INTRODUCTION

This report documents the remedial investigation (RI) on Chestnut Ridge Operable Unit (OU) 2 at the Oak Ridge Y-12 Plant. Chestnut Ridge OU 2 consists of Upper McCoy Branch (UMB), the Filled Coal Ash Pond (FCAP), and the area surrounding the sluice channel formerly associated with coal ash disposal in the FCAP. OU 2 is one of four OUs along Chestnut Ridge on the Oak Ridge Reservation (ORR). The purpose of the RI is to collect data to:

- evaluate the nature and extent of known and suspected contaminants;
- support an Ecological Risk Assessment (ERA) and a Human Health Risk Assessment (HHRA);
- support the evaluation of remedial alternatives; and
- develop a Record of Decision (ROD) for the site.

This chapter of the report summarizes the regulatory background of environmental investigations at the ORR and the approach currently being followed and provides an overview of the RI conducted at Chestnut Ridge OU 2. Subsequent chapters provide details on (1) environmental setting, (2) site history and current conditions, (3) applicable or relevant and appropriate requirements (ARARs), (4) the HHRA and ERA, (5) remedial action goals, and (6) conclusions and recommendations resulting from the RI. Figures and tables are located at the end of each chapter following the text.

## 1.1 REGULATORY INITIATIVE

The Oak Ridge Y-12 Plant, located within the ORR, is owned by the U.S. Department of Energy (DOE) and managed by Martin Marietta Energy Systems, Inc. (Energy Systems). The Y-12 Plant is one of three major DOE facilities on the ORR. The ORR contains both hazardous and mixed waste sites that are subject to regulations promulgated under the Resource Conservation and Recovery Act of 1976 (RCRA) and the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), as amended in 1986 by the Superfund Amendments and Reauthorization Act (SARA).

Under RCRA guidelines and requirements from the Tennessee Department of Environment and Conservation (TDEC), the Y-12 Plant initiated investigation and groundwater monitoring of various sites within its boundaries in the mid 1980s. The entire ORR was placed on the National Priorities List (NPL) of CERCLA sites in November 1989. Following CERCLA guidelines, sites under investigation require an RI to define the nature and extent of contamination, evaluate the risks to the public health and the environment, and determine the remedial action goals for a feasibility study (FS) of potential remedial actions.

DOE, the U.S. Environmental Protection Agency (EPA), and TDEC negotiated a Federal Facility Agreement (FFA) in response to the NPL placement of the ORR. The FFA was developed for compliance pursuant to CERCLA, RCRA, the National Environmental

Policy Act (NEPA), and the National Contingency Plan (NCP). A common goal of the parties that entered into the FFA is to ensure that past releases from operations and waste management at ORR are thoroughly investigated and that appropriate remedial action is taken for the protection of human health and the environment.

The general purposes of the FFA are to establish a framework and schedule for development, implementation, and monitoring of response actions at the ORR in accordance with applicable guidance and policy; to coordinate responses under CERCLA and RCRA to maximize flexibility and preclude redundant activity; to minimize duplication of analytical and investigative work; to ensure quality of data management; and to expedite response actions with minimal delay.

## 1.2 ORR ENVIRONMENTAL RESTORATION PROGRAM

DOE established the Environmental Restoration (ER) Program to plan and implement the investigation and remediation of the ORR in cooperation with the parties to the FFA. A fundamental goal in the implementation of the program is that remedial action be emphasized. The goal acknowledges that no investigatory process can resolve all uncertainties and that remedial actions must be able to accommodate deviation from original hypotheses after they are initiated. This approach to the process of investigation and remediation promotes early remedy selection, flexibility for the remedial action, and contingencies to account for new data obtained during investigation of the sites. Removal actions will be implemented, as necessary, to protect human health and the environment from an imminent short-term threat. Interim remedial actions also may be implemented if the site warrants quick stabilization or risk reduction.

The lead agency for the investigations and remedial actions on the ORR is the DOE Oak Ridge Operations Office (DOE-ORO). The duties of the lead agency include overseeing and managing ORR remedial actions pursuant to the FFA, serving as the primary contact and coordinator with regulators for implementing the FFA, and ensuring the availability of resources required to implement the site management plan. EPA and TDEC are working together within the terms of the FFA to initiate remedial action work and to provide regulatory oversight.

DOE-ORO has delegated remediation activities at each waste area grouping (WAG) or OU to three primary contractors in addition to Energy Systems. Jacobs Engineering Group is responsible for the FS, Environmental Assessments (EAs), proposed plans, interim Records of Decision, and RODs. ENSERCH is responsible for remedial designs. MK-Ferguson of Oak Ridge Company is responsible for remedial actions. Additional information on the ER Program is available in the *Oak Ridge Reservation Site Management Plan for the Environmental Restoration Program*, DOE/OR-1001/R2 (Energy Systems 1992).

## 1.3 FACILITY-SPECIFIC ER PROGRAM

The Oak Ridge Y-12 Plant, one of three major DOE facilities on the ORR, includes ~800 acres near the northeast corner of the reservation and adjacent to the city of Oak Ridge (Fig. 1.1). The U.S. Army Corps of Engineers built the Y-12 Plant in 1943 as part of the Manhattan Project. The plant is a manufacturing and developmental engineering facility

that produced components for various nuclear weapons systems and provides engineering support to other Energy Systems facilities. More than 200 contaminated sites created by past waste management practices have been identified at the Y-12 Plant. Many of the sites have been grouped into OUs based on priority and on investigative and remediation requirements.

A major component of the investigative and remedial strategy to be applied to the Y-12 Plant and the entire ORR is a structured approach. This approach is based on separate studies of groundwater, surface water, and contaminant sources. The hydrogeologic regime at the Y-12 Plant is complex and includes surface water drainage, groundwater flow through a dominantly fractured system, and groundwater flow through a karst system. Numerous sources may have contributed to groundwater and surface water contamination; for example, sources contributing to groundwater that is part of one OU often contribute to groundwater contamination that is part of another OU.

Given these circumstances, the parties to the FFA concluded that the most timely and cost-efficient investigations would result if sources of contamination were studied separately from the groundwater and surface water. This conclusion led to the designation of source control and integrator OUs. Source control OUs focus on the nature and extent of contaminant contributions from a specific source or group of sources. Integrator OUs, on the other hand, focus on the nature and extent of contamination in a particular medium such as surface water or groundwater independent of the specific source or sources. Some of the ORR OUs, such as the four Chestnut Ridge OUs, combine both source control and integrator aspects.

After the completion of the RI and FS process, source control OUs will be addressed initially in interim action RODs. When all proposed source control OU interim actions are complete, there should be an adequate amount of monitoring data from integrator OUs to determine if the interim actions have been effective, and, if not, additional remediation will be undertaken in the integrator OU final ROD. OUs that combine source control and integrator aspects will proceed to the final ROD, and omit interim action RODs.

## 1.4 SITE ER PROGRAM (INTENT AND SCOPE OF THE REMEDIAL INVESTIGATION/BASELINE RISK ASSESSMENT REPORT)

A RCRA Facility Investigation (RFI) Plan was prepared for the FCAP (Murphy and Loar 1988), and the investigation was later conducted at the site (Energy Systems 1991). The purpose of the investigation was to define the nature and extent of contamination at the site, gather information to assess the potential for off-site contaminant migration, and evaluate potential risk to human health from site contaminants. Because the ORR became an NPL site in 1989, the investigation conducted at the FCAP in 1990 was termed an RI instead of an RFI. All subsequent planning and reporting documents have been prepared using CERCLA terminology; for example, the area of investigation is now termed Chestnut Ridge OU 2, rather than just the FCAP.

The RFI Work Plan and the RI report for the FCAP were reviewed by EPA and TDEC and determined to require further investigation in the current regulatory environment. The Phase 1 RI data were deemed inadequate to characterize the site and risk to public health and the environment because samples of coal ash in the FCAP were not collected and analyzed; the HHRA was not complete and did not address the future homesteader scenario,

and an ERA was not completed. The scope of the Phase 1 work was developed under the RCRA Program, which is not a risk-based program nor does it require development of data quality objectives (DQOs) through a formalized process, as does CERCLA. A conceptual site model was developed; however, it did not include ash as a contaminated medium and therefore was incomplete. The Phase 1 data set did address the objectives of the study but was not adequate to meet the objectives of the CERCLA process, and therefore, a Phase 2 RI was planned and conducted.

The following are the primary comments noted by the regulators.

- There is a lack of adequate background data for all media samples.
- The HHRA did not address the future homesteader scenario.
- An ERA was not performed.
- Groundwater data did not adequately address the potential for contaminants migrating to the FCAP from the Security Pits.
- There is a lack of data on the chemical characteristics of the ash in the FCAP.
- There is a lack of data on ash volume and soil contamination associated with the sluice channel on Chestnut Ridge.

In order to address regulatory review comments, Energy Systems tasked CDM Federal Programs Corporation (CDM Federal) to develop a Phase 2 Sampling Plan for the RI to describe activities necessary to further characterize the site and potential risks to human health, the biological community, and the environment. These activities included groundwater, surface water/sediment, surface soils, and coal ash sampling; aquatic toxicity testing; benthic invertebrate, threatened and endangered species, and wetlands surveys; bioaccumulation testing of vegetation, soil invertebrates, and small mammal tissue; and soil toxicity testing.

## 1.5 SPECIAL PROBLEMS

Access to Chestnut Ridge OU 2 is restricted with the entrance blocked by a bar gate, and escorts with DOE L- or Q-clearance may be necessary, even for a site visit. In addition, any personnel who enter OU 2 for sampling or other types of fieldwork must have completed Energy Systems' General Employee Training and Occupational Safety and Health Administration (OSHA) 40-hour training with yearly 8-hour refresher sessions.

## 1.6 OBJECTIVES

The objectives of this RI report are to summarize existing site information, develop an HHRA and an ERA, identify contaminant-specific and location-specific ARARs, and provide information on waste characterization and preliminary remedial action goals for Chestnut Ridge OU 2 for use in the FS report. The RI report will also draw conclusions from the information gathered during the field investigation concerning the types and quantities of waste present at Chestnut Ridge OU 2, the potential for migration of contamination from the site, and the potential for adverse effects on human health and the environment.

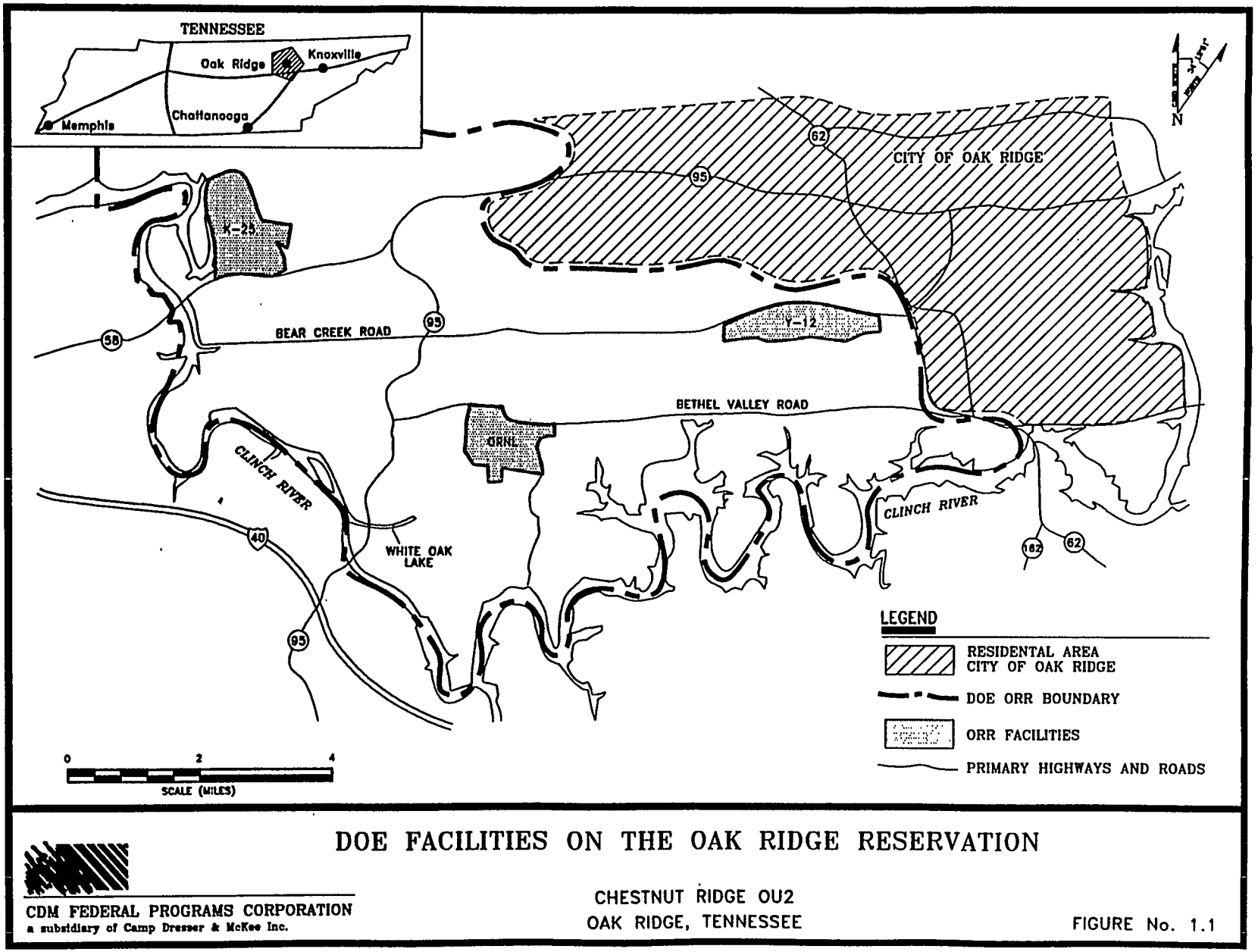
The Phase 2 RI field investigation was conducted to address deficiencies in the Phase 1 RI report and scope of work performed and to comply with the requirements of the FFA between DOE, EPA, and TDEC. The deficiencies noted in the Phase 1 RI report were addressed through an additional and expanded sampling program. The coal ash was more thoroughly characterized by drilling in the FCAP. Groundwater, surface water/sediment samples, and biological samples and measurements were collected/performed from several locations. The areal extent and thickness of the coal ash located upslope of the FCAP was measured by completing transects across the Sluice Channel Area leading into the FCAP. The data collected in Phases 1 and 2, summarized in this RI report, were used to support the HHRA and an ERA.

The primary objective of collecting biological data and performing soil and aquatic toxicity tests was to provide data to perform an ERA for the FCAP/UMB. The UMB/Rogers Quarry system has been subject to a biological monitoring program since the late 1980s related to water quality problems administered under the Tennessee Water Quality Act. Many of the samples collected and tests performed were designed to supplement the existing monitoring program. Additional samples specifically addressed direct and food chain hazards to terrestrial species. The ongoing biological monitoring program is being conducted by the Energy Systems Environmental Sciences Division at Oak Ridge National Laboratory (ORNL).

## 1.7 SCHEDULE

Table 1.1 shows the current schedule for completion of the RI/FS process at Chestnut Ridge OU 2. The RI/FS process has been somewhat accelerated for this OU; both the RI and FS reports are being prepared concurrently and reviewed concurrently. NEPA requirements will be addressed in an EA that will be combined with the FS report. The review and revision process will continue into 1995, with submittal of the final documents scheduled for March 1995.

Biomonitoring is scheduled to continue at the site through 1998. The preparation of the proposed plan is scheduled to begin during 1994, with completion in 1995, after which work will begin on the ROD. The ROD will be finalized by early 1997, and remedial design/remedial action activities should be complete by 2001.




DOE FACILITIES ON THE OAK RIDGE RESERVATION

CDM FEDERAL PROGRAMS CORPORATION
a subsidiary of Camp Dresser & McKee Inc.

CHESTNUT RIDGE OU2
OAK RIDGE, TENNESSEE

FIGURE No. 1.1

A1011/23MAR94/1=1

**Fig. 1.1 DOE facilities on the Oak Ridge Reservation**

**Table 1.1. Current schedule for completion**

| Task name | Start date | End date |
|---|---|---|
| RI report (preparation and review) | October 1, 1993 | March 2, 1995 |
| FS/EA report (preparation and review) | October 1, 1993 | March 2, 1995 |
| Biomonitoring | March 2, 1992 | October 12, 1998 |
| Proposed Plan | May 17, 1994 | October 16, 1995 |
| ROD | October 17, 1995 | January 29, 1997 |
| Remedial design | April 3, 1995 | August 14, 1997 |
| Remedial action integration | April 3, 1995 | December 7, 2001 |
| Remedial action | September 2, 1997 | December 7, 2001 |

# 2. CHARACTERIZATION OF ENVIRONMENTAL SETTING AND CONCEPTUAL SITE MODEL

## 2.1 GEOGRAPHY

The ORR includes 35,300 acres of federally owned land 20 miles west of Knoxville, Tennessee, bounded on the north and east by the city of Oak Ridge, and on the south and west by the Clinch River. The reservation is near the western margin of the Valley and Ridge physiographic province, a major subdivision of the Appalachians that stretches from Pennsylvania to Alabama. In the southern Appalachians, the Valley and Ridge province is bounded by the Blue Ridge province to the southeast and the Cumberland Plateau province to the northwest (Fig. 2.1). Characteristic features of the Valley and Ridge province include parallel valleys and ridges, typically trending northeast-southwest; topography controlled by alternating weak and strong rock units that have been folded and faulted and are now exposed by erosion; a trellis drainage pattern; abundant water gaps through resistant ridges; and many ridges with accordant summit levels that suggest former erosion surfaces.

Chestnut Ridge OU 2 includes UMB, the FCAP, and the Sluice Channel Area. UMB extends ~0.5 mile from Rogers Quarry to the toe of the FCAP dam. The FCAP is an 8- to 9-acre area on the southern slope of Chestnut Ridge, ~0.5 mile south of the main facilities of the Y-12 Plant and geographically separated from the plant by Chestnut Ridge itself (Fig. 2.2). The elevation of the FCAP is ~950 ft above mean sea level (msl) and it is relatively flat and largely vegetated. Two small ponds are usually present at the northeast and northwest corners of the FCAP. The Sluice Channel Area extends ~1000 ft from the northern margin of the FCAP to the crest of Chestnut Ridge, which has an elevation of ~1100 ft above msl.

Also located along Chestnut Ridge are the Security Pits (OU 1) and the new Y-12 landfill. These features are shown on Fig. 2.2 in relation to Chestnut Ridge OU 2.

## 2.2 DEMOGRAPHY

Chestnut Ridge OU 2 is entirely within the ORR on federally owned land. The site is currently inactive. Few Energy Systems employees use or visit Chestnut Ridge OU 2, and the site is in a restricted access area that is patrolled regularly by security personnel. The entrance is blocked to vehicles by a bar gate, and the area is posted as a controlled area. Energy Systems personnel visit the site several times each year to obtain samples for ongoing groundwater protection program monitoring.

Within 1 mile of the OU 2 boundaries, there are no residents. The South Patrol Road, a paved two-lane road that runs along the crest of Chestnut Ridge adjacent to the main Y-12 Plant area, borders the site. Access to this road is controlled via a bar gate. Several businesses are located along Bethel Valley Road, ~3/4 mile southwest of the site and 1-3/4 miles southeast of the site at the Scarboro facility. Clark Recreational Park, which has a boat ramp with access to Melton Hill Lake, is ~1-1/4 miles south-southwest of OU 2. Melton Hill Lake is open to recreational uses, including fishing, boating, and hunting, as sanctioned by the state of Tennessee.

There are no residents within 1 mile of OU 2 to the east, south, and west. The closest residents are ~1 mile north of OU 2 in a residential neighborhood north of the ORR boundary. The closest major population center is Oak Ridge, ~1 mile north-northeast of OU 2. The 1990 population of Oak Ridge was 27,310; the 1990 population of Anderson County was 68,250. Knoxville is the largest local population center, with a 1990 population of 165,121; the population of Knox County was 355,749.

## 2.3 CLIMATE

The ORR is located in a temperate climatic region. Climate in the Valley and Ridge province in the vicinity of the ORR is strongly influenced by the presence of the Great Smoky Mountains to the southeast and the Cumberland Mountains to the northwest. The mountains exert a moderating influence on the climate, which results in warm, humid summers and cool winters. Noticeable extremes in precipitation, temperature, and winds rarely occur. The Cumberland Mountains retard and weaken the force of cold winter air that commonly reaches south to these latitudes. The mountains also modify the hot summer winds that are common in the plains to the west. The Great Smoky Mountains act as an inclined plane, which lifts warm, moist air flowing northward from the Gulf of Mexico, thereby increasing the frequency of afternoon thunderstorms during summer months.

The mean annual temperature in the Oak Ridge area is 58°F (Webster and Bradley 1988). January is generally the coldest month; temperatures average ~38°F, but occasionally dip to 0°F or below. July is generally the warmest month, although average temperatures exhibit little variation between June and August. The average daily temperature in July is 77°F, with highs occasionally exceeding 100°F.

Data from the National Oceanic and Atmospheric Administration (NOAA) indicate that the mean annual precipitation in the Oak Ridge area is 55 in. The climate in the area is typically humid with frequent rain, and periods of 5 days or more without precipitation occur on average only four to five times per year (Boegly et al. 1984). Maximum monthly precipitation generally occurs between December and March and is associated with winter storms that are commonly of low intensity but long duration. Approximately 40% of the yearly precipitation falls during this interval. Snows occur between November and March, for the most part, and are usually light and accumulate only a trace. The mean annual snowfall between 1951 and 1988 was 9.8 in. (Energy Systems 1989). A second precipitation maximum occurs in July, when short, heavy rainfalls associated with thunderstorms are common (Davis et al. 1984). The driest period of the year normally occurs in September and October when slow-moving high pressure cells suppress rain and provide clear and dry weather.

Winds in the Oak Ridge area are controlled to some extent by the valley and ridge topography. The prevailing winds are either up-valley (northeasterly) or down-valley (southwesterly). Daytime winds generally blow up-valley, while nighttime winds usually blow down-valley (Davis et al. 1984). Figure 2.3 is a wind rose that shows the relative frequency (by percentage) of occurrence of winds by compass orientation and wind speed. High velocity winds and tornadoes are rare. Wind speeds are less than 7.4 mph 75% of the time, and wind speeds exceeding 18.5 mph are rare.

## 2.4 TOPOGRAPHY, GEOLOGY, AND SOILS

The Valley and Ridge province of Tennessee, in which the ORR is located, is a belt of southeast-dipping thrust faults and accompanying folds. Two regionally extensive thrust faults, the Copper Creek and Whiteoak Mountain, are exposed across the ORR along with folds and small-scale subsidiary structures (Fig. 2.4). The stratigraphic section exposed on the ORR includes rocks that range in age from Early Cambrian to Early Mississippian. The stratigraphic section exposed in the vicinity of OU 2 has been abbreviated by faulting and includes beds ranging in age from Early Cambrian to Late Ordovician (Fig. 2.5). The units comprise a complex assemblage of lithologies, including shale, sandstone, siltstone, dolostone, and limestone; total thickness is ~6560 to 8200 ft (2.0 to 2.5 km) (Hatcher et al. 1992).

Most of OU 2 is located on the southeast slope of Chestnut Ridge, at elevations ranging from ~959 to 964 ft above msl; the sluice channel extends from the northern margin of the FCAP to the crest of Chestnut Ridge at an elevation of ~1100 ft above msl. The OU is within the Whiteoak Mountain thrust sheet and is underlain entirely by the Late Cambrian-Early Ordovician Knox Group, a sequence of dolostone with interbedded limestones that ranges from ~2000 to 3000 ft (700 to 1000 m) in thickness in East Tennessee. Safford (1869) named the Knox Group for exposures in Knoxville, and a succession of subsequent workers variously described and subdivided the group (e.g., Ulrich 1911; Oder 1934; Rodgers 1953; Bridge 1956; Harris 1969; Milici 1973). The Knox Group is found throughout the Valley and Ridge province of the Southern Appalachians; stratigraphic and petrographic studies suggest that the group was deposited in peritidal environments on a broad continental shelf setting (Churnet et al. 1982). The mechanism for widespread dolomitization is not clearly understood (Hatcher et al. 1992).

The Knox Group is a prominent stratigraphic sequence on the ORR and is one of the principal ridge-forming units; it underlies Copper Ridge, Chestnut Ridge, Blackoak Ridge, and McKinney Ridge. The thickness of the Knox Group on the ORR is believed to be ~2360 ft (720 m), based on a nearly complete exposure near the Bull Run Steam Plant, just northeast of the ORR (Hatcher et al. 1992). The Knox Group is divisible into five formations across the ORR (and throughout much of East Tennessee). The formations are, in ascending stratigraphic order, the Copper Ridge Dolomite, the Chepultepec Dolomite, the Longview Dolomite, the Kingsport Formation, and the Mascot Dolomite.

Chestnut Ridge OU 2 is underlain by the Copper Ridge Dolomite, the Chepultepec Dolomite, the Longview Dolomite, and the Kingsport Formation. The Copper Ridge Dolomite is Late Cambrian in age and the Chepultepec Dolomite, the Longview Dolomite, and the Kingsport Formation are all early Ordovician in age.

The Copper Ridge Dolomite is a massively bedded cherty dolomite, with beds ranging from 65 to 230 ft (20 to 70 m) in thickness. The lower two-thirds of the Copper Ridge is medium to coarsely crystalline, thick-bedded, dark brownish-gray saccharoidal dolomite (Hatcher et al. 1992). Toward the top of the formation, medium to light gray, fine-grained, medium- to thick-bedded dolomite, typical of the entire Knox Group, is more common. Nodular chert, often oolitic and bedded, is a common feature of the Copper Ridge Dolomite, especially in the upper portions of the formation. Quartz sandstone beds also occur in the formation, particularly in the upper part. The thickness of the Copper Ridge ranges from ~800 to 1100 ft (250 to 250 m).

The Chepultepec Dolomite is a light-gray, fine-grained, medium-bedded dolomite that underlies most of the FCAP. The Chepultepec is less siliceous than the underlying Copper Ridge Dolomite, and therefore is a valley former (Hatcher et al. 1992). Chert is less abundant than in the underlying Copper Ridge, and, where present, it is lighter in color. The basal portion of the Chepultepec is characterized by a prominent zone of quartz- and dolomite-cemented sandstone that ranges in thickness from ~3 to 16 ft (1 to 5 m) in thickness. The thickness of the Chepultepec ranges from ~500 to 700 ft (165 to 225 m).

The Longview Dolomite is the thinnest unit in the Knox Group [130 to 200 ft (40 to 65 m) in thickness] and forms a prominent narrow ridge in the middle of the Knox Group outcrop belt across the ORR. The Longview consists of medium- to light-gray, thin- to medium-bedded siliceous dolomite. Finer-grained dolomite is locally replaced by coarsely recrystalline dolomite; bedded and nodular chert are common throughout the formation (Hatcher et al. 1992).

The Kingsport Formation consists of dolomites of varying color and texture interbedded with massive limestone near the base of the unit (Hatcher et al. 1992). The Kingsport dolomites include light-gray, fine- to medium-grained dolomite and fine-grained, pale-pink to grayish-pink dolomite. The limestone in the Kingsport is a massively bedded, mottled calcilutite. Chert is present in the Kingsport, but it is substantially less abundant than in the other formations of the Knox Group. The thickness of the Kingsport ranges from ~300 to 500 ft (100 to 165 m).

The bedrock at Chestnut Ridge OU 2 is deeply weathered, resulting in few exposures; strata of the Kingsport Formation are exposed along the spillway area just east of the FCAP dam. Where exposed near the site, beds generally strike N60E and dip 42° to 52° SE. Fractures are well developed in all stratigraphic units on the ORR and are the most pervasive mesoscopic structure (Hatcher et al. 1992). Both systematic and non-systematic fractures that formed at various times during the tectonic and erosional history of the Southern Appalachians are present. The fractures form a network of discontinuities and intersections that provide pathways for groundwater flow. Although there is some scatter in the orientation of the fractures, most of them tend to develop systematic orientations over a particular area, allowing patterns to be interpreted (Solomon et al. 1992).

Hatcher et al. (1992) studied the fracture system developed in bedrock across the ORR. Measurements of fractures in the Knox Group in the Whiteoak Mountain thrust sheet indicated the presence of two dominant fracture sets, one striking N68°E and dipping 56° NW and the other striking N39°W and dipping 87°SW. A third set is also present, and previous workers have described four fracture sets (Geraghty & Miller 1989). Nonsystematic fractures of varying orientation are also developed.

Fracture density is highly variable and is noted to decrease with depth; reported values on the ORR range from 5 to 200 per meter (Solomon et al. 1992). Fracture lengths generally range from several centimeters up to 1 m. Hatcher et al. (1992) discuss fracture spacing in strata on the ORR and note that dolomite has the greatest variation in fracture spacing versus bed thickness of the lithologies present; no clear trend was observed. Fracture spacing data from the Knox Group show considerable variation, with values ranging from a few inches to nearly a foot.

Haase et al. (1987) described localized solutional widening of fractures and other karst features from the core of the Knox Group at the adjacent Chestnut Ridge OU 1. Indeed, karst development is widespread in the Knox Group across the ORR (Ketelle and Huff 1984) and is responsible for features such as sinkholes, dolines, and an extensive network of subsurface cavities. Drilling during monitoring well installation during Phase 1 of this RI encountered several karst features (Energy Systems 1991). Individual subsurface cavities develop independently but enlarge through solutional processes driven by groundwater and eventually coalesce into dominant flow channels somewhat analogous to a surface drainage system (Solomon et al. 1992).

The soil and residuum mantling the Knox Group in Chestnut Ridge OU 2 is of variable thickness because of the irregular bedrock surface. In general, the soil or residuum is a reddish-brown to yellow-orange plastic to very plastic clay with locally abundant chert. A rubble-rich zone with large blocks of bedrock in a matrix of the clay is commonly encountered just above the top of the bedrock (Geraghty & Miller 1989). Hatcher et al. (1992) point out that the Knox Group soils are typically either Paleudults or Hapludults, both of which are within the Ultisol order. Ultisols have a clay-enriched subsoil but are mostly highly weathered and have <35% base saturation at a depth of 1.8 to 2.0 m.

A number of investigators have studied the permeability of the soil and residuum underlying the Knox Group along Chestnut Ridge (Mishu 1982, Ketelle and Huff 1984, Woodward-Clyde 1984). The permeability has been evaluated both in the field and in the laboratory, and results indicate widely variable values. For example, falling head permeability tests in the field resulted in values of hydraulic conductivity ranging from $7.05 \times 10^{-10}$ to $6.35 \times 10^{-8}$ ft/day at varying depths (Mishu 1982, Woodward-Clyde 1984). Laboratory values range from $2.11 \times 10^{-11}$ to $1.41 \times 10^{-9}$ ft/day at one site, with the hydraulic conductivity decreasing with depth (Woodward-Clyde 1984).

## 2.5 GROUNDWATER

The characteristics, chemistry, and movement of groundwater on the ORR are a function of the varying hydrologic properties of the underlying geologic units. Two broad groupings can be distinguished: the Knox aquifer, consisting of the formation of the Knox Group (Copper Ridge Dolomite, Chepultepec Dolomite, Longview Dolomite, Kingsport Formation, and Mascot Dolomite) and the Maynardville Limestone of the Conasauga Group; and the ORR aquitards, consisting of the remaining geologic units on the ORR (Solomon et al. 1992).

Groundwater flow in the Knox aquifer is dominated by conduit flow through solution features resulting from karst processes, although flow also occurs through fractures, especially at greater depths. The Knox aquifer stores and transmits relatively large volumes of water; it is the only true aquifer on the ORR. Flow in the ORR aquitards, on the other hand, is dominated by flow through fractures. Most of the water in the aquitards is transmitted through a layer, ~3.3-16.4 ft (1-5 m) thick, of closely spaced, connected fractures near the water table (Solomon et al. 1992). The aquitards may store relatively large quantities of water, but transmit only limited amounts. Figure 2.6 shows the distribution of the Knox aquifer and the aquitards across the ORR.

The flow system in both the Knox aquifer and the ORR aquitards have been divided into four zones: storm flow, vadose, groundwater, and aquiclude (Solomon et al. 1992). The

stormflow zone is a thin region at the surface, which corresponds approximately to the root zone of vegetation; average thickness is 31.5-47.2 in. (80-120 cm). Most of the bulk permeability in the storm flow zone results from discrete permeable features referred to as macropores; these features account for only 0.2% of the soil volume but constitute 96% of the infiltration capacity (Solomon et al. 1992). The stormflow zone transmits transient, precipitation-generated flow that corresponds to ~90% of the total subsurface water flow. The stormflow zone is a major pathway for contaminant transport from the subsurface to surficial discharge points.

The vadose zone is an unsaturated zone above the water table, commonly within regolith. The regolith includes weathered and fresh rock in many places. Following recharge events, however, saturated flow commonly occurs within discrete features in the vadose zone (Solomon et al. 1992). For example, the cherty and inhomogeneous residuum of the Knox Group can create a network of connected openings through which saturated flow can occur.

The groundwater zone is continuously saturated and is the zone in which ~10% of the total subsurface water flow occurs. The permanent water table is near the transition from regolith to bedrock at depths ranging from ~0.3-148 ft (<1-45 m). Changes in hydraulic parameters and changes in major ion chemistry suggest a subdivision of the groundwater zone into three intervals: the water table interval, the intermediate interval, and the deep interval (Solomon et al. 1992). The aquiclude zone is the region below the deep groundwater zone interval. The water is a saline, $Na^+$-$Ca^+$-Cl brine, with total dissolved solids ranging from 2,000 to 275,000 mg/L. Water movement in the aquiclude, if any, occurs on a geologic time scale (Solomon et al. 1992)

The Copper Ridge Dolomite, the Chepultepec Dolomite, the Longview Dolomite, and the Kingsport Formation (described in Sect. 2.4), which are part of the Knox aquifer, underlie Chestnut Ridge OU 2. Groundwater flow at OU 2 can be subdivided into two systems: shallow flow in the residuum overlying bedrock and deep flow in bedrock within the Knox aquifer. Flow in the shallow system occurs within the storm flow zone, described above, and also along the residuum-bedrock interface, where it approximately follows topography. Shallow groundwater discharges primarily to springs along McCoy Branch. Estimates of hydraulic conductivities of shallow water-bearing soils along Chestnut Ridge range from $5.7 \times 10^{-3}$ to 1.8 ft/day (HSW 1991).

Deep groundwater flow at OU 2 occurs in bedrock within the Knox aquifer. As stated above, the Knox aquifer is the only true aquifer on the ORR, and it is the source of sustained natural flow in many perennial streams (Solomon et al. 1992). Karst features are common in the Knox aquifer at OU 2; numerous solution features were encountered during the installation of monitoring wells during the Phase 1 RI (Energy Systems 1991). North of the FCAP, groundwater flow in bedrock appears to be toward the FCAP; south of the FCAP, groundwater in bedrock appears to flow southward toward Bethel Valley (Fig. 2.7). Given the presence of karst fatures such as solution conduits in the Knox aquifer, however, the true flow direction of deep groundwater may differ significantly from directions evaluated from hydraulic head data. Groundwater flow in the Knox aquifier at depth (i.e., greater than 100 ft) occurs through fractures similar to flow in the aquitards.

Flow in the Knox aquifer is dominated by a few well-developed cavity systems. Two wells in the Knox aquifer have produced large quantities of water (>60 L/s), but most wells yield relatively small quantities (< 1 L/min). King and Haase (1988) investigated hydraulic

conductivity in the Knox aquifer along Chestnut Ridge. Their study employed straddle packer tests in 21 intervals selected from three core holes. All intervals were in the Copper Ridge dolomite, and depths ranged from 120 to 615 ft (37-197 m) below surface. Hydraulic conductivity values from the King and Haase study ranged from $2.59 \times 10^{-11}$ to $3.98 \times 10^{-7}$ ft/day.

Transmissivity values for the Knox aquifer suggest that the bulk permeability of the Knox is ten times greater than the ORR aquitards; however, the Knox is very permeable locally due to the presence of cavities and conduits and is therefore capable of transmitting large quantities of water. The potential groundwater flow path lengths in the Knox aquifer are much greater than in the ORR aquitards, with path lengths on the order of kilometers versus tens of or a few hundred meters in the aquitards (Solomon et al. 1992).

Two dye tracer studies have been performed at the adjacent Chestnut Ridge OU 1 to clarify the hydrologic system and groundwater gradient along Chestnut Ridge. The first test was conducted during the driest part of 1990. Before the test was started, a survey of potential discharge points was conducted. Forty monitoring points were established in both Bethel Valley and Bear Creek Valley, including streams, springs, and monitoring wells. Charcoal dye detectors were placed at each monitoring point. For 3 weeks before dye introduction, the points were monitored to collect background fluorescence data.

Ten kilograms of fluorescein dye mixed with potable water were introduced into well GW-178 on July 11, 1990. Detectors at the monitoring points were sampled over a 6-month period. Dye was positively recovered from five springs, one monitoring well, and six surface water sites. Several other monitoring points were identified as having a "possible" connection. The study concluded that flow is dominantly along strike (i.e., northeast-southwest) coincident with Chestnut Ridge itself, with a potential component of flow toward the east and southeast (Geraghty & Miller 1990).

On the basis of results from the first study, Energy Systems recommended that a second test be conducted. The purpose of the second test was twofold. First, the study was undertaken to verify the hydraulic continuity between the source well and groundwater discharge points identified in the first test as containing dye (SAIC 1992). Second, it was undertaken to conduct a test during the wet season to identify any potential differences in flow characteristics between wet and dry portions of the year.

The second test started in February 1992 with a 4-week baseline monitoring period of background fluorescence. GW-178 again was selected for dye injection; 34 potential discharge points, including springs, streams, and wells, were monitored with charcoal detectors. Dyes included 255 lb of Rhodamine WT and 246 lb of Burco Fluor AR solution, and an optical brightener equivalent to Fluorescent Brightener 28. Dye injection commenced on March 13, 1992, and monitoring was to continue for 12 weeks; the monitoring period was subsequently extended to 18 weeks because no definitive results were obtained. The test results were inconclusive. No results could be characterized on a qualitative basis as positively indicating the detection of dyes at monitoring locations (SAIC 1992). The investigators concluded that GW-178 may not be integrated with the karst system and that the dye might not be available for transport even after several months (SAIC 1992).

Water chemistry data from the adjacent Chestnut Ridge OU 1 (Haase et al. 1987) show that the major cations are calcium and magnesium; the calcium-magnesium ratio was typically

0.45 to 0.55. Major anions are carbonate-bicarbonate. The major cations and anions in the groundwater reflect the chemical composition of the underlying bedrock that consists of limestone and dolostone. The concentrations of major cations and anions result from processes that occur along groundwater flow paths, such as solution, clay, alteration, ion exchange, mixing, etc. (Solomon et al. 1992). Trace amounts of sulfate are present, along with a small but relatively constant silicon concentration. Specific conductance values range from 300 to 500 $\mu$mho/cm, and pH values range from 6.9 to 8.3, typical of Knox Group groundwater on the ORR.

Solomon et al. (1992) discuss chemical characteristics of the Knox aquifer and note that a calcium-magnesium-bicarbonate type of water exists to depths of nearly 984 ft (300 m) and that a transition to sodium-bicarbonate or sodium-chloride water, observed with depth in the ORR aquitards, is not present. The difference in major ion chemistry of the Knox aquifer as compared to the aquitards may result from differences in residence times and dominant geochemical processes (Solomon et al. 1992).

## 2.6 SURFACE WATER AND SEDIMENTS

The McCoy Branch watershed encompasses ~358 acres (Bogle and Turner 1989) and MCB itself runs from the top of Chestnut Ridge across the FCAP into Rogers Quarry, and out of the quarry where it runs a short distance into Melton Hill Lake at McCoy Embayment. The portion of McCoy Branch north of Rogers Quarry, within Chestnut Ridge OU 2, is termed Upper McCoy Branch (UMB), and the portion south of Rogers Quarry, within Chestnut Ridge OU 4, is termed Lower McCoy Branch.

The original course of McCoy Branch was altered by waste disposal activities on the ORR. The coal ash pond was built in 1955 to receive coal ash from the Y-12 Steam Plant and to serve as a settling basin. During the construction of the coal ash pond, a 62-ft-high earthen dam was built across UMB to impound the ash. Additionally, UMB was diverted into Rogers Quarry in 1967 as the pond filled and sediments and ash began to overflow the earthen dam. The National Pollutant Discharge Elimination System (NPDES) outfall at the southwest corner of the quarry is the point of origin of Lower McCoy Branch.

Below the FCAP dam, UMB generates much of its flow from a spring that originates at the toe of the dam. The path of UMB is gently sloping, with an average gradient of 7%. The valley is ~250 ft wide directly below the dam and 300 ft wide where UMB enters Rogers Quarry. The valley is grass covered and moderately to heavily wooded. UMB has incised a channel ~3 ft deep into the valley floor on its present course, although during periods of heavy rainfall it is capable of overflowing its banks. At the confluence of UMB and Rogers Quarry is a relatively flat-lying marshy area that remains wet throughout most of the year (Energy Systems 1991).

Several small tributaries flow into McCoy Branch; two of these are along UMB. In addition, several springs are present that feed UMB; these occur both north and south of the FCAP dam. Craig and Tschantz (1986) estimated that the combined baseflow of the springs below the dam to be in the range of 0.11 to 0.14 Mgd. Sediments exposed in the UMB streambed appear to consist of particles that are gravel size and smaller. The sediments include Chestnut Ridge residuum and coal ash.

Evidence of karst development also occurs along McCoy Branch. For example, MCB "disappears" and then reemerges downstream, much like sinking creeks described from many karst regions (Thornbury 1969). These features are related to the karst hydrologic system developed in the Knox Group, and the system may transport to or remove contaminants from the watershed.

Pulliam (1985) collected water quality data from 34 sites near the Y-12 Plant, 3 of which were from the McCoy Branch watershed. Two of the samples from McCoy Branch are from areas draining the Chickamauga Group, while the third sample is from an area draining the Knox Group. Samples were analyzed for various parameters, including temperature (12.0-16.5°C), specific conductance (200-360 μS/cm), pH (7.2-8.7), alkalinity (one sample, 96 mg/L as $CaCO_3$), dissolved oxygen (two samples, 9.0 and 12.6 mg/L), and instantaneous discharge (<0.01-3.1 $ft^3/s$).

The samples also were submitted to a laboratory for analysis of major constituents and properties, trace constituents and compounds, and radionuclides. With respect to the major constituents, detectable concentrations were found for calcium (26-41 mg/L), magnesium (7.9-13 mg/L), sodium (1.6-3.3 mg/L), potassium (0.80-4.5 mg/L), phosphorous (<0.0100080 mg/L), and solids residue (133-231 mg/L at 180°C). Most of the trace constituents have relatively low concentrations (i.e., <10 μg/L), with the exception of barium (100 μg/L), iron (250-1600 μg/L, lithium 10 and 90 μg/L), strontium (30-770 μg/L), and zinc (20 μg/L). Radionuclide analyses detected both gross alpha and gross beta concentrations, all at relatively low concentrations (highest gross alpha hit reported as 2.6 pCi/L; highest gross beta hit reported as 1.9 pCi/L).

## 2.7 ECOLOGY

Information contained in this section describes the ecosystem existing on UMB, the FCAP, and the sluice channel (Fig. 2.8). Habitats for all locations, including Rogers Quarry, are described relative to the flora found at each site (Sect. 2.7.1). It should be noted that all vegetation growth is voluntary and no revegetative work has been conducted on this site. Habitat is described also for White Oak Creek, a reference site for media toxicity testing and benthic macro-invertebrate surveys. Identification of potential wetlands (Sect. 2.7.2) and threatened, endangered or "in need of management" plant and animal species (Sect. 2.7.3) are discussed satisfying NEPA requirements. Fauna observed (Sect. 2.7.4) at locations on Chestnut Ridge OU 2 which may be impacted by contaminants in the environment are presented. This section is used to support the ecological risk assessment (Chap. 6) and supplement the biological monitoring data (Sect. 3.2.6).

### 2.7.1 Habitat Description

#### Rogers Quarry

Rogers Quarry was surveyed from outside the existing fence due to inaccessibility of the site at the time of the survey. The quarry has very steep banks with many areas of bare rock. Red cedar (*Juniperus virginiana*) occurs sporadically throughout this area. The quarry contains a deep pond with embankments too steep to support any significant vegetation communities along its edge (Hardy 1991).

### Upper McCoy Branch

UMB extends ~0.8 km from Rogers Quarry to the base of the earthen dam on the FCAP. Immediately north of Rogers Quarry is an open treeless area ~0.25 acre in size. Significant past disturbance is evident at this site. Upstream of the open area, black willow (*Salix nigra*) and cattail (*Typha latifolia*) dominate the vegetation along both sides of the stream. Winged loosestrife (*Lythrum alatum*) also is found here, representing the only known site on the ORR for this species. Continuing north along UMB, mixed deciduous forest vegetation is dominant with extensive kudzu (*Pueraria lobata*) immediately below the earthen dam (Hardy 1991).

Stream habitat was surveyed at two sites on UMB. These sites are located 2.03 km (MCK 2.03) and 2.53 km (MCK 2.53) from the mouth of the stream (Fig. 2.8). The downstream survey site on McCoy Branch (MCK 2.03) is located ~20 m upstream from the culvert that directs the flow to Rogers Quarry. Depth generally ranges from 7 to 25 cm and the width ranges from 0.9 to 1.2 m. The substrate is comprised of a mixture of gravel and medium to small rubble and fly ash deposits are more extensive and heavier than at MCK 2.53. Riparian vegetation on the east bank is extensive and consists primarily of mixed hardwoods. The west bank is lined with a thin layer of small trees and an extensive area of grasses. There are also thick deposits of ash along the banks and in the floodplain of the west bank. Canopy cover at this site is extensive, providing more than 80% coverage over the stream.

The upstream site on McCoy Branch (MCK 2.53) is located ~200 m downstream from the FCAP dam. This site consists primarily of shallow riffles and runs with little or no pool habitat present. This site is slightly more shallow (depth range from 2.5 to 7.6 cm) and narrow (width ranges from 0.61 to 0.91 m) than at the downstream site (MCK 2.03). Substrate is composed of a mixture of medium to small rubble and gravel and a fine layer of fly ash. As at the downstream site, riparian vegetation along the east bank is extensive and is comprised of a stand of mixed hardwoods. Riparian vegetation along the west bank is comprised of a thin line of hardwoods that are covered with heavy growths of kudzu. Canopy cover over the stream provides > 75% coverage.

### White Oak Creek

The White Oak Creek site (reference site for media toxicity testing and benthic macroinvertebrate surveys) (WCK 6.8) is located north of Bethel Valley Road and approximately due north of the ORNL 7000 area. This site, slightly wider than both McCoy Branch sites (~1.2-1.5 m) consists of a mixture of runs and riffles with a few shallow pools. The substrate consists of a mixture of medium to large rubble and gravel along with extensive areas of bedrock. A narrow, gravel access road and a few small hardwoods border much of the east bank, while a mixed hardwood forest borders the west bank. Canopy cover over the stream is ~50% to 75%.

### Filled Coal Ash Pond

**Sycamore Area/FCAP Dam.** The FCAP dam is covered with kudzu (*Pueraria lobata*). Directly behind the dam, a wooded area has become established on the fly ash substrate. Sycamore (*Platanus occidentalis*) is dominant in the overstory with occasional red maple (*Acer rubrum*), eastern red cedar (*Juniperus virginiana*) and cottonwood (*Populus deltoides*). Box

elder (*Acer negundo*) is abundant in the understory. Nepal microstegium (*Eulalia viminea*) is the dominant groundcover, with Japanese honeysuckle (*Lonicera japonica*) also abundant. An alder (*Alnus serrulata*) thicket occurs just north of the cottonwood stand. Lesser ladies-tresses orchid (*Spiranthes ovalis*) is found growing in this thicket. This is an unusual find, as this plant usually occurs in drier habitats and is also classified by the state of Tennessee as a Species of Special Concern (Hardy 1991).

**Willow Area.** North of the alder thicket, the FCAP opens into a wide, scrub/shrub and herbaceous community. The vegetation is dominated by a grass, joint-headed arthraxon (*Arthraxon hispidus*) (Hardy 1991). Other dominant species include alder (*Alnus serruluta*), willows (*Salix*), wool-grass (*Scirpus cyperinus*), soft rush (*Juncus effusus*), bulrush (*Scirpus*), Canada goldenrod (*Solidago canadensis*) and tick trefoil (*Desmodium*) (Rosensteel 1992). Horsetails (*Equisetu*) and sedges (*Carex*) also occur. Scattered willows and some alders are found throughout the area and along a well-defined stream course to the east side of the ash fill. The stream course appears to be a continuation of an existing natural stream that enters the ash fill area further up the valley. Its flow appeared to be intermittent. A less well-defined stream course was also observed on the west side of the ash fill area (Hardy 1991).

Small ponds are found in the northeast and northwest corners of the the FCAP. A floating aquatic species, Larger water-starwort (*Callitriche heterophylla*) dominates the vegetation on the northwest pond (Hardy 1991). This pond dries up completely during late summer. The larger pond located on the northeast corner of the area, is dominated by cattails (*Typha latifolia*) (Rosensteel 1992, Hardy 1991). This pond receeds in size to < 0.5 acre during late summer, but never completely disappears. Sedges surrounding the edge of this pond and Leafy pondweed (*Potamogeton foliosus*) dominate vegetation on the pond surface (Hardy 1991).

Remnants of the two tributaries that form the McCoy Branch headwaters are also located in the northeast and northwest corners of the FCAP. The upper section of the northwest tributary was the only area where mineral soil was found. This mineral soil is located under ~30 cm of coal ash and is considered hydric (Rosensteel 1992).

**Sluice Channel Area**

The Sluice Channel Area is located in a young to mature upland forest. Dominant larger trees in this secondary growth forest include shortleaf pine (*Pinus echinata*), red maple (*Acer rubrum*), and red cedar (*Juniperus virginiana*). Smaller trees occasionally present include sassafras (*Sassafras albidum*), winged elm (*Ulmus alata*), and redbud (*Cercis canadensis*). The groundcover is dominated by microstegium and Japanese honeysuckle. Less abundant groundcover species include Christmas fern (*Polystichum acrostichoides*), common blackberry (*Rubus allegheniensis*), catbrier (*Smilax* sp.) and aster (*Aster* sp.).

### 2.7.2 Wetland Assessment

Executive Order 11988, Floodplain Management, and the National Environmental Policy Act of 1969 (Public Law 91-190; 91 Stat. 852; 42 U.S.C. 4321-4347) require each federal agency to ensure that any action taken in a floodplain or wetland is evaluated and that impacts are minimized. To meet this requirement, a wetlands survey and assessment were prepared to describe proposed actions, discuss the effects of those actions on the floodplain

and wetlands, and consider possible alternatives to potential wetland and floodplain disturbance.

Results from preliminary wetland surveys conducted from July to September of 1991 (Hardy 1991) and in January 1992 (Rosensteel 1992) indicate that the two ponds located in the northeast and northwest corners of the FCAP meet the wetland requirements established by the *Federal Manual for Identification and Delineating Jurisdictional Wetlands* (Federal Interagency Committee for Wetland Delineation 1989). Hydrophytic (Obligate and Facultative Wetland species) vegetation (Table 2.1), hydric soils and wetland hydrology were present (Rosensteel 1992). Personnel from the TDEC surveyed the areas on February 5, 1993, and they suggest that over half the FCAP supports wetland vegetation and is hydrologically connected to the various waters of the state at the site (Childres 1993).

Four preliminary wetland areas for jurisdictional purposes were also identified downstream in Upper McCoy Branch (Fig. 2.9) (Rosensteel 1992). One of the wetlands (Area 1) is located in a level, open area on the west side of UMB, north of Rogers Quarry. This area was also identified in the 1991 survey. The vegetation is dominated by Obligate and Facultative Wetland species. The 0.25 acre (0.101 ha) area had standing water at the surface throughout the time of this survey. The vegetation evident during this survey include species listed in Table 2.1.

A wetland (Area 2) in the wooded, riparian zone of UMB was identified upstream of Area 1. This wetland is in a small spring seep area. The vegetation at this area is found in Table 2.1. The substrate is a mix of coal ash and hydric mineral soil. The mineral soil is a gray silt with orange mottles. Due to the input from springs, this area was saturated and had several inches of flowing water.

In addition to these two wetland areas, riparian wetlands were identified on narrow, level benches (Area 3) immediately adjacent to UMB. These bench wetlands are at an elevation only slightly higher than the stream and are situated between the edge of the stream and the base of the adjacent low, steep banks. The vegetation, hydrology, and substrate are similar to wetland Area 2.

A small, grass-dominated, emergent wetland (Area 4) was identified at the base of the FCAP dam where UMB water emerges from under the dam. Water several inches deep spreads laterally in this area before continuing flow downstream into the main channel of UMB. The grasses were not identified.

The Sluice Channel Area, which leads from the Y-12 Plant to the FCAP, includes a steep-sided, deeply eroded channel located on the upper slope of Chestnut Ridge. The only area of potential wetland is in a very small area on the upstream side of the dirt road that separates the channel from the FCAP to the south (Rosensteel 1992). There were no wetlands present in the Rogers Quarry area. Vegetation found in the Sluice Channel Area is presented in Table 2.1.

Currently the wetlands identified along UMB and the FCAP may or may not be classified as jurisdictional wetlands. Although all three wetland parameters (hydrophytic vegetation, hydric soils, and wetland hydrology) are present, the FCAP area lacks soils that meet the U.S. Army Corps of Engineers definition of hydric soils. Since FCAP soil consists of ash, the hydric soil parameter is technically not fulfilled (Rosensteel 1992). The Corps of Engineers will be

consulted to determine jurisdictional wetlands present within Chestnut Ridge OU 2. A wetlands survey was conducted in March 1994 by the Corps of Engineers, and this information will be included in the FS.

### 2.7.3 Threatened and Endangered Plant and Animal Species at FCAP and Upper McCoy Branch

Surveys of threatened and endangered plant (Hardy 1991) and animal species (Kroodsma 1993) were conducted in compliance with the Threatened and Endangered Species Act of 1973, 50 CFR Part 402. During these surveys, the FCAP and UMB areas were explored for plant or animal species that are listed or proposed for listing as endangered, threatened, or of special concern by the state of Tennessee and/or the U.S. Fish and Wildlife Service and for potential habitat for these species.

Results of the threatened and endangered plant survey indicated that only one state-listed species was found in the survey area. About ten lesser ladies-tresses orchids (*Spiranthes ovalis*) were found in the FCAP above the earthen dam. This inconspicuous orchid is classified by the state of Tennessee as a species of special concern and is not a federally listed species (Hardy 1991). Two additional species of plants have been discovered along Chestnut Ridge and are eligible for state listing. Torrey's mountain mint (*Pycnanthemum torrei*) has been found along the south slope of Chestnut Ridge and is eligible for listing as a state species of special concern. Whorled balm (*Collinsonia verticillata*), found on Chestnut Ridge, is listed by the Nature Conservancy as a globally threatened species and is being considered for state listing (Hardy et al. 1992).

Results of the threatened and endangered animal surveys on FCAP and UMB indicate that no threatened or endangered animal species (aquatic and terrestrial invertebrates and vertebrates) or critical habitat listed, or proposed to be listed, by the U.S. Fish and Wildlife Service is known to be present on the ORR (Kroodsma 1987) or the FCAP and UMB areas. However, the endangered Indiana bat (*Myotis sodalis*) is a possible summer resident along East Fork Poplar Creek, the headwaters of which are located north of Chestnut Ridge. Also, the bald eagle is an occasional nonbreeding visitor to Melton Hill Lake and the Clinch River. Several animal species listed by the state of Tennessee as threatened (T), endangered (E), or in need of management (INM) are known to occur on the ORR. Table 2.2 indicates the locations where threatened, endangered, or in need of management species may occur on the ORR. The Black vulture (INM species) has been observed in UMB and sluice areas and the Red-shouldered hawk (INM species) has been observed in UMB. There is potential habitat available on the FCAP, UMB, and sluice areas for visiting or nesting T or INM species present on the ORR. The Cooper's hawk, Red-shouldered hawk, Black vulture, and the common barn owl could possibly use areas in Chestnut Ridge OU 2. Any proposed project that would not affect habitats in these locations would not be expected to affect these species.

### 2.7.4 Biomonitoring Data

#### Upper McCoy Branch

Casual observations of various terrestrial and aquatic biota at UMB included many birds, deer, salamanders, crayfish and water snakes. Table 2.3 lists all species observed on various sampling occasions.

The UMB macroinvertebrate fauna was comprised of a mixture of insect and noninsect taxa. The predominant groups included midges (Chironomidae) and the taxa within the mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera). Other invertebrate taxa that were found during sampling were aquatic worms (Oligochaeta), crayfish (Decapoda), dragonflies and damselflies (Odonata), alderflies and dobsonflies (Megaloptera), elmid beetles, and blackflies (simuliid larvae) (see Sect. 3.2.6.2).

UMB had no established fish populations. Only four fish were found during 4 years of monitoring. The quarry outfall and inflow act functionally as a barrier to fish movement into the upper section of the stream from downstream fish communities in Lower McCoy Branch and Melton Hill Reservoir. Previous fly ash disposal techniques presumably eliminated any resident fish populations in UMB, while the quarry barrier has prevented any natural recolonization. Without a fish population, any measurement of recovery of the stream following cessation of fly ash disposal was limited to improved water quality and changes in benthic invertebrate communities. Therefore, as part of the Biological Monitoring and Abatement Program, a fish introduction study has been initiated on UMB.

The study involves transplanting banded sculpins (*Cottus carolinae*) into UMB and observing population parameters. These parameters include survival, distribution, density, biomass, condition factor, feeding patterns, reproduction, and recruitment. This study will determine if UMB can support a fish population and, if so, how that population might differ from an existing population in a noncontaminated reference stream. Data collection is still in progress, however, preliminary results are included in Sect. 3.2.6.6 and Appendix A.

### Filled Coal Ash Pond

**Sycamore Area.** Few soil invertebrates were found in the Sycamore Area of the FCAP. Surveys indicated an absence of annelids; however, a small number of beetle larvae and other arthropods such as millipedes were present in the ash. The small mammal population in this area (as indicated from 525 trap-nights) consisted primarily of field mice (*Peromyscus leucopus*) and chipmunks (*Tamias striatus*). Many raccoons (*Procyon lotor*) were also present (Table 2.4) (see Sect. 3.2.6.5).

**Willow Area.** At the Willow Area of the FCAP, 675 trap-nights indicated that eastern harvest mice (*Reithrodontomys humulus*) were dominant, followed by field mice. A fulvous harvest mouse (*Reithrodontomys fulvescens*) was also trapped. Raccoons were also abundant in this area (Table 2.4) (see Sect. 3.2.6.5). During the late summer months, white-tailed deer (*Odocoileus virginianus*) use the northeast and northwest ponds for mineral licks.

### Sluice Channel Area

Soil invertebrates found in the ash primarily consisted of arthropod larvae, centipedes and millipedes. Annelids also are absent in the ash at this location. Small mammals present (as indicated from trapping 480 trap-nights) included field mice (*Peromyscus leucopus*), short-tailed shrews (*Blarina brevicauda*), and pine voles (*Microtus pinetorum*). Trapping of raccoons was not performed at this location (Table 2.4) (see Sect. 3.2.6.5).

White-tailed deer were present on the FCAP and Sluice Channel Area, as observed through deer licks, deer beds, and direct observations. Tracks of coyotes (*Canis lations*) and mink (*Mustela vison*) have been observed in the northern area of the FCAP. A snapping

turtle (*Chelydra serpentina*) has been observed on two separate occasions in the northwest pond of the FCAP (Table 2.4).

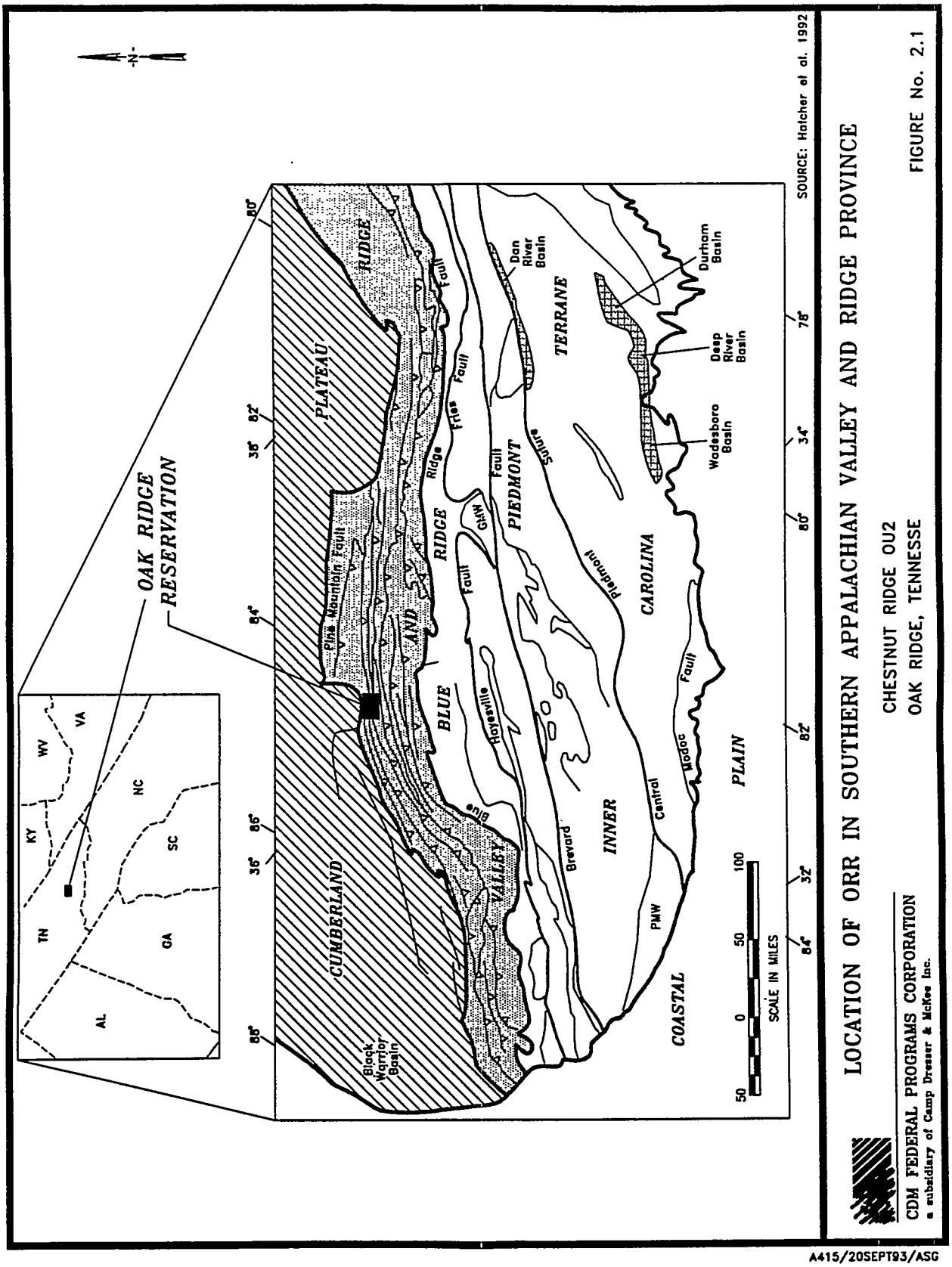



**Fig. 2.1. Location of the ORR in the Southern Appalachian Valley and the Ridge Province.**

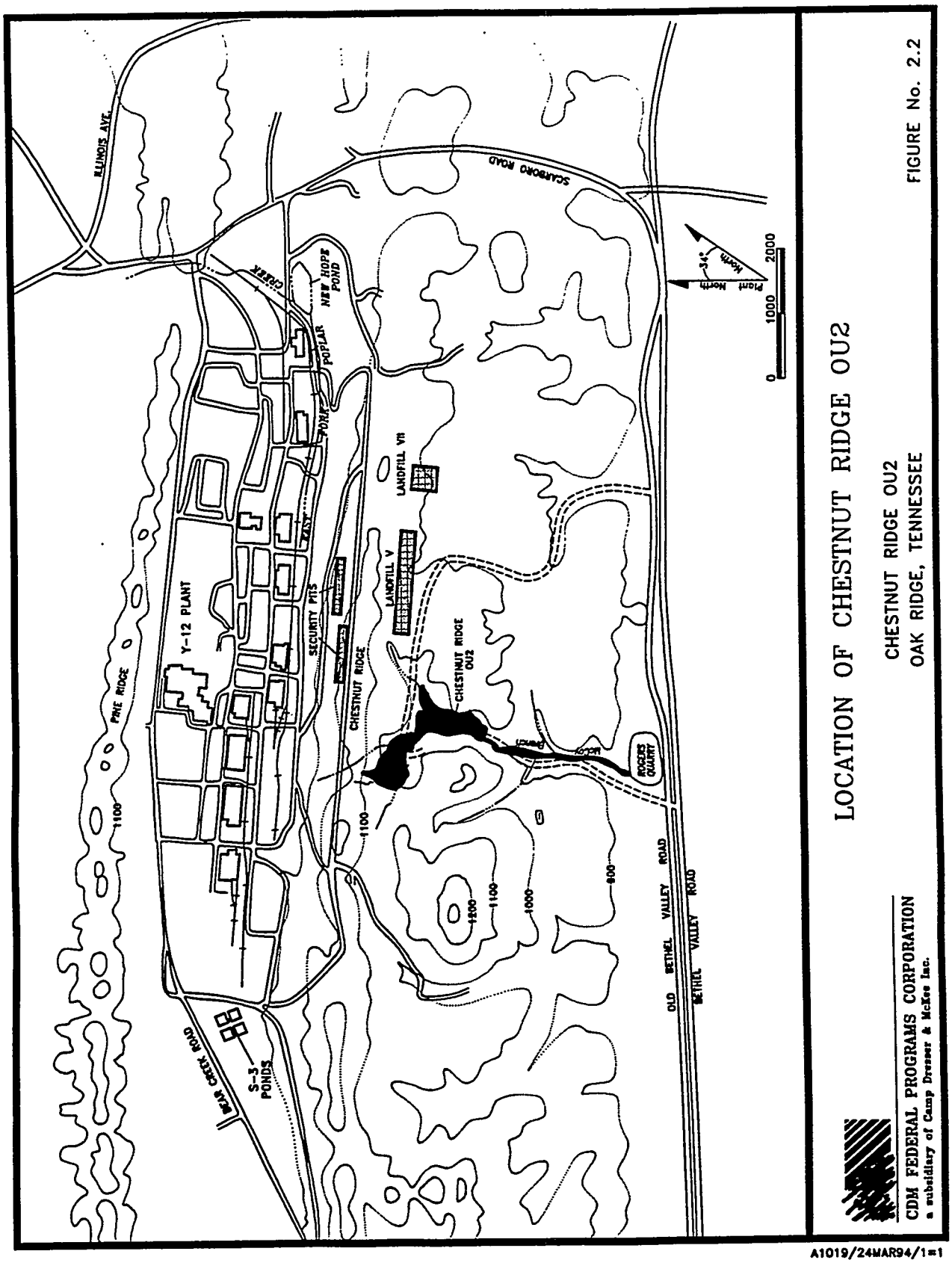
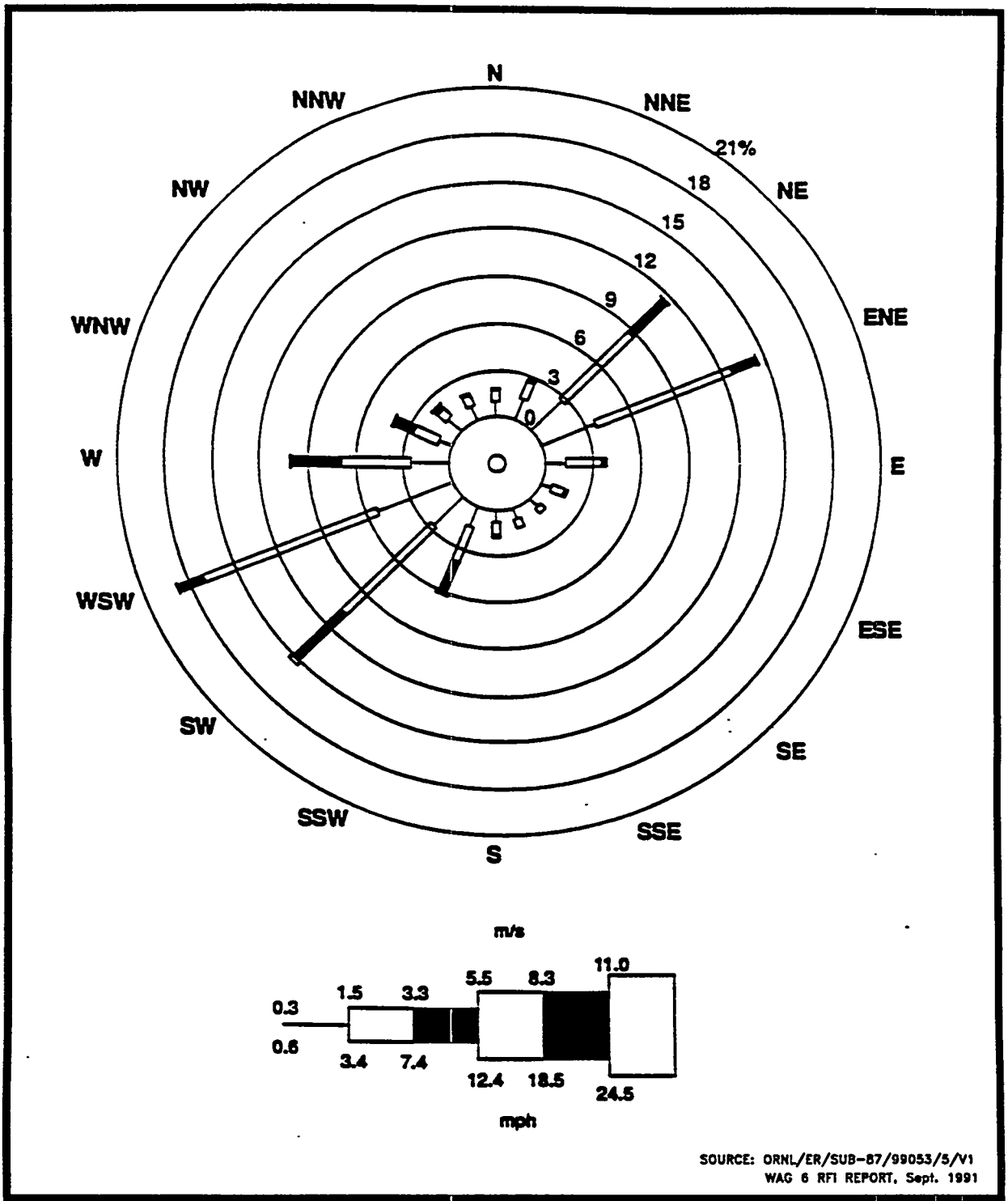



**Fig. 2.2 Location of Chestnut Ridge OU 2**




WIND ROSE DIAGRAM FOR THE OAK RIDGE RESERVATION

CDM FEDERAL PROGRAMS CORPORATION
a subsidiary of Camp Dresser & McKee Inc.

CHESTNUT RIDGE OU2
OAK RIDGE, TENNESSEE

FIGURE No. 2.3

A1012/23MA694/1=1

Fig. 2.3. Wind Rose diagram for the ORR.

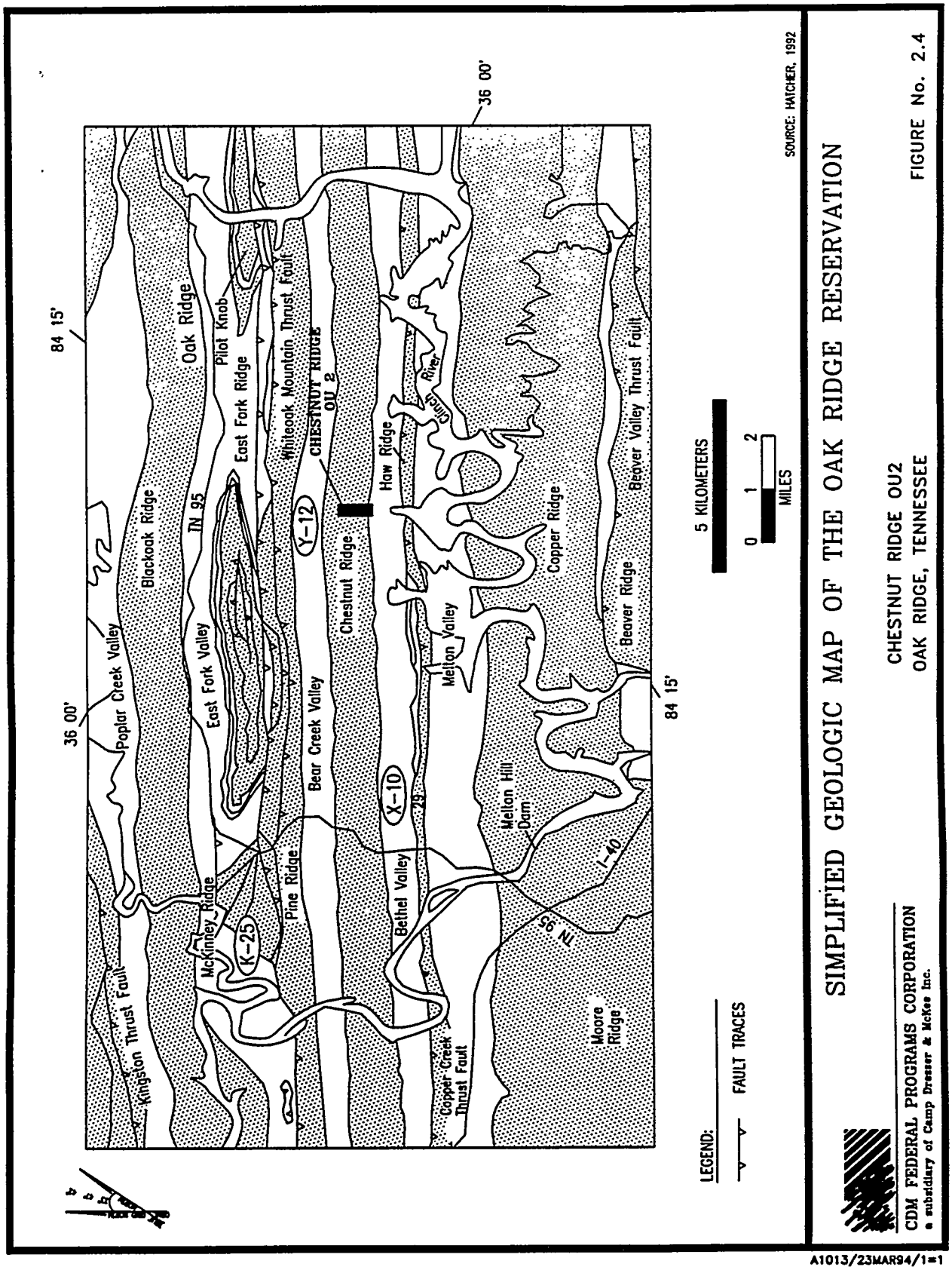
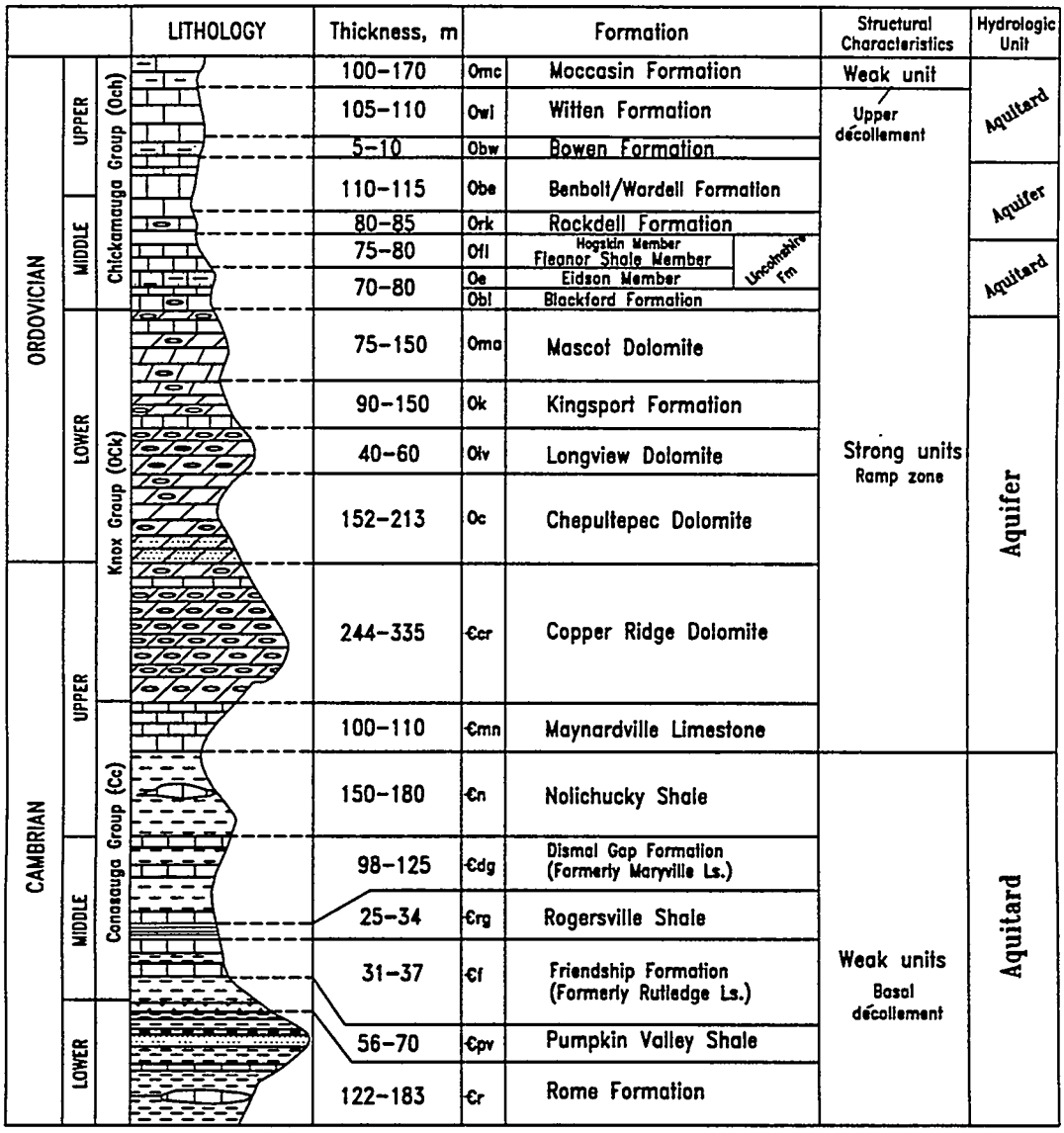



**Fig. 2.4. Simplified geologic map of the ORR.**




STRATIGRAPHIC SECTION IN THE VICINITY OF CHESTNUT RIDGE OU2, ORR

CDM FEDERAL PROGRAMS CORPORATION
a subsidiary of Camp Dresser & McKee Inc.

OAK RIDGE NATIONAL LABORTORY
OAK RIDGE, TENNESSEE

FIGURE No. 2.5

A1014/23MAR94/1=1

**Fig. 2.5. Stratigraphic section in the vicinity of Chestnut Ridge OU 2, ORR.**

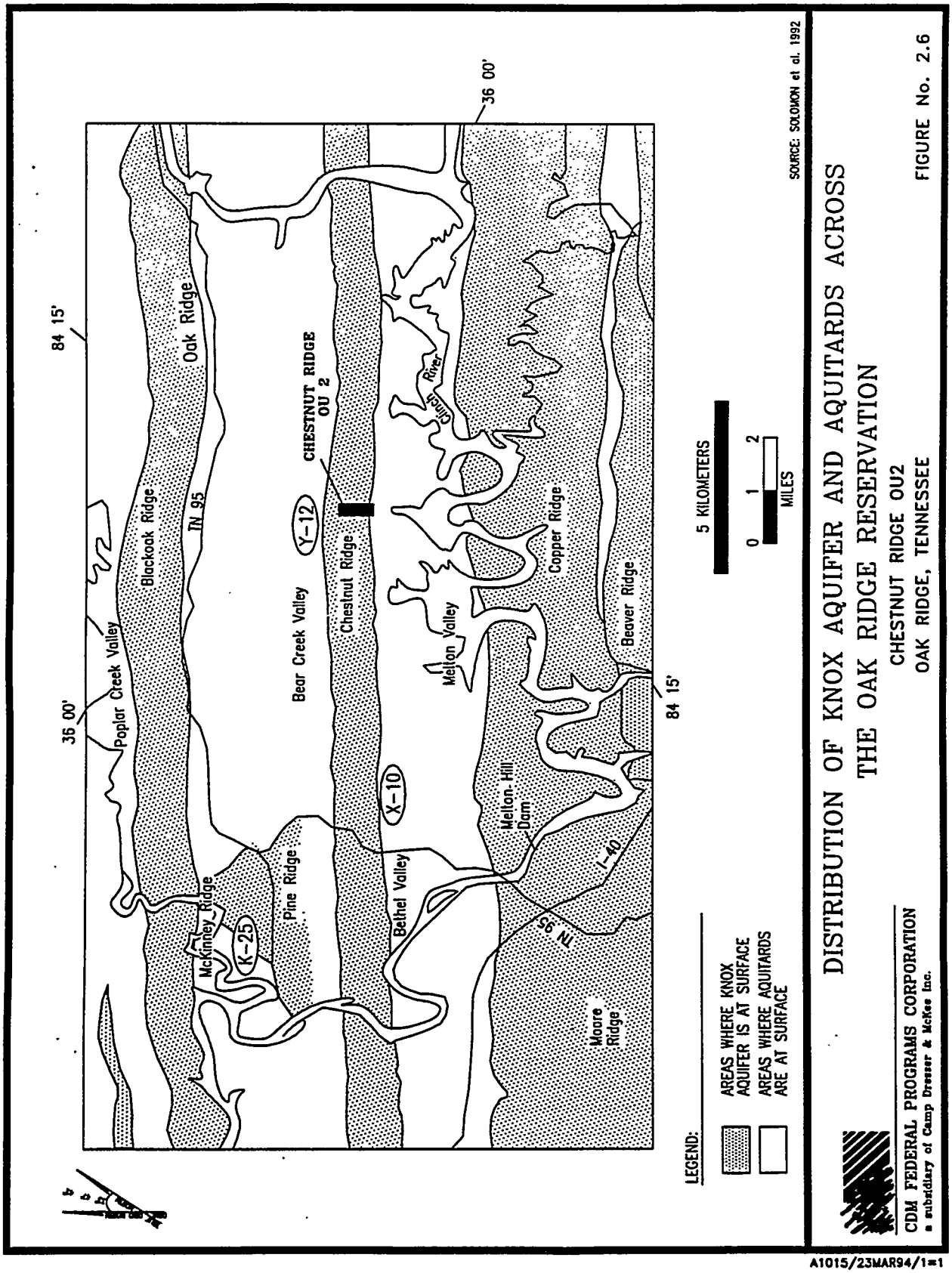



**Fig. 2.6. Distribution of Knox aquifer and aquitards across the ORR.**

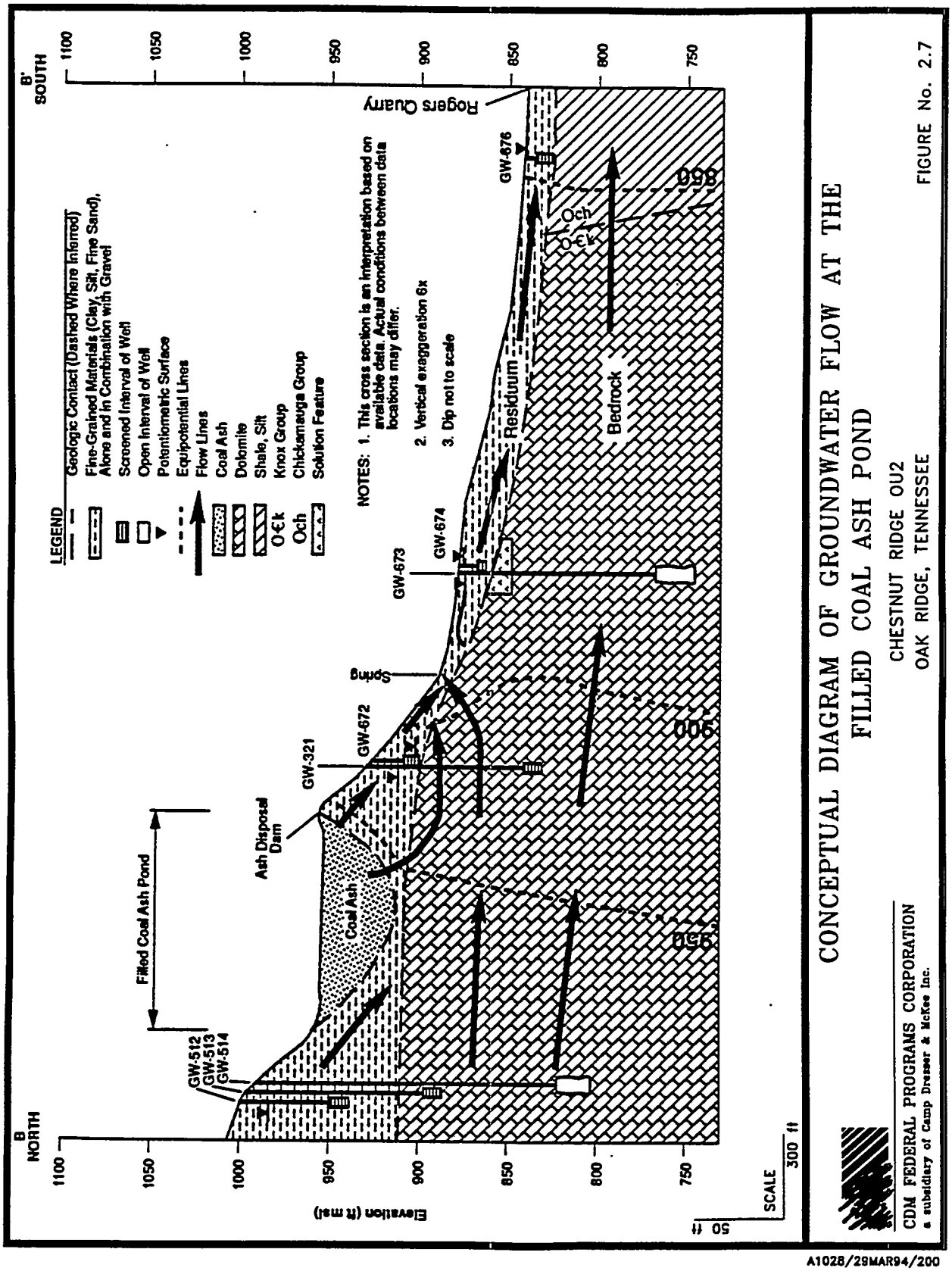



Fig. 2.7. Conceptual diagram of groundwater flow at the Filled Coal Ash Pond.

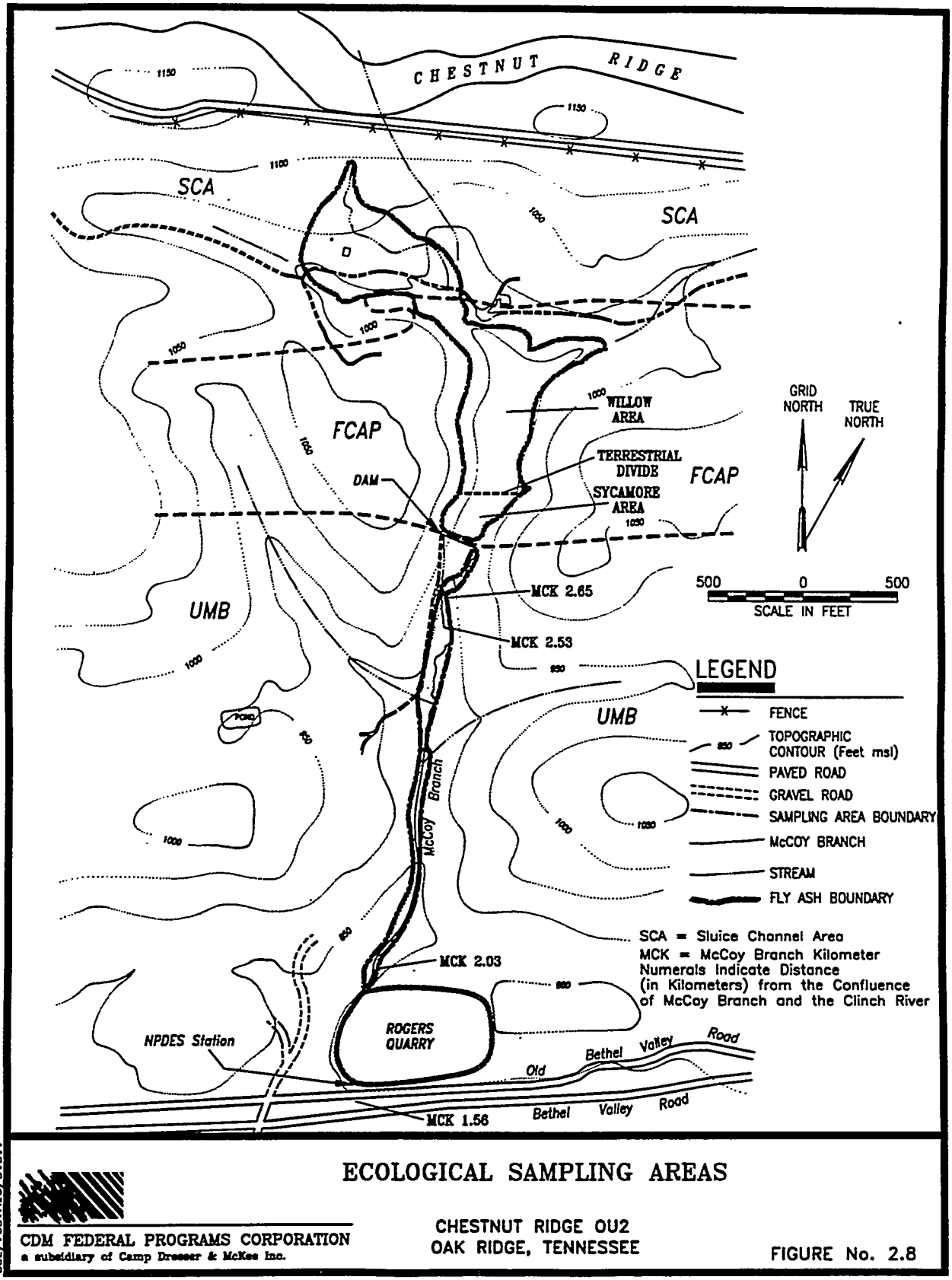



Fig. 2.8. Ecological sampling areas.

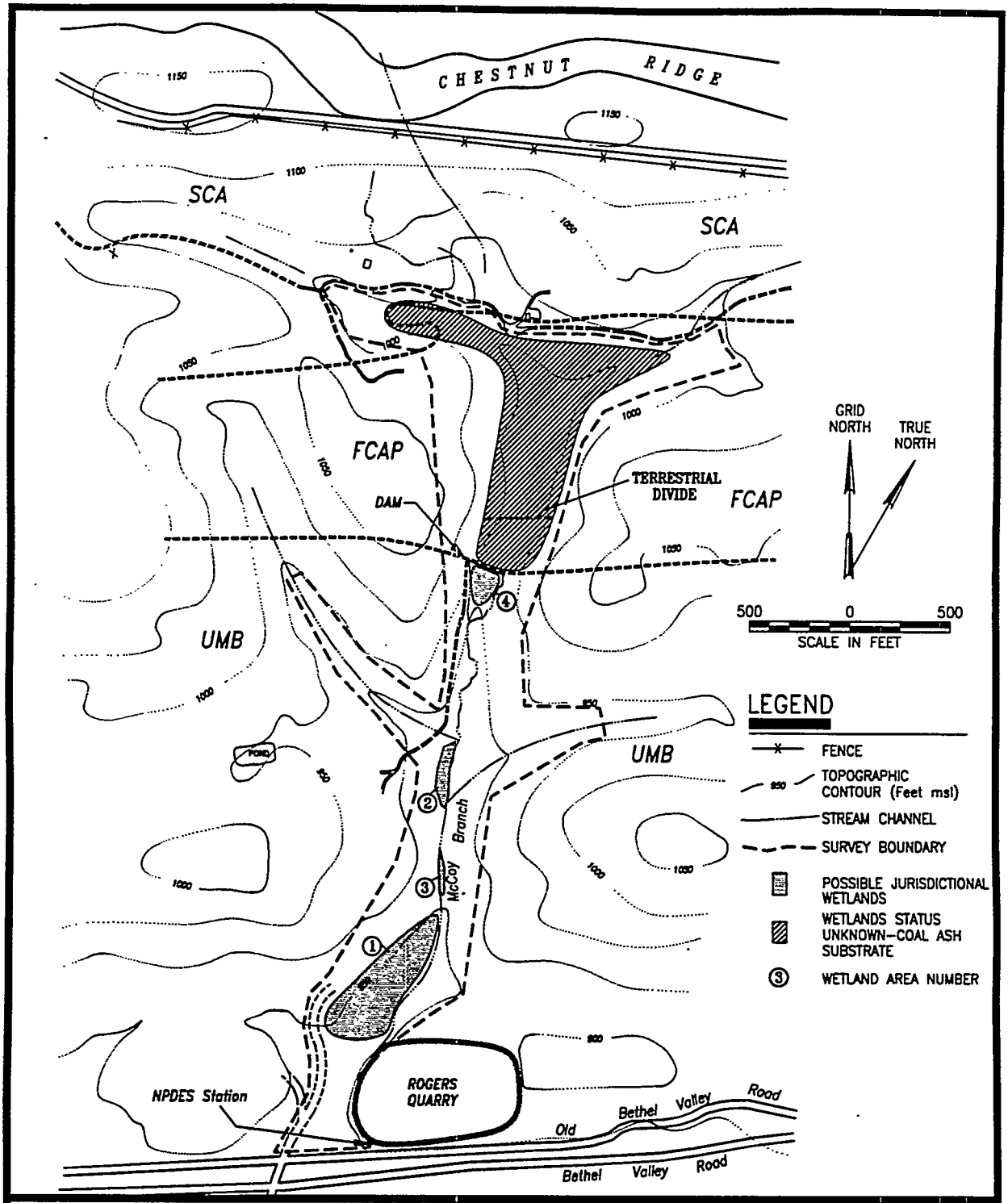



**Fig. 2.9. Possible jurisdictional wetlands.**

**Table 2.1. Abundant vegetation of Chestnut Ridge OU 2**

| Site | Species name | Wetland classification[a,b] regional indicator |
|---|---|---|
| **Sluice Channel Area** | | |
| | Shortleaf pine (*Pinus echinata*) | No Classification |
| | Red maple (*Acer rubrum*) | (NC) |
| | Eastern red cedar (*Juniperus virginiana*) | FAC |
| | Sassafras *Sassafras albidum* | FACU-[d] |
| | Winged elm (*Ulmus alata*) | FACU |
| | Eastern redbud (*Cercis canadensis*) | FACU+ |
| | Nepal microstegium (*Eulalia viminea*) | FACU |
| | Japanese honeysuckle (*Lonicera japonica*) | FAC+ |
| | Christmas fern (*Polystichum acrostichoides*) | FAC- |
| | Blackberry (*Rubus allegheniensis*) | FAC |
| | Catbrier (*Smilax* sp.)[c] | NC |
| | Aster (*Aster* sp.)[c] | FAC to OBL |
| | | FACU to OBL |
| East Subtributary | Sweet gum (*Liquidambar styraciflua*) | FAC+ |
| | Sycamore (*Platanus occidentalis*) | FACW- |
| | Tulip tree (*Liriodendron tulipifera*) | FAC |
| West Subtributary | Tulip tree (*Liriodendron tulipifera*) | FAC |
| | Sycamore (*Platanus occidentalis*) | FACW- |
| | Sweet gum (*Liquidambar styriciflua*) | FAC+ |
| | Pawpaw (*Asimina triloba*) | FAC |
| | Spicebush (*Lindera benzoin*) | FACW |
| | Dogwood (*Cornus florida*) | FACU |
| | Nepal microstegium (*Eulalia viminea*) | FAC+ |
| | Japanese honeysuckle (*Lonicera japonica*) | FAC- |
| | Wild grape (*Vitis* sp.)[c] | FAC- to FACW |
| **FCAP: Sycamore Area** | | |
| Earthen Dam | Kudzu (*Pueraria lobata*) | NC |
| Wooded Area | Sycamore (*Platanus occidentalis*) | FACW- |
| | Red maple (*Acer rubrum*) | FAC |
| | Box elder (*Acer negundo*) | FACW |
| | Eastern red cedar (*Juniperus virginiana*) | FACU- |
| | Nepal microstegium (*Eulalia viminea*) | FAC+ |
| | Cottonwood (*Populus deltoides*) | FAC+ |
| | Japanese honeysuckle (*Lonicera japonica*) | FAC- |
| | Lesser ladies-tresses orchid (*Spiranthes ovalis*) | FAC |
| Alder Thicket | Brook-side alder (*Alnus serruluta*) | FACW+ |
| | Lesser ladies-tresses orchid (*Spiranthes ovalis*) | FAC |

Table 2.1 (continued)

| Site | Species name | Wetland classification[a,b] regional indicator |
|---|---|---|
| **FCAP: Willow Area** | | |
| Open Ash-Fill Area | Joint-head Arthraxon (*Arthraxon hispidus*)<br>Sedges (*Carex* sp.)[c]<br>Horsetails (*Equisetum* sp.)[c]<br>Willows (*Salix* sp.)[c]<br>Bulrush (*Scirpus* sp.)[c]<br>Soft rush (*Juncus effusus*)<br>Wool-grass (*Scirpus cyperinus*)<br>Canada golden-rod (*Solidago canadensis*)<br>Tick-trefoil (*Desmodium* sp.)[c]<br>Little bluestem (*Schizachyrium scoparium*) or<br>Broom-sedge (*Andropogon virginicus*) | FACU+<br>OBL, FAC<br>FAC or FAC+<br>FACW or OBL<br>FACW+ or OBL<br>FACW+<br>OBL<br>FACU<br>FAC- or FACU<br>FACU<br>FAC- |
| Norhwest Pond | Water-starwort (*Callitriche heterophylla*) | OBL |
| Northeast Pond | Leafy pondweed (*Potamogeton foliosus*)<br>Broad-leaf cattail (*Typha latifolia*) | OBL<br>OBL |
| **Upper McCoy Branch** | Kudzu (*Pueraria lobata*)<br>Black willow (*Salix nigra*)<br>Broad-leaf cattail (*Typha latifolia*)<br>Winged loosestrife (*Lythrum alatum*) | NC<br>OBL<br>OBL<br>FACW+ |
| Area 1 | Black willow (*Salix nigra*)<br>Brook-side alder (*Alnus serrulata*)<br>Bulrush (*Scirpus* sp.)[c]<br>Soft rush (*Juncus effusus*)<br>Wool-grass (*Scirpus cyperinus*)<br>Alleghany monkey-flower (*Mimulus ringens*)<br>Winged loosestrife (*Lythrum alatum*[c]) | OBL<br>FACW+<br>FACW or OBL<br>FACW+<br>OBL<br>OBL<br>FACW+ |
| Areas 2 and 3 (in riparian zone) | Sycamore (*Platanus occidentalis*)<br>Brook-side alder (*Alnus serrulata*)<br>American hornbeam (*Carpinus caroliniana*)<br>Nepal microstegium (*Eulalia viminea*) | FACW-<br>FACW+<br>FAC<br>FAC+ |

Table 2.1 (continued)

| Site | Species name | Wetland classification[a,b] regional indicator |
|---|---|---|
| **Rogers Quarry Area** | Eastern red cedar (*Juniperus virginiana*) | FACU- |

[a]*National List of Plant Species that Occur in Wetlands: Tennessee*. May 1988 Biological Report, Reed Porter Jr., U.S. Fish and Wildlife Service, NERC-88/18.42 (USFWS 1988).

[b]OBL = Obligate Wetland (occur almost always; estimated probability > 99% under natural conditions in wetlands); FACW = Facultative Wetland (usually occur in wetlands; estimated probability 67%-99%, but occasionally found in nonwetlands); FAC = Facultative (equally likely to occur in wetlands or nonwetlands; estimated probability 34%-66%; FACU = Facultative Upland (usually occurs in nonwetlands; estimated probability 67%-99%, but occasionally found in wetlands; probability 1-33%;

[c]Specific Regional Indicator for wetland classification could not be determined for vegetation based solely on genus. Therefore, range of classification for genus is listed.

[d]A positive or negative sign was used with the Facultative Indicator categories to more specifically define the regional frequency of occurrence in wetlands. The positive sign indicates a frequency toward the higher end of the category (more frequently found in wetlands), and a negative sign indicates a frequency toward the lower end of the category (less frequently found in wetlands).

[e]Species identified in 1991 wetland survey.

Table 2.2. Threatened and endangered species on the Oak Ridge Reservation

| Species | Status | Possible location |
|---|---|---|
| Indiana bat[a] | E[b] | Vicinity of East Fork Poplar Creek |
| Bald Eagle | E | Melton Hill Lake and Clinch River |
| Tennessee dace | INM[c] | Bear Creek drainage, Ish Creek, and tributaries of East Fork Poplar Creek (siltation due to erosion in the vicinity of the creeks must be avoided) |
| Osprey | E | Vicinity of Melton Hill Lake and Clinch River |
| Sharp-shinned hawk | E | Area between Jones Island, Grubb Island, and 500-kV power line |
| Cooper's hawk | T[d] | Entire Oak Ridge Reservation |
| Grasshopper sparrow | T | Large grassy fields |
| Black-crowned night-heron | INM | Clinch River |
| Black vulture | INM | Entire Oak Ridge Reservation; have been observed in UMB and Sluice Channel Area |
| Red-shouldered hawk | INM | Forests and fields along the Clinch River and relatively large streams. Have been observed in UMB |
| Common barn owl | INM | Freel's bend and vicinity |

[a]Indiana bat and bald eagle are listed by the U.S. Fish and Wildlife Service. All other species are listed by the state of Tennessee;
[b]Endangered;
[c]In need of management;
[d]Threatened.

Table 2.3. Biota sightings on Upper McCoy Branch

| Bird sightings | Miscellaneous biota sightings |
|---|---|
| Black vulture (*Coragyps atratus*) | Watersnakes (*Natrix* sp.) |
| Turkey vulture (*Cathartes aura*) | Whitetail deer (*Odocoileus virginianus*) |
| Cedar waxwing (*Bombycilla cedrorum*) | Northern two-lined salamander (*Eurycea bislineata*) |
| Swamp sparrow (*Melospiza georgiana*) | Crayfish (*Cambarus*) |
| Northern cardinal (*Cardinalis cardinalis*) | |
| Magnolia warbler (*Dendroica magnolia*) | |
| American Crow (*Corvus brachyrynchos*) | |
| Blue jay (*Cyanocitta cristata*) | |
| Song sparrow (*Melospiza melodia*) | |
| Common yellow-throated warbler (*Dendroica dominica*) | |
| Red-shouldered hawk (*Buteo lineatus*) | |
| Pileated woodpecker (*Dryocopus pileatus*) | |
| Ruby-crowned kinglet (*Regulus calendula*) | |
| Eastern bluebird (*Sialia sialis*) | |
| White-throated sparrow (*Zonotrichia albicollis*) | |
| Yellow-rumped warbler (*Dendroic coronata*) | |
| Rufous-sided Towhee (*Pipilo erythnophthalmus*) | |
| Mourning dove (*Zenaida macroura*) | |
| American kestral (*Falco sparverius*) | |
| Broad-winged hawk (*Buteo platýpterus*) | |
| Northern flicker (*Colaptes auratus*) | |
| Tufted titmouse (*Parus bicolor*) | |
| Red-winged blackbird (*Agelaius phoeniceus*) | |
| Indigo bunting (*Passerina cyanea*) | |
| Carolina chickadee (*Parus carolinesis*) | |

Table 2.4. Biota sightings on the FCAP and Sluice Channel Area

| Site | Biota sightings[a] |
|---|---|
| **FCAP: Sycamore Area** | White-footed mice (*Peromyscus leucopus*) |
| | Chipmunk (*Tamias striatus*) |
| | Raccoons (*Procyon lotor*) |
| | Millipedes (Diplopoda) and beetle larvae (Coleoptera) (minimal in ash) |
| **FCAP: Willow Area** | Eastern Harvest mice (*Reithrodontomys humulus*) |
| | White-footed mice |
| | Fulvous Harvest mouse (*Reithrodontomys fulvescens*) |
| | Raccoons |
| | Coyote (*Canis lations*) |
| | Mink (*Mustela vison)* |
| | Snapping turtle (*Chelydra serpentina*) (northwest pond) |
| **Sluice Channel Area** | White-footed mice |
| | Short-tailed shrew (*Blarina brevicauda*) |
| | Pine vole (*Microtus pinetorum*) |
| | Millipedes, centipedes, arthropod larvae (minimal in ash) |

[a]Biota sightings from direct observations, animal signs (tracks), or biomonitoring data.

# 3. SITE HISTORY AND CURRENT CONDITIONS

## 3.1 SITE DESCRIPTION

### 3.1.1 Geographic Information

Chestnut Ridge OU 2 is located within the DOE's ORR in Anderson County, Tennessee, ~24 miles west of Knoxville. The OU is an 8.5-acre impoundment area on the southern slope of Chestnut Ridge, 0.5 mile south of the main Y-12 Plant and geographically separated from the Y-12 Plant by Chestnut Ridge (Figs. 2.1 and 2.2). Elevation at the top of the dam is 950 ft, and elevation at the ash slurry pipe discharge atop the ridge is 1100 ft. The ash pond is fairly flat and vegetation is slight to heavy across the pond and the dam.

The 100-year floodplain of Upper East Fork Poplar Creek occurs within the Y-12 Plant boundary but is confined by Chestnut Ridge to the southeast and does not extend above an elevation of 910 ft. OU 2 is situated in the McCoy Branch Watershed, which is not hydrologically connected to EFPC. Perennial streams on the southeast flank of Chestnut Ridge (e.g., McCoy Branch) have a relatively steep gradient and have not developed a discernible floodplain. During heavy storm events, any overbank flow from these streams would accumulate in Bethel Valley, where the base elevation is approximately 240 ft below the elevation of the FCAP.

A number of waste management sites are situated along Chestnut Ridge. The closest sites to OU 2 are the Chestnut Ridge Security Pits, approximately 1/4 mile to the northeast; the East Chestnut Ridge Waste Pile and Landfill V, approximately 3/4 mile to the northeast; the United Nuclear Corporation Site, approximately 1 mile southeast along Chestnut Ridge; and the Steam Plant Ash Disposal and Landfill VII, ~1 1/2 miles to the northeast. Existing data from the FCAP (discussed in sections that follow) do not suggest that these nearby sites are potential contaminant sources to OU 2.

### 3.1.2 Background

The DOE Oak Ridge Y-12 Plant disposed of coal ash from its steam plant operations as a slurry that was discharged into an ash retention impoundment. This impoundment is the FCAP. The FCAP was constructed in 1955 by building an earthen dam across the northern tributary of McCoy Branch. By July 1967, ash had consumed the impoundment storage behind the dam to within 4 ft of the top. As the sediment trap efficiency decreased, the slurry was released directly into UMB (Tschantz 1986). In 1967 and 1968, UMB was diverted into Rogers Quarry. Ash continued to flow through the FCAP and UMB to the quarry until 1989, when a bypass pipe was constructed to carry it directly from the steam plant to Rogers Quarry.

### 3.1.3 Operational Information

The Y-12 Plant operates a coal-fired steam plant. Coal used in the plant is from the eastern United States, generally Tennessee and Kentucky. The coal ash from the steam plant operations was disposed as a slurry into an impoundment constructed in a natural drainway. This impoundment is the FCAP.

During construction of the earthen dam across the northern tributary of McCoy Branch, a principal spillway consisting of a standpipe and 36-in.-diameter reinforced concrete pipe was installed. An emergency spillway was constructed by excavating a channel into the consolidated bank at the eastern abutment of the dam. The dam was designed to provide a storage volume of approximately 129-acre-ft of ash slurry and was expected to have sufficient capacity to hold 20 years of Y-12 steam plant ash; however, storage capacity was reached in 12-13 years. Vegetation was not grubbed or removed from within the impoundment, except as required to accomplish construction activities.

The ash slurry was piped to the crest of Chestnut Ridge and released to gravity flow approximately 1000 ft to the impoundment pool. The area of overland ash slurry flow from the pipe discharge to the impoundment is the Sluice Channel Area. Flow cut a channel in the natural terrain that conveyed the slurry to the FCAP.

Coal ash was intermittently sluiced from the steam plant by flushing ash from the furnace bottoms (bottom ash) and bag house hoppers (fly ash) with potable water. The fly ash was removed using a wet collection system, and the bottom ash is collected from the bottom boiler in wet form. About 20% of the ash produced is bottom ash (Turner et al. 1986). Until November 1989, the ash sluice water was pumped through an 8-in.-diameter iron pipe to the crest of Chestnut Ridge and discharged. In November 1989, additional pipeline was installed to bypass McCoy Branch and the coal ash pond altogether by piping the ash wash directly into Rogers Quarry (Turner et al. 1993).

In recent years, several steps have been taken to reduce the amount of ash discharged. First, the plant started using a higher grade of washed coal in 1986. This change increased the burning efficiency and decreased the total volume of ash. Second, the plant began using natural gas as a fuel in the winter of 1988. The goal is to use gas for 80% of the fuel needs at the plant. Third, a chemical vacuum system was installed in 1990 to collect fly ash. Fly ash is now disposed in a landfill and is no longer discharged to Rogers Quarry. Finally, a bottom ash dewatering system has been installed at the plant. The bottom ash is also put into a landfill, washdown water is discharged through the Y-12 Plant wastewater treatment system, and all discharges into Rogers Quarry have stopped (Turner et al. 1986).

All discharges from the steam plant to FCAP stopped in 1989 when the bypass pipe was installed. Ash deposits behind the dam, including those above the pond and along the McCoy Branch stream channel below the dam, have been left in place. Since ash deposition in these areas ceased, the site has become heavily vegetated.

### 3.1.4 Releases

By July 1967, ash had filled up the impoundment storage behind the 64-ft-high dam to within 4 ft of the dam crest. As the settling efficiency of the impoundment decreased, the slurry began to flow over the settled residue in the impoundment to the emergency spillway and into McCoy Branch (Tschantz 1986). In 1967 and 1968, UMB was diverted into Rogers Quarry, located about 0.5 mile downstream from the dam. Ash slurry continued to flow across the impoundment into McCoy Branch until 1989, when a pipeline was constructed around the filled coal ash pond to Rogers Quarry.

Significant erosion of the spillway has occurred cutting into the dam, and an erosion channel several feet wide has appeared in the ash itself. These conditions have led to slow, intermittent releases of ash into UMB.

## 3.2 SUMMARY OF EXISTING DATA

The remedial investigation was conducted in two phases. The Phase 1 effort was conducted by CH2M HILL in 1990. Surface waters, soils and ash sampling was limited to the UMB zone in Phase 1. The Phase 2 investigation, conducted by CDM Federal, was designed to fill data gaps about the surface waters, soils, and ash in the FCAP and Sluice Channel Area zones. Groundwater monitoring wells were sampled during both phases of the investigation. The Sluice Channel Area, FCAP, and UMB zones of OU 2 are delineated in Fig. 3.1. The location of the ash disposal area in relation to White Oak Creek and Walker Branch, the reference surface water and sediment sampling sites, is shown in Fig. 3.2. The complete listing of the analytical results from the Phase 1 effort are included in Appendix B; analytical results from the Phase 2 effort are included in Appendix C. Because the Phase 1 and Phase 2 data are not entirely comparable, the discussions and presentations of the data are separated within the following sections.

This section presents the available data concerning the nature and extent of the contamination at the Chestnut Ridge OU 2 site. Since the composition and distribution of the ash is fundamental to ascertaining the nature and extent of contamination, coal ash is discussed in Sect. 3.2.1 before the discussions on the contaminated environmental media. Surface water data, sediment data, groundwater data, soil data, and biomonitoring data are discussed in Sects. 3.2.2, 3.2.3, 3.2.4, 3.2.5, and 3.2.6, respectively.

Radiological survey data and air data were not collected. Historical site information did not indicate a significant source of external exposure to radiation, and direct analysis of the soil samples for radiological COPCs was planned, therefore, the information available through a surficial radiological survey would be of no additional benefit. Air monitoring was not performed because, as the conceptual model for the site (Fig. 5.2) indicates, any air contamination would be a result of soil and/or water contamination, through dust or volatilization, and both these media were already to be sampled and analyzed.

### 3.2.1 Coal Ash Data

Coal ash samples were collected from UMB during the Phase 1 effort and from the FCAP and the Sluice Channel Area during the Phase 2 effort. Five ash samples were collected in Phase 1 from sample points A2 through A6 along UMB as shown in Fig. 3.3. Coal ash samples were obtained in the FCAP area by boring through the ash to the native soil at the locations shown in Fig. 3.4. The borings ranged from 17 ft to 38 ft in depth. The saturated, slurry nature of the ash approximately 3 ft below the surface made the collection of discrete interval samples impossible. Each of the nine ash borings produced a single composite sample for analysis. Three composite hand auger samples of the ash in the Sluice Channel Area were collected for analysis. The Sluice Channel Area composite samples were comprised of three aliquots collected along a transect. The Sluice Channel Area ash sampling locations are shown in Fig. 3.5. The list of analytes for which the Phase 1 coal ash samples were analyzed is presented in Table 3.1. The Phase 2 list of coal ash analytes is in Table 3.2.

Coal ash samples from deer lick sites were not collected. Attempts to locate active deer lick sites were unsuccessful. The previously noted deer licks were obliterated by the weather and the deer were not actively using the site for mineral licks during the Phase 2 field effort.

The field pH data for the ash, reference soils, and sediments are presented in Table 3.3. The pH data ranged from 5.0 to 8.4. The ash samples in the FCAP borings tended to have higher pH values than the rest of the samples, suggesting that the fly-ash-to-bottom-ash ratio is higher in the FCAP than elsewhere in the OU. The analytical results of the ash are presented in Tables 3.4 and 3.5. The ash contains high levels of metals and low estimated levels of three organic compounds. Three ash samples were analyzed for pesticides and polychlorinated biphenyls (PCBs). Three qualified pesticide results were reported above the instrument detection limit but below the practical quantitation limit. The maximum concentration reported for each analyte is shown in bold type in the tables.

The chemical analysis of the OU 2 coal ash is typical of coal ash from the combustion of eastern United States coals. The ash radiological results are also consistent with typical coal ash composition and not the result of augmentation by Y-12 Plant processes. As shown in Table 3.6 the analytical results of the OU 2 coal ash are within the range of eastern United States coal ash for all the parameters reported.

The quantity and distribution of the coal ash was determined by measuring the depth and lateral extent of the ash along surveyed transects. The Phase 1 effort established transects perpendicular to UMB, as shown in Fig. 3.6, and measured the depth of the ash along them. Cross-sections developed from the field measurements are presented in Appendix D.

The Phase 2 effort established transects perpendicular to the dam in the FCAP zone and measured the depth of the ash in nine boreholes (see Fig. 3.4). Also in the Phase 2 effort, transects were established in the Sluice Channel Area zone perpendicular to the channel (see Fig. 3.5). The depth and extent of the ash were measured along the sluice channel transects. Cross-sections of the Sluice Channel Area ash are presented in Appendix E.

It is estimated from the measurements of the ash depth and areal extent that there are ~236,000 $yd^3$ of coal ash present in the FCAP, 6000 $yd^3$ in the Sluice Channel Area, and 4300 $yd^3$ in UMB (Radian 1993). The total ash volume is estimated to be 246,300 $yd^3$.

### 3.2.2 Surface Water Data

Surface waters were characterized by sampling and analysis in Phases 1 and 2 of the remedial investigation. During Phase 1, surface waters were sampled at three points along UMB. The Phase 2 investigation included surface water samples at 12 locations: 3 points along UMB, 3 points at the north end of the FCAP, 2 reference locations in Walker Branch, and 1 reference location in White Oak Creek. The surface water sampling locations within OU 2 are shown on Fig. 3.7. The reference surface water sample points in White Oak Creek and Walker Branch are shown on Figs. 3.8 and 3.9, respectively. The analytes for Phase 1 and Phase 2 surface water samples are listed in Tables 3.7 and 3.8, respectively.

The Phase 1 sample locations were sampled twice, during dry weather (July 30, 1990) and during wet weather (October 8, 1990). Volatile and semivolatile organics were analyzed in the dry weather samples only. Phase 1 surface water analytical results are presented in Table 3.9. Only those parameters that were detected above the detection limit in at least one of the

samples are contained in Table 3.9. The values that exceed the ambient water quality standards are shown in bold in the table. There is a clear trend of reduced contaminant concentrations with distance downstream.

The Phase 2 surface water sample points were sampled once, except for SW-3 (which was sampled three times) and SW-12 (which was sampled twice). SW-3 samples are of the leachate from the bottom of the FCAP. The spring at the northwest corner of the FCAP was not observed during the Phase 2 field activities and therefore not sampled. However, sediment samples (SD1-1 SD1-2, and SD1-3) were collected at the approximate location of the spring. A summary of the Phase 2 surface water analytical results, which contains only those parameters that were detected above the detection limit in at least one of the samples, is presented in Table 3.10. The values that exceed ambient water quality standards are shown in bold in the table. The analytical results of some typical coal ash leachate parameters from Phase 2 sampling points SW-2, SW-3, SW-4, SW-8, SW-9, and SW-12 are presented in Fig. 3.10. Even with the addition of the contaminated spring waters at SW-4, a clear trend of reduced contaminant concentrations downstream is observed in Phase 1 and 2.

The UMB Phase 2 surface water data indicate that reference concentrations were exceeded for six metals, bis(2-ethylhexyl)phthalate, and diethylphthalate. Radiological results for UMB and the reference streams are primarily nondetections with the UMB data not exceeding the reference data.

The Phase 2 FCAP pond water data (Table 3.10) show that reference pond water concentrations were exceeded for 14 metals, $^{228}$Th, $^{230}$Th, $^{234}$U, and $^{238}$U. No organic compounds were detected in the FCAP pond water or the reference pond water.

The Phase 2 spring water data (Table 3.10) indicate that reference spring concentrations are exceeded for 11 metals, gross alpha, gross beta, 6 radioactive isotopes, and bis(2-ethylhexyl)phthalate. No other organic compounds were detected in the OU 2 springs. The OU 2 spring data (aluminum 1460 μg/L, arsenic 229 μg/L, iron 3310 μg/L, and manganese 898 μg/L) exceed the chronic ambient water quality criteria (Sect. 4.2.1) (aluminum 750 μg/L, arsenic 190 μg/L, iron 1000 μg/L, and manganese 50 μg/L).

### 3.2.3 Sediment Data

Sediment samples were collected and analyzed in both the Phase 1 and Phase 2 investigations of OU 2. Sediments were not sampled from the UMB stream bed during the Phase 1 effort, but ash deposited beside the UMB and the soil beneath the ash were sampled. The results of the ash and soils beneath the ash analyses are discussed in Sect. 3.2.1 along with similar data from the FCAP and Sluice Channel Area zones.

During the Phase 2 investigation, sediment samples were collected from 12 points adjacent to springs or seeps as shown in Fig. 3.11. Four composite sediment samples were created by combining three samples from each location. Background reference samples were collected from Walker Branch and the headwaters of White Oak Creek. The sediment samples were analyzed for the parameters shown in Table 3.11.

All the Phase 2 sediment analyses results above detection level are presented in Table 3.12. Aluminum, arsenic, barium, iron, manganese, potassium, and sodium in the sediments of OU 2 are well above the reference sediment results. High levels of these metals

are typically associated with coal and coal ash. The $^{238}U$ and the $^{232}Th$ results for the on-site sediments are elevated relative to the reference sediments results. These are the most common isotopes of uranium and thorium and being typical trace elements of coal and coal ash are expected to occur in elevated concentrations, along with the other heavy metals in the on-site sediments. Four organic compounds were detected in the sediments: 4-methylphenol, butylbenzylphthalate, di-n-butylphthalate, and 2-butanone; all of the organic detections are estimated values. The detections of 4-methylphenol in the sediments occurred at three different locations, in concentrations ranging from 99.0 μg/kg to 1100 μg/kg.

### 3.2.4 Groundwater Data

Both phases of the OU 2 RI characterized groundwater quality through sampling of monitoring wells and laboratory analysis of the samples. Eight monitoring wells are located within OU 2; four of these are completed in overburden and four are completed in bedrock (Fig. 3.12). Phase 1 activities included the installation of four of the monitoring wells (GW-672, GW-673, GW-674, and GW-676) and six piezometers. The Phase 1 well and piezometer logs are provided in Appendix F. The piezometers, which were constructed of 2-in.-diameter polyvinyl chloride screen and riser, were installed to evaluate groundwater flow direction in the vicinity of the FCAP. The piezometers were not sampled for groundwater quality during either the Phase 1 or Phase 2 RI. The other monitoring wells at OU 2 were installed during the 1980s in conjunction with studies of the FCAP (GW-321) and the Chestnut Ridge Security Pits (GW-512, GW-513, and GW-514). Table 3.13 provides construction information on the OU 2 monitoring wells.

Groundwater samples collected during Phase 1 were analyzed for total and dissolved metals, radiological parameters, common ions, phenols, total organic halides, and total organic carbon (TOC). Table 3.14 lists the specific analytes within these groups of parameters. Before sampling, each monitoring well was purged of at least three well volumes using a Grundfos submersible stainless steel pump or a stainless steel bailer, following Energy Systems' procedure ESP 302-2 (Kimbrough et al. 1988). During purging, specific conductance, pH, and temperature were measured for each well volume to determine stability and ambience. After purging, groundwater samples were collected from each monitoring well using a stainless steel bailer. Nylon line used to lower the bailer was dedicated for each monitoring well.

Groundwater samples collected during Phase 2 were analyzed for total and dissolved metals, radiological parameters, volatile organic compounds (VOCs), and semivolatile organic compounds (SVOCs). Table 3.15 lists the specific analytes within these groups of parameters. The analyte list for Phase 2 included a greater number of parameters than the list in Phase 1, and it also eliminated some of the analytes from the Phase 1 list (common ions, total organic halides, TOC). During review of the Phase 1 RI report and preparation of the Phase 2 sampling plan, all participants agreed upon the need to include VOCs and SVOCs in the list. The VOCs were deemed necessary to evaluate any contributions from the adjacent Chestnut Ridge Security Pits, and the SVOCs were deemed necessary to evaluate any contributions of polycyclic aromatic hydrocarbons (PAHs) from the coal ash.

The technique used for groundwater sampling in Phase 2 differed somewhat from that used in Phase 1. The monitoring wells were purged and sampled using a Purge Mizer™ bladder pump manufactured by QED Environmental Systems, Inc. The advantage of using the bladder pump is that the pump can be adjusted to a relatively low (i.e., 100 - 500 mL/min) flow rate for purging and sampling. The benefit of the low flow rate bladder pump is that it

allows purging and sampling of the monitoring well without excessive agitation of the water, which would elevate the turbidity. Increased turbidity can result in inaccurate concentrations of compounds in the results of analysis (Puls and Powell 1992). This is particularly true for metals, which are potential contaminants of concern at OU 2.

A minimum of three well volumes was evacuated from each of the monitoring wells during purging. In addition, pH, temperature, specific conductance, and turbidity were monitored and recorded; data recorded during purging are in Appendix F. Field personnel followed Energy Systems' procedure ESP 302-5 (Kimbrough et al. 1988) during purging and sampling.

Several of the monitoring wells were purged and sampled with the bladder pump as described above, but an inflatable packer, also manufactured by QED Environmental Systems, Inc., was used in conjunction with the pump assembly. The packer was not involved in sample collection; rather, it facilitated the sampling process. The packer was positioned and inflated just above the top of the screened interval. Teflon tubing from the bladder pump ran through the packer assembly but the groundwater did not contact the packer materials; the pump intake was 5 ft or more below the packer. The benefit of using the packer was twofold. First, it isolated the screened interval, thereby substantially reducing the amount of water purged from the monitoring well; this was especially beneficial in the deeper wells and was consistent with waste minimization. Second, it eliminated or virtually eliminated exchanges between the atmosphere and the groundwater to be sampled, thereby minimizing the effects of aeration and other processes that may compromise the integrity of the sample..

Groundwater quality data from Phase 1 were evaluated by comparing concentrations in samples from the monitoring wells south of the FCAP (GW-321, GW-672, GW-673, GW-674, and GW-676) with concentrations in samples from the monitoring wells north of the FCAP (GW-512, GW-513, GW-514). The three monitoring wells north of the FCAP were used as background; this may not be appropriate given the proximity of the wells to the Chestnut Ridge Security Pits and actual groundwater flow direction created by the karst system developed on Chestnut Ridge.

Using the comparison described above, the Phase 1 groundwater data (filtered) indicated that background concentrations were exceeded for 12 metals, gross alpha, gross beta, and total uranium. The metals included aluminum, barium, calcium, cobalt, copper, iron, potassium, magnesium, manganese, sodium, selenium, and zinc. Few standards were exceeded for any of the parameters analyzed. TDEC secondary maximum contaminant levels (MCLs) (Sect. 4.2.1) were exceeded for aluminum (GW-672, GW-674, and GW-676), iron (GW-672 and GW-673), and manganese (GW-672, GW-674, and GW-676). The federal maximum contaminant level (MCL) for gross alpha was exceeded in samples from GW-672, GW-673, and GW-674 and in a duplicate sample from GW-676. Table 3.16 summarizes the analytes detected in Phase 1 groundwater samples and their concentrations. Figure 3.13 shows concentrations of selected indicator parameters from the Phase 1 data; the contaminants provided in the figure are from unfiltered samples. The parameters selected typically are associated with coal ash leachate.

The Phase 1 data indicate that the concentrations of analytes are quite low and typically do not exceed primary and secondary standards. The overall impact of OU 2 on groundwater, therefore, is relatively small. Concentrations of parameters in Phase 1 were generally higher in samples from the overburden wells than in samples from the bedrock wells. According to

the Phase 1 RI report, the overall extent of contamination from the FCAP could not be accurately established using the existing monitoring well network (Energy Systems 1991).

Groundwater quality data from Phase 2 were evaluated by comparison with applicable standards. Although the three background monitoring wells north of the FCAP were sampled during the Phase 2 RI, these were not considered true background locations, but rather reference locations. The combination of both overburden and bedrock wells, the proximity of the Chestnut Ridge Security Pits, and the uncertainties associated with the groundwater flow direction along Chestnut Ridge preclude the use of these wells for background. Table 3.17 summarizes the analytes detected in Phase 2 groundwater samples and their concentrations. Figure 3.14 shows concentrations of selected indicator parameters from the Phase 2; note that the parameters match those shown in Fig. 3.13. As in Fig. 3.13, the concentrations provided are from unfiltered samples.

The organics analysis detected few parameters, and those that were detected were at relatively low concentrations; none of the parameters exceeded either primary or secondary standards. Among the VOCs, only trichloroethene was detected, and it was detected in only one well (GW-513) at a concentration of 2.0 μg/L, which is below the SW-876 quantitation level of 5.0 ppb. Among the SVOCs, bis (2-ethylhexyl)phthalate was detected in five wells (GW-512, GW-513, GW-514, GW-672, and GW-674) at concentrations ranging from 1.0 to 24.0 μg/L; diethylphthalate was detected in one well (GW-512) at 2.0 μg/L; di-n-butylphthalate in two wells (GW-512 and GW-672) at concentrations of 1.0 and 3.0 μg/L; and phenol in two wells (GW-512 and GW-513) at concentrations of 2.0 and 5.0 μg/L, respectively. The data suggest that the presence and extent of organics in groundwater at OU 2 are quite limited.

Radionuclides were detected in the Phase 2 groundwater samples but did not exceed applicable MCLs. Gross alpha was detected in two wells (GW-321 and GW-512) at concentrations of 7.2J and 9.1 pCi/L; gross beta was detected in three wells (GW-321, GW-512, and GW-673) at concentrations ranging from 4.9 to 12.0 pCi/L. Isotopic thorium analysis indicated the presence of $^{228}$Th and $^{232}$Th in one well (GW-512) at concentrations of 0.32 and 0.22 pCi/L, respectively. Isotopic uranium analysis indicated the presence of $^{234}$U in all wells at concentrations ranging from 0.22 to 0.98 pCi/L, $^{235}$U in one well (GW-673) at 0.054 pCi/L, and $^{238}$U in seven wells (all wells but GW-321) at concentrations ranging from 0.11 to 0.44 pCi/L. The radionuclide parameters are present at varying concentrations in both overburden and bedrock wells both north and south of the FCAP. It would be difficult to attribute the concentrations to specific sources, but, in general, the data suggest that the presence and extent of radionuclides in groundwater at OU 2 is limited.

The metals analysis of Phase 2 groundwater samples detected Al, As, Ba, Be, Ca, Cu, Fe, Pb, Mg, Mn, K, Se, Na, V, and Zn. Some of the metals, such as Ca, Mg, and Na are typical of groundwater on this portion of the ORR. Other metals, such as As, Pg, Se, Ba, and Mn, may be related to former activities at OU 2. Arsenic was detected in two wells (GW-512 and GW-513) at concentrations of 3.8 and 5.0 μg/L. Lead was detected in three wells (GW-512, GW-513, and GW-673) at concentrations ranging from 3.1 to 11.7 μg/L. Selenium was detected in one well (GW-674) at 5.8 μg/L. Barium was detected in all wells at concentrations ranging from 8.3 to 186.0 μg/L, and manganese was detected in all wells at concentrations ranging from 1.3 to 1360 μg/L. None of these concentrations exceeded respective MCLs. The TDEC SMCL for manganese was exceeded in four wells (GW-512, GW-672, GS-674, and GW-676) The presence of arsenic and lead in GW-512 and GW-513 cannot be attributed

entirely to ash disposal activities at OU 2 because of the location of these wells with respect to the FCAP and the assumed groundwater flow direction (see Sect. 2.5). As stated above, however, groundwater flow direction(s) along Chestnut Ridge are not completely understood. As with organics and radionuclides, apparently the presence and extent of metals in groundwater at OU 2 are relatively limited.

Groundwater data from Chestnut Ridge OU 2 suggest that former activities at the site have had some impact on the groundwater, but the impact is quite limited. Data from both phases included four MCL exceedences for one analyte; Phase 1 samples from GW-672, GW-673, and GW-674 and a duplicate sample from GW-676 exceeded the gross alpha MCL of 15 pCi/L. In general, the presence and extent of organics, metals, and radionuclides in groundwater at OU 2 is quite limited.

### 3.2.5 Soil Chemical Data

Soils beneath the ash in the Sluice Channel Area, FCAP, and UMB zones were sampled and analyzed to ascertain whether contaminants are leaching from the ash into the soils. Soil samples were collected from beneath the ash at five locations (A2 through A6; see Fig. 3.3) along UMB in Phase 1 of the investigation. The Phase 1 reference soil samples, samples from location A1, were rejected as reference samples due to close proximity to the contaminated site and the lack of organic analytical data. New reference soil samples were collected and analyzed in the Phase 2 effort. The soil samples of the Phase 1 effort were tested for the analytes listed in Table 3.18. Soils beneath the ash in the FCAP and in the Sluice Channel Area and soils from an uncontaminated reference area adjacent to the site were collected in the Phase 2 investigation. Samples from soils beneath the ash in the FCAP were collected using a split-spoon sampler in conjunction with hollow stem auger drilling. All other soil samples were collected following Energy Systems, Procedures ESP 303-1 and ESP 308-1 (Kimbrough et al. 1988). The Phase 2 soil analytes are listed in Table 3.19. The locations of the Phase 2 soil beneath the ash samples are shown on Figs. 3.4, 3.5, and 3.6. The reference soil samples locations (S010 through S012) are depicted on Fig. 3.15.

The analytical results of the Phase 1 and Phase 2 soils beneath the ash samples are presented in Tables 3.20 and 3.21. Iron is the only contaminant that exceeded reference soil mean in every sample in both phases. Arsenic (maximum 43.6 mg/kg) was significantly elevated in 8 of the 12 samples. The lower contaminant concentrations in UMB, relative to the soils data for Sluice Channel Area and FCAP zones, indicate that the ash has been subjected to leaching processes before migrating into the UMB zone.

Results for Phase 2 samples from soil beneath the ash in the FCAP and Sluice Channel Area are shown in Table 3.21. A summary of the Phase 2 data and the reference soils is presented in Table 3.22. The analytical results for FCAP and Sluice Channel Area soils beneath the ash are similar. The mean of the FCAP and Sluice Channel Area sample results for each of the metals, except mercury and uranium, exceed the reference soil mean. Aluminum, arsenic, and iron concentrations in all the FCAP and Sluice Channel Area samples exceeded the reference soil means. Four organic compounds were detected at trace levels: bis(2-ethylhexyl)phthalate; butylbenzylphthalate; 2-methylnaphthalene; and 4-nitrophenol.

Surface ash samples were collected in the FCAP zone because these represent the present contaminated surface soil media. The 18 surface ash samples collected at the locations shown on Fig. 3.15 were combined into 6 composite samples and analyzed for the analytes

in Table 3.19. The geotechnical data for the reference soils, sediments, and surface ash are presented in Table 3.23. The results of the analyses, all results above the detection level, are presented in Table 3.24. In all cases, except manganese, the maximum metal concentration in the surface ash exceeded the maximum metal concentration in the reference soil. Arsenic and iron concentrations in every surface ash sample were an order of magnitude higher than in the reference soils. Three organic compounds were detected in the surface ash in low, estimated concentrations.

### 3.2.6 Biomonitoring Data

This section summarizes all biological monitoring data including media toxicity (aquatic, soil/sediment/ash) (Sect. 3.2.6.1), benthic macroinvertebrates survey of UMB (Sect. 3.2.6.2), ash invertebrate survey (Sect. 3.2.6.3), and heavy metal bioaccumulation in vegetation (Sect. 3.2.6.4) and small mammals (Sect. 3.2.6.5). The following data give information on both aquatic and terrestrial ecosystems; considering the toxicity of the site, presence of aquatic and terrestrial fauna, and the extent of contaminant bioaccumulation in food sources and potential predators. All information provided in this section supports the ecological risk assessment (Chap. 6).

#### 3.2.6.1 Media toxicity data

Various kinds of tests were used to assess the toxicity of water and sediment samples from three sites in McCoy Branch and of ash samples from the FCAP. Tests used to routinely assess water quality included 7-day laboratory tests with fathead minnow (*Pimephales promelas*) larvae, and 7-day laboratory tests with *Ceriodaphnia dubia* (a freshwater microcrustacean). These two tests commonly are used to estimate effluent and ambient water toxicity. On several occasions, a 4-day laboratory feeding-rate test with a pleurocerid (gill-breathing) snail (*Elimia clavaeformis*) was also used. During two of the feeding-rate tests, sediment collected from MCK 1.56 and MCK 2.03 (Fig. 3.16) was tested with the snails, as well. Two types of 14-day tests with earthworms (*Eisenia foetida*) were used to assess the biological quality of ash from the FCAP and Sluice Channel Area. One of these was conducted in situ; the other involved quantifying responses of the worms to FCAP and Sluice Channel Area ash samples under laboratory conditions. The results of various other biological or toxicological tests of water in McCoy Branch [e.g., in situ snail-release study (at MCK 1.92 and MCK 1.60); a life-cycle test with *Ceriodaphnia* (using water from MCK 1.92); and 7-day fathead minnow larvae and *Ceriodaphnia* toxicity tests for January 1989 to April 1990 (MCK 1.92)] are given in another report (Ryon 1992; Appendix G).

Information about the locations of the sites that were tested, the month and date when the tests were initiated, and the types of tests that were used to assess biological conditions, is summarized in Table 3.25. In each test used to assess water quality, measurements were also made of pH, conductivity, alkalinity, and hardness. In tests used to assess ash samples, measurements were made of ash pH and organic content (by loss of mass with ignition). The water quality data obtained during the tests above are also summarized in this section, as is information about the FCAP ash samples that were tested with *Eisenia foetida*. A more detailed description of the methods used for the biological and physicochemical measurements is provided in the following section.

### 3.2.6.2 Materials and methods

**Fathead minnow larvae survival and growth tests.** Seven-day static-renewal tests with fathead minnow larvae (*Pimephales promelas*) are commonly used to estimate the toxicity of effluents and receiving waters. This test has been standardized by the EPA (method 1000.0; Weber et al. 1989); it involves rearing newly hatched replicate groups of larval minnows in beakers containing control water and beakers containing water or effluent samples, under controlled temperature, lighting, and food conditions in the laboratory. Water samples for the tests are collected and handled using chain-of-custody procedures in accordance with Standard Operating Procedures for the Environmental Sciences Division's Toxicology Laboratory Quality Assurance Program (QAP-X-89-ES). Generally, ten larvae are used in each replicate beaker, with four replicates per test. The minnows are inspected daily for mortality; dead fish, if any, are removed, and the old test solution is then siphoned off and replaced with fresh test solution. At the end of a 7-day test period, survival and growth of the minnow larvae in the effluent or ambient water samples are compared to the survival and growth of the minnow larvae in the control water. Statistically significant reductions in minnow survival or growth in the effluent or ambient water samples are used as evidence for acute and chronic toxicity, respectively. In some ambient applications, low mean survival of fathead minnow larvae test can occur due to the presence of facultative pathogenic microorganisms; in these instances, the test may yield "false positives" for acute toxicity (Kszos and Stewart 1992). In most effluent applications, the fathead minnow test appears to perform reliably. Examples of the use of 7-day fathead minnow tests to assess ambient toxicity in streams can be found in Norberg-King and Mount (1986) and Stewart et al. (1990).

In each test period in which fathead minnow larvae were used to conduct water quality assessments, various concentrations of water (e.g., 80%, 60%, 40%, etc.) were tested in addition to full-strength water. Dilutions, when used, were prepared using the same water as was used in the control (i.e., diluted mineral water).

***Ceriodaphnia dubia* survival and reproduction tests.** Seven-day static-renewal tests with the freshwater microcrustacean *Ceriodaphnia dubia* are also used to estimate toxicity of effluents and receiving waters. This test has been standardized for use in effluent testing by the EPA (method 1002.0; Weber et al. 1989); it involves rearing neonate *Ceriodaphnia* (<24 hours old) individually in small beakers containing either control water or effluent or ambient water, under controlled temperature, lighting, and food conditions in a laboratory. These daphnids are typically parthenogenic (i.e., females can produce offspring in the absence of males) and achieve reproductive maturity in two to four days. In a typical 7-day test, *Ceriodaphnia* neonates used to start the test can produce three or more broods of new neonates. The contents of each beaker are inspected visually each day of the test; when new neonate *Ceriodaphnia* are present in a test beaker, they are counted and discarded; only adult daphnids are transferred individually to fresh test solution. *Ceriodaphnia* survival and reproduction in control water in a concurrently conducted test is compared to survival and reproduction of the daphnids in the test solutions to provide estimates of acute and chronic toxicity, respectively.

*Ceriodaphnia* in general are more sensitive than fathead minnow larvae to most toxicants. Inherent differences in procedures for the fathead minnow larvae and *Ceriodaphnia* test methods exist (e.g., groups of ten fish per replicate, versus single daphnids per replicate; measurements of weight change for the fish, as estimates of growth, versus numbers of neonates produced per daphnid in the *Ceriodaphnia* test). In addition, the difference in

number of replicates typically used in each test method (four for the fish, ten for the daphnids) also tend to increase the resolution of the *Ceriodaphnia* test, relative to the fathead minnow test. Reproduction of *Ceriodaphnia* and other daphnids is closely linked to food quality and quantity. Thus, mean reproduction of *Ceriodaphnia* in ambient water samples that contain bacteria, algae or detritus can exceed mean reproduction of *Ceriodaphnia* in the controls (Kszos and Stewart 1992). Additionally, water quality factors such as conductivity, hardness, alkalinity, and pH also can affect survival or reproduction of *Ceriodaphnia*, or virtually any other type of bioassay. Thus, in ambient applications, test-to-test variation in *Ceriodaphnia* mean reproduction is often nearly as great as site-to-site variation in *Ceriodaphnia* reproduction (Stewart and Loar 1994). These findings show that the results of *Ceriodaphnia* "toxicity" tests must be considered carefully if they are to provide meaningful information about water quality in ambient applications. Examples of the use of *Ceriodaphnia* tests to assess ambient water quality are provided by Nimmo et al. (1990), Stewart et al. (1990) and Kszos and Stewart (1992).

In each test period in which *Ceriodaphnia* were used to conduct water quality assessments, various concentrations of water (e.g., 80%, 60%, 40%, etc.) were tested in addition to full-strength water. Dilutions, when used, were prepared using the same water as was used in the control (i.e., diluted mineral water).

**Snail feeding-rate tests.** Pleurocerid (gill-breathing) snails are key grazers and influential competitors in many streams (Hawkins and Furnish 1987, Burris et al. 1990, Hill 1992, Stewart et al. 1993). These animals can occur in high densities, sometimes comprising more than 90% of the macroinvertebrate biomass (Hawkins and Furnish 1987, Elwood et al. 1981). Snails respond behaviorally to differences in water quality. When toxicants are present, for example, they may cease feeding, slough their tentacles or retract their foot, or actively attempt to avoid the conditions by moving downstream or crawling out of the water (Burris et al. 1990, Harry and Aldrich 1963, Arthur and Leonard 1970, Winger et al. 1984, Burris 1987). These findings suggest that quantifiable changes in snail activities could be used to describe water quality.

Laboratory tests that used measurements of the feeding rates *Elimia clavaeformis*, a pleurocerid snail that is abundant in many streams on the ORR, were conducted to provide information about water quality in McCoy Branch. These tests were used to assess samples of water collected both upstream and downstream of Rogers Quarry.

*Elimia clavaeformis* is common and abundant in most streams within the White Oak Creek drainage basin, and are present in several other streams that drain the south slope of Chestnut Ridge (e.g., Walker Branch, Scarboro Creek, Ish Creek). Although *Elimia* do not presently occur in McCoy Branch, snail distribution patterns on the ORR suggest that these snails may have inhabited the stream in the past and may eventually recolonize the stream, with continued improvements in water quality. Specimens used for the laboratory tests reported here were collected from minimally disturbed areas in upper White Oak Creek [near White Oak Creek kilometer (WCK) 6.8]. A 20% concentration of degassed mineral water was used as a negative control. Grab samples of water were collected daily from McCoy Branch for use in the tests.

In the laboratory, the snails were held in 12-L aquaria supplied with a constant flow of dechlorinated tap water. They were fed lettuce *ad libitum* and acclimated stepwise to the testing temperature (25°C). The tests were conducted under controlled conditions (25°C and

a 16:8 L:D photoperiod) using a static-renewal procedure similar to the fathead minnow larvae test procedure. Twelve snails, selected randomly, were placed in a 600-mL beaker and exposed to 250 mL of treatment water; four replicate beakers were used to assess each treatment. Three green leaf lettuce disks (total area of 3 $cm^2$) were added to each beaker each day for food; the disks were weighted with two staples so that they would not float. Initial and final wet weights of the disks were recorded each day. The open top of each beaker was covered with mesh to prevent the snails from escaping.

Snail feeding-rate tests were conducted on three dates (Table 3.25). The tests differed slightly with respect to objectives and methodology. Relevant differences among the tests are described below:

Test 1. Water was collected from McCoy Branch kilometer (MCK) 2.03 (upstream of Rogers Quarry at the weir) and MCK 1.56 (downstream of Bethel Valley Road). Snails were exposed to full-strength water and water diluted by 50% with degassed mineral water. The tests were conducted at 25°C.

Test 2. Water was collected from MCK 2.03 and MCK 1.56. Sediment was collected from MCK 2.53. Snails were exposed to full-strength water, and full-strength water plus sediment (1 g sediment/L) from each site. The tests were conducted at 25°C.

Test 3. Water and associated sediment were collected from MCK 2.03 and MCK 1.56. Snails were exposed to full-strength water with ~1.5 g of sediment from the site. Each treatment was conducted at 15°C and 25°C.

In each test, feeding rate (expressed as mean number of grams of wet weight of lettuce eaten within each beaker per day) of the snails was used as the response variable. The data were analyzed using PROC GLM (SAS Institute 1985). Whenever possible, the tests were conducted concurrently with toxicity tests performed with fathead minnow larvae (*Pimephales promelas*) and *Ceriodaphnia dubia* (Table 3.25). The snail-feeding assay described above is not yet an EPA-approved procedure, but has been used to assess water quality at the inlet and outfall of Lake Reality in upper East Fork Poplar Creek (Hinzman 1993), and to evaluate stormwater quality entering White Oak Creek (Hinzman et al. 1993).

**Earthworm tests.** Earthworm tests are used for solid waste and soil toxicity assessments. These soil-dwelling invertebrates are ecologically significant, both because they modify texture and fertility of soil, and are important as food to many small mammals and birds. Earthworms are exposed to contaminants in soils both externally (through direct contact with soil) and internally (through ingestion of soil particles). These animals also grow and reach reproductive age rapidly. Thus, they are potentially excellent indicators of a soil's environmental condition and may be useful in ecological risk assessments. An EPA-approved method for testing with earthworms is available (Greene et al. 1989). This method allows assessment of soils by quantifying effects of soils on earthworm survival and growth.

Two types of earthworm tests were used to assess ash in the FCAP. One test, conducted in July 1992 (Table 3.25), involved in situ testing at six locations in the FCAP. The other test, conducted in the laboratory during August 1993, was used to assess ash samples collected from three locations in the FCAP. The procedures for these two tests are described below.

**In situ earthworm test.** Six sites, running along a north-south transect, were selected for testing. It was speculated that there might be a north-south gradient in ash toxicity caused by differences in the weathering duration of the ash, with "newer" ash being more likely near the north end of the FCAP, and "older" ash predominating at the south end of the FCAP. The in situ tests were considered to be exploratory in nature; an experimental design that used randomly selected sites (and a broader range in reference sites) might have been more appropriate for assessing spatial heterogeneity in toxicity of FCAP ash. The locations of the six sites that were used in the in situ tests are given as follows: Site 1 was ~25 m north of the earthen dam. Successive sites (2 through 6) were separated by 23 m, 61 m, 35 m, 38 m, and 33 m, respectively. At each site, four treatments (one replicate each) were evaluated: (1) non-altered site ash, (2) site ash augmented with fermented alfalfa (used as *Eisenia* food), (3) pH-adjusted peat moss, and (4) pH-adjusted peat moss augmented with fermented alfalfa. Each replicate consisted of a 0.5-gal plastic bucket filled with test medium. The buckets were partially buried in the FCAP ash, deep enough so that their tops were nearly level with the ash surface. Six earthworms, pre-weighed as a batch from laboratory cultures, were placed on top of the ash in each bucket. The tops of the buckets were then covered with mesh to prevent the worms from escaping. Buckets containing FCAP ash were filled with ash from the hole dug to bury the buckets. The peat moss was obtained as a commercial product. Enough $CaCO_3$ was added to the peat moss (about 1%, on a dry mass basis) before the test to neutralize the peat moss's acidity. The peat moss also was saturated with distilled water prior to starting the test. Because rain events occurred throughout the 14-day testing period, no water was added to any of the treatments during the test. The fermented alfalfa used as a food supplement in the food-augmented combinations was prepared using the method described by Greene et al. (1989).

After 14 days, the 24 buckets were removed from the FCAP and taken to the laboratory. The ash in each bucket was then hand sorted to harvest the surviving earthworms, and analyzed for pH, water content (by drying at 100°C), and organic matter content (by ashing at 500°C). Surviving earthworms were purged of their gut contents before being weighed.

The number of worms surviving in each bucket (expressed as a proportion; arcsine square-root transformed) was used as the response variable in this test. The transformed survival data were then analyzed using a random effects ANOVA for a nested design, with ash type (FCAP soil versus peat moss) and food-amendment (yes or no) serving as independent variables. The experimental design was not appropriate for distinguishing differences in worm responses among sites; it was suitable for detecting differences in worm survival in fly-ash soil versus pH-adjusted peat moss, and differences in worm survival in the two ash types attributable to the presence of food. Growth data could not be analyzed for reasons discussed in the results section.

**Laboratory earthworm test.** This test also used *Eisenia foetida*, and generally followed the EPA-approved test procedure described by Greene et al. (1989). The test was conducted during August 1-14, 1993 (Table 3.25). Three samples of ash were tested; the samples were from FCAP-1, FCAP-2, and SLUICE-1 (Fig. 3.16). A reference site and an artificial soil were tested concurrently, using the same methods. An additional treatment was introduced to the test by comparing autoclaved versus non-autoclaved ash from each site. The autoclaved samples were sterilized at 1.6 kg/cm$^2$ for 15 min to rid the ash of possible microbial pathogens. The experimental design described above involved testing ten types of materials (autoclaved versus non-autoclaved samples of artificial soil, a reference site soil, FCAP-1, FCAP-2, and SLUICE-1 ash).

After the samples had been collected and autoclaved, distilled water was added to each soil (or ash) type to achieve approximately a 50% moisture fraction (the consistency of a thoroughly wet, but not muddy soil). Each ash or soil type was tested by adding 400 g of hydrated soil (or ash) to each of four 1-pint canning jars. Ten earthworms (pre-weighed as a group) were then placed on top of the material in each jar. Lids (with a 1/16-in. hole to permit gas exchange) were screwed onto the jars, and each jar was placed in an environmental chamber. The temperature in the chamber was 25 $\pm$ 2°C; the light regime was 16 hours light and 8 hours dark. Fourteen days after the test had started, the jars were removed and the material in each jar was hand-sorted to remove all surviving worms. The worms in each replicate were rinsed with distilled water to remove adhering soil (or ash) particles before being weighed.

The number of worms surviving in each replicate was analyzed using a one-way ANOVA; the survivorship data were transformed (arcsine square root) prior to analysis. A multiple-comparison T-test (equivalent to Fischer's least significant difference test) was used to determine which differences, if any, were statistically significant (SAS Institute 1988).

Growth of the worms in each replicate was estimated by subtracting the mean final mass of the earthworms (g mean individual fresh mass per jar) from the mean initial weight of the earthworms (g mean individual fresh mass per replicate). One-way ANOVA was then used to determine the influence of autoclaving and soil (or ash) source on earthworm growth.

**Physicochemical water quality factors analyses.** Water samples for use in the fathead minnow tests, *Ceriodaphnia* tests, and snail-feeding tests described above were collected as grab samples daily for 7 days in most of the test periods. The water temperature was measured at the time of sample collection, and portions of each sample were analyzed for pH, conductivity, alkalinity (EPA method 130.1), and hardness (EPA method 130.2) in the laboratory. The methods used for each of these analyses are described by Kszos et al. (1989). Because multiple samples were collected from each site during each test period, it was possible to analyze the results of the water quality measurements using two-way ANOVA, using test periods and sites as class variables. Response variables in this analysis included temperature, pH, conductivity, alkalinity, and hardness.

### 3.2.6.3 Results

**Fathead minnow and *Ceriodaphnia* tests.** The results of the fathead minnow tests and the *Ceriodaphnia* tests are summarized in Table 3.26. In most cases tests with these two species were conducted concurrently, used the same water samples, and involved the same dilution series. In this table, test date refers to the date the test was initiated. In general, *Ceriodaphnia* has been shown to be more sensitive than fathead minnows to diverse contaminants (Constable and Orr 1994, Stewart et al. 1990). Thus, interpretation of the results of the ambient toxicity tests for the McCoy Branch sites in this section emphasizes the results the *Ceriodaphnia* tests, compared to the results of the fathead minnow tests.

During the 11 test periods, the distribution of survival values for *Ceriodaphnia* in full-strength water samples from McCoy Branch sites was as follows: 100% survival in 17 of 24 cases, 90% in 4 of 24 cases, 70% in 2 of 24 cases, and 60% in one case. The distribution of *Ceriodaphnia* survival values in the McCoy Branch tests is very similar to the distribution of *Ceriodaphnia* survival values reported for noncontaminated reference sites in headwater

streams on the ORR (Loar 1991). Thus, *Ceriodaphnia* overall survival patterns in tests of full-strength water samples from McCoy Branch do not provide evidence for acute toxicity.

During the 11 test periods, mean reproduction of *Ceriodaphnia* in the controls exceeded 20 offspring per female in nine cases, and was lower than 20 offspring per female in two cases. Mean reproduction of *Ceriodaphnia* in full-strength water from the McCoy Branch sites was greater than 20 offspring per female in 18 of 24 cases. All three McCoy Branch sites had low values for *Ceriodaphnia* reproduction during the test that started on March 25, 1993; reproduction in the control for these tests was also low. When the tests for March 25, 1993, were excluded from consideration, mean reproduction of *Ceriodaphnia* was less than 20 offspring per female in one of ten cases for controls, and less than 20 offspring per female in 3 of 21 cases involving full-strength water from McCoy Branch. In the three cases where *Ceriodaphnia* mean reproduction was less than 20 offspring per female in full-strength McCoy Branch water, one value was 19.7 offspring per female (July 11, 1991; MCK 2.03), one was 18.3 (July 30, 1990; MCK 1.56) and one was 11.1 (July 30, 1990; MCK 2.03). In each of the three cases, minnow survival was 100% and minnow growth in full-strength water from McCoy Branch differed from minnow growth in the corresponding control by less than 10%.

Minnow tests of McCoy Branch water from one or more sites were conducted during nine test periods (Table 3.26). Minnow survival and growth data were available for 19 cases involving full-strength water from McCoy Branch. In 2 of the 19 cases, mean survival of the minnows was less than 70%, a value that may be used as an arbitrary but reasonable criterion to suggest toxicity. These "low survival" outcomes occurred at MCK 1.56 during the April 2, 1992, test (survival was 47.5 $\pm$ 18.9%), and at MCK 2.53 during the March 25, 1993, test (55.0 $\pm$ 36.9%). In both of these "low survival" tests, minnow growth in the potentially toxic water was greater than minnow growth in the corresponding control.

In various instances, minnow survival in one or more samples of diluted McCoy Branch water samples was lower than 70% (Table 3.26). In several of these instances, there was no consistent concentration-response pattern, such that greater mortality occurred in beakers that contained higher concentrations of McCoy Branch water. Examples of this situation can be found by inspecting the results of the minnow tests for MCK 2.03 for January 31, 1992, April 2, 1992, and July 9, 1992 (Table 3.26), and for MCK 1.56 for October 24, 1991, and April 2, 1992.

The results of the *Ceriodaphnia* and fathead minnow tests described above do not provide consistent evidence for either acute or chronic toxicity to either fathead minnows or *Ceriodaphnia*.

**Snail feeding-rate tests.** No inhibition of feeding rate of the snails was detected in water from any of the McCoy Branch sites in any test period, relative to feeding rates of snails in the controls (Table 3.27). In the first test, snails in full-strength water from MCK 2.03 and the 50% concentration of water from MCK 1.56 ate approximately twice as much as the controls (0.07 mg per day for controls, versus 0.17 mg per day for MCK 1.56 and MCK 2.03; Table 3.27). This difference was significant ($p = 0.05$). In the second test, the mean feeding rate of snails in the controls was virtually identical to that of snails in water either from MCK 1.56 or MCK 2.03 (0.17 mg per day). In this test, too, the addition of sediment to the water from the two McCoy Branch sites did not affect snail feeding rates (Table 3.27). In the third test, a temperature effect on feeding rate was very evident: at 25°C, snails ate 2.5 to 3.5 times more lettuce per day than they did at 15°C. Water temperature alone explained about

92.4% of variation in feeding rates; the site/temperature interaction was not significant ($p = 0.829$).

Collectively, the results of the three feeding-rate tests suggest that water and sediment quality in McCoy Branch may not be factors that limit the recolonization of the stream by *Elimia*. Longer term tests that consider the possibility of effects of water or sediment on snail growth and/or reproduction would be needed to assess this possibility.

**In situ earthworm test.** Survival was higher in the peat moss soil than in the FCAP ash samples, and mean survival in the food-added treatments was greater than in the non-fed treatments of the same soil (or ash) type (Table 3.28). The ANOVA for nested designs using the transformed survival data revealed significant differences among survival means due to soil (or ash) type ($p = 0.029$). The differences in means attributed to food were not statistically significant ($p = 0.428$).

The measurements of soil (or ash) pH, moisture content and organic matter content showed that the FCAP ash was slightly acidic (pH 6.7-6.9) (Table 3.29). This ash had a water content of 47% to 48%, and an organic content of 30% to 34% (dry weight basis; Table 3.29). These conditions are within the ranges considered tolerable for earthworm survival (Lee 1988). The food-addition treatment slightly increased the pH in both the ash and the peat-moss soil (by 0.2 units, on average; Table 3.29).

Because survivorship in the FCAP soils was so low, there were not enough individuals from which to obtain growth data to perform a valid statistical analysis.

**Laboratory earthworm test.** Results of the 14-day laboratory test are summarized in Table 3.30. Mean survival of the worms in the reference and control (artificial) soil treatments was > 90%, both for autoclaved and nonautoclaved treatments. Mean survival of worms among the three FCAP and Sluice Channel Area ash samples ranged from 32.5% [nonautoclaved ash from FCAP-1 (Sycamore area)] to 95% [autoclaved ash from SLUICE-1 and FCAP-2 (Willow area)] (see Fig. 3.16). ANOVA revealed statistically significant treatment-related differences in survival ($F = 4.55$, $p < 0.0008$). The multiple-level T-test showed that worm survival in ash from the FCAP-1 (both autoclaved and non-autoclaved) was significantly lower than either the control or reference soils ($p < 0.05$), and that worm survival in ash from the FCAP-2 and SLUICE-1 was not distinguishably different from survival in the control or references soils (Table 3.30). ANOVA also was used to inspect the relation between earthworm survival for autoclaved versus nonautoclaved ash. The results of this analysis indicated that apparent increases in worm survival in the autoclaved samples was not statistically significant ($p = 0.152$). Thus, the mean effect of autoclaving on worm survival, for all sites compared simultaneously, was not great.

The results of the 14-day laboratory test described above suggest that ash at the FCAP-1 may have been toxic to earthworms. In the FCAP-1 ash, mean survival of the worms increased considerably (by 61%; from 32.5% to 52.5%; see Table 3.30) with autoclaving. The increase in survival in the FCAP-1 sample caused by autoclaving does not eliminate the possibility that worm survival was affected by pollutants, because autoclaving can rid soils of volatile or semivolatile contaminants, in addition to killing potential pathogens or parasites. The results of this test can only suggested that pathogenic organisms probably do not account for the apparent toxicity of FCAP-1 ash.

On average, earthworms tended to lose more body weight in the ash treatments than they did in any of the control- or reference-soil treatments (Table 3.30). The greatest loss in body weight (about 50.5%) occurred in non-autoclaved ash from the FCAP-1 (Table 3.30). The losses in body weight among the soil (or ash) types, though, did not differ significantly when all treatments were considered simultaneously ($p = 0.066$). In general, reproduction is considered to be more sensitive than growth to toxic chemicals. Soil toxicity tests that incorporate earthworm reproduction as a response variable might be more useful than tests that include only survival and growth as response variables.

**Physicochemical water quality factors.** Measurements of pH, conductivity, alkalinity and hardness were made on water from two or more sites in McCoy Branch on 11 of the 13 test periods shown in Table 3.25. Summary statistics for these parameters, by test, for MCK 1.56 and MCK 2.03, are shown in Tables 3.31 and 3.32.

The chemical data from MCK 1.56 and MCK 2.03 were extensive, compared to sites MCK 2.53 and MCK 2.65 (the latter two sites had seven observations only; see Table 3-8 in Ryon 1992, Appendix G). Data from MCK 1.56 and MCK 2.03 were analyzed statistically by SAS Institute (PROC GLM) to quantify spatial and temporal differences in pH, conductivity, alkalinity, hardness, and the ratios of alkalinity to conductivity (A:C) and alkalinity to hardness (A:H). This analysis used site and test period (see Table 3.25) as class variables. Site, test, and the interaction between site and test explained 81.9%, 92.8%, 96.0%, and 75.6% of the variation in pH, conductivity, alkalinity, and hardness, respectively (Table 3.33). Site, test, and the interaction between site and test explained 81.8% of the variation in A:C, and 58.7% of the variation in A:H. Each of the three explanatory variables included in the model (test, site, and test: site) was highly significant ($p < 0.0001$) for each water quality variable alone, and for both ratios. Thus, for data pooled temporally, the two sites were distinguishable statistically ($p < 0.0001$, with $N = 76$) on the basis of any one of the four physicochemical parameters alone. The two sites could also be resolved statistically based on A:C and A:H, as well, but these two ratios were less characteristic of the sites than measurements of pH, alkalinity, hardness, or conductivity measurements alone. Based on the magnitude of the means computed by pooling data temporally, alkalinity (28.5%), hardness (16.8%), and conductivity (14.3%) differences between the two sites were more easily discerned than differences in pH (1.9%), A:C (11.8%), or A:H (8.7%) (Table 3.33).

Conductivity, alkalinity, and hardness are considered to be conservative properties of water, in that they are influenced more by fundamental geochemical processes and hydrological conditions than by biological processes (Stewart 1988). Inputs of wastewaters, or significant inputs of groundwater, or seepage water enriched in any constituent that affects pH, conductivity, or hardness, can alter the "chemical fingerprint" of stream water; the inputs are discernible by examining the relationships among conductivity, alkalinity, and hardness conditions. This concept was used previously to develop and explore application of a Chemical Perturbation Index (CPI) (see Hinzman 1993, p. 55). The CPI is the sum of the Pearson (rank) correlations between conductivity, alkalinity and hardness; it has a maximum value of 3.0, indicating perfect correlations among all three factors. In nonperturbed headwater sites in streams on the ORR, the CPI typically is as high as 2.6 to 2.8; at sites influenced by wastewater discharges or other plant operations, the CPI sometimes is lower than 1.0, indicating major disruption of "normal" relationships among water quality factors. Data from MCK 1.56 and MCK 2.03 were used to compute CPI; the CPI was 2.24 for MCK 1.56, and 2.70 for MCK 2.03, with 76 observations being used at each site. The CPI for MCK 1.56 was 17% lower than that for MCK 2.03. This difference (17%) is great enough to suggest that the

differences were statistically significant, based on CPI data for other streams on the ORR (Hinzman 1993). Based on the CPI, MCK 1.56 could be characterized as similar to conditions that can be found in White Oak Creek between sites WCK 5.1 and WCK 4.4, and much more typical of natural conditions than water at any site in East Fork Poplar Creek.

### 3.2.6.4 Discussion and conclusions

**Fathead minnow and *Ceriodaphnia* tests.** The fathead minnow tests provide no strong evidence for acute or chronic toxicity of water at any of the tested sites. In three of eleven test periods, survival of fathead minnow larvae was lower than 80% in full-strength water from one or more sites. The site-date combinations in which minnow survival was suspect included January 31, 1992 (MCK 2.03, 75% survival), April 2, 1992 (MCK 1.56, 47.5% survival), and March 25, 1993, where survival at three sites was 55.0% to 77.5% (Table 3.26). Growth of the minnow larvae was high in each of the tests where survival was suspect. In each test in which minnow survival was low, among-replicate variation in survival was atypically high (Table 3.26). Other studies have found that low mean survival of fathead minnow larvae is fairly common in ambient toxicity testing situations; low mean survival, if accompanied by an unusually large variation in survival among replicates, may indicate the presence of pathogenic bacteria or fungi (Kszos and Stewart 1992). The unusually large variation in survival among replicates for the minnows in each of the suspect site-date combinations provides better evidence for pathogens than for chemical contaminants.

The results of the *Ceriodaphnia* tests also provided no strong evidence for toxicity. In 17 of the 24 site-test combinations that used full-strength McCoy Branch water, survival of *Ceriodaphnia* was 100%; in four of the 24 site-test combinations, it was 90%; in two cases, survival was 70%, and in one case, it was 60% (Table 3.26). This distribution of survival values is very similar to that obtained when using *Ceriodaphnia* to assess non-contaminated stream sites on the ORR (Loar 1990). In April 1992, when two site-date combinations yielded *Ceriodaphnia* survival values that were $\leq$ 70%, there was no evidence of a dose-response relationship, even though multiple concentrations were tested. Additionally, the *Ceriodaphnia* that survived in the two full-strength water samples produced large numbers of offspring.

In 18 of the 24 site-test combinations involving full-strength water from McCoy Branch sites, *Ceriodaphnia* reproduction was $\geq$ 20 offspring per female, which is well above the EPA's criterion for acceptability of controls (15 offspring per female). In 7 of the 24 site-test combinations, *Ceriodaphnia* reproduction in full-strength McCoy Branch water exceeded reproduction in the controls. In the July 30, 1990, test period, mean reproduction of *Ceriodaphnia* in full-strength water from MCK 2.03 was low, but no dose-response pattern in MCK 2.03 dilutions was evident (Table 3.26). During the March 1993 test, reproduction of the daphnids was low both in the controls (7.0 offspring per female) and in McCoy Branch water (MCK 1.56, MCK 2.03, and MCK 2.53). In this instance, the mean numbers of offspring produced by daphnids in water from the three McCoy Branch sites was greater than that of the control. This response does not rule out the possibility that water from McCoy Branch was toxic during the March 1993 test, but the more conservative explanation is that the daphnids used in the test were stressed inadvertently either before or during the test.

**Snail feeding-rate tests.** The feeding rates of *Elimia* in water from MCK 1.56 and MCK 2.03 were very similar to feeding rates of the snails in diluted mineral water controls in all three test periods (Table 3.27). At 25°C, the mean feeding rates ranged from about 0.14 mg to 0.18 mg of wet weight of lettuce eaten per day (Table 3.27). In two of the tests, water

from MCK 1.56 and MCK 2.03 was spiked with a high concentration of McCoy Branch sediment (1 g/L and 1.5 g/L) to determine if sediment contaminants or physical nature of the sediments interfered with snail feeding. The feeding rates of the snails were not inhibited by the sediment in either of these tests (Table 3.27). Water temperature strongly influenced the feeding rates of the snails, as would be expected based on simple physiological principles, but there was no significant interaction between the temperature and sediment treatments. Thus, we conclude that the snail test results provide no evidence for acute toxicity either from the water or sediments of MCK 1.56 or MCK 2.03. It is possible that longer exposures to these media would reveal snail behavioral differences that might be related to the presence of contaminants.

**Laboratory and in situ earthworm tests.** The two earthworm tests showed that FCAP and Sluice Channel Area ash might be problematic for soil organisms. In the in situ test, earthworm survival in FCAP ash at six sites was substantially lower than in peat moss (Table 3.28). The addition of food increased survival in both the ash and peat moss soil, but survival even in food-augmented FCAP ash was still very low compared to nonamended peat moss (Table 3.28). The low survival of the worms in the FCAP ash did not appear to be explained by pH or water or organic matter content. In the laboratory test, ash from the FCAP-1 site adversely affected earthworm survival. On the average, worms lost more body weight in all ash treatments than they did in any of the control- or reference-soil treatments (Table 3.30). This difference was nearly significant ($p = 0.066$). On average, autoclaving the ash samples used in this test increased mean survival of the earthworms, but this effect was not statistically significant ($p = 0.152$). Additionally, mean survival in autoclaved FCAP-1 site ash was only 52.5%. Ash texture, or ash-associated contaminants, are factors that could account for earthworm mortality in the in situ and laboratory tests and/or weight-loss patterns noted in the laboratory test.

**Physicochemical water quality factors.** Water quality data from MCK 1.56 and MCK 2.03 were evaluated by ANOVA and by inspecting correlative relationships among conductivity, alkalinity, and hardness. The results of these analyses indicate that water quality conditions at MCK 1.56 and MCK 2.03 are similar to other stream sites on the ORR where impacts could be described as slight to moderate. The correlative analysis that was used provides a good means for detecting major chemical perturbations (i.e., those that perturb naturally established relationships among conductivity, alkalinity, and hardness), but may not be very sensitive to the presence of low concentrations of toxic contaminants. The analyses of the water quality factors showed that the McCoy Branch sites that were tested for toxicity could be distinguished from one another statistically based on factors such as conductivity, alkalinity, and hardness. However, these analyses do not provide evidence for or against the presence of toxicants. Additionally, since no strong evidence for toxicity was detected, no relationships could be established between toxicity and measurements of pH, conductivity, alkalinity, or hardness. Computations of CPI suggested that water quality conditions in McCoy Branch were more similar to noncontaminated reference streams on the ORR than they were to conditions typical for streams such as East Fork Poplar Creek.

### 3.2.6.5 Benthic macroinvertebrates

Benthic macroinvertebrates are those organisms that live on or in the substrate of flowing and nonflowing bodies of water and are large enough to be seen without the aid of magnification. The limited mobility and relatively long life spans (a few months to more than a year) of most taxa make them ideal for evaluating the ecological effects of various

disturbances to streams (Platts et al. 1983). The composition and structure of the benthic community reflect the relatively recent past, and can be considerably more informative than alternative methods that rely solely on water quality analyses but miss the potential synergistic effects often associated with multiple perturbations. The objective of this survey was to describe the composition and structure of the benthic macroinvertebrate community in upper McCoy Branch and evaluate these attributes for evidence of possible impacts associated with past discharges of fly ash into McCoy Branch.

### 3.2.6.6. Materials and methods

Benthic macroinvertebrates were surveyed at two sites on McCoy Branch above Roger's Quarry located 2.03 km (MCK 2.03) and 2.53 km (MCK 2.53) from the mouth of the stream (Fig. 3.16). Because the FCAP is situated in the headwaters of the stream, a single site in upper White Oak Creek, located ~6.8 km (WCK 6.8) from the mouth of the stream served as a reference site and was surveyed concurrently (Fig. 3.17). Samples from surveys were taken on July 15, 1992, and again on March 30, 1993.

One sample was taken on each date from each site with an aquatic dip net (mesh of 500 $\mu$m) by disturbing all identifiable habitat types (e.g., riffles, pools, leaf packs, undercut banks, root mats, etc.) by foot and/or hand and then holding the net downstream of, or sweeping the net through, the suspended materials and organisms. A reach of ~30 to 35 m was sampled over a 10 to 15 min period. After collection of a sample, it was preprocessed in the field by placing it into a large bucket partially filled with water, vigorously swirling it to suspend the lighter debris and organisms, and then pouring the suspended materials and water through a fine-meshed net (mesh of 368 $\mu$m), being careful not to allow large pieces of debris and rocks to enter the net. This process was repeated three or more times for each sample until the entire contents of the dipnet had been processed. After pouring the suspended material through the net, the material remaining in the bucket was poured into an 8- by 10-in., white plastic tray and visually scanned for additional organisms, particularly for mollusks (snails and clams) and insects that use rocks and other debris for construction of cases. Any additional organisms collected were placed into a sample container. Each sample was placed into a prelabeled, polypropylene-coated glass jar, preserved in 95% ethyl alcohol, and returned to the laboratory for processing.

In the laboratory, the contents of each sample were initially placed in a #60 sieve (250 $\mu$m) and washed with water following the same procedures as described in Wojtowicz and Smith (1992) for washing ethanol from quantitative benthos samples. Each sample was then processed in two stages: semiquantitative and qualitative. For the first stage (i.e., semiquantitative stage), a small amount of sample (i.e., $\approx$1 teaspoon) was randomly removed from the sieve and placed into an 8- by 10-in., white plastic tray containing water, and *all* organisms were removed from the debris under 2x magnification. After this process was repeated nine additional times, the organisms from these first 10 trays were identified to the lowest practical taxon and the total number of individuals of each taxon counted. The remainder of the sample was included in the second processing stage (i.e., qualitative stage). This stage entailed placing a small portion of the remaining unprocessed sample into a white plastic tray and then removing only specimens of those taxa *not* observed during the first processing stage; this process was repeated until the entire sample had been sorted. The organisms removed during the second stage of processing were identified only. The organisms *and* data from the qualitative and semiquantitive portions of each sample were maintained separately.

Management and analyses of semiquantitive and qualitative data were done on computer with the Statistical Analysis System (SAS Institute 1985). Analyses of semiquantitative data included simple, nonstatistical, site comparisons of parameters such as relative abundances of major taxa, total richness, and Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness. Data from the qualitative portion of each sample was used to supplement the taxonomic richness and composition data obtained from the semiquantitative portion of the sample.

### 3.2.6.7 Results

A checklist of the invertebrates collected from each site in McCoy Branch and White Oak Creek appears in Table 3.34. Also included in Table 3.34 is an estimate of the relative abundance for each taxon collected. During both collection periods, the predominant groups of organisms at all sites were the Chironomidae and the taxa within the Ephemeroptera, Plecoptera, and Trichoptera (EPT; Fig. 3.18). Both of these groups combined accounted for > 75% of all invertebrates collected during both sampling periods.

In July 1992, the chironomids were the most abundant taxa collected at MCK 2.03 and MCK 2.53, comprising > 60% at both sites (Fig. 3.18). Particularly abundant were the Chironomini at MCK 2.03 and Orthocladiinae at MCK 2.53 (Table 3.34). In contrast, chironomids comprised < 25% of the invertebrates at WCK 6.8 during the same period. The relative numbers of the EPT taxa comprised 23.7% at MCK 2.03 and 16.0% at MCK 2.53, while this group comprised 66.2% at WCK 6.8. The dominant EPT taxa at the three sites were *Hydropsyche depravata* at MCK 2.03, and *Leuctra* at MCK 2.53 and WCK 6.8.

In March 1993, the relative abundance of chironomids was less at all sites than in July (Fig. 3.18). Most notable was the difference at MCK 2.53 where the relative abundance of the chironomids was ≈ 60% less than in July, and where the group's relative abundance was similar to that of the group's relative abundance at WCK 6.8. Although the relative abundance of the chironomids at MCK 2.03 was less in March than in July, the change was considerably less than at MCK 2.53 (≈ 20%) and the group's relative abundance at MCK 2.03 was still considerably higher than at WCK 6.8 (9.6%). At WCK 6.8, chironomid relative abundance differed little between sampling periods. The relative abundance of EPT taxa at the McCoy Branch sites changed considerably from July with values of 50.6% at MCK 2.03 and 74% at MCK 2.53. The EPT taxa with the greatest numbers were *Eurylophella* at MCK 2.03 and *Amphinemura* at both MCK 2.03 and MCK 2.53. At WCK 6.8, the relative numbers of EPT taxa were somewhat higher in March than in July, representing 73.4% of the total taxa collected. The genera *Baetis, Amphinemura*, and *Diplectrona* were the most numerous EPT taxa.

The total number of taxa (i.e., total richness) and the combined richness of the Ephemeroptera, Plecoptera, and Trichoptera (i.e., EPT richness) exhibited some differences between sampling dates (Fig. 3.19). In July 1992, total richness was 30 at both McCoy Branch sites, while the total richness at WCK 6.8 was 45. This is in contrast to March 1993, when total richness at the McCoy Branch sites was 46 and 50 at MCK 2.03 and MCK 2.53, respectively, which was comparable to the 48 taxa collected at WCK 6.8. EPT richness showed a similar trend between sampling periods, in that during July, 11 EPT taxa were collected at MCK 2.03 and MCK 2.53, while 19 were collected at WCK 6.8 (Fig. 3.19). In March, EPT richness was more similar among the three sites, with totals of 21 at MCK 2.03, 26 at MCK 2.53, and 27 at WCK 6.8.

### 3.2.6.8 Conclusions

Data from the two benthic macroinvertebrate surveys conducted in July 1992 and March 1993 exhibited no evidence of strong impact at either McCoy Branch site. Except for minor differences in taxonomic composition and the relative abundance of existing taxa, the communities existing at both McCoy Branch sites and WCK 6.8 in March 1993 appeared to be similarly balanced (i.e., a variety of taxa) and to contain several taxa within groups that overall, are relatively intolerant of pollutants, such as Ephemeroptera, Plecoptera, and Trichoptera (Lenat 1988). The only exception to this may have been at MCK 2.03 where chironomids comprised a much greater proportion of the community than at either MCK 2.53 or WCK 6.8. As conditions become more stressed the proportion of chironomids generally increase (Wiederholm 1984, Smith 1993). However, since this was a general survey with only one sample being collected in March, this difference could be due to sampling error and thus an artifact.

Overall in March, taxa richness values were comparable among the three sites, which would suggest that if any impacts were occurring, they were relatively minor. In July 1992, differences between WCK 6.8 and the two McCoy Branch sites were much greater. The benthic community in McCoy Branch during summer low flow was dominated by midge taxa with reduced total and EPT richness. Under higher flow conditions during March, when contaminant or water quality stressors would be diluted, the community was more balanced with values for total and EPT richness comparable to those for White Oak Creek. During summer, when flows are lowest and temperatures are highest, the aquatic communities may be exposed to worst case conditions, thus any stress associated with contaminants could be amplified. Since arsenic and selenium have previously been implicated as two possible contaminants in McCoy Branch contributing to moderate stress to the invertebrates (Tolbert and Smith 1992) it is possible that these data may also be indicative of stress from these metals or some other unidentified factor(s). Thus, these results suggest that water quality in McCoy Branch may be impacted by past operations or existing conditions in the watershed.

As an alternative hypothesis, it is also possible that the differences between WCK 6.8 and the McCoy Branch sites are reflective of natural regional variability. The differences in taxa composition may also show that White Oak Creek has more habitat and more stable flow conditions than McCoy Branch. Considerable variability exists not only among relatively undisturbed streams flowing south off of Chestnut Ridge (Smith 1993), it also occurs among several streams in and near the ORR (Smith 1992). Since McCoy Branch is unique among streams in the area in having had fly ash discharged into its headwaters, the ability to distinguish natural variability from changes due to impact will be difficult without more rigorously designed field and laboratory studies.

### 3.2.6.8 Soil (ash) invertebrate survey

A preliminary survey of invertebrates living in the ash was conducted in July 1993. The purpose of this survey was to determine the abundance of earthworms present at the site to perform an earthworm bioaccumulation study (as described in the Sampling and Analysis Plan). Five samples, consisting of 1 $ft^3$ of soil, were hand-sorted using a 2.36-mm sieve. Samples were excavated ~15 ft apart in a transect within FCAP-1. The surface ash was primarily barren with the exception of a few larval and adult arthropods. Specimens collected include:

sample 1: 0 macroinvertebrates,
sample 2: 2 small beetle larvae (*Coleoptera*),
sample 3: 1 millipede (*Diplopoda*),
sample 4: 1 millipede, and
sample 5: 3 millipedes.

Two additional samples (1 $ft^3$ soil) were collected and hand-sorted from SLUICE-1. Similar organisms with comparable abundance were present ( sample #1: two millipedes and one beetle larvae; sample #2: three millipedes). No earthworms were present in the ash in the FCAP or the Sluice Channel Area samples. The specimens collected from the ash were not sufficient for contaminant analysis due to low numbers and sample weight. The apparent absence of earthworms in the ash suggests that this may not be a significant pathway for current contaminant exposure for certain avian species (i.e., woodcocks, robins) and other vermivores.

### 3.2.6.9 Heavy metal bioaccumulation in vegetation

Ash typically contains nutrient elements for plant survival and growth; however, toxic elements may also be present (Furr et al. 1976). Plants living on the FCAP are subjected to elevated metal concentrations (Al, As, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Mg, Na, Ni, Pb, Se, Tl, V, and Zn) found in the fly ash (Sect. 3.2.1). Although contaminants may not reach phytotoxic levels, heavy metals (arsenic and selenium) accumulated in plant tissue may exist at potentially toxic levels for herbivorous wildlife (Adriano et al. 1980; Furr et al. 1977) (i.e., white-tailed deer, small mammals). Many studies have shown that selenium is readily absorbed and accumulated by plants (Arthur et al. 1992a, 1992b; El-Mogazi et al. 1988; Adriano et al. 1980; Scanlon and Duggan 1979; and Furr et al. 1977). Plants should ideally contain between 0.05 and 0.3 ppm (dry weight) of selenium for sufficient animal metabolism supplied by dietary intake. Higher concentrations (3 to 40 ppm) of selenium may result in chronic toxicity to foraging animals (Ohlendorf 1989). The primary objective of this study was to determine the degree of bioaccumulation of certain heavy metals (As, Cd, Cr, Pb, Se, and Tl) in vegetation at the FCAP, thus providing information for contaminant exposure and risk to wide-ranging and resident herbivorous wildlife (Sect. 6.2).

### 3.2.6.10 Materials and methods

Plants were sampled at two distinct vegetative zones on the FCAP (FCAP-1: Sycamore area and FCAP-2: Willow area) to determine concentrations of arsenic, cadmium, chromium, lead, selenium, and thallium. The plants sampled in mid-September 1992 were red maple (*Acer rubrum*), box elder (*Acer negundo*), eastern cottonwood (*Populus deltoides*), eastern red cedar (*Juniperus virginiana*), sycamore (*Platanus occidentalis*), willow (*Salix* sp.), a grass (*Eulalia viminea*), and japanese honeysuckle (*Lonicera japonica*). The Sluice Channel Area (SLUICE-1), located on a hillside north of the FCAP in the vicinity of the former effluent channel from the Y-12 Plant, was sampled at the same time. Plants sampled at the sluice included red maple, sycamore, eastern red cedar, and *Eulalia*. Individual trees were sampling units, and only leaves were collected from each tree. Grasses (*Eulalia* sp.) were sampled by compositing both leaves and stems at each sampling point. Honeysuckle was sampled in a manner similar to grasses. The samples were not washed and were frozen during storage. Washed samples would not be representative of the plant material consumed by herbivores or litter detritivores. Controls were selected from archived, dry foliage samples collected in August or September (1988 or 1991) from Walker Branch Watershed. The control samples

included leaves from dogwood (*Cornus florida*), sourwood (*Oxydendrum arboreum*), American beech (*Fagus grandifolia*), sweet gum (*Liquidambar styraciflua*), eastern red cedar, sycamore, box elder, red maple, and *Eulalia*.

Plant samples were freeze dried and ground to a fine, homogenous powder using a Tecator sample mill. Samples were then submitted to Roy F. Weston, Inc., for metals analysis. The samples were prepared for analysis as follows: (1) a 200-mg portion was weighed into a Teflon acid digestion bomb (Parr Model 4782); (2) concentrated nitric acid (2.5 mL) was added to the sample, which was irradiated in a microwave oven for 90 seconds at 350 watts; and (3) the resultant solution was diluted with deionized water to give a final volume of 100 mL. Total metals analysis (As, Cd, Cr, Pb, Se, and Tl) was performed by Graphite Furnace Atomic Absorption. Mercury was analyzed by Cold Vapor Atomic Absorption. Quality was assured for these analyses by the use of standard procedures, calibration of instruments, and maintenance of laboratory control charts for all instruments. Statistical comparisons were made using analysis of variance (SAS Institute 1988).

### 3.2.6.11 Results

Heavy metal concentrations for individual plant species collected at Walker Branch Watershed (Control), FCAP-1, FCAP-2, and the SLUICE-1 are presented in Table H.1 (Appendix H). Average concentrations of metals in deciduous trees, eastern red cedar, and ground cover collected from all sites are summarized in Table 3.35.

**Thallium.** Concentrations of thallium in plants from the United States do not normally exceed 0.5 μg/g ash (Gough et al. 1979), or ≈7 μg/g dry mass (based on an ash weight:dry weight ratio of 0.075 for leaves). Thallium was not present above the detection limit of 1 μg/g dry mass in any of the 70 plant samples analyzed (Table H.1).

**Lead.** Lead was not above the detection limit of 1 μg/g dry mass in any control samples or in samples from the sluice site (Table H.1). Two vegetation samples from FCAP-1 had lead concentrations between 1 and 2 μg/g dry mass, but 95% of the vegetation samples from the FCAP were less than the detection limit.

**Arsenic.** Arsenic in control samples was not above the detection limit (1.00 μg As/g dry mass). Slightly elevated mean arsenic concentrations were found in deciduous tree leaves (1.53 to 1.83 μg/g) at the FCAP-1 and FCAP-2. These mean arsenic concentrations were significantly different from controls ($p < 0.01$).

**Cadmium.** Cadmium was detected in 1 of 19 control samples, and the concentration (0.30 μg/g) was slightly above the detection limit (0.25 μg Cd/g dry mass). Cadmium was also detected in 1 of 12 samples collected at SLUICE-1 (the concentration was 0.60 μg/g; all other samples were at or below the detection limit). Under natural conditions, cadmium concentrations in plants are low (Shacklette 1972), and concentrations exceeding 3 μg Cd/g in plant tissues are associated with reduced growth (Gough et al. 1979). Mean cadmium concentrations in deciduous tree leaves from the FCAP-2 were seven times greater than cadmium concentrations in control plants and were significantly different from those measured at other locations ($p < 0.001$) (Table 3.35). However, concentrations at the FCAP were not outside of the range expected for vegetation from natural environments (Connor and Shacklette 1975).

**Chromium.** Chromium was not above the detection limit of 3 μg/g dry mass in plant samples from the control or SLUICE-1 locations. With the exception of a few samples, chromium was not found above the analytical detection limit in leaves from trees sampled on the FCAP. However, all samples of ground cover (honeysuckle and grasses) at the FCAP had detectable levels of chromium. The average concentration was 10.07 μg/g. Based on plants growing in serpentine soils, toxicity symptoms can occur at foliar concentrations exceeding 18 μg Cr/g dry mass (Gough et al. 1979).

**Mercury.** Mercury concentrations in plants from the FCAP and Sluice Channel Area sites did not exceed those found in deciduous tree leaves from the control site. In most instances, mercury concentrations in plants from the FCAP were at or below the limit of detection (0.05 μg Hg/g dry matter). Mercury concentrations in tree leaves from the control site were actually greater than those at FCAP-1 or SLUICE-1 ($p < 0.001$). The reported range of natural mercury concentrations in plants from Missouri is <0.025 to 0.050 μg/g dry matter (Connor and Shacklette 1975).

**Selenium.** Selenium concentrations in leaves from deciduous trees at FCAP-1 and FCAP-2 were significantly greater than those found in trees from the control and SLUICE-1 sites ($p < 0.001$). The highest selenium levels were found in leaves from deciduous trees at FCAP-2 (mean = 29.1 μg/g, Table 3.35). Ground covers and eastern red cedar at the FCAP exhibited selenium concentrations that were less than those found in leaves from deciduous trees ($p < 0.05$), but were higher than reference site concentrations. Sensitivity to selenium toxicity is so variable among plants that general toxicity levels cannot be established (Gough et al. 1979). Vegetation growing in seleniferous areas of Canada contain from 3 to 4000 μg Se/g dry matter (Gough et al. 1979).

### 3.2.6.12 Discussion

Chromium and selenium were the metals found in highest concentrations in plants from sites at the FCAP. Following root uptake, there is little translocation of chromium to aboveground plant parts (Parr and Taylor 1980). Therefore, elevated concentrations of chromium in ground covers on the FCAP are probably due to the resuspension of fine dust to leaf surfaces (leaves were not washed prior to analysis in this study). Previous studies have shown no increase in chromium and lead uptake by woody plants living in fly ash, which indicates that these elements are fixed in fly ash and are present in available forms (Scanlon and Duggan 1979).

The highest selenium concentration observed for tree leaves from the FCAP was 97 μg/g dry mass. For the point of comparison, severe toxicity (more than 80% yield reduction) in bush beans and alfalfa is associated with selenium concentrations of 900 to 1000 μg/g dry leaf mass (Wallace et al. 1980, Wan et al. 1988). However, for many agricultural plants the level of tolerance to selenium is <50 μg/g dry mass (Wan et al. 1988). Alfalfa yields can be reduced by 10% when selenium concentrations in plant tissues exceed 25 to 30 μg/g dry mass (Soltanpour and Workman 1980). Comparable reductions in yield have been observed in bush beans when foliar levels reach 25 μg Se/g dry mass (Wallace et al. 1980). Based on measured concentrations and reported toxicity levels in agricultural plants, there is some potential for selenium toxicity in trees growing on the FCAP. However, due to the widely varying resistance to selenium toxicity in plants (Gough et al. 1979) and because internal and external contamination were not distinguished in this first survey, the importance of potential selenium toxicity to plants at the FCAP is not known. This possibility needs to be evaluated in

follow-up studies of selenium toxicity to tree seedlings. It is unlikely that the selenium concentrations measured in tree leaves can be attributed to surface contamination by resuspended dust because leaves were usually sampled at a height of more than 1 m from ground level.

Concentrations of selenium in plants on the FCAP are high enough to warrant further study of possible toxicity effects on small mammals and soil invertebrates. The influx of selenium to soil by means of autumn leaf fall may present some potential for toxicity to litter fauna. Complexation with organic matter, adsorption of selenium to iron and aluminum oxides and oxyhydroxides, and high sulfate concentrations in soil all tend to reduce the root uptake of selenium (Logan et al. 1987). Wild plants (white sweet clover) growing in fly ash have been found to accumulate over 200 ppm of selenium (Gutenmann 1976). Selenium accumulates in protein-containing tissues of plants, like seeds, because of biochemical and metabolic similarities to sulfur (Adriano 1986). No information is currently available on the levels of selenium in seeds from the FCAP. Concentrations of selenium in seeds comparable to those measured in tree leaves ($\approx$20 $\mu$g/g) could be of concern from the standpoint of toxicity to small mammals. As a point of reference, selenium concentrations in feeds > 5 $\mu$g/g dry mass are considered potentially toxic to animals (Gough et al. 1979, Adriano 1986).

### 3.2.6.13 Heavy metal bioaccumulation in small mammals

Small mammals (i.e., mice, shrews, voles) inhabiting the FCAP and Sluice Channel Area are exposed to elevated concentrations of heavy metals. Exposure may occur through consumption of vegetation, surface water, and incidental exposure to ash while foraging or burrowing (Suter 1993). Small mammals are known to bioaccumulate a variety of heavy metals from contaminated environments (Talmage and Walton 1991). Talmage and Walton (1991) found a general relationship between concentrations of contaminants in soil or food and concentrations in tissues of several species, especially for nonessential heavy metals such as Cd, Pb, and Hg. Heavy metals may reach potentially toxic concentrations to the small mammals or their wide-ranging predators (i.e., birds of prey, snakes, foxes). The primary objective of this study was to determine the degree of bioaccumulation of heavy metals (As, Cd, Cr, Hg, Pb, Se, and Tl) in small mammals at Chestnut Ridge OU 2 sites and the potential for accumulation or biomagnification into higher trophic levels.

### 3.2.6.14 Materials and methods

Small mammals were trapped from three areas within Chestnut Ridge OU 2 (FCAP-1, FCAP-2, SLUICE-1) from June to October 1993. Small mammals were also trapped from Walker Branch Watershed, a reference site (Fig. 3.20). Seventy-five Sherman live traps were placed on a 11 by 7 trap grid, ~3 m apart, at both FCAP-1 and FCAP-2 (only 5 traps in 11th row). Forty pit-fall traps, spaced ~5 m apart, were installed at FCAP-1. One hundred Sherman live traps were placed on a 10 by 10 trap grid ~3 m apart at both the Sluice Channel Area and Walker Branch sites. The traps were baited with a mixture of peanut butter and oats and checked daily. Trapped specimens were euthanized by suffocation in plastic bags with dry ice. Small mammals were identified to species according to Burt and Grossenheider (1976), and body measurements (weight, body length, and tail length) were recorded. Whole-body samples were then washed with Prell® shampoo to remove external contamination and thoroughly rinsed with deionized water. Samples were then stored at -20°C in a locked freezer until submission for laboratory analysis. One laboratory rinsate sample per

ten small mammal samples was also analyzed to determine if the processing procedure (i.e., use of Prell® shampoo) was a source of contamination.

Samples were shipped overnight in a cooler containing dry ice to Southwest Research Institute for analysis. Staff at Southwest Research Institute homogenized each whole-body sample using a stainless steel blender. Three 2-g aliquots of homogenate were completely digested with boiling nitric and perchloric acid until total dissolution. The digested samples were analyzed for As, Cd, Cr, Pb, Se, and Tl by graphite furnace atomic absorption; mercury analyses were performed by cold vapor atomic absorption. Metal concentrations were reported on both a wet-weight and dry-weight corrected basis.

### 3.2.6.15 Statistical analyses

Heavy metal concentrations, standard deviations, and the coefficient of variation were calculated from the three aliquots analyzed for each animal. The coefficient of variation is the standard deviation expressed as a fraction of the mean, and is used to compare variation among aliquots independent of the magnitude of their means. Coefficient of variation values were used to evaluate the degree of homogeneity achieved by the grinding procedure.

For risk assessment purposes, upper tolerance limits were determined for all small rodents collected at Chestnut Ridge OU 2. Animals used to calculate tolerance limits and total rodent means at Chestnut Ridge OU 2 include white-footed mice (*Peromyscus leucopus*), harvest mice (*Reithrodontomys* sp.), and a pine vole (*Microtus pinetorum*) collected from FCAP-1, FCAP-2, and SLUICE-1. Insectivores, such as the short-tailed shrew (*Blarina brevicauda*), were excluded from these calculations for various reasons. These reasons include (1) shrews tended to have much higher metal concentrations than other genera; (2) shrews have different dietary habits; (3) shrews comprise a small percentage of the diet of most mammalian predators (Greenburg et al. 1988 and Story et al. 1982); and (4) shrews may have increased exposure due to a fossorial lifestyle. Tolerance limits were not calculated for rodents collected at Walker Branch due to extremely low sample size.

Tolerance limits are used to indicate the limits within which one can expect to find a proportion (P) of a population. Knowing the population mean (X) and standard deviation (S), it is possible to determine a constant K such that one can assert, with 95% confidence, that the proportion of the population contained between X - KS and X + KS is at least P (Hines and Montgomery 1980). With 95% confidence, the conclusion could be reached that a proportion of the population (e.g., 99%) would be less than the upper tolerance limit. The tolerance limits were calculated using the log-transformed means of metal concentrations, which were found to more nearly satisfy the assumptions for normality and homogeneity of variance required for conventional parametric statistical analyses. Thus, one can state with 95% certainty that 99% of the heavy metal distribution is less than the upper limit. However, there are limitations for the use of tolerance limits. Small sample sizes in conjunction with large variability of heavy metal concentrations may result in unrealistic tolerance limits.

Analysis of variance procedures (PROC GLM; SAS Institute 1988) were performed to determine if metal contamination differed among sites or species (*Peromyscus* vs *Reithrodontomys*). Because *Peromyscus* was the most commonly trapped small mammal from the FCAP, Sluice Channel Area, and Walker Branch sites, a comparison of significant differences of heavy metal concentrations between sites was conducted using the log-transformed means for this species.

### 3.2.6.16 Results

A total of 24 small mammals were trapped from the FCAP, Sluice Channel Area, and Walker Branch locations in 2980 trap-nights (Table 3.36). No animals were caught in the pitfall traps at FCAP-1. Tolerance limits for rodents at Chestnut Ridge OU 2 ranged from 0.165 mg/kg for cadmium to 26.98 mg/kg for selenium (Table 3.37). Mean concentrations of metals (wet-weight), derived from individual animal means are also summarized in Table 3.37. Elevated mean mercury (Sluice Channel Area and Walker Branch) and chromium (Walker Branch) concentrations found in *Blarina*, exceed maximum levels for all other small mammal species collected (Table 3.38).

Wet and dry-weight corrected raw data and summary statistics for whole body metal concentrations from small mammals from the FCAP, Sluice Channel Area, and Walker Branch locations are provided in Tables H.2 - H.5. Most wet-weight metal concentrations were qualified as estimated during the validation process (Table H.2).

Variation among the three aliquots from each animal is due to analytical variability (including homogenization). The EPA specifies that the relative percent difference values for duplicate samples in a solid matrix should be less than 35% if the reported value is at least five times the contract required detection limit or minimum detection limit (Energy Systems 1992). A coefficient of variation value of 25% is equivalent to a relative percent difference of 35%. Distribution percentiles were calculated using all 24 small mammal samples for those analytes where there were no values below the detection limit:

Se: 99% of the coefficient of variation values were < 20.1%,
Pb: 75% of the coefficient of variation values were < 30.5%,
Hg: 95% of the coefficient of variation values were < 24.2%, and
Cr: 75% of the coefficient of variation values were < 23.1%.

Except for lead, coefficient of variation distributions for analytes indicate that analytical variability was within acceptable limits, thus suggesting that the homogenization process was adequate.

**Thallium.** Thallium was detected in only two small mammals collected from FCAP-1 (one *Peromyscus*) and FCAP-2 (one *Reithrodontomys*). One aliquot for each animal had detectable thallium values (mean= 0.243 mg/kg and 0.202 mg/kg, respectively; Table H.2). Thallium was not detected in any animals from Walker Branch.

**Lead.** Lead was detected in small mammals (n= 24; 72 aliquot samples) from all locations. Concentrations ranged from 0.094 mg Pb/kg to 10.27 mg Pb/kg (Table H.3). Lead concentrations found in rodents at Chestnut Ridge OU 2 were slightly greater than that found in *Peromyscus* at Walker Branch (1.030 $\pm$ 2.408 mg/kg and 0.334 $\pm$ 0.228 mg/kg, respectively; Table 3.37). Although lead concentrations in rodents collected from Chestnut Ridge OU 2 locations were relatively low (< 0.82 mg Pb/kg for 15 of 17 animals), one animal (a *Reithrodontomys* from FCAP-2) had an average concentration of 10.27 mg Pb/kg. When excluding the outlier, the tolerance limit (20.10 mg/kg; Table 3.37) would be substantially lower.

**Arsenic.** Arsenic was detected in 37 of 54 and 3 of 18 small mammal aliquots (3 aliquots per small mammal) collected from Chestnut Ridge OU 2 and Walker Branch locations,

respectively (Table H.2). The mean arsenic concentration measured in *Peromyscus* at the FCAP is significantly higher ($p < 0.05$) than that found in *Peromyscus* at the Sluice Channel Area or Walker Branch locations (mean= 0.171 ± 0.121 mg As/kg (FCAP), 0.051 ± 0.001 mg As/kg (Sluice Channel Area), and 0.054 ± 0.007 mg As/kg (Walker Branch), respectively; Table 3.37).

**Cadmium.** Cadmium was detected in 36 of 54 and 17 of 18 small mammal aliquots (3 aliquots per small mammal) collected from Chestnut Ridge OU 2 and Walker Branch locations, respectively (Table H.2). Mean cadmium concentrations in rodents from Chestnut Ridge OU 2 sites (0.022 ± 0.016 mg Cd/kg) was not significantly different ($p= 0.749$) from levels in rodents collected at Walker Branch (0.022 ± 0.007 mg Cd/kg) (Table 3.37). Cadmium concentrations in small mammals found at Chestnut Ridge OU 2 and Walker Branch are below reported concentrations found at other reference sites (0.1 to 1.4 mg/kg; Talmage and Walton 1991).

**Chromium.** Chromium was detected in small mammals collected (n= 24; 72 aliquot samples) at all locations having mean concentrations ranging from 0.679 mg Cr/kg to 5.4 mg Cr/kg (Table H.3). Levels of chromium in rodents at Chestnut Ridge OU 2 locations were not significantly different ($p= 0.687$) from concentrations found in rodents at Walker Branch (mean= 1.256 ± 0.354 mg/kg and 1.587 ± 1.498 mg/kg, respectively). Acute and chronic adverse effects of chromium to mammals are caused mainly by $Cr^{+6}$ compounds (Eisler 1986). Since these analyses resulted in total chromium values, the amount of $Cr^{+6}$ that is present in the soil and biota was not determined.

**Mercury.** Mercury was detected in small mammals collected (n=24; 72 aliquot samples) at all locations. Mean concentrations ranged from 0.012 mg Hg/kg to 0.481 mg Hg/kg. Mercury concentrations found in rodents from Chestnut Ridge OU 2 sites were higher than concentrations found in rodents from Walker Branch. However, this difference was not statistically significant ($p = 0.573$). Mercury concentrations found in *Blarina* (Table 3.38), at both the Sluice Channel Area (0.481 ± 0.121 mg Hg/kg) and Walker Branch (0.337 ± 0.145 mg Hg/kg) locations, were substantially higher (~13 times greater for *Blarina* at the Sluice Channel Area) than concentrations found in rodents (0.037 ± 0.025 mg Hg/kg) at Chestnut Ridge OU 2.

**Selenium.** Selenium was detected in small mammals collected (n=24; 72 aliquot samples) at all locations with mean concentrations ranging from 0.248 mg Se/kg to 5.64 mg Se/kg. The mean selenium concentration found in rodents (*Peromyscus* and *Reithrodontomys*) at the FCAP was significantly higher ($p < 0.05$) than concentrations found in *Peromyscus* at the Sluice Channel Area and Walker Branch locations (Table 3.37).

### 3.2.6.17 Discussion

Cadmium, chromium, lead, mercury, and thallium concentrations in small mammals collected from Chestnut Ridge OU 2 were similar to concentrations in animals collected from the reference site. However, one small mammal living on FCAP-2 had an average lead concentration of 10.27 mg/kg; the tolerance limit for the population was 20.1 mg Pb/kg. Potentially high lead concentrations found in a small proportion of the population may exceed documented toxicity levels known to cause adverse effects in experimental animals (Sects. 6.3.1 and 6.4.1). Lead adversely affects survival, growth, reproduction, development, behavior, learning and metabolism. Specifically, lead inhibits blood delta aminolevulinic acid dehydratase

activity in mice at concentrations of ≥ 0.05 mg Pb/kg body weight (Eisler 1988a). Cellular alterations were also associated with renal lead levels of 8.1 mg Pb/kg in small mammals (Goyer et al. 1970). Additionally, survival was reduced by acute oral doses as low as 5 mg/kg body weight in rats (Eisler 1988a).

Arsenic and selenium were found in significantly higher ($p < 0.05$) concentrations in *Peromyscus* at the FCAP compared to *Peromyscus* collected at reference and Sluice Channel Area locations. *Riethrodontomys* sp. at FCAP-2 also had significantly higher metal concentrations ($p < 0.05$) as compared to *Peromyscus* at the reference site. The mean concentration (1.83 ± 1.48 mg/kg) and upper tolerance limit for selenium (26.98 mg/kg) in rodents at Chestnut Ridge OU 2 exceed toxicological benchmarks and toxicity values for small mammals with potential repercussions for predators (Sects. 6.3.1 and 6.4.1). Fishbein (1977) reported minimum toxic concentrations of selenium for a lifetime exposure in rats of 0.35 mg Se/kg; this concentration in the diet caused changes in liver chemistry. Longevity and histology of the heart, kidney, and spleen were also modified when rats were exposed to 0.75 mg Se/kg diet. Selenium can cause congenital malformations in mice, rats, swine, and cattle (Harr 1978).

Arsenic was also found in concentrations (Chestnut Ridge OU 2 rodent mean= 0.126 ± 0.010 mg/kg; tolerance limit = 1.087 mg/kg) which may cause adverse effects in rodents and their predators (Sects. 6.3.1 and 6.4.1). Blakely et al. (1980) showed immuno-suppressive effects from a dose of 0.10 mg As/kg from mice drinking water containing 0.5 mg As/L (sodium arsenite) for 3 weeks. NRCC (1978) has shown that a maternal dose of 5 mg $As^{+5}$/kg BW (arsenate) in hamsters (*Cricetus* sp.) and 10 mg $As^{+5}$/kg in mice (*Mus* sp.) caused some fetal mortality and fetal malformations (mice only). Mice that were fed 5 mg sodium arsenite for three generations showed a reduction in litter size (Pershagen and Vahter 1979, as cited by Eisler 1988b).

### 3.2.6.18 Species and site specific differences

Statistical analyses (PROC GLM/Tukeys; SAS Institute) showed that selenium and arsenic concentrations in *Peromyscus* at FCAP-1 were higher ($p < 0.05$) than concentrations in *Peromyscus* at the Sluice and Walker Branch sites. Metal concentrations in *Peromyscus* at the Sluice Channel Area and Walker Branch sites were similar ($p > 0.05$). Arsenic and selenium concentrations found in *Reithrodontomys* were similar to that found in *Peromyscus* at the FCAP; however, arsenic was not statistically different from *Peromyscus* at the Sluice Channel Area or Walker Branch sites. Therefore, it appears that mice in the FCAP area are bioaccumulating higher levels of selenium than at other areas on Chestnut Ridge OU 2 or the reference site.

A nonstatistical comparison was made between *Blarina brevicauda* and rodents collected. Heavy metal concentrations, with the exception of As, in *Blarina* (Sluice Channel Area and Walker Branch sites) (Table 3.38) exceeded those in most *Peromyscus*. Average mercury concentrations in shrews were at least 9 times greater than average concentrations found in other species (Table 3.37; Table 3.38). The shrew has the potential to ingest larger concentrations of metals in the diet. Shrews are voracious feeders, ingesting an equivalent of their body weight each day. The shrews diet consists of earthworms, insects, millipedes, snails, slugs, and, occasionally, other small mammals (Talmage and Walton 1991, Burt and Grossenheider 1976). *Peromyscus* are opportunistic feeders, primarily herbivorous, whose diet consists of seeds, insects, fruit and green vegetation (Brown 1964). Boggess (1977) found

higher metal body burden concentrations (e.g., lead) in insectivorous rodents relative to herbivores. Heavy metals present in the soil may also contribute to increased exposure in *Blarina* due to their burrowing lifestyle. Mean selenium, and mercury concentrations in *Blarina*, collected at Walker Branch watershed and the sluice area exceed toxicological benchmarks and toxicity levels known to cause adverse effects in other organisms (Sects. 6.3.1 and 6.4.1).

### 3.2.6.19 Conclusions

Average concentrations of As, Se, and Pb found in small mammals collected at Chestnut Ridge OU 2 are higher than concentrations of those metals found in small mammals from the reference site. Concentrations and tolerance limits exceed levels which may pose a toxicological risk to the small mammals and their predators (Sect. 6.4.1). Although Cd, Hg, and Cr bioaccumulated in small mammals on Chestnut Ridge OU 2 sites, concentrations were relatively low. Since heavy metals, with the exception of thallium, were found in small mammals at the FCAP, these animals serve as a potential pathway for contaminant transfer to predators and other wide-ranging species.

### 3.2.6.20 Miscellaneous Biological Monitoring and Abatement Program projects

**Banded Sculpin Reintroduction study.** The Biological Monitoring and Abatement Program has initiated a fish introduction study on UMB. The study involves transplanting banded sculpins (*Cottus carolinae*) into UMB and observing population parameters. These parameters include survival, distribution, density, biomass, condition factor, feeding patterns, reproduction, and recruitment. This study will determine if UMB can support a fish population and if so, how might that population differ from an existing population in a noncontaminated reference stream.

Banded sculpins were obtained from Hansard Mill Branch (Hansard Mill km 1.23) for introduction into UMB. One Smith-Root model 15-A backpack gas powered electrofisher was used to collect the sculpins, on three sampling dates. The collections were made on August 25, 1992 (110 sculpins), August 28, 1992 (75 sculpins), and October 2, 1992 (52 sculpins). Fish were transferred to the laboratory and placed in a flow-through tank for recovery from the electrofishing process. Following recovery, the sculpins were anesthetized with FINQUEL Tricane Methanesulfonate (MS 222), and were injected with Liquitex nontoxic acrylic polymer emulsions as per Lotrich and Merideth (1974) for identification marking. The paint was injected in the loose subcutaneous area on the ventral portion of the caudal peduncle. Vivid lime green and Brilliant orange paint were used to destinguish between fish introduced at the upstream (MCK 2.12) and downstream (MCK 1.97), respectively.

Initial fish population surveys at UMB took place on April 10, 1992, and October 2, 1993, before sculpin introductions. Four adult (8-12 cm total length) sculpins were collected. Extensive sampling of all tributaries above the quarry was performed but no potential refuges were discovered.

On October 3, 1992, 90 marked sculpins were released at each site in UMB. A second group of sculpins were captured from Hansard Mill km 1.23 and injected with Liquitex Brilliant Blue (Phthalocyanine Blue) (downstream) and Liquitex Medium Magenta (Quinarcidone Magenta). Nine blue and nine magenta marked sculpins were released in the

MCK 1.97 and MCK 2.12, respectively, on August 2, 1993. Recovery of fish was achieved by electrofishing 50 to 300 m of stream in three to six passes. The number of sculpins recovered from UMB on four sampling occasions, to date, is presented in Table 3.39. For additional information on materials and methods and detailed future analyses (food availability and consumption; fish community; and reproductive) see Appendix A.

The number of sculpins recovered from UMB may vary due to a number of factors. These factors include: (1) age-dependent mortality, (2) starvation, (3) predation, (4) cannibalism, (5) introduction-related stress mortality, (6) immigration, (7) emigration, (8) reproduction, (9) pollutant-related stress mortality, (10) changes in sampling techniques, or (11) ineffective sampling of certain habitats.

(Note: The order of above factors does not reflect any ranking of probability.)

**Microbes in Fly Ash.** The effects of fly ash on the biomass of bacteria involved in symbiotic relationships with leguminous plants were examined (Appendix I). *Rhizobium leguminosarum* was cultured in four mixtures of sand containing 0, 33, 50, and 100% fly ash. Microbial phospholipids were extracted, isolated and quantified using thin layer chromatography-flame ionization detection. Detection of these phospholipids was used to determine bacterial biomass.

The concentration of all phospholipids decreased with increasing fly ash in the growth medium (see Appendix I, Fig. 3). Total phospholipids decreased by ~50%, with 0 and 33% fly ash in the growth medium. Fly ash alone (100%) was able to support less than 10% of the biomass of control cultures (0% fly ash).

These results suggest that fly ash is toxic to these microorganisms or is physically unsuitable as a growth substrate. This may have implications for the ability of leguminous plants to colonize nitrogen-poor ash deposits that do not support populations of *Rhizobium* inoculum. However, the deep ash of FCAP-2 at Chestnut Ridge OU 2 supports the leguminous tick trefoil (*Desmodium* sp.).

For additional information see Appendix I.

### 3.2.7 Data Quality

The usefulness of the data described in the preceding sections for preparing the HHRA, ERA, and FS is predicated on meeting data quality objectives for the data set and the project. Analytical data quality is evaluated via the data validation process, which provides an independent review of analytical laboratory results/performance and results in verification or qualification of the laboratory results. Aside from the data validation issues related to data quality, there is a goal or objective for completeness of the data set. Completeness is generally identified as a percentage of the valid measurements versus the total measurements planned. Data rejected in the validation process impact the completeness of the data set, as well as samples not collected because of operational problems or unexpected conditions/ circumstances. Completeness of the data set must also be evaluated in terms of whether or not the data population is adequate to verify the conceptual model and then meet specific error tolerance for risk-based decisions. The goal of any remedial investigation is to adequately characterize the nature and extent of contamination, and not necessarily just to

collect a specific number of valid samples. These objectives must be balanced in reaching the overall goals of the project.

### 3.2.7.1 Phase 1 RI data

Data quality and validation issues related to the Phase 1 Remedial Investigation conducted by CH2M HILL are presented in Appendix B along with the Phase 1 data. This information was essentially excerpted from the Phase 1 RI report in total to provide a full explanation of those data.

The Phase 1 RI preceded development of Energy Systems guidelines for quality control of data collected under the Environmental Restoration Program described in ES/ER/TM-16 (Energy Systems 1992b). Validation of the Phase 1 data followed other existing guidelines that are similar to the requirements in ES/ER/TM-16, but are not as rigorous as the current requirements. Not withstanding the differences, in general, the Phase 1 data, as qualified, have been deemed usable and are included in the data set used for the risk assessments and the feasibility study.

The Phase 1 RI data were deemed inadequate to characterize the site and risk to public health and the environment because sample of coal ash in the FCAP were not collected and analyzed; the HHRA was not complete and did not address the future homesteader scenario; and an ERA was not completed. The scope of the Phase 1 work was developed under the RCRA Program that is not a risk-based program nor does it require development of DQOs through a formalized process, as does CERCLA. A conceptual site model was developed; however, it did not include ash as a contaminated medium and therefore was incomplete. The Phase 1 data set did address the objectives of the study but was not adequate to meet the objectives of the CERCLA process; therefore, a Phase 2 RI was planned and conducted under CERCLA.

### 3.2.7.2 Phase 2 RI data

The Phase 2 RI was planned and conducted under CERCLA and included development of site conceptual models and DQOs specific for a risk-based evaluation. Data quality issues associated with this investigation are addressed in terms of laboratory analytical data quality and the completeness and quality of the overall sampling effort in terms of providing adequate data to support the decision making process, (i.e., the risk assessments and the FS).

**Laboratory data quality.** The Phase 2 RI was planned and conducted in accordance with the requirements of ES/ER/TM-16 (Energy Systems 1992b). All of the analytical data were subjected to Level C data validation as described in ES/ER/TM-16. The Phase 2 data presented in Appendix C include the qualifiers designated by the data validators. Generally, the vast majority of the laboratory data are acceptable and usable, and the DQO for completeness (90%) was met or exceeded. A total of 194 results were rejected for a variety of reasons out of 12,677 total results (<2%).

Appendix C also includes all QC sample results followed by two tables summarizing the number of results qualified by each validation code (i.e., U, J, R, etc). A total of 9104 results of 12,677 (~72%) were nondetects (U's); another 1138 results (9%) were qualified as UJ, which means the analyte was not detected above the sample quantitation limit but the quantitation limit is estimated; 1873 results (14.8%) were not qualified at all; 325 results

(2.5%) were estimated (J) results; and as previously noted, 194 results (1.5%) were rejected (R). The remaining 43 results were qualified by less common qualifiers, such as B, BJ, E, etc. This analysis shows that over 80% of the results of analysis were nondetects, with only 14.8% of the results showing contamination levels above quantitation limits. The following is a brief description of specific quality issues identified during the data validation process and their effects on usability of the data set. The discussion is organized by major categories of parameters (i.e., organics, inorganics, radionuclides) and secondarily by medium (soil, water, sediment, ash).

**Inorganics–water samples.** There were two analytical quality issues related to water samples (surface water, groundwater and water QC samples). The first was the presence of common anions and cations in the blanks, including, Fe, Mn, Ca, Zn, K, and Na. The presence of these constituents is not related to site activities and their occurrence in natural waters is well known. This does not represent a quality issue for the data set.

The second quality issue was related to noncompliant matrix spike recoveries for Se, As, Th and, to a lesser extent, Pb. For a majority of the groundwater and surface water samples these constituents were present at low levels (at or below the detection limit) and appear to be subject to matrix related interferences. A majority of the values were qualified as estimated ("J") in addition to undetected ("U"), due to the spike recovery problem. Because these constituents were generally present at or below the detection limit in batches where the spike was acceptable, and because they are known to be difficult to quantify in natural waters, the fact that the data have been qualified as estimated does not affect the useability of the data. There were no other quality issues related to measurement of inorganics in water samples collected from Chestnut Ridge OU 2. All of the data, as qualified, are included in the data set for use, as appropriate. There were no rejected data in this category.

**Inorganics–soil/sediment/ash samples.** The primary quality issue for the soils/sediment/ash - inorganic parameters data was matrix spike recoveries for the same parameters as the waters (Se, Th, As, and Pb) and others, including Sb, Cd, and Ag. In every case where there was a spike recovery problem with Sb, Cd, and Ag the concentrations of contaminants in the samples were at or below the detection limit. The problems with spike recovery of these and the other parameters is due to matrix-related interference. There were three samples (soil/sediment) for which matrix spike recoveries for selenium were less than 30% and, therefore, these data were qualified as rejected.

A secondary quality issue related to the soil/sediment/ash samples was duplicate differences greater than specified in the Functional Guidelines. Parameters affected in this area included arsenic, selenium, and iron. In all cases, this led to qualification of the data as estimated; none of the data were rejected. A factor that contributed to this problem was the high water content of the ash and sediment samples. Because many of these samples were less than 50% solids, it was difficult to obtain homogeneous aliquots of a sample for analysis. Nonetheless, all but six of the results were deemed usable and are included in the data set.

**Organics–water samples.** Organics analysis include the Target Compound List (TCL) volatile and semivolatile compounds and pesticides and PCBs. Pesticides and PCBs were analyzed on only a small percentage of the samples. Quality issues for analyses of organic compounds on aqueous samples included nonconforming instrument Relative Response Factors, method blank contamination, holding time violations, and spike recovery problems.

Problems meeting the instrument Relative Response Factor requirements in ER/ES/TM-16 resulted in rejection of the results for 1,1,2,2-Tetrachloroethane. Those requirements in ES/ER/TM-16 are stricter than National Functional Guidelines and EPA's CLP requirements; these data would not have been rejected under those other programs. Nonetheless, the rejected values were all below the detection limit as were almost all of the results for organics analyses on surface water and groundwater samples.

Various volatile and semivolatile organics were found in method blanks, field blanks (potable, deionized, and high performance liquid chromatography water) and equipment blanks. These contaminants were present in trace amounts (estimate of values near the detection limit) in a few samples or undetected; the most common analyte present in the blanks and the samples was bis(2-ethylhexyl)phthalate.

For three field blank samples for pesticide and PCB analyses the holding times for the analyses were exceeded ($\sim$45 d = more than twice the holding time), and, therefore, the results (84) were rejected. Pesticides and PCBs were measured on a small percentage of ash samples only. The aqueous samples analyzed were field blanks and equipment blanks only. The first set of field blanks were analyzed within the holding times and showed that pesticides and PCBs were not present above the detection limits.

Low spike recoveries resulted in rejection of mix results for 4-chloro-methylphenol and one result for benzo(g,h,i)perylene; all of these results were below detection, as were the nonrejected values for the remaining samples. Although a number of organic analytes were estimated or rejected based on the aforementioned analytical problems/occurrences, the data as qualified are considered useable for the risk assessment and feasibility study; organic contaminants are not prevalent or expected at this site, as verified by the data.

**Organics–soil/sediment/ash.** The vast majority of results for organic analysis were nondetects. Surrogate recovering problems resulted in rejection of 50 semivolatile results for one ash sample, and internal standard performance problems resulted in rejection of 20 volatile results for a different ash sample. A number of nondetect results were qualified as estimated based on low matrix spike recoveries. All of these problems are related to matrix interferences associated primarily with the ash samples. A number of the sediment and ash samples were nearly 50% water, which also created a problem obtaining homogeneous aliquots for analysis. A majority of the data, as qualified, were usable for performing the risk assessment and feasibility study.

**Radiochemical–water samples.** The analytical quality issues for radiochemical analyses of aqueous samples included the presence of analytes ($^{230}$Th, $^{234}$U, and, to a lesser extent, gross alpha activity) in the preparation blanks, where the blank results were greater than the minimum detectable activities (MDAs). These radionuclides were also present in the field blanks. Consistently low spike recoveries were noted for gross alpha analyses, resulting in estimation of most results. Matrix duplicate differences also resulted in estimation of some of the results for $^{238}$U and $^{230}$Th. Finally, there were a few (four) samples for which internal standard recoveries were low and results were estimated. A vast majority (75%) of the radiochemical results were below the MDAs and few values were greater than the associated uncertainties. Many of the "S" qualifiers added to the data were for non-detects. Further, many of the values that were above the MDA were within a factor of two of the MDA. None of the radiochemical results were rejected. The data, as qualified, are all usable for the risk assessment and the feasibility study.

**Radiochemical-soil/sediment/ash samples.** Matrix spike recoveries exceeding contract specified limits resulted in estimation of many of the gross alpha and gross beta results. For many of the soil results, recoveries were biased high, which may result in false positive values. The presence of $^{230}Th$ in the preparation blank resulted in qualification of many of the $^{230}Th$ results as nondetects; other $^{230}Th$ results were estimated based on matrix duplicate differences greater than specified in the contract requirements. None of the radiochemical results for soil/sediments/ash were rejected. The data, as qualified, are usable for the risk assessment and feasibility study.

**Sampling quality issues.** Generally, the investigation was conducted according to the approach specified in the approved Phase 2 Sampling and Analysis Plan. One area where this was not possible was the sampling of coal ash in the FCAP. Coal ash characterization was required for the risk assessments and FS. The original approach was designed to evaluate whether or not the coal ash was homogenous with depth. There was some thought that the coal ash present a greater depths may have different chemical characteristics due to the lack of weathering. To evaluate this question samples of coal ash were to be collected every 10 ft from the surface to the bottom and composited across the transects. Because of the soupy consistency of the coal ash beneath the vegetative zone, it was not technically feasible to collect discreet samples from depth and therefore a single sample was collected from each borehole location across the entire depth of the FCAP at that location. The conceptual model for the site did not specifically address homogeneity or heterogeneity of the coal ash in the FCAP and therefore, the fact that the coal ash had to be assumed to be homogenous based on its physical characteristics and conditions within the FCAP did not affect the overall outcome or uncertainty of the risk assessments or the FS. The representativeness of sample results is an issue relative to the uncertainty of the risk assessments, specifically. Coal ash samples were collected in a manner to provide horizontal and vertical spatial coverage representative of the entire FCAP. Although the sample size was less than originally planned (18 planned, 9 collected), the nine sample collected from throughout the FCAP (vertical and horizontal coverage) are considered representative of the contents of the FCAP and therefore provided a reasonable basis for preparation of the risk assessments and the FS.

Appendix C includes a table summarizing the relative percent difference (RPD) calculations for field duplicates. The table of RPD results for the Phase 2 data is followed by a summary table that presents average RPD values and the range of RPD values for each group of analytes (i.e., metals, organics, etc.) and each medium (i.e., ash, soil, etc.). This information provides a check on the field sampling program. Generally, for field duplicates an RPD of 35 % or less is considered excellent performance and a value of 50% or less is acceptable. The average RPDs for all analyte groups and media were less than 35% with the simple exception of radionuclides measured in surface waters. The average RPD for this category was 43%. The range of RPDs for each category is also provided in the summary table.

**Field problems.** Problems encountered in the field during implementation of the RI were minimal and did not have a substantial impact on the quality of the investigation or the sample collected. Site access was the most consistently encountered problem. Several weeks before the start of field work, construction of the SPAD landfill just east of OU 2 initiated. Truck traffic from the landfill construction often rendered the primary road to OU 2 impassable. The road dunning from Bethel Valley Road to the FCAP dam provided access only to the southern end of the site and not to the FCAP or the Sluice Channel Area. In addition, Y-12 Plant security personnel had to be contacted to unlock the gate at the

southern end of the road. The Y-12 Plant agreed to allow field personnel access to a gravel haul road to the northwest of OU 2 off of which a road to the northern portions of OU 2 branches. Access to the haul road, however, was restricted to the early morning and late afternoon. The overall impact of access problems on the investigation was time delay.

Twelve issues arose during the investigation that resulted in deviations from the approved Phase 2 Sampling and Analysis Plan. Field change request forms document these changes, all of which were discussed and approved before implementation. Five of the 12 change requests involved clarification of statements or procedures in the Sampling and Analysis Plan. For example, the Health and Safety Plan portion of the approved plan stated that all sampling equipment would be rinsed with isopropanol once during decontamination. The approved procedure, however, specifies a double rinse with isopropanol. The approved field change corrects and clarifies the procedure.

Several field changes resulted from unanticipated problems in the field. For example, problems associated with sampling the coal ash resulted in two field change requests. The first change occurred because vegetation at the southern end of the FCAP precluded drilling in all the locations specified in the Sampling and Analysis Plan. The requested change moved the southernmost boring locations northward. The second change occurred because the saturated nature of the coal ash below a depth of several feet (described in Sect. 3.2.1) precluded collection of samples from discrete depths. The requested change modified both the method of sample compositing to accommodate the saturated condition of the coal ash and the total number of coal ash samples to be collected from the FCAP.

Representatives from DOE conducted an audit of paperwork and field documentation related to the OU 2 RI several weeks after the conclusion of field activities. The auditors noted several problems concerning the completeness of logbook entries. For example, a summary of weather conditions was omitted from entries for 1 day, and the serial number of a field instrument was omitted from entries for 1 or 2 days. DOE auditors also raised a concern about the construction materials of the packer used to facilitate sampling in several of the monitoring wells. The packer construction was primarily Viton, a material nearly as inert as Teflon but more elastic and therefore appropriate for use as an inflatable packer. The validated data from the monitoring well sampling do not suggest that the packer introduced false positive analytical results.

The CDM Federal and Energy Systems QA programs addressed issues raised during the DOE audit; follow-up included the preparation and implementation of a corrective action plan. The corrective action plan also assessed the impact of the issues on the investigation and concluded that there was no substantial impact. No resampling or qualification of the results of analysis was required; however, planning and implementation of subsequent field investigations will address these issues.

### 3.2.7.3 Summary of findings

Overall, a very small percentage of the analytical results were rejected (<2%) during the data validation process based on analytical problems or missed holding times. A majority of the analytical problems were due to matrix-related interferences, especially for soil/sediment/ash samples. Very few inorganic results were rejected and a majority of the rejected results were for selenium, which is difficult, at best, to quantify in solid matrices. The occurrence of organics was negligible as expected for a combusted material (i.e., coal ash).

A vast majority of qualified results were nondetects. Rejected data were also largely associated with matrix interference, except for the three field blanks for pesticides and PCBs for which holding times were exceeded. None of the radiochemical results were rejected and many results were at or below the MDAs. Many valid results were obtained for soil and ash samples. Ash sample results were typical of other coal ash constituents.

The number and quality of sample results appear to be adequate to address the site conceptual model with a reasonable degree of uncertainty. The overall data set is representative of site conditions and provides the analytical results required for preparation of the risk assessments and FS.

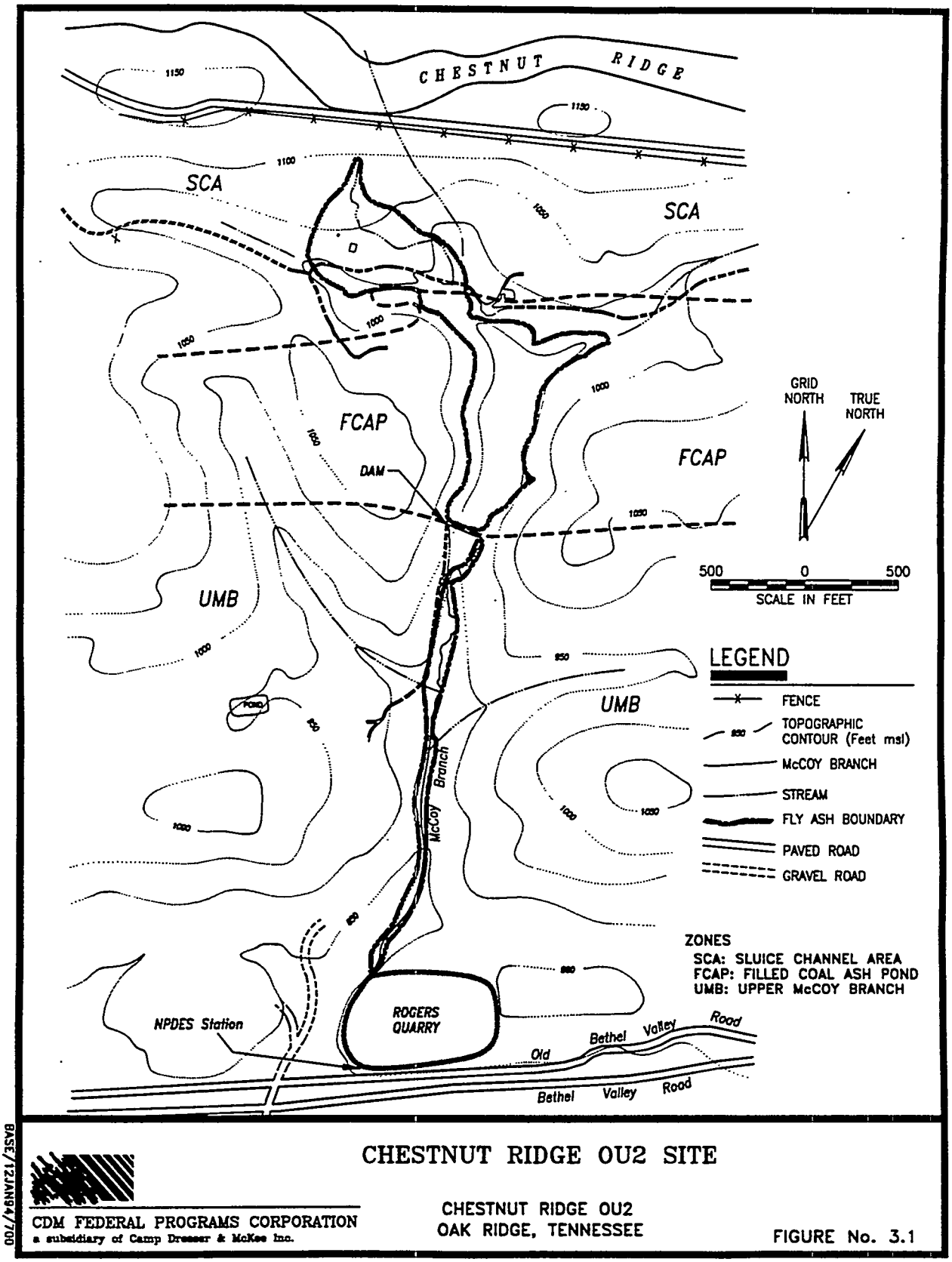



Fig. 3.1. Chestnut Ridge OU 2 Site.

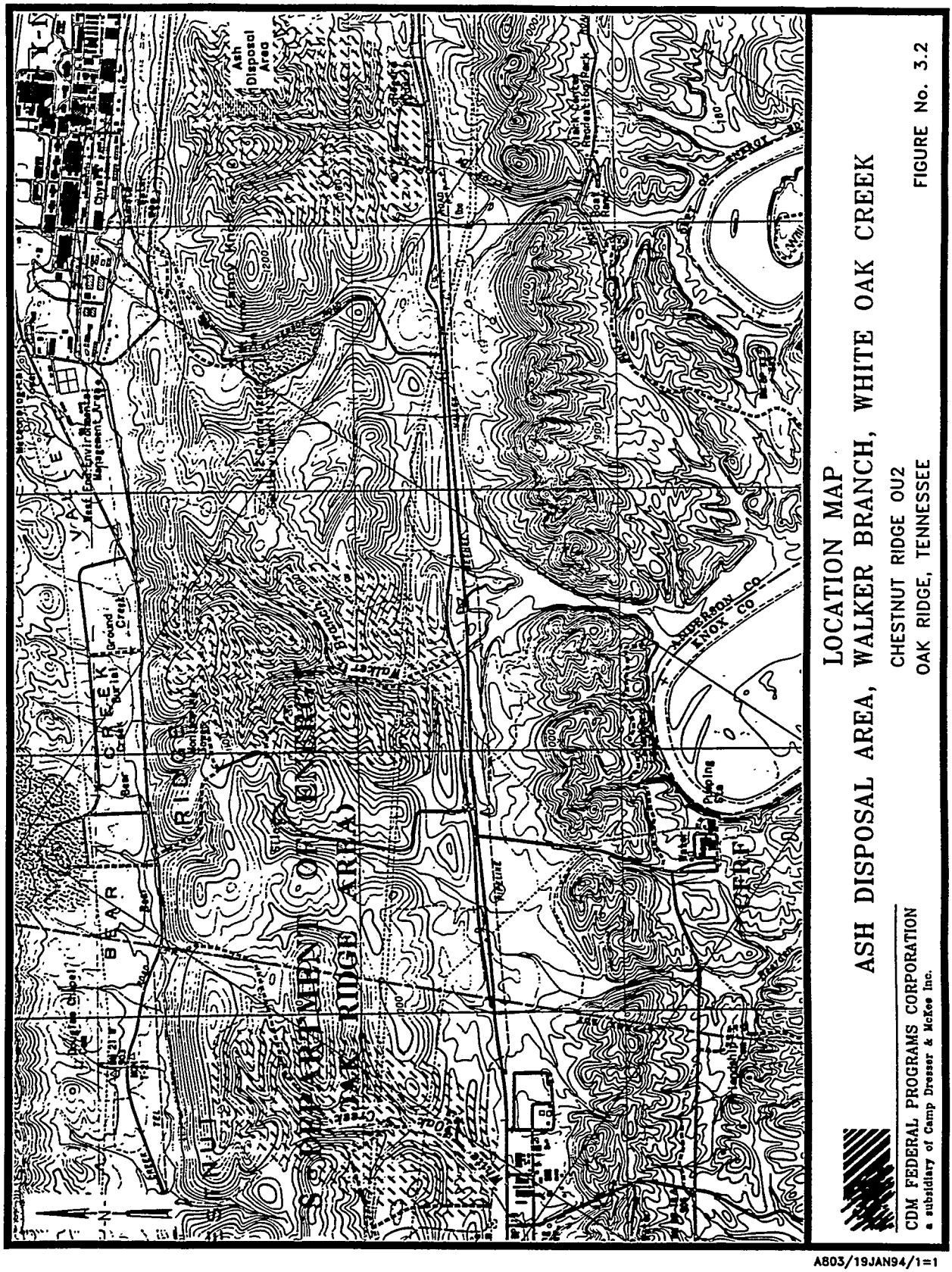



**Fig. 3.2. Location Map—Ash Disposal Area, Walker Branch, White Oak Creek.**

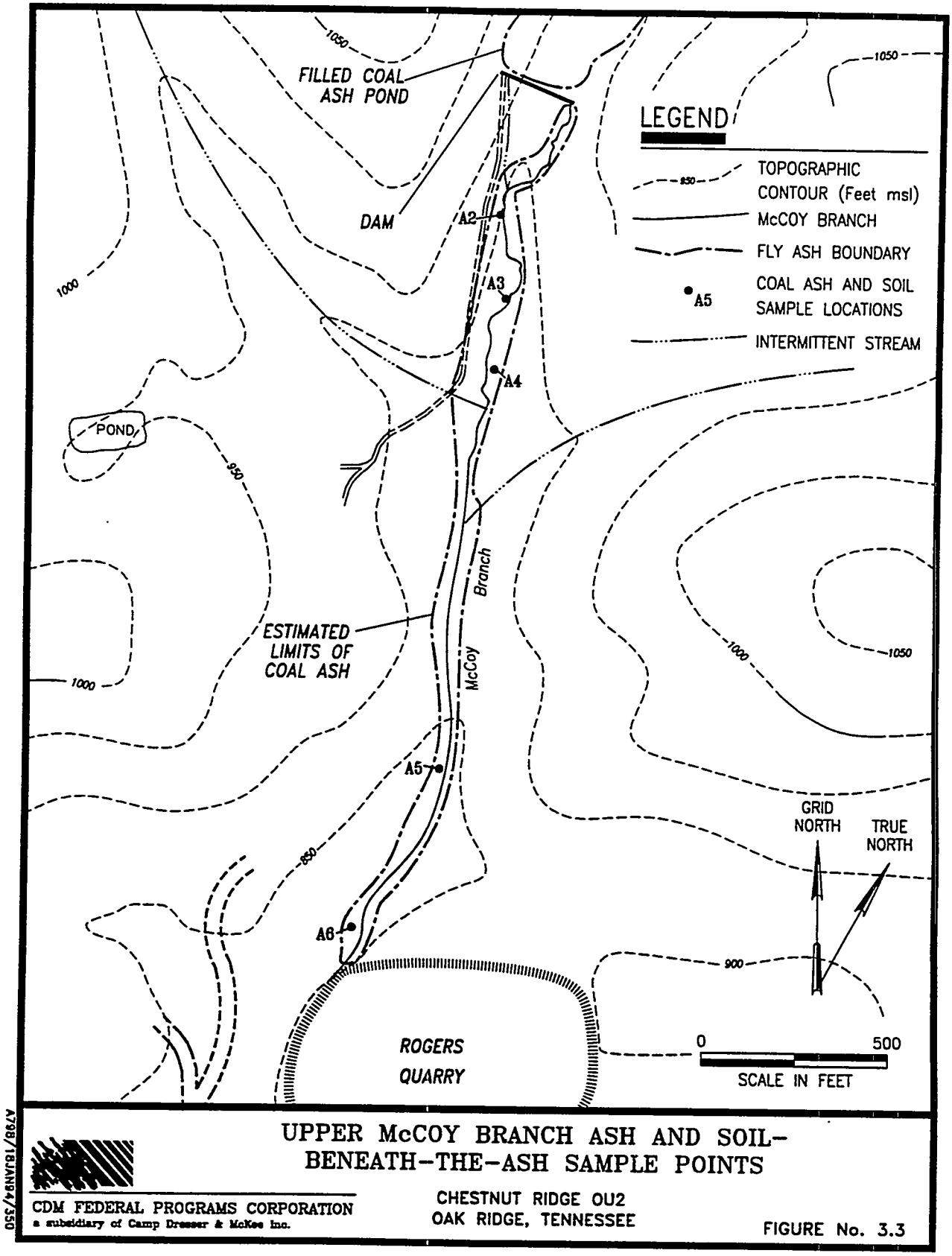



**Fig. 3.3. Upper McCoy Branch ash and soil beneath the ash sample points.**

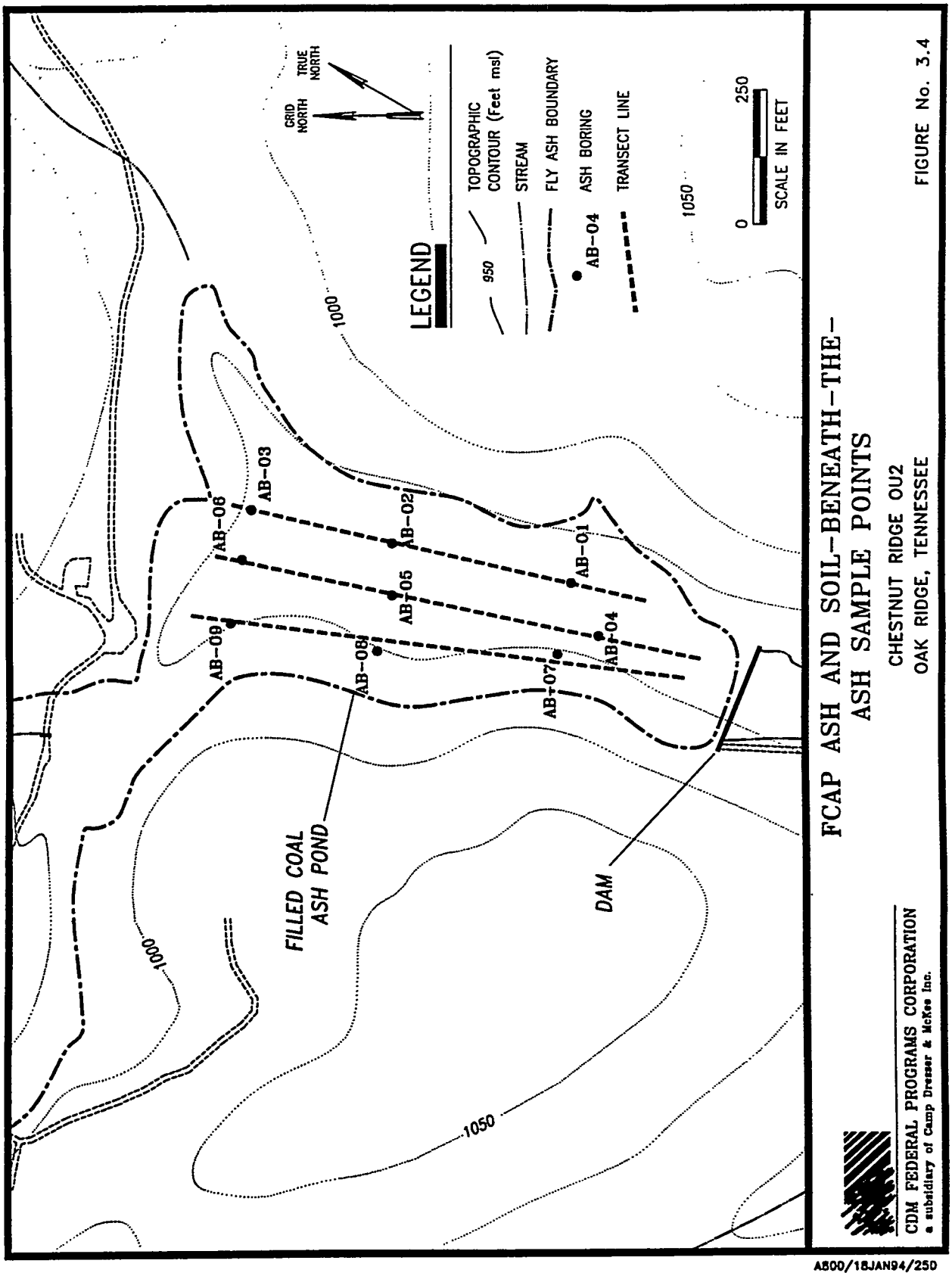



**Fig. 3.4. FCAP ash and soil beneath the ash sample points.**

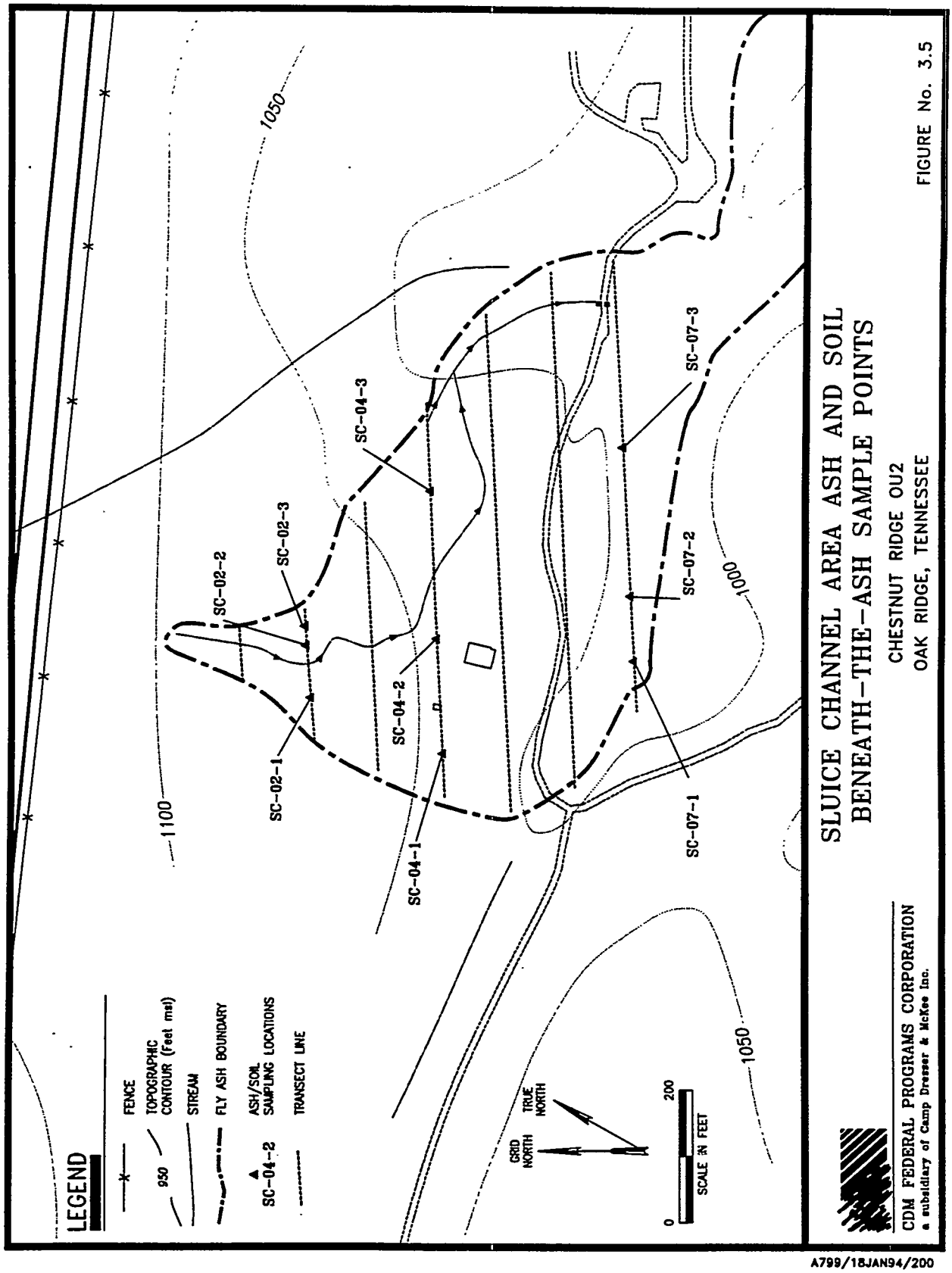



**Fig. 3.5. Sluice Channel Area ash and soil beneath the ash sample points.**

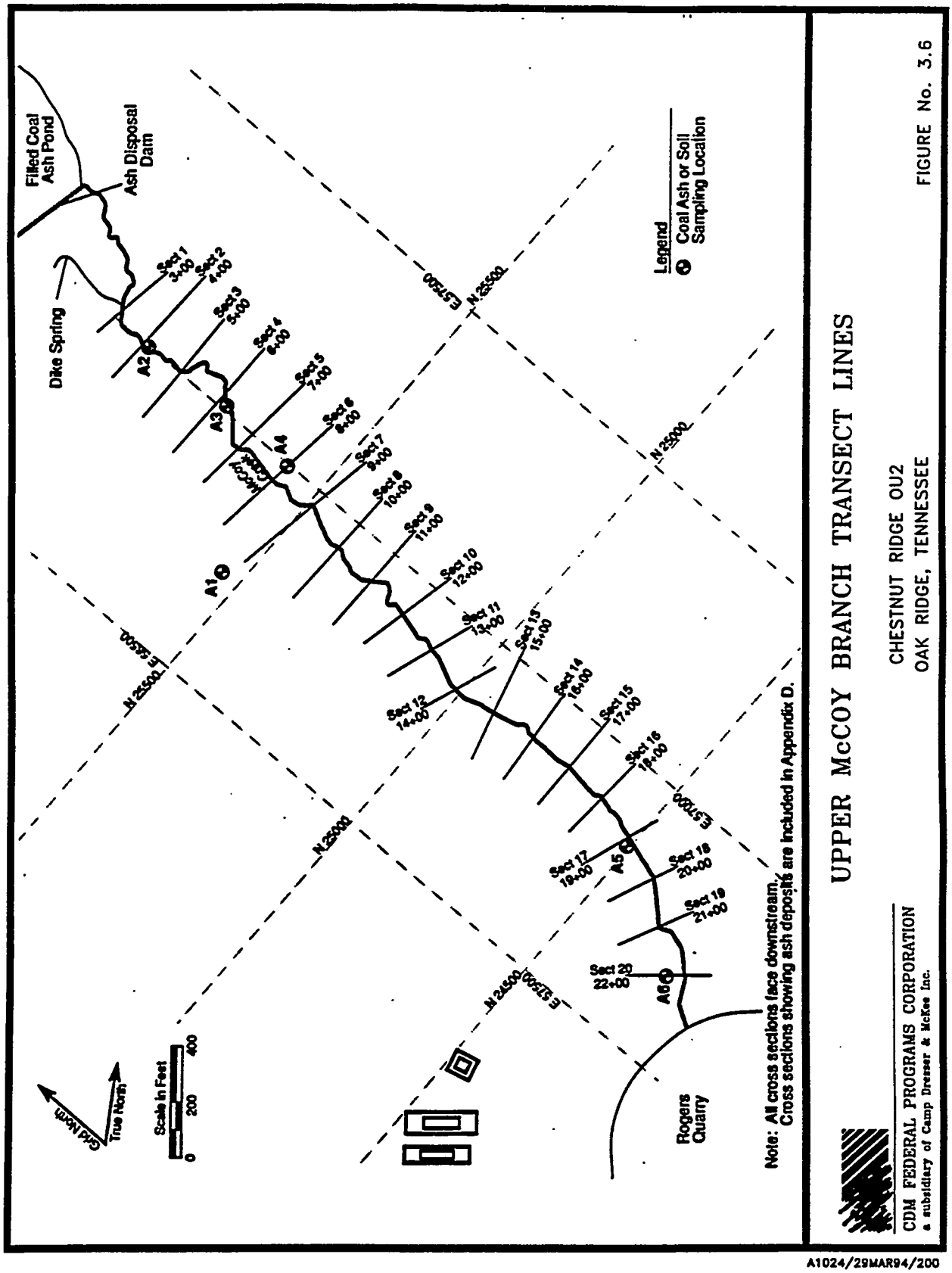



Fig. 3.6. Upper McCoy Branch transect lines.

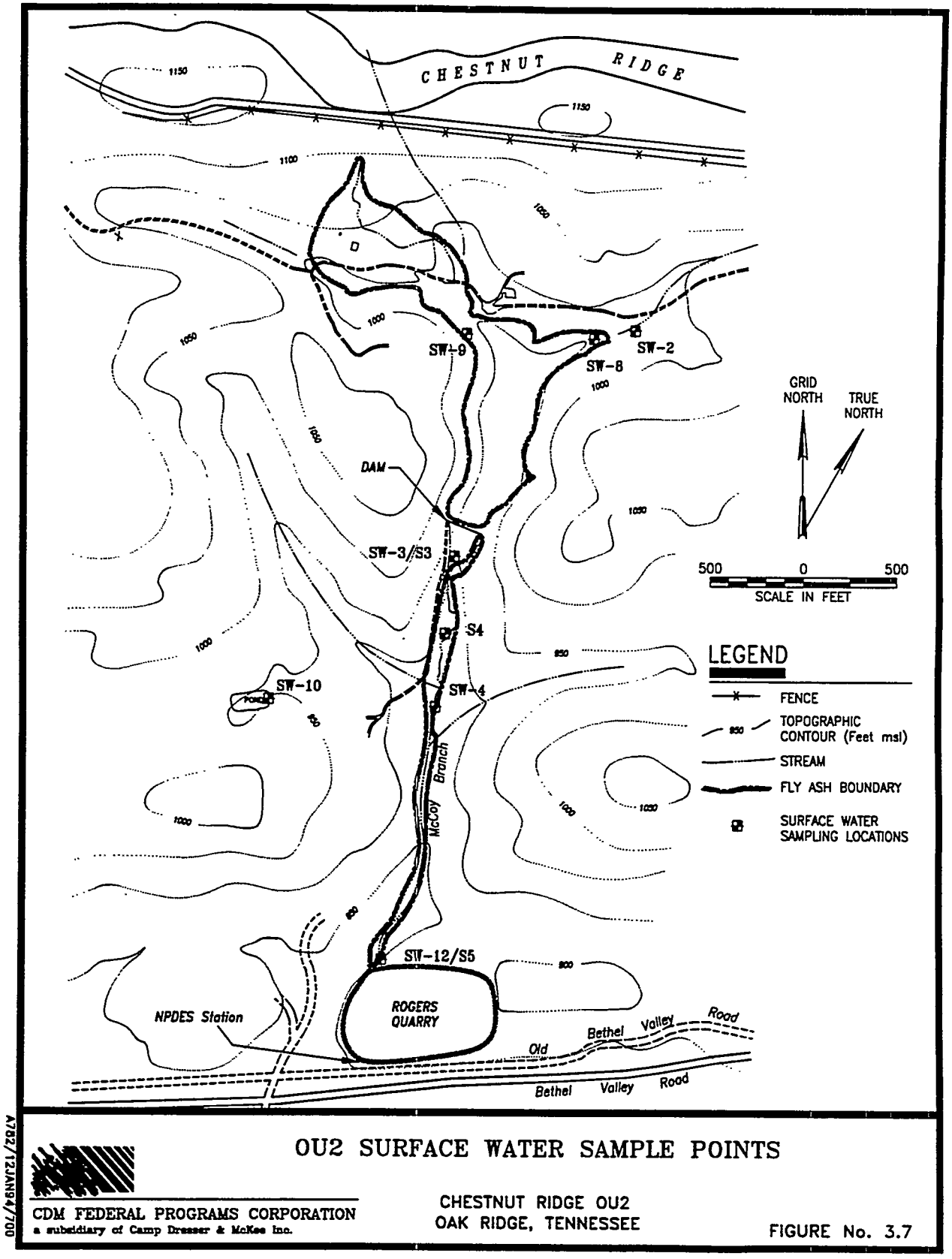



Fig. 3.7. OU 2 surface water sample points.

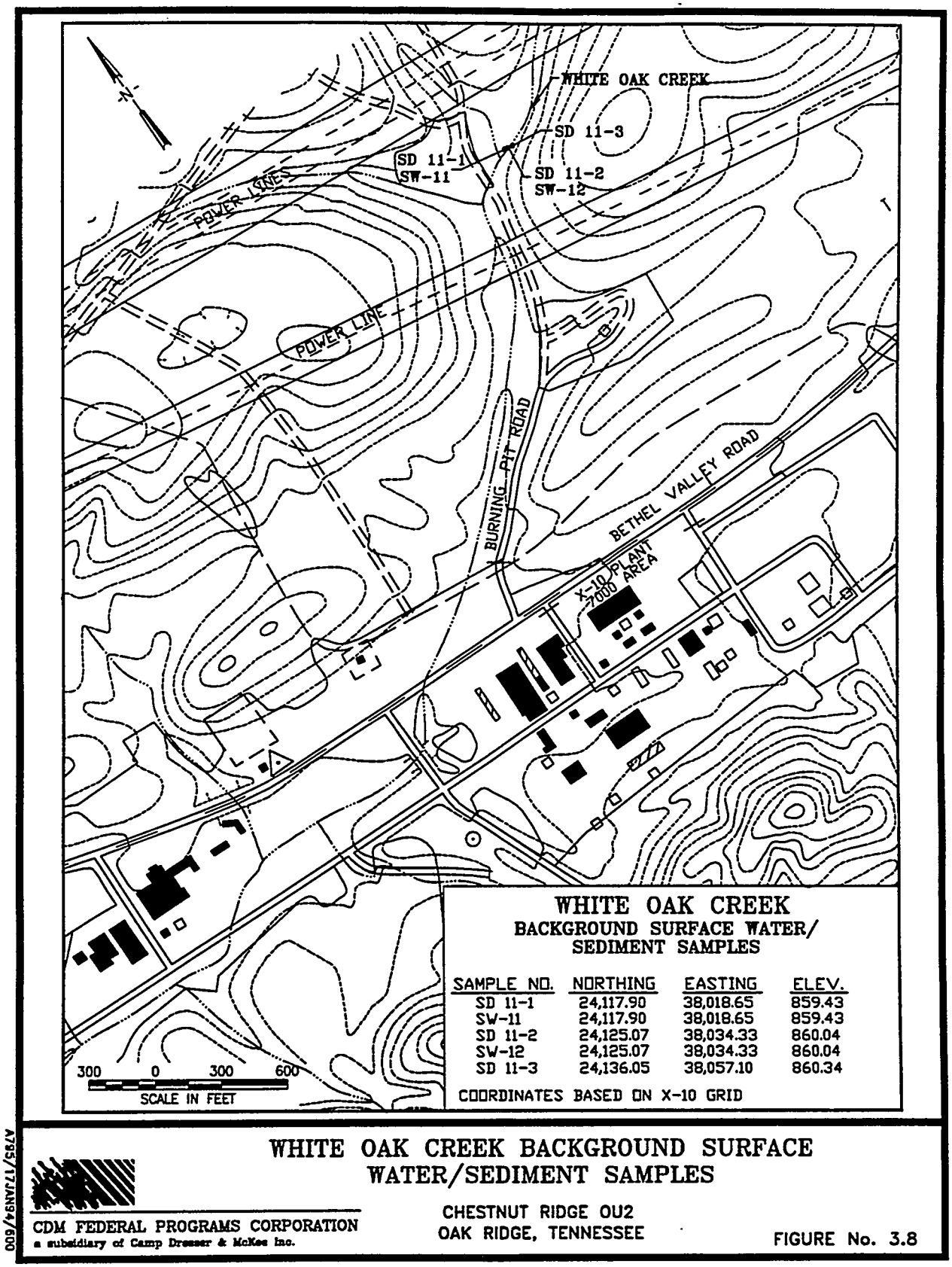



Fig. 3.8. White Oak Creek background surface water/sediment samples.

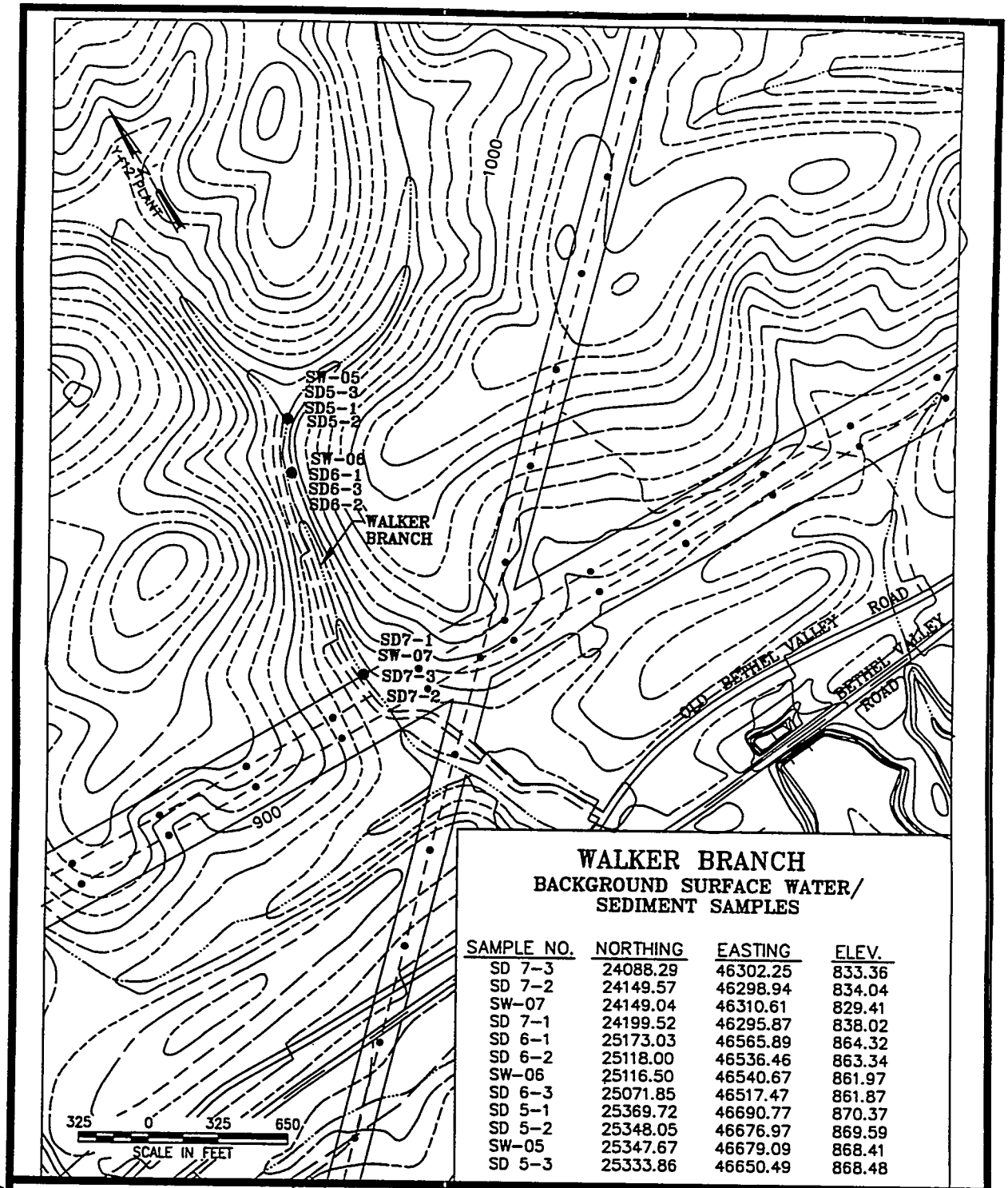



**Fig. 3.9. Walker Branch background surface water/sediment samples.**

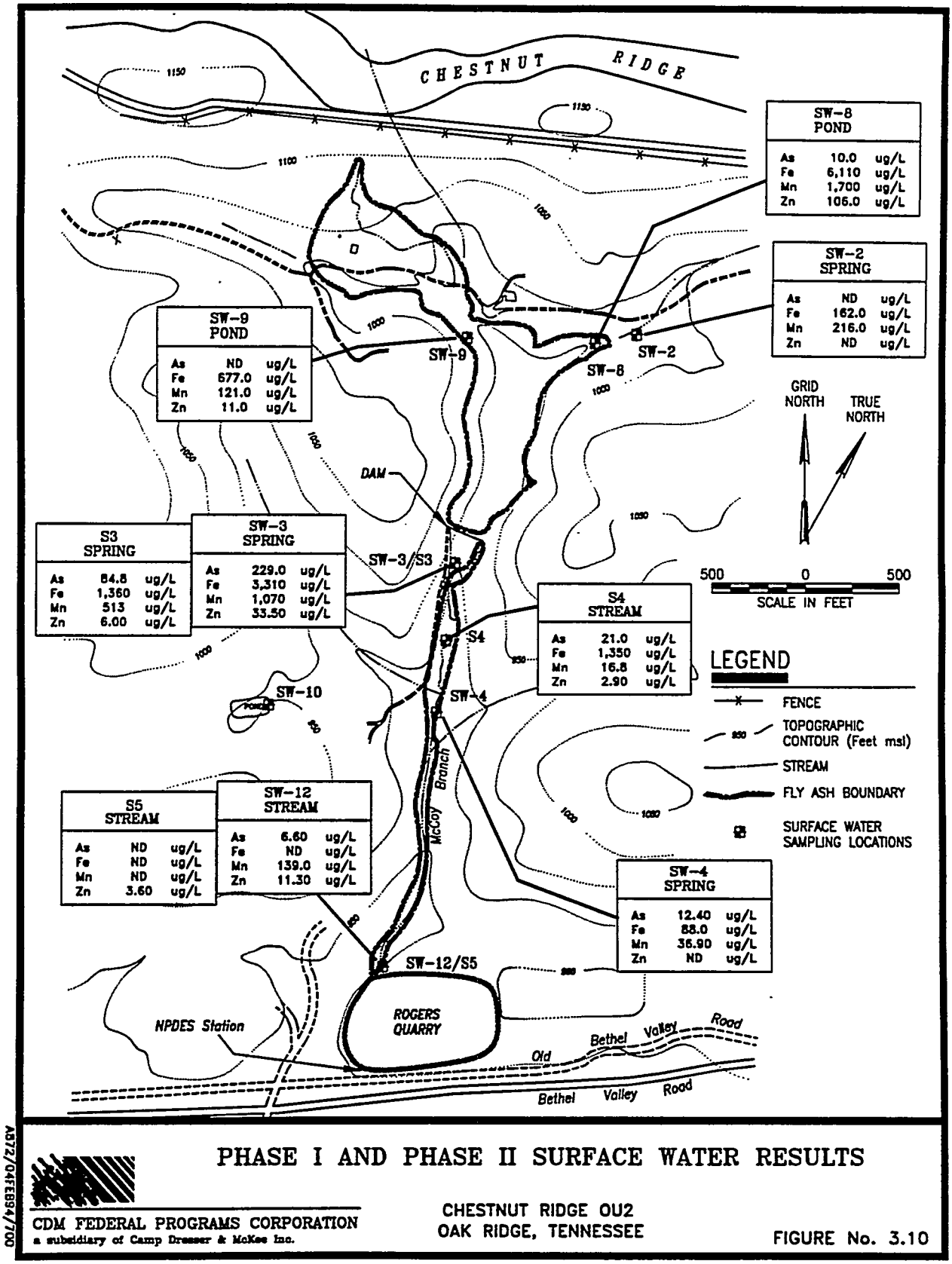



Fig. 3.10. Phase 1 and Phase 2 surface water results.

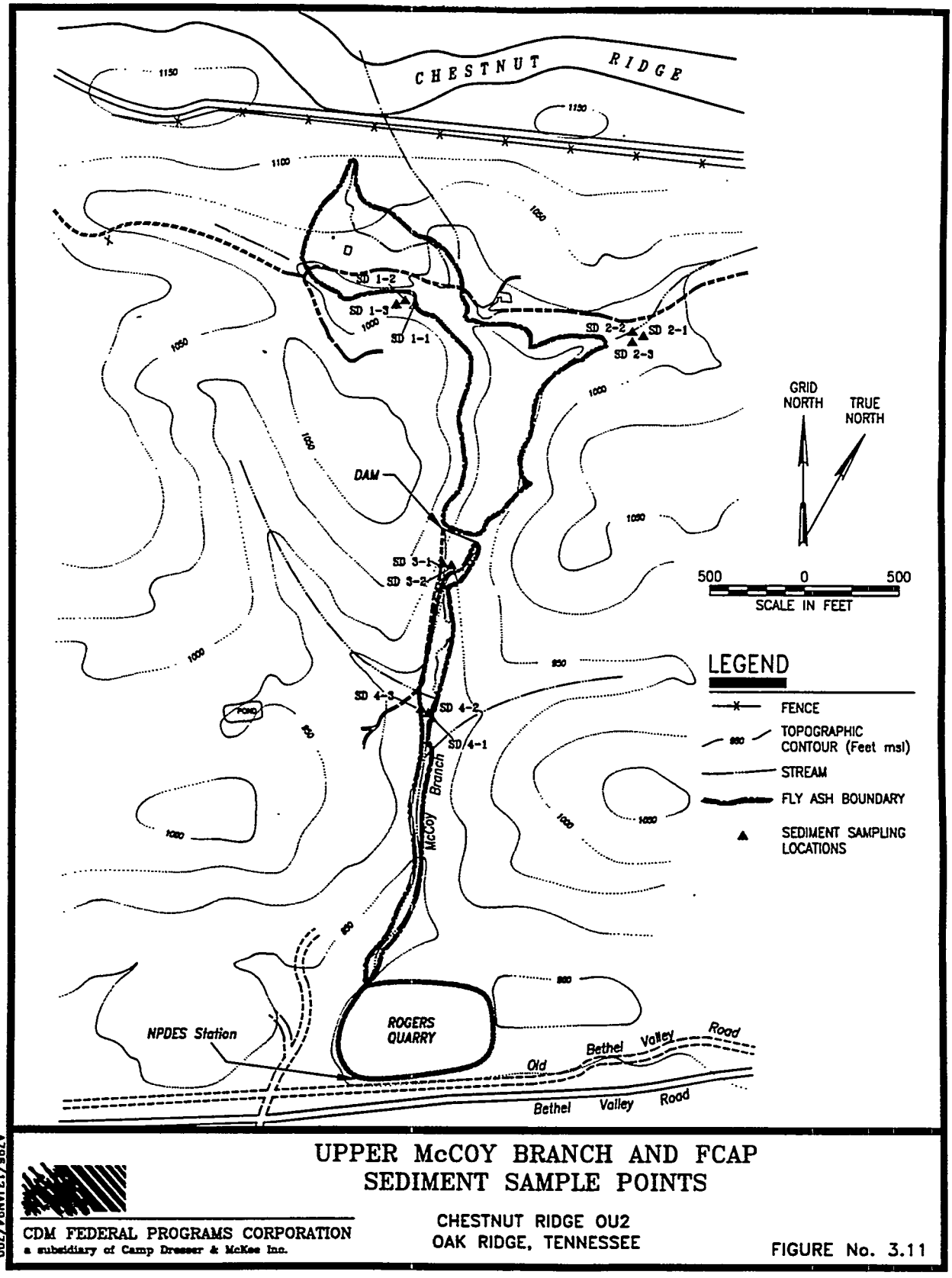



**Fig. 3.11. Upper McCoy Branch and FCAP sediment sample points.**

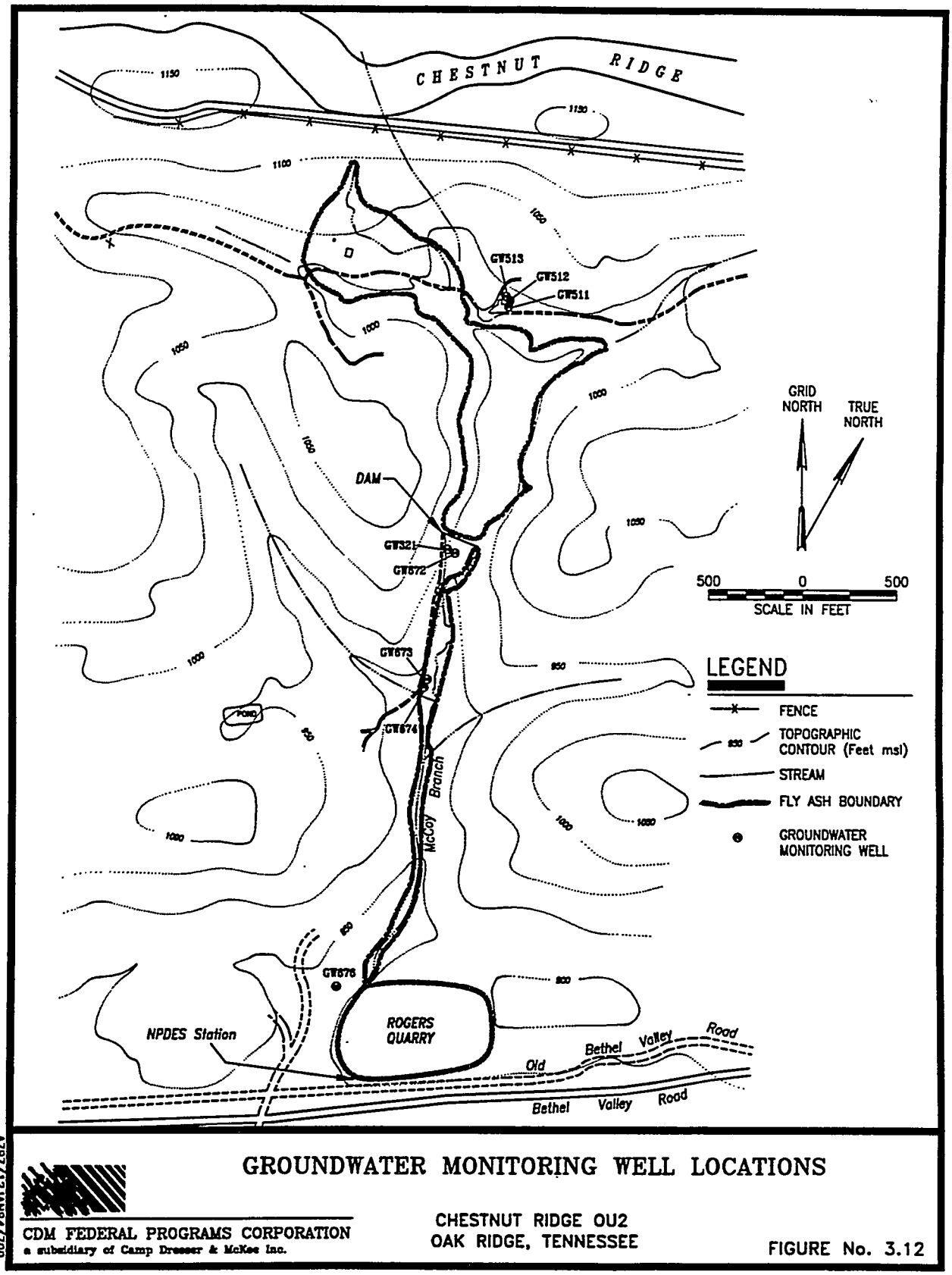



**Fig. 3.12. Groundwater monitoring well locations.**

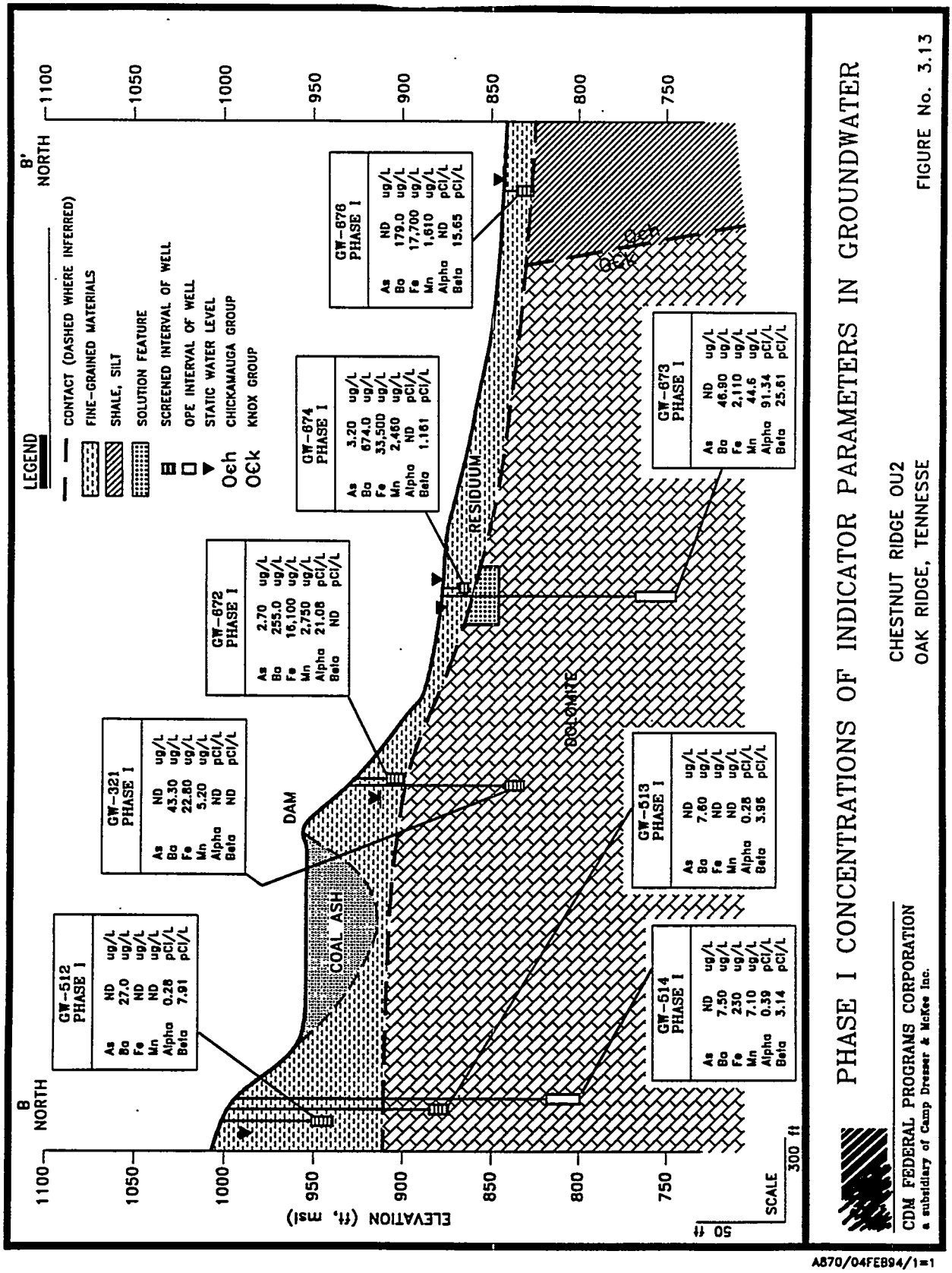



**Fig. 3.13. Phase 1 concentrations of indicator parameters in groundwater.**

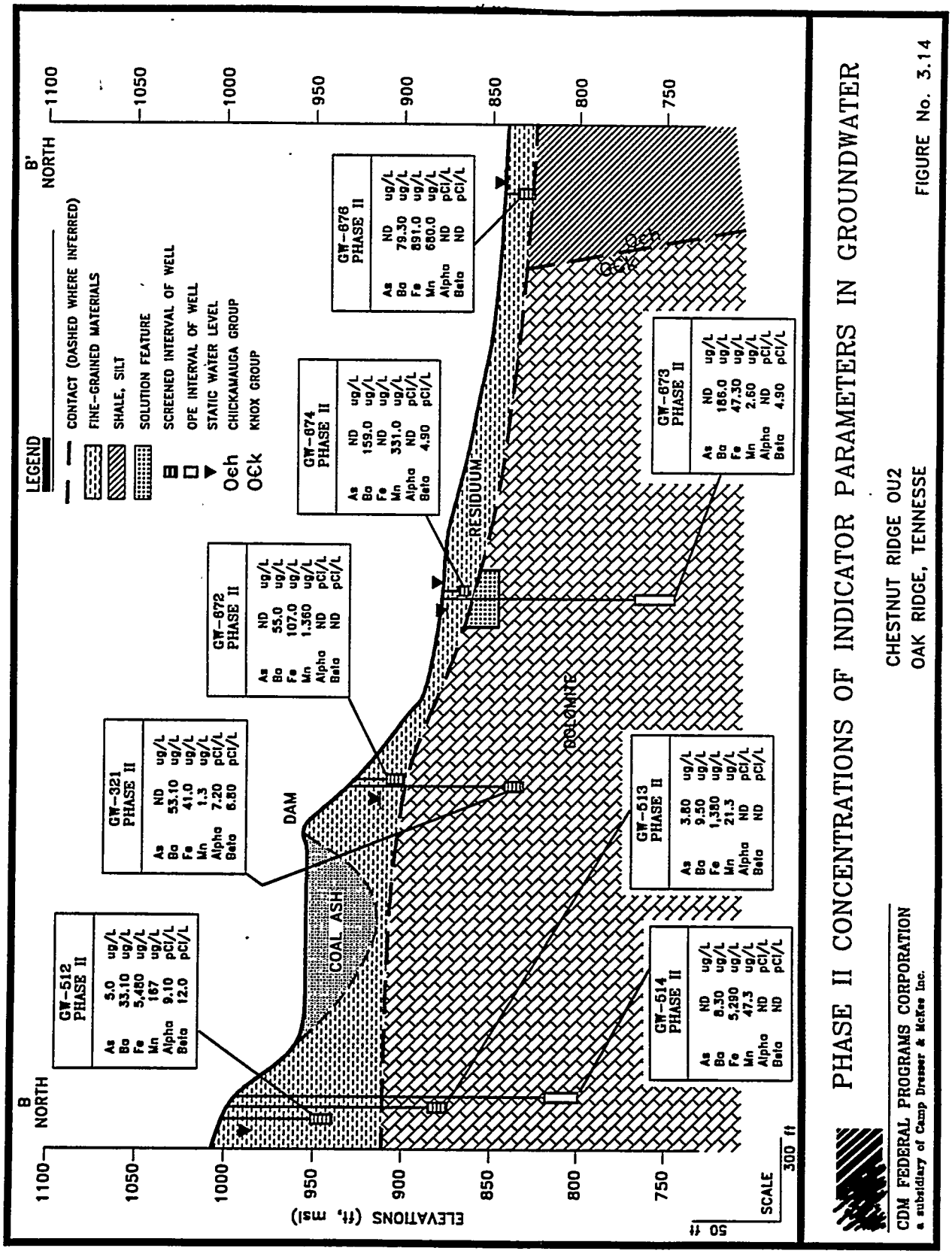
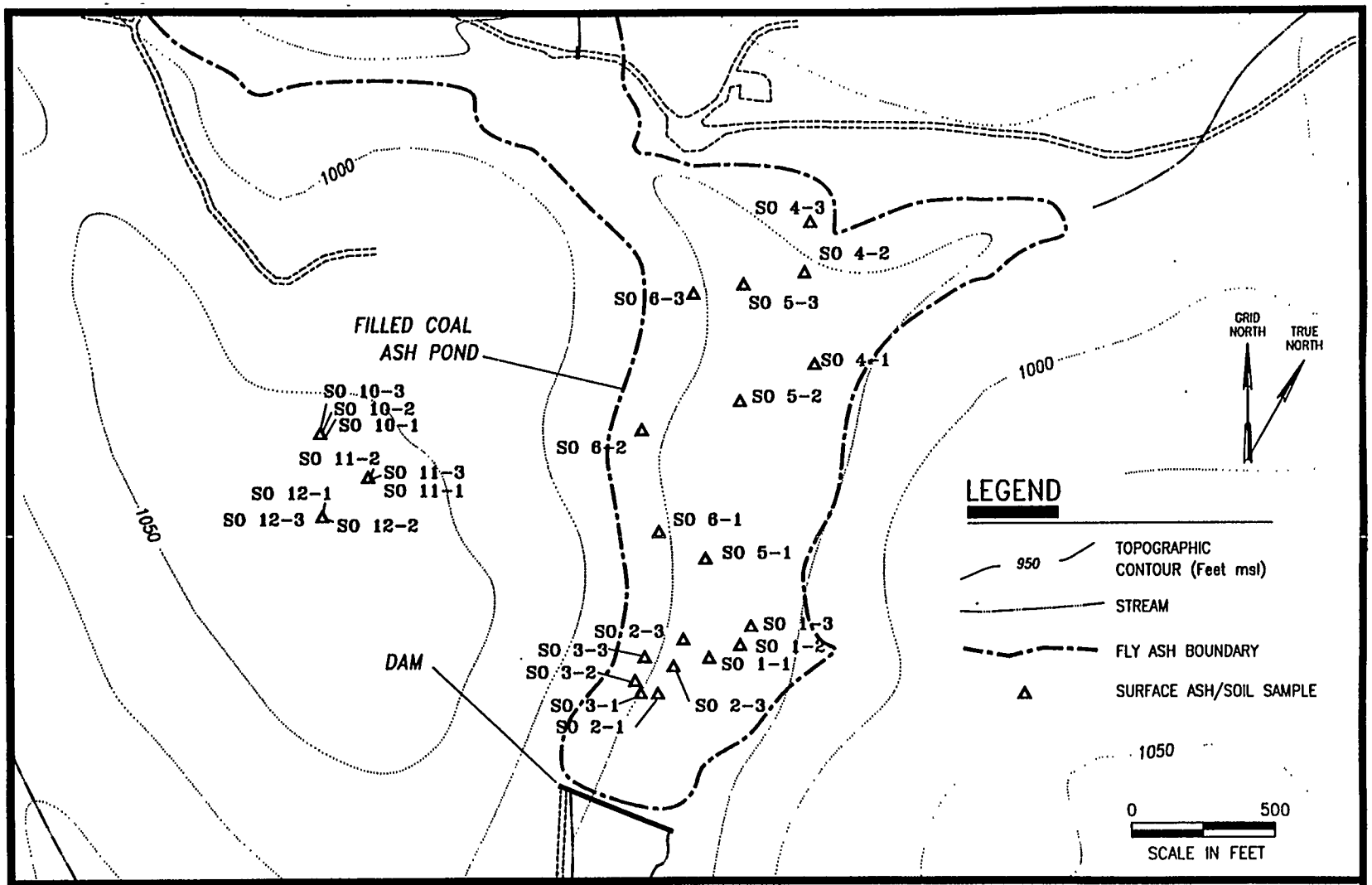



Fig. 3.14. Phase 2 concentrations of indicator parameters in groundwater.




FCAP ASH AND BACKGROUND SURFACE SOIL SAMPLE POINTS

CHESTNUT RIDGE OU2
OAK RIDGE, TENNESSEE

FIGURE No. 3.15

CDM FEDERAL PROGRAMS CORPORATION
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AS01/18JAN94/250

Fig. 3.15. FCAP ash and background surface soil sample points.

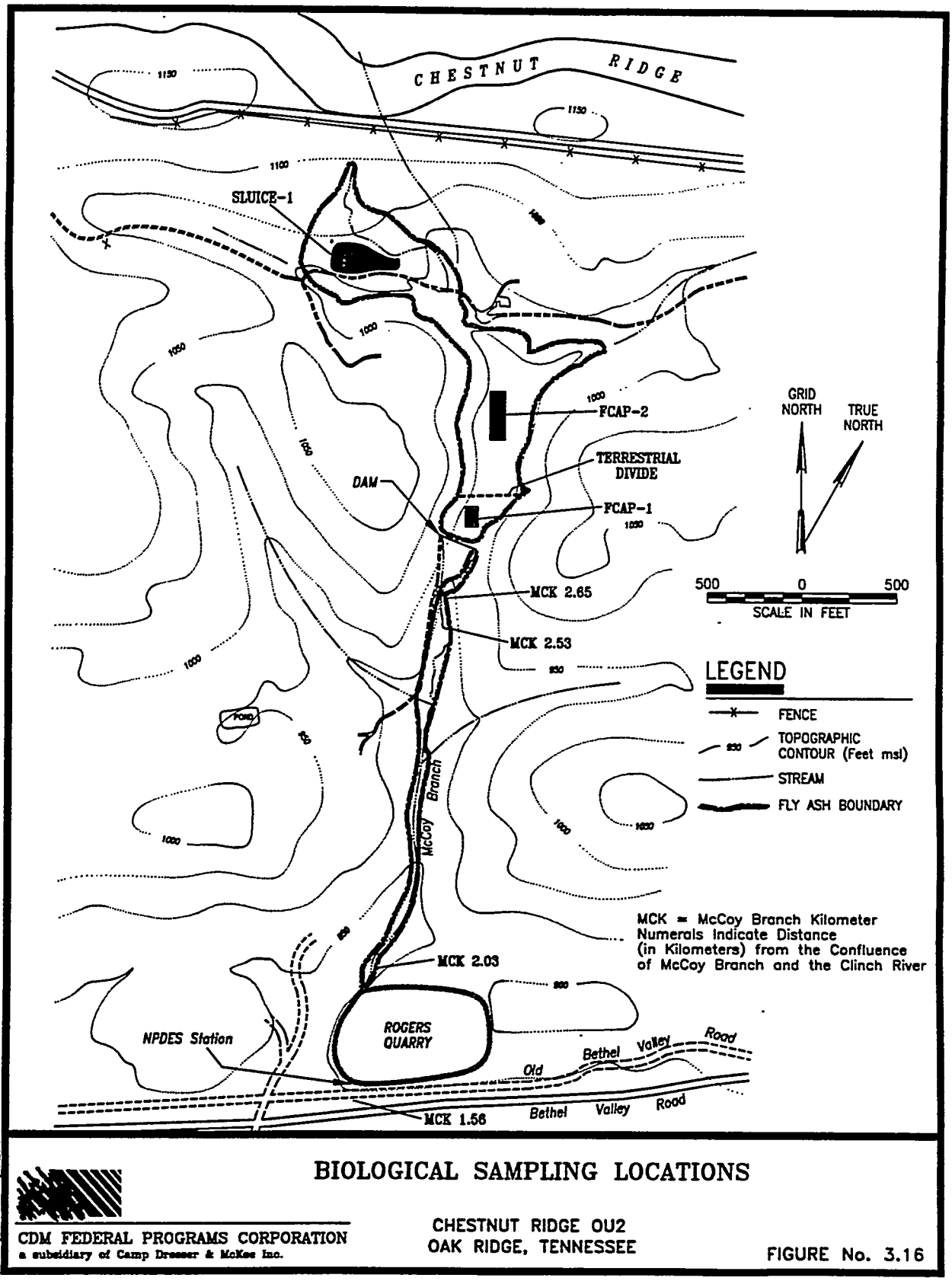



Fig. 3.16. Biological sampling locations.

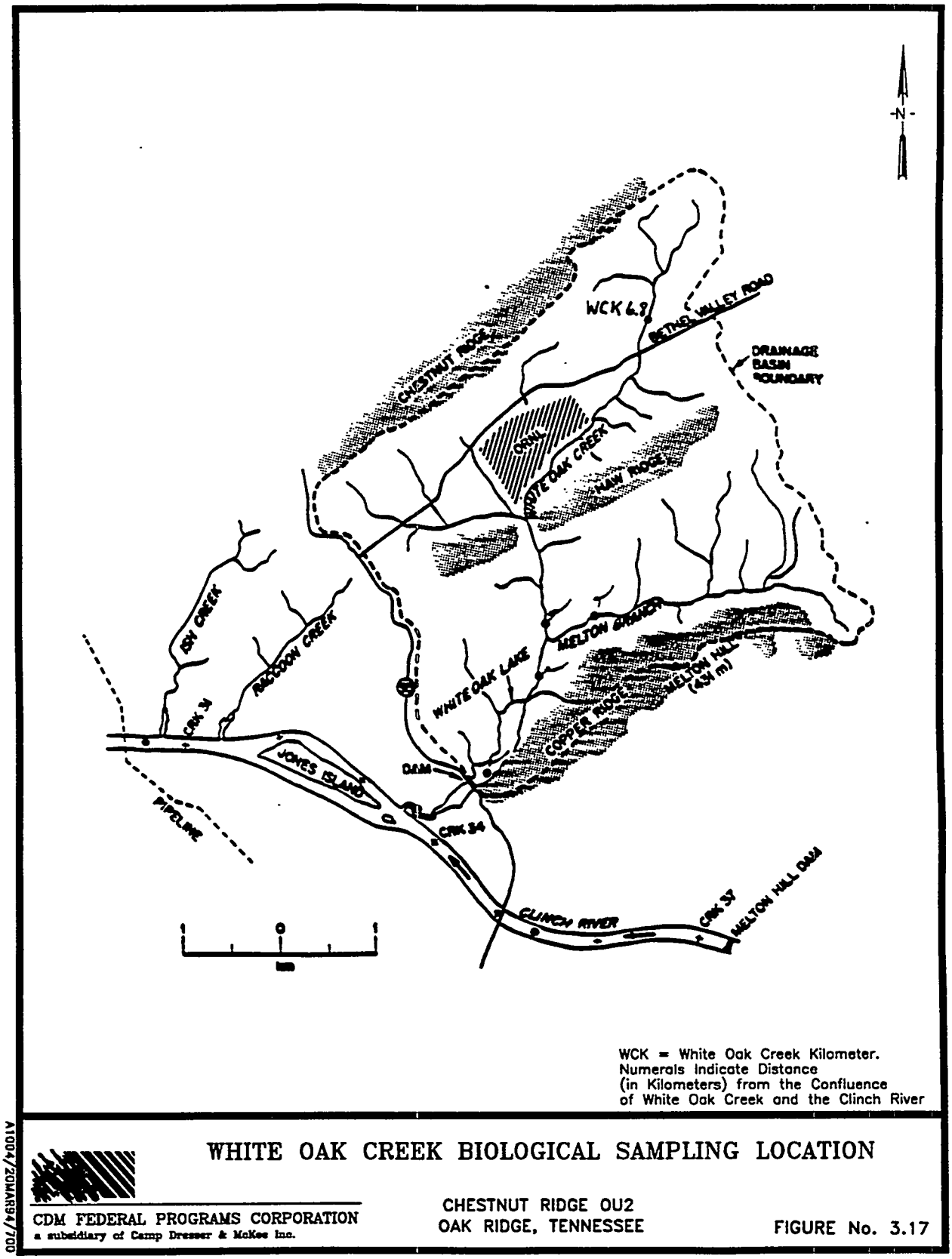
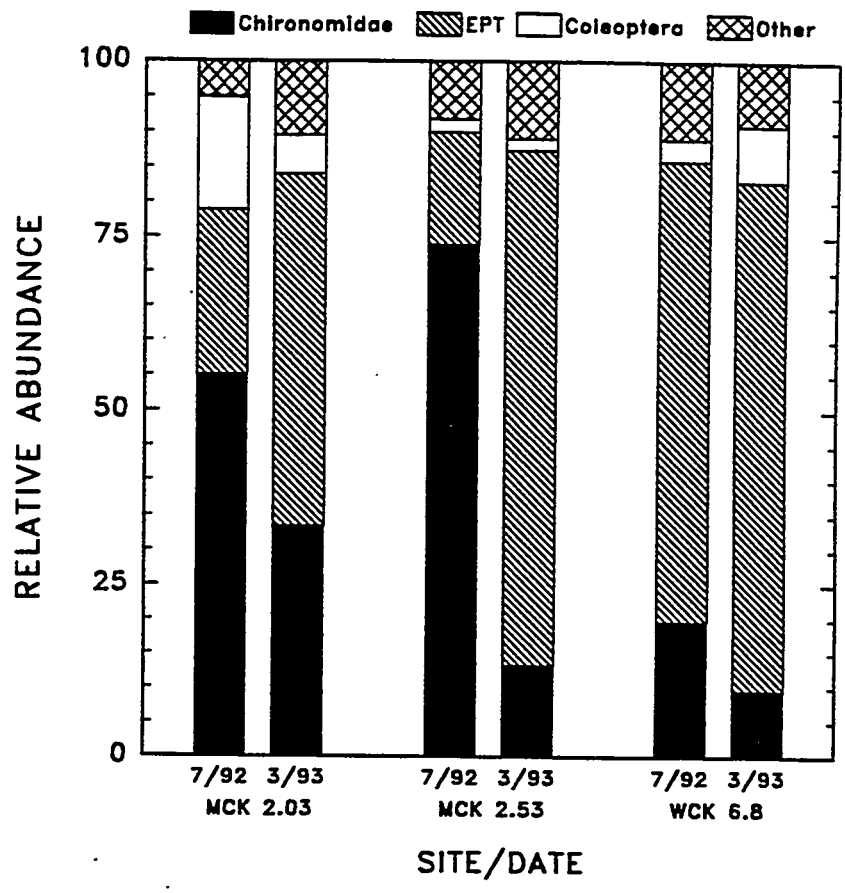



**Fig. 3.17. White Oak Creek biological sampling location.**




EPT = Ephemeroptera, Plecoptera, and Trichoptera.
Other = Total of all other taxa excluding Chironomidae, EPT, and Coleoptera. MCK = McCoy Branch Kilometer; WCK = White Oak Creek Kilometer.

RELATIVE ABUNDANCE OF MAJOR GROUPS OF BENTHIC MACROINVERTEBRATES IN McCOY BRANCH AND WHITE OAK CREEK

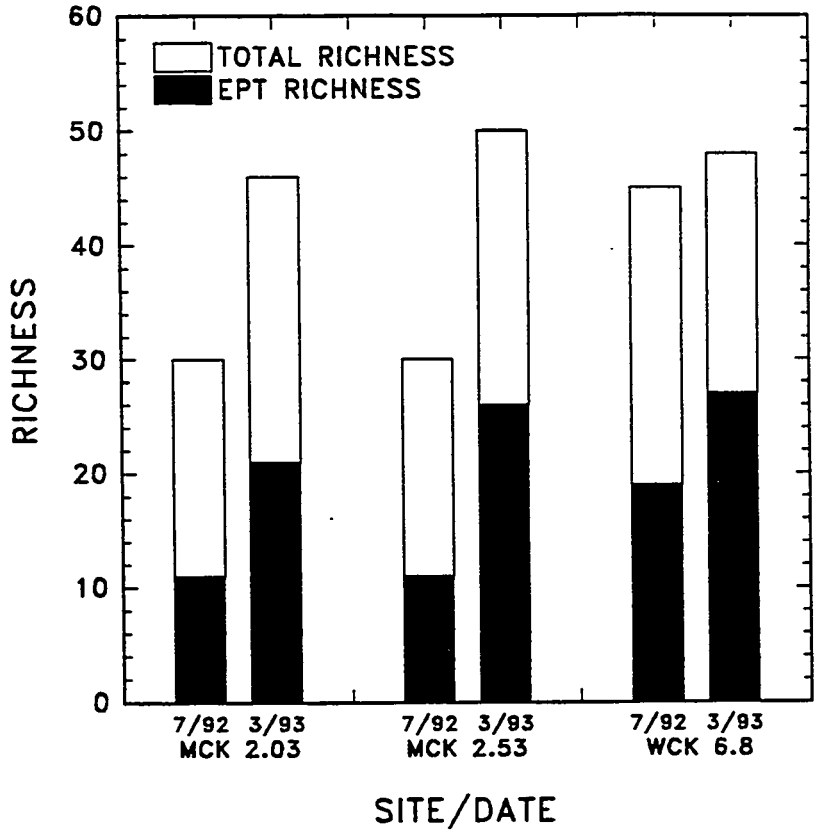
CDM FEDERAL PROGRAMS CORPORATION
a subsidiary of Camp Dresser & McKee Inc.

CHESTNUT RIDGE OU2
OAK RIDGE, TENNESSEE

FIGURE No. 3.18

A1005/20MAR94/700

Fig. 3.18. Relative abundance of major groups of benthic macroinvertebrates in McCoy Branch and White Oak Creek.




EPT = Ephemeroptera, Plecoptera, and Trichoptera.
Other = Total of all other taxa excluding Chironomidae, EPT, and Coleoptera. MCK = McCoy Branch Kilometer; WCK = White Oak Creek Kilometer.

COMBINED RICHNESS OF THE EPT TAXA, AND TOTAL RICHNESS OF BENTHIC MACROINVERTEBRATES IN McCOY BRANCH AND WHITE OAK CREEK

CDM FEDERAL PROGRAMS CORPORATION
a subsidiary of Camp Dresser & McKee Inc.

CHESTNUT RIDGE OU2
OAK RIDGE, TENNESSEE

FIGURE No. 3.19

A1006/20MAR94/700

**Fig. 3.19. Combined richness of the EPT TAXA and total richness of benthic macroinvertebrates in McCoy Branch and White Oak Creek.**

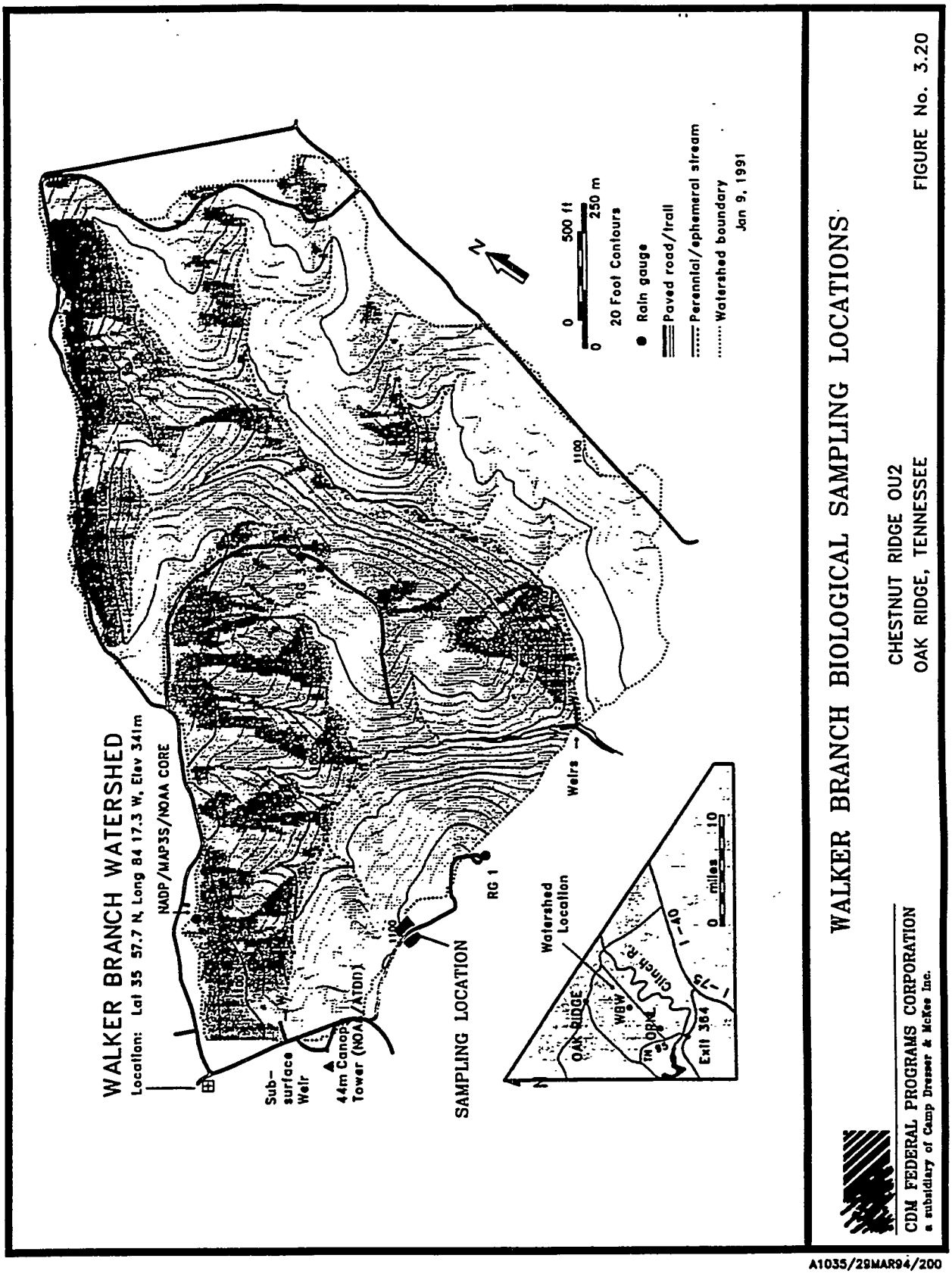



Fig. 3.20. Walker Branch biological sampling locations.

Table 3.1. Phase 1 analytes for ash at Chestnut Ridge OU 2

| Inorganics | |
|---|---|
| Aluminum | Magnesium |
| Antimony | Manganese |
| Arsenic | Mercury |
| Barium | Nickel |
| Beryllium | Potassium |
| Cadmium | Selenium |
| Calcium | Silver |
| Chromium | Sodium |
| Cobalt | Thallium |
| Copper | Vanadium |
| Iron | Zinc |
| Lead | Uranium (fluorometric) |
| **Physical properties** | |
| Oil and grease (%) | |
| **Radiological parameters** | |
| Gross alpha | Thorium-234 |
| Gross beta | Uranium-235 |
| Cesium-137 | |
| Thorium-228 | |

Table 3.2. Phase 2 analytes for ash at Chestnut Ridge OU 2

| Volatile organic compounds (CLP) | |
|---|---|
| 1,1,1-Trichloroethane | Carbon disulfide |
| 1,1,2,2-Tetrachloroethane | Carbon tetrachloride |
| 1,1,2-Trichloroethane | Chlorobenzene |
| 1,1-Dichloroethane | Chloroethane |
| 1,2-Dichloroethane | Chloroform |
| 1,1-Dichloroethene | Chloromethane |
| 1,2-Dichloroethene (total) | Cis-1,3-dichloropropene |
| 1,2-Dichloroethane | Dibromochloromethane |
| 1,2-Dichloropropane | Ethylbenzene |
| 2-Butanone | Methylene chloride |
| 2-Hexanone | Styrene |
| 4-Methyl-2-pentanone | Tetrachloroethene |
| Acetone | Toluene |
| Benzene | Trans-1,3-dichloropropene |
| Bromodichloromethane | Trichloroethene |
| Bromoform | Vinyl chloride |
| Bromomethane | Xylene (total) |
| **Base neutral acid compounds (CLP)** | |
| 1,2,4-Trichlorobenzene | 4-Nitroaniline |
| 1,2-Dichlorobenzene | 4-Nitrophenol |
| 1,3-Dichlorobenzene | Acenaphthene |
| 1,4-Dichlorobenzene | Acenaphthylene |
| 2,2-Oxybis(1-chloropropane) | Anthracene |
| 2,4,5-Trichlorophenol | Benzo(a)anthracene |
| 2,4,6-Trichlorophenol | Benzo(a)pyrene |
| 2,4-Dichlorophenol | Benzo(b)fluoranthene |
| 2,4-Dimethylphenol | Benzo(g,h,i)perylene |
| 2,4-Dinitrophenol | Benzo(k)fluoranthene |
| 2,4-Dinitrotoluene | Bis(2-chloroethoxy)methane |
| 2,6-Dinitrotoluene | Bis(2-chloroethyl)ether |
| 2-Chloronaphthalene | Bis(2-ethylhexyl)phthalate |
| 2-Chlorophenol | Butyl benzyl phthalate |
| 2-Methylnaphthalene | Carbazole |
| 2-Methylphenol | Chrysene |
| 2-Nitroaniline | Di-n-butyl phthalate |
| 2-Nitrophenol | Di-n-octyl phthalate |
| 3-Nitroaniline | Dibenzo(a,h)anthracene |
| 3,3-Dichlorobenzidine | Dibenzofuran |
| 4,6-Dinitro-2-methylphenol | Diethyl phthalate |
| 4-Bromophenyl phenyl ether | Dimethyl phthalate |
| 4-Chloro-3-methylphenol | Fluoranthene |
| 4-Chloraniline | Fluorene |
| 4-Chlorophenyl phenyl ether | Hexachlorobenzene |
| 4-Methylphenol | Hexachlorobutadiene |

Table 3.2 (continued)

| Base neutral acid compounds (CLP—continued) | |
|---|---|
| 4-Methylphenol | Hexachlorobutadiene |
| Hexachlorocyclopentadiene | Naphthalene |
| Hexachloroethane | Nitrobenzene |
| Indeno(1,2,3-cd)pyrene | Pentachlorophenol |
| Isophorone | Phenanthrene |
| N-nitrosi-di-n-propylamine | Phenol |
| N-nitrosodiphenylamine | Pyrene |
| **Inorganics (TAL—total)** | |
| Aluminum | Magnesium |
| Antimony | Manganese |
| Arsenic | Mercury |
| Barium | Nickel |
| Beryllium | Potassium |
| Cadmium | Selenium |
| Calcium | Silver |
| Chromium | Sodium |
| Cobalt | Thallium |
| Copper | Vanadium |
| Iron | Zinc |
| Lead | |
| **Physical properties** | |
| pH | |
| **Radiological parameters** | |
| Gross alpha | Isotopic thorium |
| Gross beta | Isotopic uranium |
| **Pesticides and PCBs** | |
| Alpha-BHC | 4,4'-DDT |
| Beta-BHC | Methoxychlor |
| Delta-BHC | Endrin ketone |
| Gamma-BHC (Lindane) | Endrin aldehyde |
| Heptachlor | Alpha-Chlordane |
| Aldrin | Gamma-Chlordane |
| Heptachlor epoxide | Toxaphene |
| Endosulfan I | Aroclor 1016 |
| Dieldrin | Aroclor 1221 |
| 4,4'-DDE | Aroclor 1232 |
| Endrin | Aroclor 1242 |
| Endosulfan II | Aroclor 1248 |
| 4,4'-DDD | Aroclor 1254 |
| Endosulfan sulfate | Aroclor 1260 |

Table 3.3. Phase 2 pH data at Chestnut Ridge OU 2

| Location | Sample identification | pH |
|---|---|---|
| **Ash** | | |
| FCAP Borings | AB-001-01 | 7.2 |
| | AB-002-01 | 7.6 |
| | AB-003-01 | 7.9 |
| | AB-004-11 | 7.7 |
| | AB-004-11D | 7.3 |
| | AB-005-01 | 8.4 |
| | AB-006-01 | 7.7 |
| | AB-007-01 | 8.0 |
| | AB-008-01 | 6.1 |
| | AB-009-11 | 7.4 |
| | AB-009-11D | 7.3 |
| FCAP Ground Surface | SO-001-11 | 7.0 |
| | SO-001-11D | 6.2 |
| | SO-002-01 | 6.3 |
| | SO-003-01 | 6.1 |
| | SO-004-01 | 6.4 |
| | SO-005-01 | 6.5 |
| | SO-006-01 | 5.0 |
| Sluice Channel Area | SC-002-11 | 6.5 |
| | SC-002-11D | 6.3 |
| | SC-004-01 | 6.1 |
| | SC-007-01 | 6.9 |
| **Soil** | | |
| FCAP | AT-001-01 | 6.5 |
| | AT-002-01 | 6.8 |
| | AT-003-01 | 6.9 |
| Sluice Channel Area | SC-002-12 | 6.3 |
| | SC-002-12D | 6.1 |
| | SC-004-02 | 5.8 |
| | SC-007-12 | 6.9 |
| Near FCAP (background) | SO-010-11 | 6.5 |
| | SO-010-11D | 6.4 |
| | SO-011-01 | 6.4 |
| | SO-012-01 | 7.0 |
| **Sediment** | | |
| FCAP | SD-001-11 | 6.3 |
| | SD-001-11D | 6.3 |
| | SD-002-01 | 6.6 |
| UMB | SD-003-01 | 6.3 |
| | SD-003-02 | 7.4 |
| | SD-004-11 | 7.0 |
| | SD-004-11D | 6.8 |
| Walker Branch (background) | SD-005-01 | 7.4 |
| | SD-006-01 | 7.1 |
| | SD-007-01 | 7.3 |
| White Oak Creek (background) | SD-011-01 | 6.7 |

**Table 3.4. Phase 1 ash results at Chestnut Ridge OU 2**

| Analysis | Analyte | Units | MCB-A2-A | MCB-A3-A | MCB-A3-A7-A | MCB-A4-A | MCB-A5-A | MCB-A6-A |
|---|---|---|---|---|---|---|---|---|
| METAL | Aluminum | mg/kg | 18,600.00 | 20,800.00 | 14,900.00 | 20,200.00 | 12,300.00 | 15,000.00 |
| METAL | Arsenic | mg/kg | 54.50 N* J | 46.20 N* J | 59.20 BN* J | 54.20 N* J | 53.20 N* J | 15.60 BN* J |
| METAL | Barium | mg/kg | 684.00 | 548.00 | 425.00 | 355.00 | 250.00 | 252.00 |
| METAL | Beryllium | mg/kg | 3.20 | 2.90 | 2.40 | 2.30 | 1.10 B | 1.20 B |
| METAL | Cadmium | mg/kg | ND | 4.20 | 2.60 | 1.60 B | 4.20 | 1.40 B |
| METAL | Calcium | mg/kg | 3,460.00 | 5,130.00 | 3,830.00 | 3,210.00 | 3,300.00 | 6,950.00 |
| METAL | Chromium | mg/kg | 15.50 | 18.20 | 11.50 | 16.40 | 9.10 | 14.30 |
| METAL | Cobalt | mg/kg | 12.00 B | 21.90 | 11.90 B | 15.70 B | 12.50 B | 20.00 |
| METAL | Copper | mg/kg | 74.90 | 53.5 | 40.80 | 37.70 | 27.60 | 26.00 |
| METAL | Iron | mg/kg | 18,300.00 | 64,300.00 | 36,800.00 | 29,900.00 | 51,500.00 | 25,200.00 |
| METAL | Lead | mg/kg | 13.00 W* | 15.40 W* | 16.60 W* | 14.30 W* | 10.40 W* | 15.10 W* |
| METAL | Magnesium | mg/kg | 1,140.00 B | 1,690.00B | 1,210.00 B | 1,620.00 | 1,090.00 B | 1,610.00 B |
| METAL | Manganese | mg/kg | 81.30 | 672.00 | 194.00 | 575.00 | 577.00 | 1,200.00 |
| METAL | Mercury | mg/kg | 0.73 | 0.66 | 0.30 | 0.19 | 0.13 | 0.19 |
| METAL | Nickel | mg/kg | 29.10 | 40.70 | 27.80 | 32.30 | 26.40 | 21.00 |
| METAL | Phosphorus | mg/kg | 2,180.00 J | 1,060.00 J | 1,000.00 J | 767.00 J | 262.00 J | 375.00 J |
| METAL | Potassium | mg/kg | 3,570.00 | 3,670.00 | 2,930.00 | 3,900.00 | 2,100.00 | 2,800.00 |
| METAL | Selenium | mg/kg | 9.10 EN J | 5.30 N J | 1.40 BEN J | 1.70 BEN J | 2.70 BN J | 1.50 BWNJ |
| METAL | Sodium | mg/kg | 326.00 B | 384.00 B | 332.00 B | 355.00 B | 257.00 B | 290.00 B |
| METAL | Vanadium | mg/kg | 83.70 | 74.20 | 55.80 | 52.70 | 37.70 | 40.20 |
| METAL | Zinc | mg/kg | 39.30 | 75.70 | 34.20 | 66.50 | 46.20 | 72.30 |
| PHYS | Uranium, Total | mg/kg | 7.00 | 8.00 | 13.00 | 7.00 | 6.00 | 8.00 |
| RAD | Alpha activity | pCi/g | ND | 1.35 J | 1.21 J | ND | 2.390 | 5.750 |
| RAD | Beta activity | pCi/g | ND | ND | ND | ND | 2.010 J | 7.270 |
| RAD | Cesium-137 | pCi/g | 0.152 J | 0.139 J | 0.0505 J | 0.0265 | 0.0935 J | 0.0695 J |
| RAD | Thorium-228 | pCi/g | ND | ND | ND | ND | ND | 9.20 |

**Table 3.4 (continued)**

| Analysis | Analyte | Units | MCB-A2-A | MCB-A3-A | MCB-A3-A7-A | MCB-A4-A | MCB-A5-A | MCB-A6-A |
|---|---|---|---|---|---|---|---|---|
| RAD | Thorium-234 | pCi/g | ND | ND | ND | ND | ND | **6.30** |
| RAD | Uranium-235 | WT% | ND | **0.76** | ND | ND | ND | ND |

J,E = estimated value; W = furnace atomic absorption (AA) postdigestion spike not within control limits, while sample absorbance is less than spike; N = spiked sample recovery not within control limits absorption. * Duplicate analysis not within control limits; B = below the required detection limit, but at or above the instrument detection limit. **Bold** = highest value detected for that analyte.

## Table 3.5. Phase 2 ash results at Chestnut Ridge OU 2

| Analysis | Analyte | Units | Composite coal ash samples from FCAP ash borings | | | | | | | | | Composite coal ash samples from Sluice Channel Area | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | AB-001-01 | AB-002-01 | AB-003-01 | AB-004-01 | AB-005-01 | AB-006-01 | AB-007-01 | AB-008-01 | AB-009-01 | SC-002-01 | SC-004-01 | SC-007-01 |
| METAL | Aluminum | mg/kg | 15,600.00 | 11,600.00 | 13,800.00 | 15,700.00 | 17,700.00 | 17,200.00 | 17,200.00 | 18,500.00 | 17,900.00 | 14,100.00 | 10,200.00 | 8,120.00 |
| METAL | Arsenic | mg/kg | 46.50 J | 65.70 J | 59.30 J | 59.10 | 66.20 J | 113.00 J | 98.90 | 100.00 | 64.90 J | 153.00 J | 159.00 | 88.50 |
| METAL | Barium | mg/kg | 343.00 | 238.00 | 281.00 | 352.00 | 314.00 | 371.00 | 410.00 | 376.00 | 376.00 | 421.00 | ND | 334.00 |
| METAL | Beryllium | mg/kg | 2.00 | 1.10 | 1.80 | 2.60 | 2.20 | 3.10 | 2.60 | 2.60 | 1.80 | 1.60 | 1.10 | 1.60 |
| METAL | Calcium | mg/kg | 5,280.00 | 4,260.00 | 2,970.00 | 3,930.00 | 5,050.00 | 4,190.00 | 4,340.00 | 3,410.00 | 3,520.00 | 4,610.00 | 4,180.00 | 4,920.00 |
| METAL | Chromium | mg/kg | 15.70 | 8.90 | 14.20 | 18.40 | 18.40 | 17.30 | 18.70 | 21.20 | 19.60 | 17.10 J | 10.00 | 8.20 |
| METAL | Cobalt | mg/kg | 9.90 | 5.20 | 8.40 | 10.70 | 9.00 | 10.10 | 14.90 | 16.20 | 19.70 J | 28.60 J | 5.30 | 9.00 |
| METAL | Copper | mg/kg | 40.20 | 25.10 | 36.50 | 47.60 | 46.80 | 45.70 | 53.70 | 52.90 | 54.20 | 70.10 J | 31.10 | 19.30 |
| METAL | Iron | mg/kg | 20,100.00 | 25,100.00 | 12,300.00 | 12,900.00 | 25,600.00 | 19,500.00 | 27,600.00 | 24,300.00 | 38,200.00 | 18,000.00 | 16,300.00 J | 13,200.00 J |
| METAL | Lead | mg/kg | 12.60 J | 9.70 J | 10.50 J | 10.00 | 12.20 J | 12.50 J | 14.40 | 9.80 | 8.60 | 10.60 | 17.40 J | 17.80 J |
| METAL | Magnesium | mg/kg | 1,450.00 | 1,190.00 | 1,090.00 | 1,300.00 | 1,640.00 | 1,360.00 | 1,440.00 | 1,480.00 | 1,550.00 | 940.00 | 524.00 | 601.00 |
| METAL | Manganese | mg/kg | 124.00 | 143.00 | 98.20 | 70.50 | 102.00 | 136.00 | 167.00 | 132.00 | 288.00 | 245.00 | 313.00 J | 344.00 J |
| METAL | Mercury | mg/kg | 0.48 | 0.20 | 0.33 | 0.29 | 0.21 | ND | 0.36 | 0.28 | ND | 0.17 | 0.60 | 0.61 |
| METAL | Nickel | mg/kg | ND | 12.30 | 16.50 | 21.10 | 25.80 | 15.40 | 27.00 | 27.00 | 38.70 | 39.90 | 13.40 | 16.90 |
| METAL | Potassium | mg/kg | 2,590.00 | 2,860.00 | 3,410.00 | 3,420.00 | 3,720.00 | 3,600.00 | 3,830.00 | 4,500.00 | 4,240.00 | 1,920.00 | 858.00 | ND |
| METAL | Selenium | mg/kg | 15.70 J | 4.90 J | 4.90 J | 13.00 J | 9.80 J | 8.30 J | 26.10 J | 5.80 J | 5.80 J | 3.60 J | 4.70 J | 2.50 J |
| METAL | Sodium | mg/kg | 668.00 | 546.00 | 578.00 | 566.00 | 765.00 | 659.00 | 598.00 | 613.00 | 560.00 | 342.00 | 119.00 | 111.00 |
| METAL | Vanadium | mg/kg | 56.90 | 33.70 | 53.50 | 68.50 | 69.50 | 74.00 | 73.90 | 72.30 | 32.90 J | 9.00 J | 34.90 | 33.10 |
| METAL | Zinc | mg/kg | 38.20 | 22.20 | 29.00 | 34.90 | 41.30 | 35.20 | 37.30 | 42.20 | 37.10 | 40.00 | 27.70 | 28.20 |
| PEST | 4,4'-DDD | µg/kg | ---- | ND | ---- | 0.52 J | ---- | ---- | ---- | ND | ---- | ---- | ---- | ---- |
| PEST | Alpha-BHC | µg/kg | ---- | ND | ---- | 0.26 JP | ---- | ---- | ---- | ND | ---- | ---- | ---- | ---- |
| RAD | Alpha | pCi/g | 30.00 J | 41.00 J | 35.00 J | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- |
| RAD | Beta | pCi/g | 36.00 J | 45.00 J | 36.00 J | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- |

## Table. 3.5 (continued)

| | | | Composite coal ash samples from FCAP ash borings | | | | | | | | | Composite coal ash samples from Sluice Channel Area | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| RAD | Thorium-228 | pCi/g | 2.000 | 2.600 | 2.100 | 2.700 | 1.900 | 2.300 | 1.500 | 2.300 | 1.400 | 1.100 | 1.100 | 0.970 |
| RAD | Thorium-230 | pCi/g | 4.200 | 4.400 | 3.700 | 5.800 | 3.600 | 4.000 | 3.500 | 3.000 | 2.900 | 2.900 | 2.700 | ---- |
| RAD | Thorium-232 | pCi/g | 2.300 | 2.400 | 2.300 | 2.700 | 1.900 | 2.300 | 1.600 | 1.500 | 1.700 | 0.970 | 1.300 | 0.950 |
| RAD | Uranium-234 | pCi/g | 3.500 | 4.600 | 4.100 | 4.700 | 4.100 | 2.600 | 3.600 | 3.800 | 2.700 | 3.400 | 4.400 | 3.200 |
| RAD | Uranium-235 | pCi/g | 0.150 | 0.350 | 0.220 | 0.290 | 0.170 | 0.140 | 0.210 | 0.250 | 0.150 | 0.150 | 0.170 | 0.099 |
| RAD | Uranium-238 | pCi/g | 3.500 | 4.500 | 3.800 | 4.800 | 4.300 | 3.700 | 3.000 | 3.200 | 2.300 | 2.600 | 3.300 | 2.100 |
| SVOA | bis(2-Ethylhexyl)phthalate | µg/kg | ND | 180.0 J | ND | ND | ND | ND | ND | ND | ND | 70.0 J | ND | ND |
| SVOA | Butylbenzylphthalate | µg/kg | 89.0 J | ND | ND | ND | ND | 110.0 J | ND | ND | ND | 66.0 J | ND | ND |
| VOA | Chloroform | µg/kg | ---- | ND | ---- | 5.0 J | ---- | ---- | ---- | 2.0 J | ---- | ---- | ---- | ---- |

ND = not detected; ---- = not analyzed; J = estimated value; P = lower of two pesticide results, when the two C column results differ >25%; **BOLD** = highest value detected for that analyte.

Analysis Analyte Units

Table 3.6. Comparison of FCAP ash to Eastern U.S. coal ash (mg/kg)

| Analyte | Eastern coal[a] | | Chestnut Ridge OU2 Phase II | |
|---|---|---|---|---|
| | Range | Median | Range | Median |
| Arsenic | 2.0–279 | 75 | 46.5–193 | 100 |
| Barium | 52–2200 | 892 | ND–630 | 371 |
| Beryllium | --- | --- | 1.1–7.5 | 2.20 |
| Cadmium | 0.10–8.24 | 1.59 | ND | ND |
| Chromium | 34–437 | 165 | 8.2–42.1 | 18.40 |
| Cobalt | 6.22–79 | 40.6 | 5.2–29.8 | 14.45 |
| Copper | 3.7–349 | 136 | 19.30–101 | 52.75 |
| Lead | 1.3–222 | 18.0 | 8.6–27.30 | 13.50 |
| Manganese | 79–430 | 190 | 34.10–344 | 134 |
| Mercury | 0.02–4.2 | 0.192 | 0.17 – 1.3 | 0.42 |
| Nickel | 6.6–258 | 78 | 12.3–56.0 | 27.00 |
| Selenium | 0.36–19.0 | 8.05 | 2.5–30.8 | 6.40 |
| Silver | 0.25–8.0 | 0.695 | ND | ND |
| Thallium | 7.0–28.0 | 25.0 | 1.7–3.2 | 2.40 |
| Thorium[b] | 13–68 | 45 | 13.39–24.11 | 20.53 |
| Uranium (total)[b] | 0.8–19.0 | 7.3 | 4.32–9.6[c] | 5.40 |
| Vanadium | 110–551 | 269 | 9.0–12.3 | 69.00 |
| Zinc | 16–1420 | 163 | 22.2–83.1 | 39.10 |

ND = Not detected; --- = data not available.

[a]Report to Congress: Wastes from the Combusion of Coal by Electric Utility Power Plants, EPA/530-SW-88-002 (EPA 1988c).

[b]Furr et al. 1977, *National Survey of Elements and Radioactivity of Fly Ash. Absorption of Elements by Cabbage Grown in Fly Ash–Soil Mixture,* Vol. 11, No. 13, December 1977, pp. 1194 - 1201.

[c]Phase 1 data only.

Table 3.7. Phase 1 analytes for surface water at Chestnut Ridge OU 2

| Volatile organic compounds (CLP) | |
|---|---|
| 1,1,1-Trichloroethane | Carbon tetrachloride |
| 1,1,2,2-Tetrachloroethane | Chlorobenzene |
| 1,1,2-Trichloroethane | Chloroethane |
| 1,1-Dichloroethane | Chloroform |
| 1,1-Dichloroethene | Chloromethane |
| 1,2-Dichloroethene | Cis-1,3-dichloropropene |
| 1,2-Dichloropropane | Dibromochloromethane |
| 2-Butanone | Ethylbenzene |
| 2-Hexanone | Methylene chloride |
| 4-Methyl-2-pentanone | Styrene |
| Acetone | Tetrachloroethene |
| Benzene | Toluene |
| Bromodichloromethane | Trans-1,3-dichloropropene |
| Bromoform | Trichloroethene |
| Bromomethane | Vinyl acetate* |
| Carbon disulfide | Vinyl chloride |
| **Base neutral acid compounds (CLP)** | |
| 1,2,4-Trichlorobenzene | 4-Nitroaniline |
| 1,2-Dichlorobenzene | 4-Nitrophenol |
| 1,3-Dichlorobenzene | 4,6-Dinitro-2-methylphenol |
| 1,4-Dichlorobenzene | Acenaphthene |
| 2,4,5-Trichlorophenol | Acenaphthylene |
| 2,4,6-Trichlorophenol | Anthracene |
| 2,4-Dichlorophenol | Benzo(a)anthracene |
| 2,4-Dimethylphenol | Benzo(a)pyrene |
| 2,4-Dinitrophenol | Benzo(b)fluoranthene |
| 2,4-Dinitrotoluene | Benzo(g,h,i)perylene |
| 2,6-Dinitrotoluene | Benzo(k)fluoranthene |
| 2-Chloronaphthalene | Benzoic acid* |
| 2-Chlorophenol | Benzyl alcohol* |
| 2-Methylnaphthalene | Bis(2-chloroethoxy)methane |
| 2-Methylphenol | Bis(2-chloroethyl)ether |
| 2-Nitroaniline | Bis(2-chloroisopropyl)ether* |
| 2-Nitrophenol | Bis(2-ethylhexyl)phthalate |
| 3,3-Dichlorobenzidine | Butyl benzyl phthalate |
| 3-Nitroaniline | Chrysene |
| 4-Bromophenyl phenyl ether | Di-n-butyl phthalate |
| 4-Chloro-3-methylphenol | Di-n-octyl phthalate |
| 4-Chloraniline | Dibenzo(a,h)anthracene |
| 4-Chlorophenyl phenyl ether | Dibenzofuran |
| 4-Methylphenol | Diethyl phthalate |

Table 3.7 (continued)

| Base neutral acid compounds (continued) | |
|---|---|
| Dimethyl phthalate | N-nitroso-di-n-propylamine |
| Fluoranthene | N-nitrosodiphenylamine |
| Fluorene | Naphthalene |
| Hexachlorobenzene | Nitrate* |
| Hexachlorobutadiene | Nitrobenzene |
| Hexachlorocyclopentadiene | Pentachlorophenol |
| Hexachloroethane | Phenanthrene |
| Indeno(1,2,3-cd)pyrene | Phenols* |
| Isophorone | Pyrene |
| | Sulfate* |
| | TOC* |
| | TOX* |
| **Inorganics** | |
| Aluminum | Manganese |
| Antimony | Mercury |
| Arsenic | Nickel |
| Barium | Niobium* |
| Beryllium | Magnesium |
| Cadmium | Phosphorus |
| Calcium | Potassium |
| Chloride IC* | Selenium |
| Chromium | Silver |
| Cobalt | Sodium |
| Copper | Total uranium (fluorometric)* |
| Flouride IC* | Vanadium |
| Iron | Zinc |
| Lead | |
| **Physical properties** | |
| Alkalinity* | |
| Conductivity* | |
| Dissolved solids* | |
| pH* | |
| Suspended solids* | |
| **Radiological parameters** | |
| Gross alpha | Total radium* |
| Gross beta | |

*Not included in Phase 2 analysis.

Table 3.8. Phase 2 analytes for surface water at Chestnut Ridge OU 2

| VOCs (CLP) | |
|---|---|
| 1,1,1-Trichloroethane | Carbon disulfide |
| 1,1,2,2-Tetrachloroethane | Carbon tetrachloride |
| 1,1,2-Trichloroethane | Chlorobenzene |
| 1,1-Dichloroethane | Chloroethane |
| 1,2-Dichloroethane* | Chloroform |
| 1,1-Dichloroethene | Chloromethane |
| 1,2-Dichloroethene (total) | Cis-1,3-dichloropropene |
| 1,2-Dichloroethane | Dibromochloromethane |
| 1,2-Dichloropropane | Ethylbenzene |
| 2-Butanone | Methylene chloride |
| 2-Hexanone | Styrene |
| 4-Methyl-2-pentanone | Tetrachloroethene |
| Acetone | Toluene |
| Benzene | Trans-1,3-dichloropropene |
| Bromodichloromethane | Trichloroethene |
| Bromoform | Vinyl chloride |
| Bromomethane | Xylene (total)* |
| **Base neutral acid compounds (CLP)** | |
| 1,2,4-Trichlorobenzene | 4-Methylphenol |
| 1,2-Dichlorobenzene | 4-Nitroaniline |
| 1,3-Dichlorobenzene | 4-Nitrophenol |
| 1,4-Dichlorobenzene | Acenaphthene |
| 2,2-Oxybis(1-chloropropane)* | Acenaphthylene |
| 2,4,5-Trichlorophenol | Anthracene |
| 2,4,6-Trichlorophenol | Benzo(a)anthracene |
| 2,4-Dichlorophenol | Benzo(a)pyrene |
| 2,4-Dimethylphenol | Benzo(b)fluoranthene |
| 2,4-Dinitrophenol | Benzo(g,h,i)perylene |
| 2,4-Dinitrotoluene | Benzo(k)fluoranthene |
| 2,6-Dinitrotoluene | Bis(2-chloroethoxy)methane |
| 2-Chloronaphthalene | Bis(2-chloroethyl)ether |
| 2-Chlorophenol | Bis(2-ethylhexyl)phthalate |
| 2-Methylnaphthalene | Butyl benzyl phthalate |
| 2-Methylphenol | Carbazole* |
| 2-Nitroaniline | Chrysene |
| 2-Nitrophenol | Di-n-butyl phthalate |
| 3,3-Dichlorobenzidine | Di-n-octyl phthalate |
| 3-Nitroaniline | Dibenzo(a,h)anthracene |
| 4,6-Dinitro-2-methylphenol | Dibenzofuran |
| 4-Bromophenyl phenyl ether | Diethyl phthalate |
| 4-Chloro-3-methylphenol | Dimethyl phthalate |
| 4-Chloraniline | Fluoranthene |
| 4-Chlorophenyl phenyl ether | Fluorene |

Table 3.8 (continued)

| Base neutral acid compounds (CLP—continued) | |
|---|---|
| Hexachlorobenzene | N-nitrosodiphenylamine |
| Hexachlorobutadiene | Naphthalene |
| Hexachlorocyclopentadiene | Nitrobenzene |
| Hexachloroethane | Pentachlorophenol |
| Indeno(1,2,3-cd)pyrene | Phenanthrene |
| Isophorone | Phenol |
| N-nitroso-di-n-propylamine | Pyrene |
| **Inorganics (TAL—total)** | |
| Aluminum | Iron |
| Antimony | Lead |
| Arsenic | Magnesium |
| Barium | Potassium |
| Beryllium | Selenium |
| Cadmium | Silver |
| Calcium | Sodium |
| Chromium | Thallium* |
| Cobalt | Vanadium |
| Copper | Zinc |
| Hexavalent chromium* | |
| **Radiological parameters** | |
| Gross alpha | Isotopic thorium* |
| Gross beta | Isotopic uranium* |

* Not included in Phase 1 analysis.

Table 3.9. Phase 1 surface water results[a] at Chestnut Ridge OU 2

| Analysis | Analyte | Units | Dry weather | | | |
|---|---|---|---|---|---|---|
| | | | MCB-S3-DW-S3 | MCB-S4-DW-S4 | MCB-S5-DW-S5 | MCB-S6-DW-S6 |
| METAL | Aluminum | µg/L | 129.00 B J | 39.40 B J | 39.40 B J | 48.70 J |
| METAL | Arsenic | µg/L | 84.80 | 21.00 | 2.10 B | ND |
| METAL | Barium | µg/L | 60.60 B | 49.60 B | 78.50 B | 70.20 B |
| METAL | Calcium | µg/L | 43,900.00 | 45,500.00 | 44,700.00 | 32,300.00 |
| METAL | Copper | µg/L | ND | ND | 4.10 B | ND |
| METAL | Iron | µg/L | 1,360.00 | 5.80 B | ND | ND |
| METAL | Magnesium | µg/L | 13,600.00 | 13,500.00 | 14,700.00 | 13,600.00 |
| METAL | Manganese | µg/L | 513.00 | 16.80 | ND | ND |
| METAL | Niobium | µg/L | ND | ND | 7.90 J | ND |
| METAL | Potassium | µg/L | 4,940.00 B | 4,530.00 B | 2,070.00 B | 2,040.00 B |
| METAL | Sodium | µg/L | 2,780.00 B | 2,920.00 B | 1,850.00 B | 1,690.00 B |
| METAL | Zinc | µg/L | 3.30 B J | 2.90 B J | 3.60 B J | 2.10 B J |
| METAL(F) | Aluminum | µg/L | 32.00 B | 38.20 B | 38.20 B | ND |
| METAL(F) | Arsenic | µg/L | 32.00 | 19.90 | ND | --- |
| METAL(F) | Barium | µg/L | 65.60 B | 48.90 B | 78.20 B | --- |
| METAL(F) | Calcium | µg/L | 43,200.00 | 44,200.00 | 45,300.00 | --- |
| METAL(F) | Copper | µg/L | 7.10 B | 4.50 B | 4.90 B | --- |
| METAL(F) | Iron | µg/L | 316.00 | 14.70 B J | 18.30 B J | --- |
| METAL(F) | Lead | µg/L | ND | ND | ND | 0.80 B J |
| METAL(F) | Magnesium | µg/L | 13,200.00 | 12,900.00 | 14,400.00 | --- |
| METAL(F) | Manganese | µg/L | 1,000.00 | 29.60 | 28.10 | --- |
| METAL(F) | Niobium | µg/L | ND | 8.50 J | ND | --- |
| METAL(F) | Potassium | µg/L | 4,430.00 B | 4,450.00 B | 2,130.00 B | --- |
| METAL(F) | Sodium | µg/L | 2,740.00 BE | 3,160.00 BE | 1,730.00 BE | --- |
| METAL(F) | Zinc | µg/L | 6.00 B J | 3.10 B J | 5.00 B J | --- |
| PHYS | Alkalinity | µg/L | 148,000.00 | 159,000.00 | 164,000.00 | 161,000.00 |
| PHYS | Chloride | µg/L | 3,000.00 | 2,000.00 | 2,000.00 | 2,000.00 |
| PHYS | Conductivity | µmhos/cm | ND | ND | ND | 174,000.00 |
| PHYS | Fluoride | µg/L | 200.00 | 200.00 | 200.00 | 200.00 |
| PHYS | Nitrate | µg/L | ND | ND | 1,000.00 J | 1,000.00 J |
| PHYS | pH | STD UNITS | R | R | 7.90 | ND |
| PHYS | Phenols (PIP) | µg/L | 30.00 J | ND | ND | ND |
| PHYS | Sulfate | µg/L | 24,000.00 | ND | 14,000.00 | 14,000.00 |
| PHYS | Total dissolved solids | µg/L | 180,000.00 | 180,000.00 | 168,000.00 | 174,000.00 |
| PHYS | Total Organic Carbon (TOC) | ug/L | ND | ND | ND | ND |
| PHYS | Total Organic Halides (TOX) | µg/L | 10.00 J | ND | ND | 16.00 J |
| PHYS | Total suspended solids | µg/L | 5,000.00 | 8,000.00 | 1,000.00 | 2,000.00 |
| PHYS | Uranium, Total | µg/L | 1.00 | 1.00 | ND | ND |
| RAD | Alpha activity | pCi/L | 378.67 | 352.13 | ND | ND |
| RAD | Beta activity | pCi/L | 936.17 | 580.70 | ND | ND |

Table 3.9 (continued)

| Analysis | Analyte | Units | Dry weather | | | |
|---|---|---|---|---|---|---|
| | | | MCB-S3-DW-S3 | MCB-S4-DW-S4 | MCB-S5-DW-S5 | MCB-S6-DW-S6 |
| RAD | Total Radium | pCi/L | 0.5400 J | 0.5400 J | 0.5400 J | 0.5406 |
| SVOA | Di-n-octylphthlate | µg/L | ND | 27.0 J | ND | ND |
| METAL | Aluminum | µg/L | 79.80 B J | 76.70 B J | 76.70 B J | 33.20 B J |
| METAL | Arsenic | µg/L | 35.40 | 20.80 | 2.40 B | ND |
| METAL | Barium | µg/L | 71.70 B | 53.40 B | 84.10 B | 1.30 B |
| METAL | Calcium | µg/L | 46,700.00 | 46,100.00 | 51,200.00 | 93.60 B J |
| METAL | Copper | µg/L | ND | ND | 8.90 B | ND |
| METAL | Iron | µg/L | 550.00 | 170.00 | 122.00 | ND |
| METAL | Magnesium | µg/L | 14,400.00 | 13,900.00 | 14,500.00 | 12.20 B J |
| METAL | Manganese | µg/L | 1,090.00 | 204.00 | 53.30 | ND |
| METAL | Niobium | µg/L | 9.50 J | ND | ND | ND |
| METAL | Potassium | µg/L | 4,700.00 B | 4,720.00 B | 2,410.00 B | ND |
| METAL | Sodium | µg/L | 2,960.00 B | 2,780.00 B | 2,020.00 B | 248.00 B |
| METAL | Zinc | µg/L | 7.00 B J | 15.30 B | 9.10 B J | 2.40 B J |
| METAL(F) | Aluminum | µg/L | 48.70 B J | 26.90 B J | 61.10 B J | --- |
| METAL(F) | Arsenic | µg/L | 20.90 | 23.90 S S | 2.40 B | --- |
| METAL(F) | Barium | µg/L | 67.20 B | 48.90 B | 82.80 B | --- |
| METAL(F) | Calcium | µg/L | 45,900.00 | 45,600.00 | 50,400.00 | --- |
| METAL(F) | Copper | µg/L | ND | ND | ND | --- |
| METAL(F) | Iron | µg/L | 97.30 B | ND | 60.80 B | --- |
| METAL(F) | Lead | µg/L | ND | ND | ND | --- |
| METAL(F) | Magnesium | µg/L | 14,100.00 | 13,700.00 | 14,200.00 | --- |
| METAL(F) | Manganese | µg/L | 940.00 | ND | 47.40 | --- |
| METAL(F) | Niobium | µg/L | 8.70 J | ND | 8.10 J | --- |
| METAL(F) | Potassium | µg/L | 4,510.00 B | 4,920.00 B | 2,490.00 B | --- |
| METAL(F) | Sodium | µg/L | 2,800.00 B | 2,890.00 B | 1,890.00 B | --- |
| METAL(F) | Zinc | µg/L | 5.40 B J | 4.70 B J | 5.10 B J | --- |
| PHYS | Alkalinity | µg/L | 155,000.00 | 152,000.00 | 171,000.00 | 2,000.00 |
| PHYS | Chloride | µg/L | 3,000.00 | 3,000.00 | 2,000.00 | ND |
| PHYS | Conductivity | µmhos/cm | 342.00 | ND | ND | 1.20 |
| PHYS | Fluoride | µg/L | 100.00 | 100.00 | 200.00 | ND |
| PHYS | Nitrate | µg/L | ND | ND | ND | ND |
| PHYS | pH | STD UNITS | 7.63 | R | --- | 6.55 |
| PHYS | Phenols (PIP) | µg/L | ND | ND | ND | ND |
| PHYS | Sulfate | µg/L | 24,000.00 | 26,000.00 | 18,000.00 | ND |
| PHYS | Total dissolved solids | µg/L | 226,000.00 | 214,000.00 | 194,000.00 | ND |
| PHYS | Total Organic Carbon (TOC) | ug/L | ND | 1,000.00 | --- | 2,000.00 |
| PHYS | Total Organic Halides (TOX) | µg/L | ND | ND | 1,000.00 | ND |
| PHYS | Total suspended solids | µg/L | ND | ND | ND | ND |
| PHYS | Uranium, Total | µg/L | 1.00 | 1.00 | ND | ND |
| RAD | Alpha activity | pCi/L | ND | 2.37 J | ND | 0.69 J |
| RAD | Beta activity | pCi/L | ND | ND | ND | ND |

## Table 3.9 (continued)

| | | | | | | |
|---|---|---|---|---|---|---|
| METAL | Aluminum | µg/L | 79.80 B J | 76.70 B J | 76.70 B J | 33.20 B J |
| RAD | Total Radium | pCi/L | 0.4860 J | ND | 0.5670 J | 0.4320 J |
| SVOA | Di-n-octylphthlate | µg/L | ND | ND | ND | ND |

[a]For comparison, Ambient Water Quality Criteria are given in Table 4.2 (pages 4-11).
J, E = Estimated value;
S = Estimated by the method of standard additions;
R = Rejected;
B = Below the required detection limit but at or above the instrument detection limit;
**BOLD** = Exceeds water quality criteria.

**Table 3.10. Phase 2 surface water results at Chestnut Ridge OU 2**

| Analysis | Analyte | Units | Upper McCoy Branch (SW-012-02 taken at SW-012-01 location, 7 days later) | | Background creek samples | | | |
|---|---|---|---|---|---|---|---|---|
| | | | | | Walker Branch | | | White Oak Creek |
| | | | SW-012-01 | SW-012-02 | SW-005-01 | SW-006-01 | SW-007-01 | SW-011-91 |
| METALS | Aluminum | µg/L | 133.00 | 54.60 | ND | ND | ND | ND |
| METALS | Arsenic | µg/L | 6.60 J | 5.40 | ND | ND | ND | ND |
| METALS | Barium | µg/L | 77.10 | 77.20 | 53.40 | 63.00 | 63.70 | 146.00 |
| METALS | Calcium | µg/L | 47,100.00 | 46,700.00 | 27,000.00 | 27,200.00 | 27,600.00 | 35,600.00 |
| METALS | Chromium | µg/L | ND | ND | ND | ND | ND | ND |
| METALS | Copper | µg/L | ND | ND | ND | ND | ND | ND |
| METALS | Iron | µg/L | ND | ND | 41.90 | 41.90 | ND | 54.00 |
| METALS | Lead | µg/L | ND | ND | ND | ND | ND | ND |
| METALS | Magnesium | µg/L | 15,100.00 | 15,400.00 | 14,300.00 | 14,300.00 | 14,200.00 | 17,400.00 |
| METALS | Manganese | µg/L | 139.00 | 75.50 | 14.60 | 8.60 | 6.40 J | 38.10 |
| METALS | Mercury | µg/L | ND | ND | ND | ND | ND | ND |
| METALS | Potassium | µg/L | 2,360.00 | 2,560.00 | ND | ND | ND | ND |
| METALS | Sodium | µg/L | 1,400.00 | 1,410.00 | 833.00 | 791.00 | 785.00 | 548.00 |
| METALS | Vanadium | µg/L | ND | ND | ND | ND | ND | ND |
| METALS | Zinc | µg/L | 11.30 | 6.50 | 7.60 J | 5.20 J | 5.40 J | 4.90 |
| RAD | Alpha | pCi/g | ND | ---- | ---- | ---- | ---- | ---- |
| RAD | Beta | pCi/g | ND | ---- | ---- | ---- | ---- | ---- |
| RAD | Thorium-228 | pCi/g | ND | ND | ND | ND | ND | ND |
| RAD | Thorium-230 | pCi/g | ND | ND | ND | ND | ND | ND |
| RAD | Thorium-232 | pCi/g | ND | ND | ND | ND | ND | ND |
| RAD | Uranium-234 | pCi/g | 0.320 | 0.500 | 0.210 | 0.170 | 0.140 | 0.510 J |
| RAD | Uranium-235 | pCi/g | ND | ND | ND | ND | ND | ND |
| RAD | Uranium-238 | pCi/g | 0.180 | ND | ND | ND | ND | 0.190 J |
| SVOA | bis(2-Ethylhexyl)Phthalate | µg/L | ND | 48.00 | 6.00 J | 6.00 J | ND | 2.00 J |
| SVOA | Di-n-Butylphthalate | µg/L | ND | ND | ND | ND | 1.00 J | ND |
| SVOA | Phenol | µg/L | ND | ND | ND | ND | ND | 2.00 J |
| SVOA | Diethylphthalate | µg/L | ND | 2.00 J | ND | ND | ND | ND |

**Table 3.10 (continued)**

| Analysis | Analyte | Units | Ponds—FCAP | | Pond—BKGD Reference Pond | Springs—Upper McCoy Branch area | | | | Spring—BKGD Upper east end of FCAP |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Upper east end of FCAP | Upper west end of FCAP | | Toe of dam | | | McCoy Branch | |
| | | | | | | (Wet season) | (Dry season) | (rain event) | | |
| | | | SW-008-01 | SW-009-01 | SW-010-01 | SW-003-01 | SW-003-02 | SW-003-03 | SW-004-01 | SW-002-01 |
| METALS | Aluminum | µg/L | 7,790.00 | ND | 556.00 | 1,460.00 | 420.00 | 38,500.00 | ND | ND |
| METALS | Arsenic | µg/L | 10.00 J | ND | ND | 229.00 | 77.40 | 5,430.00 | 12.40 | ND |
| METALS | Barium | µg/L | 68.60 | 33.20 | 16.40 | 77.6 | 76.90 | 1,010.00 | 153.00 | 15.50 |
| METALS | Beryllium | µg/L | ND | ND | ND | ND | ND | 8.400 | ND | ND |
| METALS | Calcium | µg/L | 18,500.00 | 6,000.00 | 2,140.00 | 48,900.00 | 49,900.00 | 61,300.00 | 40,000.00 | 27,000.00 |
| METALS | Chromium | µg/L | 8.80 | ND | ND | ND | ND | 45.200 J | ND | ND |
| METALS | Copper | µg/L | 7.30 | ND | ND | ND | ND | 60.800 | ND | ND |
| METALS | Iron | µg/L | 6,110.00 | 677.00 | 2,340.00 | 3,310.00 | 1,230.00 | 147,000.00 | 88.00 | 162.00 |
| METALS | Lead | µg/L | 3.50 | ND | ND | 3.00 J | ND | 37.10 | ND | ND |
| METALS | Magnesium | µg/L | 5,590.00 | 2,450.00 | 617.00 | 13,900.00 | 15,700.00 | 21,100.00 | 15,400.00 | 15,300.00 |
| METALS | Manganese | µg/L | 1,700.00 | 121.00 | 435.00 | 898.00 | 1,070.00 | 20,000.00 | 36.90 | 216.00 |
| METALS | Mercury | µg/L | ND | ND | ND | ND | 0.24 | 0.41 J | ND | ND |
| METALS | Nickel | µg/L | ND | ND | ND | ND | ND | 72.70 | ND | ND |
| METALS | Potassium | µg/L | 6,540.00 | ND | 2,160.00 | 3,200.00 | 3,720.00 | 11,300.00 | 2,480.00 | ND |
| METALS | Sodium | µg/L | 1,190.00 | 876.00 | 329.00 | 2,360.00 | 2,180.00 | 3,290.00 | 1,660.00 | ND |
| METALS | Vanadium | µg/L | 14.50 | ND | ND | ND | ND | 103.00 J | ND | ND |
| METALS | Zinc | µg/L | 106.00 J | 11.10 J | 6.20 J | 28.50 | 33.5 | 594.00 | ND | ND |
| RAD | Alpha | pCi/g | ---- | ---- | ---- | 18.000 J | ND | 17.00 J | ND | ND |
| RAD | Beta | pCi/g | ---- | ---- | ---- | 6.900 | 3.400 | 18.000 | ND | ND |
| RAD | Thorium-228 | pCi/g | 0.220 | ND | ND | 0.041 | 0.005 | 1.300 J | ND | ND |
| RAD | Thorium-230 | pCi/g | ND | 2.000 J | ND | 0.620 J | 0.180 | 1.700 J | ND | ND |
| RAD | Thorium-232 | pCi/g | ND | ND | 0.630 | 0.077 | 0.032 | 0.880 J | ND | ND |
| RAD | Uranium-234 | pCi/g | 0.330 | ND | 0.072 | 0.290 | 0.460 | 1.500 J | 0.250 | 0.210 |
| RAD | Uranium-235 | pCi/g | ND | ND | ND | ND | 0.009 | 0.072 | ND | ND |
| RAD | Uranium-238 | pCi/g | 0.099 J | ND | ND | 0.330 J | 0.220 | 0.600 J | 0.220 J | 0.068 J |

**Table 3.10 (continued)**

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| SVOA | bis(2-Ethylhexyl)Phthalate | µg/L | ND | ND | ND | **1.00 J** | ND | ND | **4.00 J** | ND |
| SVOA | Di-n-Butylphthalate | µg/L | ND | ND | ND | ND | ND | ND | ND | ND |
| SVOA | Phenol | µg/L | ND | ND | ND | ND | ND | ND | ND | ND |
| SVOA | Diethylphthalate | µg/L | ND | ND | ND | ND | ND | ND | ND | ND |

ND = not detected; ---- = not analyzed for J = estimated value; **BOLD** = exceeds water quality standards.

Table 3.11. Phase 2 analytes for sediment at Chestnut Ridge OU 2

| VOCs (CLP) | |
|---|---|
| 1,1,1-Trichloroethane | Carbon tetrachloride |
| 1,1,2,2-Tetrachloroethane | Chlorobenzene |
| 1,1,2-Trichloroethane | Chloroethane |
| 1,1-Dichloroethane | Chloroform |
| 1,2-Dichloroethane | Chloromethane |
| 1,1-Dichloroethene | Cis-1,3-dichloropropene |
| 1,2-Dichloroethene (total) | Dibromochloromethane |
| 1,2-Dichloropropane | Ethylbenzene |
| 2-Butanone | Methylene chloride |
| 2-Hexanone | Styrene |
| 4-Methyl-2-pentanone | Tetrachloroethene |
| Acetone | Toluene |
| Benzene | Trans-1,3-dichloropropene |
| Bromodichloromethane | Trichloroethene |
| Bromoform | Vinyl chloride |
| Bromomethane | Xylene (total) |
| Carbon disulfide | |
| **Base neutral acid compounds (CLP)** | |
| 1,2,4-Trichlorobenzene | 4-Nitrophenol |
| 1,2-Dichlorobenzene | 4,6-Dinitro-2-methylphenol |
| 1,3-Dichlorobenzene | Acenaphthene |
| 1,4-Dichlorobenzene | Acenaphthylene |
| 2,2-Oxybis(1-chloropropane) | Anthracene |
| 2,4,5-Trichlorophenol | Benzo(a)anthracene |
| 2,4,6-Trichlorophenol | Benzo(a)pyrene |
| 2,4-Dichlorophenol | Benzo(b)fluoranthene |
| 2,4-Dimethylphenol | Benzo(g,h,i)perylene |
| 2,4-Dinitrophenol | Benzo(k)fluoranthene |
| 2,4-Dinitrotoluene | Bis(2-chloroethoxy)methane |
| 2,6-Dinitrotoluene | Bis(2-chloroethyl)ether |
| 2-Chloronaphthalene | Bis(2-ethylhexyl)phthalate |
| 2-Chlorophenol | Butyl benzyl phthalate |
| 2-Methylnaphthalene | Carbazole |
| 2-Methylphenol | Chrysene |
| 2-Nitroaniline | Di-n-butyl phthalate |
| 2-Nitrophenol | Di-n-octyl phthalate |
| 3,3-Dichlorobenzidine | Dibenzo(a,h)anthracene |
| 3-Nitroaniline | Dibenzofuran |
| 4-Bromophenyl phenyl ether | Diethyl phthalate |
| 4-Chloro-3-methylphenol | Dimethyl phthalate |
| 4-Chloraniline | Fluoranthene |
| 4-Chlorophenyl phenyl ether | Fluorene |
| 4-Methylphenol | Hexachlorobenzene |
| 4-Nitroaniline | Hexachlorobutadiene |

Table 3.11 (continued)

| Base neutral acid compounds (CLP—Continued) | |
|---|---|
| Hexachlorocyclopentadiene | Naphthalene |
| Hexachloroethane | Nitrobenzene |
| Indeno(1,2,3-cd)pyrene | Pentachlorophenol |
| Isophorone | Phenanthrene |
| N-nitroso-di-n-propylamine | Phenol |
| N-nitrosodiphenylamine | Pyrene |
| | TOC |
| **Inorganics (TAL—total)** | |
| Aluminum | Manganese |
| Antimony | Mercury |
| Arsenic | Nickel |
| Barium | Magnesium |
| Beryllium | Potassium |
| Cadmium | Selenium |
| Calcium | Silver |
| Chromium | Sodium |
| Cobalt | Thallium |
| Copper | Vanadium |
| Iron | Zinc |
| Lead | |
| **Physical properties** | |
| Particle size distribution | |
| **Radiological parameters** | |
| Gross alpha | Isotopic thorium |
| Gross beta | Isotopic uranium |

## Table 3.12. Phase 2 sediment sample results at Chestnut Ridge OU 2

| | | | Springs - Upper McCoy Branch area | | | | Springs - FCAP | | Background stream sediments | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | McCoy Branch | Toe of dam | | | West upper end of FCAP | East upper end of FCAP | Walker Branch | | | White Oak Creek |
| Analysis | Analyte | Units | SD-004-01 | (Wet season) SD-003-01 | (Dry season) SD-003-02 | (Rain event) SD-003-03 | SD-001-01 | SD-002-01 | SD-005-01 | SD-006-01 | SD-007-01 | SD-011-01 |
| METAL | Aluminum | mg/kg | 9,320.00 | 22,700.00 | 18,600.00 | 10,200.00 J | 11,500.00 | 10,600.00 | 6,030.00 | 3,870.00 | 4,900.00 | 2,810.00 |
| METAL | Arsenic | mg/kg | 167.00 J | 2,300.00 J | 774.00 | 43.50 J | 53.50 J | 10.00 J | 7.60 | 4.40 | 13.40 | 3.50 J |
| METAL | Barium | mg/kg | 309.00 | 596.00 | 235.00 | 26.10 | 98.10 | 43.60 | 218.00 | 118.00 | 68.00 | 87.80 |
| METAL | Beryllium | mg/kg | 1.40 | 3.40 | 4.70 | ND | ND | 2.10 | 0.77 | 0.80 | 1.20 | ND |
| METAL | Calcium | mg/kg | 3,640.00 | 8,190.00 | 30,400.00 | 3,080.00 | 1,450.00 | 3,710.00 | 10,200.00 | 1,020.00 | 17,900.00 | 859.00 |
| METAL | Chromium | mg/kg | 8.50 | 21.70 | 20.90 | 15.40 J | 10.40 | 14.50 | 8.30 | 27.30 | 24.50 | 16.60 |
| METAL | Cobalt | mg/kg | 20.70 | 86.20 | 29.10 | 10.30 | 9.60 | 8.50 | 13.20 | 13.40 | 17.60 | 8.10 |
| METAL | Copper | mg/kg | 19.10 | 23.10 | 20.00 | 22.40 J | 12.10 | ND | 14.80 | 2.00 | 2.70 | ND |
| METAL | Iron | mg/kg | 21,900.00 | 63,300.00 | 42,700.00 | 26,300.00 J | 18,600.00 | 9,550.00 | 11,600.00 | 19,400.00 | 20,900.00 | 20,300.00 J |
| METAL | Lead | mg/kg | 6.70 J | 11.60 J | 17.10 J | 24.70 J | 16.90 J | 24.80 J | 9.90 | 1.70 | 11.80 | 10.30 J |
| METAL | Magnesium | mg/kg | 1,020.00 | 3,810.00 | 18,000.00 | 1,330.00 | 727.00 | 1,230.00 | 5,890.00 | 467.00 | 3,130.00 | 189.00 |
| METAL | Manganese | mg/kg | 4,030.00 | 13,700.00 | 4,900.00 | 495.00 J | 603.00 | 363.00 | 3,370.00 | 2,040.00 | 1,490.00 | 1,310.00 J |
| METAL | Nickel | mg/kg | 30.70 | 45.10 | 37.20 | 16.50 | 13.00 | ND | 11.50 | 5.90 | 9.20 | ND |
| METAL | Potassium | mg/kg | 1,750.00 | 2,600.00 | 2,280.00 | 815.00 | 748.00 | ND | 785.00 | ND | ND | ND |
| METAL | Selenium | mg/kg | ND | ND | 2.20 J | ND | 1.00 J | ND | ND | ND | ND | ND |
| METAL | Sodium | mg/kg | 385.00 | 520.00 | 159.00 | 49.10 | 218.00 | 541.00 | ND | ND | ND | 32.40 |
| METAL | Vanadium | mg/kg | 27.30 | 48.70 | 47.10 | 48.00 | 29.20 | 23.70 | 13.60 | 33.70 | 33.90 | 35.00 |
| METAL | Zinc | mg/kg | 157.00 | 251.00 | 265.00 J | 67.60 J | 40.00 | 58.20 | 321.00 | 106.00 | 106.00 | 24.10 |
| RAD | Alpha | pCi/g | 18.00 J | 74.00 J | 13.00 J | 11.00 | 12.00 J | 11.00 J | ---- | ---- | ---- | ---- |
| RAD | Beta | pCi/g | 16.00 J | 49.00 J | 22.00 | ND | 16.00 J | 16.00 J | ---- | ---- | ---- | ---- |
| RAD | Thorium-228 | pCi/g | 1.000 | 4.300 | 0.810 J | 0.67 J | 0.730 | 0.630 | 0.840 | 0.360 | 0.720 | 0.620 |
| RAD | Thorium-230 | pCi/g | ND | ND | ND | 1.60 | 2.200 | ND | ND | ND | ND | ND |
| RAD | Thorium-232 | pCi/g | 1.100 | 0.930 | ND | 1.50 | 0.630 | 0.450 | 0.580 | 0.260 | 0.710 | 0.660 |
| RAD | Uranium-234 | pCi/g | 0.930 | 1.700 | 1.000 | ND | 1.700 | 3.900 | 1.000 | 0.650 | 0.950 | 0.870 |
| RAD | Uranium-235 | pCi/g | ND | 0.140 | ND | ND | 0.290 | 0.250 | 0.180 | 0.220 | 0.120 | ND |
| RAD | Uranium-238 | pCi/g | 1.200 | 1.900 | 1.200 J | 1.90 | 1.600 | 3.000 | 1.400 | 0.790 | 0.720 | 0.800 |
| SVOA | 4-Methylphenol | µg/kg | ND | 99.000 J | 210.000 J | ND | ND | 1,100.000 J | ---- | ---- | ---- | ---- |
| SVOA | Butylbenzylphthalate | µg/kg | ND | ND | ND | ND | ND | 170.00 J | ---- | ---- | ---- | ---- |
| SVOA | Di-n-Butylphthalate | µg/kg | ND | ND | ND | ND | ND | 160.00 J | ---- | ---- | ---- | ---- |
| VOA | 2-Butanone | µg/kg | ND | ND | 16.00 J | ND | ND | ND | ---- | ---- | ---- | ---- |
| VOA | Acetone | µg/kg | ND | ND | ND | 64.00 J | ND | ND | ---- | ---- | ---- | ---- |
| SVOA | 4-Nitrophenol | µg/kg | ND | ND | ND | 87.00 J | ND | ND | ---- | ---- | ---- | ---- |

ND = not detected; ---- = not analyzed; J = estimated value.

**Table 3.13. Monitoring wells at Chestnut Ridge OU 2**

| Well | Total Depth | Screened Interval | Casing I.D. | Material | Date |
|---|---|---|---|---|---|
| GW-321 | 98.60 ft | 87.3 - 98.0 ft | 4.0 in. | stainless | 9/9/87 |
| GW-672 | 28.00 ft | 18.0 - 28.0 ft | 4.0 in. | stainless | 9/5/90 |
| GW-673 | 133.00 ft | Casing to 112 ft; open hole | 4.0 in. | steel | 10/1/90 |
| GW-674 | 16.5 ft | 9.0 - 14.0 ft | 4.0 in. | stainless | 9/3/90 |
| GW-676 | 17.0 ft | 7.0 - 17.0 ft | 4.0 in. | stainless | 9/2/90 |
| GW-512 | 61.00 ft | 50.5 - 60.5 ft | 4.0 in. | stainless | 4/4/88 |
| GW-513 | 125.30 ft | 114.8 - 124.8 ft | 4.0 in. | stainless | 3/30/88 |
| GW-514 | 195.00 ft | Casing to 172.3 ft; open hole | 6.6 in. | steel | 3/24/88 |

Table 3.14. Phase 1 analytes for groundwater at Chestnut Ridge OU 2

| Inorganics | |
|---|---|
| Aluminum | Manganese |
| Antimony | Mercury |
| Arsenic | Molybdenum |
| Barium | Niobium |
| Beryllium | Phosphorus |
| Boron | Potassium |
| Cadmium | Selenium |
| Calcium | Silicon |
| Chromium | Silver |
| Cobalt | Sodium |
| Copper | Strontium |
| Iron | Thorium |
| Lead | Vanadium |
| Magnesium | Zinc |
| **Physical properties** | |
| Alkalinity | pH |
| Bicarbonate | Phenols |
| Carbonate | Redox |
| Chloride IC | Sulfate |
| Specific conductance | Suspended solids |
| Dissolved oxygen | Temperature (°C) |
| Dissolved solids | TOC |
| Fluoride FIA | TOX |
| Fluoride IC | Total uranium (fluorometric) |
| Nitrate | Turbidity |
| Nitrate nitrogen | |
| **Radiological parameters** | |
| Gross alpha | Total radium |
| Gross beta | |

Table 3.15. Phase 2 analytes for groundwater at FCAP

| VOCs (CLP) | |
|---|---|
| 1,1,1-Trichloroethane | Carbon tetrachloride |
| 1,1,2,2-Tetrachloroethane | Chlorobenzene |
| 1,1,2-Trichloroethane | Chloroethane |
| 1,1-Dichloroethane | Chloroform |
| 1,2-Dichloroethane | Chloromethane |
| 1,1-Dichloroethene | Cis-1,3-dichloropropene |
| 1,2-Dichloroethane (total) | Dibromochloromethane |
| 1,2-Dichloropropane | Ethylbenzene |
| 2-Butanone | Methylene chloride |
| 2-Hexanone | Styrene |
| 4-Methyl-2-pentanone | Tetrachloroethene |
| Acetone | Toluene |
| Benzene | Trans-1,3-dichloropropene |
| Bromodichloromethane | Trichloroethene |
| Bromoform | Vinyl chloride |
| Bromomethane | Xylene (total) |
| Carbon disulfide | |
| **Base neutral acid compounds (CLP)** | |
| 1,2,4-Trichlorobenzene | 4-Nitroaniline |
| 1,2-Dichlorobenzene | 4-Nitrophenol |
| 1,3-Dichlorobenzene | 4,6-Dinitro-2-methylphenol |
| 1,4-Dichlorobenzene | Acenaphthene |
| 2,2-Oxybis(1-chloropropane) | Acenaphthylene |
| 2,4,5-Trichlorophenol | Anthracene |
| 2,4,6-Trichlorophenol | Benzo(a)anthracene |
| 2,4-Dichlorophenol | Benzo(a)pyrene |
| 2,4-Dimethylphenol | Benzo(b)fluoranthene |
| 2,4-Dinitrophenol | Benzo(g,h,i)perylene |
| 2,4-Dinitrotoluene | Benzo(k)fluoranthene |
| 2,6-Dinitrotoluene | Bis(2-chloroethoxy)methane |
| 2-Chloronaphthalene | Bis(2-chloroethyl)ether |
| 2-Chlorophenol | Bis(2-ethylhexyl)phthalate |
| 2-Methylnaphthalene | Butyl benzyl phthalate |
| 2-Methylphenol | Carbazole |
| 2-Nitroaniline | Chrysene |
| 2-Nitrophenol | Di-n-butyl phthalate |
| 3,3-Dichlorobenzidine | Di-n-octyl phthalate |
| 3-Nitroaniline | Dibenzo(a,h)anthracene |
| 4-Bromophenyl phenyl ether | Dibenzofuran |
| 4-Chloro-3-methylphenol | Diethyl phthalate |
| 4-Chloraniline | Dimethyl phthalate |
| 4-Chlorophenyl phenyl ether | Fluoranthene |
| 4-Methylphenol | Fluorene |

Table 3.15 (continued)

| Base neutral acid compounds (CLP—continued) | |
|---|---|
| Hexachlorobenzene | N-nitrosodiphenylamine |
| Hexachlorobutadiene | Naphthalene |
| Hexachlorocyclopentadiene | Nitrobenzene |
| Hexachloroethane | Pentachlorophenol |
| Indeno(1,2,3-cd)pyrene | Phenanthrene |
| Isophorone | Phenol |
| N-nitroso-di-n-propylamine | Pyrene |
| **Inorganics (TAL—total and dissolved)** | |
| Aluminum | Iron |
| Antimony | Lead |
| Arsenic | Magnesium |
| Barium | Potassium |
| Beryllium | Selenium |
| Cadmium | Silver |
| Calcium | Sodium |
| Chromium | Thallium |
| Cobalt | Vanadium |
| Copper | Zinc |
| **Radiological parameters** | |
| Gross alpha | Isotopic thorium |
| Gross beta | Isotopic uranium |

## Table 3.16. Phase 1 groundwater results at Chestnut Ridge OU 2

| Analysis | Analyte | Units | Sluice Channel Area monitoring wells | | | Upper McCoy Branch area monitoring wells | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | GW-512 | GW-513 | GW-514 | GW-321 | GW-672 Sample 5001 | GW-672 Sample 5007 | GW-673 | GW-674 | GW-676 Sample 5004 |
| METAL | Aluminum | µg/L | ND | ND | ND | 42.90 BE | 12,000.00 | ND | 1,060.00 | 21,800.00 | 11,700.00 E |
| METAL | Arsenic | µg/L | ND | ND | ND | ND | 2.70 B | ND | ND | 3.20 B | ND |
| METAL | Barium | µg/L | 27.00 | 7.60 | 7.50 | 43.30 B | 255.00 | 3.10 B | 46.90 B | 674.00 | 179.00 B |
| METAL | Beryllium | µg/L | 0.33 | ND | ND | ND | 1.30 B | ND | ND | 4.00 B | 0.50 B |
| METAL | Boron | µg/L | 220.00 | 240.00 | 15.00 | ---- | ---- | ---- | ---- | ---- | ---- |
| METAL | Cadmium | µg/L | ND | ND | ND | ND | ---- | ---- | ND | 8.00 | ND |
| METAL | Calcium | µg/L | 35,000.00 | 35,000.00 | 34,000.00 | 24,300.00 E | 114,000.00 | 152.00 B | 8,310.00 | 93,200.00 | 159,000.00 E |
| METAL | Chromium | µg/L | ND | ND | ND | ND | 24.90 | ---- | 74.70 | 31.40 | 22.00 |
| METAL | Cobalt | µg/L | ND | ND | ND | ND | 21.10 B | ND | ND | 27.60 B | 13.10 B |
| METAL | Copper | µg/L | ND | ND | 22.00 | ND | 30.10 | ND | 52.50 | 73.50 | 15.00 B |
| METAL | Iron | µg/L | ND | ND | 230.00 | 22.80 BE | 16,100.00 | 14.60 B | 2,110.00 | 33,500.00 | 17,700.00 E |
| METAL | Lead | µg/L | ND | ND | ND | ND | 6.50 NJ | ND | 10.40 BWNJ | 52.90 WNJ | 8.10 WNJ |
| METAL | Magnesium | µg/L | 22,000.00 | 22,000.00 | 21,000.00 | 13,900.00 | 33,900.00 | 23.40 J | 12,100.00 | 37,400.00 | 21,200.00 |
| METAL | Manganese | µg/L | ND | ND | 7.10 | 5.20 B | 2,750.00 B | 2.20 B | 44.60 | 2,460.00 | 1,610.00 |
| METAL | Mercury | µg/L | ND | ND | ND | ND | 0.26 | ND | ND | 0.33 | ND |
| METAL | Nickel | µg/L | ND | ND | 14.00 | ND | 25.40 B | ND | 32.90 B | 41.10 | 13.40 B |
| METAL | Niobium | µg/L | ---- | ---- | ---- | 8.40 J | ND | ND | ND | ND | 10.20 |
| METAL | Phosphorus | µg/L | ---- | ---- | ---- | ND | ND | ND | ND | ND | 530.00 |
| METAL | Potassium | µg/L | 3,600.00 | 1,700.00 | 1,300.00 | 775.00 B | 4,960.00 B | ND | 67,900.00 | 7,710.00 | 4,100.00 B |
| METAL | Silicon | µg/L | 6,500.00 | 4,500.00 | 3,900.00 | ---- | ---- | ---- | ---- | ---- | ND |
| METAL | Sodium | µg/L | 1,200.00 | 1,500.00 | 700.00 | 660.00 B | 3,870.00 B | 378.00 B | 16,800.00 | 3,510.00 B | 8,250.00 |
| METAL | Strontium | µg/L | 23.00 | 22.00 | 21.00 | ---- | ---- | ---- | ---- | ---- | 8,250.00 |
| METAL | Vanadium | µg/L | ND | ND | ND | ND | 34.30 B | ND | ND | 62.70 | 18.50 B |
| METAL | Zinc | µg/L | 6.90 | ND | 12.00 | 50.00 | 14.60 B | 244.00 | 134.00 | 2,070.00 | 87.10 |
| METAL(F) | Aluminum | µg/L | ND | 50.00 | 24.00 | ND | 910.00 | ---- | 48.70 BJ | 70.40 BJ | 52.10 BE |
| METAL(F) | Barium | µg/L | 29.00 | 7.90 | 7.30 | ND | 133.00 B | ---- | 29.60 B | 236.00 | 62.80 B |
| METAL(F) | Boron | µg/L | ---- | 32.00 | 19.00 | ---- | ---- | ---- | ---- | ---- | ND |
| METAL(F) | Calcium | µg/L | 34,000.00 | 35,000.00 | 34,000.00 | ND | 95,200.00 | ---- | 5,290.00 | 62,200.00 | 76,600.00 E |
| METAL(F) | Cobalt | µg/L | ND | ND | ND | ND | 5.60 B | ---- | ND | ND | ND |
| METAL(F) | Copper | µg/L | 4.60 | 6.10 | 5.40 | ND | 11.10 B | ---- | 6.30 B | 6.60 B | ND |
| METAL(F) | Iron | µg/L | 150.00 | ND | 63.00 | ND | 1,110.00 | ---- | 37.10 B | 20.60 B | 26.30 BE |

## Table 3.16 (continued)

| Analysis | Analyte | Units | Sluice Channel Area monitoring wells | | | Upper McCoy Branch area monitoring wells | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | GW-512 | GW-513 | GW-514 | GW-321 | GW-672 Sample 5001 | GW-672 Sample 5007 | GW-673 | GW-674 | GW-676 Sample 5004 |
| METAL(F) | Lead | µg/L | ND | ND | ND | 3.90 WNJ | 2.40 BNJ | ---- | ND | ND | 3.10 WNJ |
| METAL(F) | Magnesium | µg/L | 21,000.00 | 22,000.00 | 21,000.00 | ND | 25,500.00 | ---- | 11,200.00 | 21,100.00 | 16,400.00 |
| METAL(F) | Manganese | µg/L | 2.90 | ND | 7.10 | ND | 1,380.00 | ---- | 3.80 B | 224.00 | 680.00 |
| METAL(F) | Nickel | µg/L | ND | 10.00 | ND | ND | ND | ---- | ND | ND | ND |
| METAL(F) | Niobium | µg/L | ---- | ---- | ---- | ND | ND | ---- | ND | 8.80 J | ND |
| METAL(F) | Potassium | µg/L | 3,400.00 | 1,400.00 | 1,100.00 | ND | 3,960.00 B | ---- | 65,500.00 | 4,230.00 B | 1,470.00 B |
| METAL(F) | Selenium | µg/L | ND | ND | ND | ND | 2.20 BW | ---- | ND | ND | ND |
| METAL(F) | Silicon | µg/L | 5,400.00 | 3,700.00 | 3,800.00 | ---- | ---- | ---- | ---- | ---- | ND |
| METAL(F) | Sodium | µg/L | 700.00 | 750.00 | 630.00 | ND | 4,200.00 B | ---- | 17,000.00 | 3,540.00 B | 7,720.00 |
| METAL(F) | Strontium | µg/L | 22.00 | 22.00 | 21.00 | ---- | ---- | ---- | ---- | ---- | ND |
| METAL(F) | Zinc | µg/L | 8.30 | ND | 4.40 | ND | 56.20 | ---- | 16.30 B | 73.50 | 32.30 |
| PHYS | Alkalinity | µg/L | 118,000.00 | 176,000.00 | 172,000.00 | 119,000.00 | 282,000.00 | 3,000.00 | 121,000.00 | 224,000.00 | 185,000.00 |
| PHYS | Chloride | µg/L | 1,000.00 | 1,000.00 | 1,000.00 | 1,000.00 | 3,000.00 | ND | 2,000.00 | 2,000.00 | 5,000.00 |
| PHYS | Dissolved Oxygen | ppm | 5.30 | 5.40 | 4.60 | ---- | ---- | ---- | ---- | ---- | ---- |
| PHYS | Fluoride | µg/L | ND | ND | ND | ND | 100.00 | ND | 1,000.00 | 200.00 | 200.00 |
| PHYS | Nitrate | µg/L | ND | ND | ND | 1,000.00 J | 3,000.00 J | ND | 1,000.00 J | ND | 7,000.00 J |
| PHYS | Redox | MV | 197.00 | 181.00 | 138.00 | ND | ---- | ---- | ---- | ---- | ND |
| PHYS | Sulfate | µg/L | 1,000.00 | 2,000.00 | 4,000.00 | 2,000.00 | 37,000.00 | ND | 27,000.00 | 27,000.00 | 10,000.00 |
| PHYS | Total dissolved solids | µg/L | 170,000.00 | 166,000.00 | 188,000.00 | 148,000.00 | 382,000.00 | 14,000.00 | 222,000.00 | 260,000.00 | 336,000.00 |
| PHYS | Total Organic Carbon | µg/L | ND | ND | ND | 4,000.00 | 2,000.00 J | ND | 10,000.00 | 7,000.00 | 1,000.00 |
| PHYS | Total Organic Halides | µg/L | ND | ND | ND | ND | 54.00 J | ND | 351.00 J | ND | ND |
| PHYS | Total suspended solids | µg/L | 3,000.00 | 3,000.00 | ND | 38,000.00 | 1,170,000.00 | ND | ND | ND | 526,000.00 |
| PHYS | Uranium, Total | µg/L | ND | ND | 1.00 | ND | 2.00 | ND | 1.00 | 2.00 | 1.00 |
| PHYS(F) | Alkalinity | µg/L | ND | ND | ND | 119,000.00 | 279,000.00 | ---- | 143,000.00 | 226,000.00 | 216,000.00 |
| PHYS(F) | Chloride | µg/L | ND | ND | ND | 1,000.00 | 3,000.00 | ---- | 2,000.00 | 2,000.00 | 4,000.00 |
| PHYS(F) | Fluoride | µg/L | ND | ND | ND | ND | 100.00 | ---- | 1,000.00 | 200.00 | 200.00 |
| PHYS(F) | Nitrate | µg/L | ND | ND | ND | 1,000.00 J | 4,000.00 J | ---- | ND | ND | 5,000.00 J |
| PHYS(F) | Sulfate | µg/L | ND | ND | ND | 2,000.00 | 37,000.00 | ---- | 27,000.00 | 27,000.00 | 8,000.00 |
| PHYS(F) | Total dissolved solids | µg/L | ND | ND | ND | 138,000.00 | 374,000.00 | ---- | 226,000.00 | 268,000.00 | 314,000.00 |

## Table 3.16 (continued)

| Analysis | Analyte | Units | Sluice Channel Area monitoring wells | | | Upper McCoy Branch area monitoring wells | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | GW-512 | GW-513 | GW-514 | GW-321 | GW-672 Sample 5001 | GW-672 Sample 5007 | GW-673 | GW-674 | GW-676 Sample 5004 |
| PHYS(F) | Total Organic Carbon | µg/L | ND | ND | ND | 2,000.00 | 2,000.00 J | ---- | 9,000.00 | 8,000.00 | 2,000.00 |
| PHYS(F) | Total Organic Halides | µg/L | ND | ND | ND | ND | 120.00 J | ---- | ND | ND | 13.00 J |
| PHYS(F) | Total suspended solids | µg/L | ND | ND | ND | ND | 2,000.00 | ---- | ND | 2,000.00 | ND |
| PHYS(F) | Uranium, Total | µg/L | ND | ND | 1.00 | ND | 2.00 | ---- | 1.00 | 2.00 | 2.00 |
| RADS | Alpha activity | pCi/L | 0.28 | 0.28 | 0.39 | ND | 21.08 J | ---- | 91.34 | 46.07 | ND[a] |
| RADS | Beta activity | pCi/L | 7.91 | 3.96 | 3.14 | ND | ND | ---- | 25.61 J | ND | 15.65 |
| RADS | Total Radium | pCi/L | ND | ND | ND | 0.297 J | 1.269 | ---- | ND | 1.161 | 0.540 J |
| RAD(F) | Alpha activity | pCi/L | ND | ND | ND | 27.10 J | 11.45 J | ---- | 1.61 J | ND | ND |
| RAD(F) | Total Radium | pCi/L | ND | ND | ND | 0.19 J | 0.38 J | ---- | 0.35 J | 0.81 J | 0.54 |

ND = not detected; ---- = not analyzed; E,J = estimated value; S = estimated by method of standard additions; W = AA postdigestion spike not within control limits; B = Below the required detection limit, but at or above the instrument detection limit.

[a]A duplicate sample of GW-676 exceeded the gross alpha MCL.

## Table 3.17. Phase 2 groundwater results at Chestnut Ridge OU 2

| Analysis | Analyte | Units | Sluice Channel Area monitoring wells | | | Upper McCoy Branch area monitoring wells | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | GW-512 | GW-513 | GW-514 | GW-321 | GW-672 | GW-673 | GW-674 | GW-676 |
| METAL | Aluminum | µg/L | 4,230.00 | 1,290.00 | 45.10 | ND | ND | 45.90 | ND | ND |
| METAL | Arsenic | µg/L | 5.00 J | 3.80 | ND | ND | ND | ND | ND | ND |
| METAL | Barium | µg/L | 33.10 | 9.50 | 8.30 | 53.10 | 55.00 | 186.00 | 159.00 | 79.30 |
| METAL | Beryllium | µg/L | 2.10 | ND | ND | ND | ND | ND | ND | ND |
| METAL | Calcium | µg/L | 29,600.00 | 37,600.00 | 41,000.00 | 26,600.00 | 76,400.00 | 42,600.00 | 49,900.00 | 92,500.00 |
| METAL | Copper | µg/L | 8.80 | 4.20 | ND | ND | ND | ND | ND | ND |
| METAL | Iron | µg/L | 5,480.00 | 1,380.00 | 5,920.00 | ND | 107.00 | 47.30 | ND | 891.00 |
| METAL | Lead | µg/L | 8.60 | 3.10 | ND | ND | ND | 11.70 J | ND | ND |
| METAL | Magnesium | µg/L | 17,200.00 | 21,400.00 | 21,800.00 | 14,400.00 | 23,300.00 | 20,700.00 | 16,100.00 | 17,800.00 |
| METAL | Manganese | µg/L | 167.00 | 21.30 | 47.30 | 1.30 J | 1,360.00 | 2.60 | 331.00 | 680.00 |
| METAL | Potassium | µg/L | 4,430.00 | 1,580.00 | ND | ND | 1,400.00 | ND | 4,010.00 | ND |
| METAL | Selenium | µg/L | ND | ND | ND | ND | ND | ND | 5.80 | ND |
| METAL | Sodium | µg/L | 1,300.00 | 981.00 | 494.00 | 712.00 | 1,880.00 | 1,490.00 | 2,100.00 | 5,610.00 |
| METAL | Vanadium | µg/L | 12.70 | ND | ND | ND | ND | ND | ND | ND |
| METAL | Zinc | µg/L | 114.00 J | 62.70 J | 21.30 | 51.50 J | 4.60 J | 22.80 | 43.10 J | 38.30 |
| METAL(F)* | Aluminum | µg/L | 50.30 | 50.60 | 65.30 | 67.10 | 72.80 | 69.90 | 54.90 | 50.10 |
| METAL(F) | Barium | µg/L | 25.60 | 8.20 | 6.00 | 56.90 | 46.30 | 186.00 | 168.00 | 77.20 |
| METAL(F) | Calcium | µg/L | 32,700.00 | 41,000.00 | 43,300.00 | 28,400.00 | 81,500.00 | 43,500.00 | 53,900.00 | 91,700.00 |
| METAL(F) | Copper | µg/L | ND | ND | ND | ND | 4.30 | ND | ND | ND |
| METAL(F) | Iron | µg/L | ND | ND | ND | ND | ND | ND | ND | 521.00 |
| METAL(F) | Magnesium | µg/L | 18,700.00 | 23,700.00 | 23,800.00 | 15,700.00 | 24,700.00 | 21,900.00 | 17,000.00 | 18,000.00 |
| METAL(F) | Manganese | µg/L | 1.80 | ND | 38.00 | 3.80 | 925.00 | 2.20 | 423.00 | 675.00 |
| METAL(F) | Potassium | µg/L | 4,160.00 | ND | ND | ND | 1,640.00 | 2,190.00 | 4,220.00 | 1,530.00 |
| METAL(F) | Selenium | µg/L | ND | ND | ND | ND | ND | ND | 7.10 J | ND |
| METAL(F) | Sodium | µg/L | 1,130.00 | 775.00 | 793.00 | 891.00 | 2,120.00 | 1,730.00 | 2,590.00 | 5,820.00 |
| METAL(F) | Zinc | µg/L | 5.70 | ND | ND | 29.10 | ND | 8.10 | 23.10 | ND |
| RAD | Alpha | pCi/L | 9.10 | ND | ND | 7.20 J | ND | ND | ND | ND |
| RAD | Beta | pCi/L | 12.00 | ND | ND | 6.80 | ND | 4.90 | ND | ND |
| RAD | Thorium-228 | pCi/L | 0.320 | ND | ND | ND | ND | ND | ND | ND |
| RAD | Thorium-232 | pCi/L | 0.220 | ND | ND | ND | ND | ND | ND | ND |
| RAD | Uranium-234 | pCi/L | 0.220 | 0.230 | 0.200 | 0.320 | 0.440 | 0.980 | 0.220 | 0.350 |
| RAD | Uranium-235 | pCi/L | ND | ND | ND | ND | ND | 0.054 | ND | ND |
| RAD | Uranium-238 | pCi/L | 0.170 | 0.081 | 0.160 | ND | 0.320 | 0.440 | 0.110 | 0.190 |
| SVOA | bis(2-Ethylhexyl)Phthalate | µg/L | 24.00 | 2.00 J | 2.00 J | ND | 2.00 J | ND | 1.00 J | ND |

**Table 3.17 (continued)**

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| SVOA | Diethylphthalate | µg/L | 2.00 J | ND | ND | ND | ND | ND | ND | ND |
| SVOA | Di-n-Butylphthalate | µg/L | 3.00 J | ND | ND | ND | 1.00 J | ND | ND | ND |
| SVOA | Phenol | µg/L | 5.00 J | 2.00 J | ND | ND | ND | ND | ND | ND |
| VOA | Trichloroethene | µg/L | ND | 2.00 J | ND | ND | ND | ND | ND | ND |

ND = not detected; ---- = not analyzed; J = estimated value; *(F) = filtered sample.

Table 3.18. Phase 1 analytes for soil at Chestnut Ridge OU 2

| Inorganics | |
|---|---|
| Aluminum | Magnesium |
| Antimony | Manganese |
| Arsenic | Mercury |
| Barium | Nickel |
| Beryllium | Potassium |
| Cadmium | Selenium |
| Calcium | Silver |
| Chromium | Sodium |
| Cobalt | Thallium |
| Copper | Vanadium |
| Iron | Zinc |
| Lead | |
| **Physical properties** | |
| Oil and grease (%) | Total uranium (fluorometric) |
| **Radiological parameters** | |
| Gross alpha | $^{234}$Th |
| Gross beta | $^{235}$U |
| $^{137}$Cs | Uranium (fluorometric) |
| $^{228}$Th | |

Table 3.19. Phase 2 analytes for soil at Chestnut Ridge OU 2

| VOCs (CLP) | |
|---|---|
| 1,1,1-Trichloroethane | Carbon tetrachloride |
| 1,1,2,2-Tetrachloroethane | Chlorobenzene |
| 1,1,2-Trichloroethane | Chloroethane |
| 1,1-Dichloroethane | Chloroform |
| 1,2-Dichloroethane | Chloromethane |
| 1,1-Dichloroethene | Cis-1,3-dichloropropene |
| 1,2-Dichloroethene (total) | Dibromochloromethane |
| 1,2-Dichloropropane | Ethylbenzene |
| 2-Butanone | Methylene chloride |
| 2-Hexanone | Styrene |
| 4-Methyl-2-pentanone | Tetrachloroethene |
| Acetone | Toluene |
| Benzene | Trans-1,3-dichloropropene |
| Bromodichloromethane | Trichloroethene |
| Bromoform | Vinyl chloride |
| Bromomethane | Xylene (total) |
| Carbon disulfide | |
| **Base neutral acid compounds (CLP)** | |
| 1,2,4-Trichlorobenzene | 4-Nitroaniline |
| 1,2-Dichlorobenzene | 4-Nitrophenol |
| 1,3-Dichlorobenzene | Acenaphthene |
| 1,4-Dichlorobenzene | Acenaphthylene |
| 2,2-Oxybis(1-chloropropane) | Anthracene |
| 2,4,5-Trichlorophenol | Benzo(a)anthracene |
| 2,4,6-Trichlorophenol | Benzo(a)pyrene |
| 2,4-Dichlorophenol | Benzo(b)fluoranthene |
| 2,4-Dimethylphenol | Benzo(g,h,i)perylene |
| 2,4-Dinitrophenol | Benzo(k)fluoranthene |
| 2,4-Dinitrotoluene | Bis(2-chloroethoxy)methane |
| 2,6-Dinitrotoluene | Bis(2-chloroethyl)ether |
| 2-Chloronaphthalene | Bis(2-ethylhexyl)phthalate |
| 2-Chlorophenol | Butyl benzyl phthalate |
| 2-Methylnaphthalene | Carbazole |
| 2-Methylphenol | Chrysene |
| 2-Nitroaniline | Di-n-butyl phthalate |
| 2-Nitrophenol | Di-n-octyl phthalate |
| 3-Nitroaniline | Dibenzo(a,h)anthracene |
| 3,3-Dichlorobenzidine | Dibenzofuran |
| 4,6-Dinitro-2-methylphenol | Diethyl phthalate |
| 4-Bromophenyl phenyl ether | Dimethyl phthalate |
| 4-Chloro-3-methylphenol | Fluoranthene |
| 4-Chloraniline | Fluorene |
| 4-Chlorophenyl phenyl ether | Hexachlorobenzene |
| 4-Methylphenol | Hexachlorobutadiene |

Table 3.19 (continued)

| Base neutral acid compounds (CLP—continued) | |
|---|---|
| Hexachlorocyclopentadiene | Naphthalene |
| Hexachloroethane | Nitrobenzene |
| Indeno(1,2,3-cd)pyrene | Pentachlorophenol |
| Isophorone | Phenanthrene |
| N-nitroso-di-n-propylamine | Phenol |
| N-nitrosodiphenylamine | Pyrene |
| **Inorganics (TAL—total)** | |
| Aluminum | Magnesium |
| Antimony | Manganese |
| Arsenic | Mercury |
| Barium | Nickel |
| Beryllium | Potassium |
| Cadmium | Selenium |
| Calcium | Silver |
| Chromium | Sodium |
| Cobalt | Thallium |
| Copper | Vanadium |
| Iron | Zinc |
| Lead | |
| **Physical properties** | |
| Particle size distribution | |
| **Radiological parameters** | |
| Gross alpha | Isotopic thorium |
| Gross beta | Isotopic uranium |

**Table 3.20. Phase 1 results from soil beneath the ash at Chestnut Ridge OU 2**

| Analysis | Analyte | Units | MCB-A1-S1 | MCB-A1-S2 | MCB-A2-S1 | MCB-A3-S1 | MCB-A3-S2 | MCB-A3-A7-S1 | MCB-A3-A7-S2 |
|---|---|---|---|---|---|---|---|---|---|
| METAL | Aluminum | mg/kg | 9,030.00 | 9,550.00 | 15,300.00 | 5,830.00 | 7,480.00 | 6,670.00 | 8,540.00 |
| METAL | Arsenic | mg/kg | 2.20 BN*J | 2.20 BN*J | 24.30 N*J | 28.30 N*J | 22.00 N*J | 43.60 BN*J | 41.70 BN*J |
| METAL | Barium | mg/kg | 89.00 | 118.00 | 109.00 | 71.40 | 77.10 | 96.00 | 76.60 |
| METAL | Beryllium | mg/kg | 0.60 B | 0.71 B | 1.20 B | 0.49 B | 0.56 B | 0.56 B | 0.62 B |
| METAL | Cadmium | mg/kg | 2.10 | 0.88 B | 1.50 | 1.20 | 1.70 | 1.80 | 1.20 |
| METAL | Calcium | mg/kg | 2,230.00 | 686.00 B | 1,880.00 | 858.00 B | 906.00 B | 1,030.00 B | 963.00 B |
| METAL | Chromium | mg/kg | 25.00 | 17.40 | 12.00 | 5.60 | 10.10 | 6.80 | 7.40 |
| METAL | Cobalt | mg/kg | 18.00 | 25.00 | 27.50 | 13.00 | 11.10 B | 17.00 | 18.90 |
| METAL | Copper | mg/kg | 6.10 | 5.50 B | 24.10 | 11.80 | 11.50 | 15.10 | 11.90 |
| METAL | Iron | mg/kg | 19,200.00 | 16,900.00 | 19,800.00 | 20,100.00 | 19,600.00 | 24,900.00 | 16,700.00 |
| METAL | Lead | mg/kg | 33.40 W* | 36.20 W* | 26.00 W* | 12.50 W* | 6.50 W* | 11.40 W* | 11.90 W* |
| METAL | Magnesium | mg/kg | 597.00 B | 525.00 B | 970.00 B | 473.00 B | 614.00 B | 653.00 B | 694.00 B |
| METAL | Manganese | mg/kg | 2,240.00 | 2,600.00 | 1,830.00 | 965.00 | 702.00 | 820.00 | 1,020.00 |
| METAL | Mercury | mg/kg | 0.10 | 0.12 | 0.11 | 0.05 | 0.06 | 0.07 | 0.07 |
| METAL | Nickel | mg/kg | 6.70 B | 8.30 B | 34.20 | 22.30 | 15.50 | 21.90 | 18.40 |
| METAL | Phosphorus | mg/kg | 326.00 J | 211.00 J | 272.00 J | 129.00 J | 145.00 J | 143.00 J | 126.00 J |
| METAL | Potassium | mg/kg | 544.00 B | 651.00 B | 1,960.00 | 911.00 B | 1,310.00 | 1,070.00 B | 1,370.00 |
| METAL | Selenium | mg/kg | ND | ND | 2.20 BENJ | 0.47 BWNJ | ND | 0.89 BENJ | 0.50 BWNJ |
| METAL | Sodium | mg/kg | 49.60 BJ | 88.80 BJ | 141.00 BJ | 141.00 BJ | 135.00 BJ | 86.00 BJ | 106.00 BJ |
| METAL | Vanadium | mg/kg | 39.40 | 34.30 | 50.00 | 22.10 | 33.40 | 26.00 | 26.00 |
| METAL | Zinc | mg/kg | 34.40 | 33.90 | 102.00 | 56.10 | 43.50 | 59.10 | 55.30 |
| PHYS | Uranium, Total | mg/kg | 4.00 | 3.00 | 5.00 | 4.0 | 3.00 | 3.00 | 3.00 |
| RAD | Alpha activity | pCi/kg | 620.00 J | 460.00 J | 620.00 J | 2,820.00 | 3,440.00 | ND | 6,300.00 |
| RAD | Beta activity | pCi/kg | ND | ND | 207.00 | 2,460.00 J | 2,910.00 J | ND | 1,680.00 J |
| RAD | Cesium-137 | pCi/kg | 207.00 | 330.00 | ND | 64.50 J | 85.50 J | 44.70 J | 103.00 J |
| RAD | Thorium-228 | pCi/kg | ND | ND | ND | ND | ND | ND | ND |
| RAD | Thorium-234 | pCi/kg | ND | ND | ND | ND | ND | ND | ND |
| RAD | Uranium-235 | WT% | ND | ND | ND | ND | ND | ND | ND |

**Table 3.20 (continued)**

| Analysis | Analyte | Units | MCB-A4-S1 | MCB-A4-S2 | MCB-A5-S1 | MCB-A5-S2 | MCB-A6-S1 | MCB-A6-S2 |
|---|---|---|---|---|---|---|---|---|
| METAL | Aluminum | mg/kg | 11,200.00 | 9,500.00 | 6,080.00 | 18,600.00 | 9,210.00 | 9,460.00 |
| METAL | Arsenic | mg/kg | 15.30 N*J | 10.60 BN*J | 10.30 BN*J | 11.50 BN*J | 3.20 BWN*J | 3.00 BN*J |
| METAL | Barium | mg/kg | 48.10 | 39.30 B | 71.00 B | 196.00 | 79.50 B | 71.60 |
| METAL | Beryllium | mg/kg | 0.72 B | 0.59 B | ND | 1.40 B | ND | 0.69 B |
| METAL | Cadmium | mg/kg | 1.20 | 1.20 B | 22.80 | 2.30 | 7.00 B | 1.50 |
| METAL | Calcium | mg/kg | 620.00 B | 518.00 B | 12,400.00 B | 17,800.00 | 250,000.00 | 220,000.00 |
| METAL | Chromium | mg/kg | 8.60 | 8.60 | 50.60 | 21.50 | ND | 8.90 |
| METAL | Cobalt | mg/kg | 18.80 | 21.40 | 29.50 B | 10.20 B | ND | 8.60 B |
| METAL | Copper | mg/kg | 11.20 | 10.70 | 70.20 B | 4.90 B | ND | 23.10 |
| METAL | Iron | mg/kg | 14,500.00 | 13,600.00 | 372,000.00 | 21,300.00 | 13,700.00 | 13,000.00 |
| METAL | Lead | mg/kg | 8.60 W* | 15.10 W* | 14.50 W* | 13.80 W* | 21.70 W* | 27.10 W* |
| METAL | Magnesium | mg/kg | 977.00 B | 732.00 B | 3,740.00 B | 3,320.00 | 11,200.00 B | 7,880.00 |
| METAL | Manganese | mg/kg | 1,030.00 | 1,050.00 | 2,380.00 | 248.00 | 425.00 | 320.00 |
| METAL | Mercury | mg/kg | 0.07 | 0.07 | 0.09 | 0.09 | 0.05 | 0.05 |
| METAL | Nickel | mg/kg | 22.10 | 18.00 | 40.40 B | 15.30 | ND | 9.90 |
| METAL | Phosphorus | mg/kg | 97.30 J | ND | ND | 269.00 J | ND | 227.00 J |
| METAL | Potassium | mg/kg | 1,780.00 | 1,320.00 | ND | 2,190.00 | 2,140.00 B | 1,990.00 |
| METAL | Selenium | mg/kg | ND | ND | 0.30 BWNJ | ND | ND | ND |
| METAL | Sodium | mg/kg | 95.90 BJ | 45.40 BJ | 626.00 B | 125.00 BJ | 177.00 B | 189.00 B |
| METAL | Vanadium | mg/kg | 27.70 | 25.20 | 94.00 B | 29.40 | 12.70 B | 16.20 |
| METAL | Zinc | mg/kg | 90.20 | 79.60 | 81.70 | 46.60 | 84.50 | 79.30 |
| PHYS | Uranium, Total | mg/kg | 3.00 | 3.00 | 3.00 | 7.00 | ND | 2.00 |
| RAD | Alpha activity | pCi/kg | ND | 5,990.00 | 1,210.00 J | 3,380.00 | 2.00 | 1,600.00 |
| RAD | Beta activity | pCi/kg | ND | 3,240.00 J | ND | 3,130.00 J | 600.00 J | 4,140.00 |
| RAD | Cesium-137 | pCi/kg | ND | 31.50 J | 268.00 | 59.00 J | 110.00 J | ND |
| RAD | Thorium-228 | pCi/kg | ND | ND | ND | ND | ND | ND |
| RAD | Thorium-234 | pCi/kg | ND | 5,150.00 | ND | ND | 5,100.00 | ND |
| RAD | Uranium-235 | WT% | ND | ND | ND | ND | ND | ND |

J,E = estimated value; N = spiked sample recovery not within control limits; * duplicate analysis not within control limits; W = furnace atomic absorption (AA) postdigestion spike not within control limits, while sample absorbance is less than spike absorption; B = below the required detection limit, but at or above the instrument detection limit.

**Table 3.21. Phase 2 results from soil beneath the ash at Chestnut Ridge OU 2**

| Analysis | Analyte | Units | Soil from FCAP ash transects | | | Soil from Sluice Channel Area ash transects | | |
|---|---|---|---|---|---|---|---|---|
| | | | AT-001-01 | AT-002-01 | AT-003-01 | SC-002-02 | SC-004-02 | SC-007-02 |
| METAL | Aluminum | mg/kg | 9,250.00 | 14,000.00 | 13,200.00 | 7,460.00 | 5,670.00 | 6,060.00 |
| METAL | Arsenic | mg/kg | 32.90 J | 15.10 | 43.50 | 14.80 J | 9.40 J | 50.00 |
| METAL | Barium | mg/kg | 126.00 | 52.00 | 46.30 | 112.00 | 80.80 | 61.90 |
| METAL | Beryllium | mg/kg | 0.87 | 0.54 | 1.50 | ND | ND | ND |
| METAL | Calcium | mg/kg | 2,230.00 | 1,270.00 | 657.00 | 628.00 | 517.00 | 762.00 |
| METAL | Chromium | mg/kg | 9.60 | 16.80 | 13.30 | 6.90 | 15.10 | 16.60 |
| METAL | Cobalt | mg/kg | 11.60 | 21.90 | 52.20 | 9.70 | 7.40 | 12.40 |
| METAL | Copper | mg/kg | 18.10 | 12.60 | 16.40 | ND | 6.10 | 6.10 |
| METAL | Iron | mg/kg | 16,700.00 | 18,000.00 | 17,000.00 | 8,950.00 J | 14,000.00 J | 16,400.00 J |
| METAL | Lead | mg/kg | 10.80 J | 17.10 | 25.00 | 24.70 J | 17.50 J | 15.30 J |
| METAL | Magnesium | mg/kg | 860.00 | 937.00 | 816.00 | 213.00 | 125.00 | 222.00 |
| METAL | Manganese | mg/kg | 216.00 | 400.00 | 1,020.00 | 1,920.00 J | 1,430.00 J | 674.00 J |
| METAL | Mercury | mg/kg | ND | ND | ND | ND | 0.14 | ND |
| METAL | Nickel | mg/kg | 9.20 | 10.80 | 27.50 | 8.00 | ND | 8.10 |
| METAL | Potassium | mg/kg | 929.00 | 1,240.00 | 1,130.00 | ND | ND | ND |
| METAL | Selenium | mg/kg | 2.70 J | 0.88 J | 2.60 J | ND | ND | 0.97 J |
| METAL | Sodium | mg/kg | 300.00 | ND | ND | 33.60 | ND | 34.90 |
| METAL | Thallium | mg/kg | ND | ND | ND | ND | ND | ND |
| METAL | Vanadium | mg/kg | 27.90 | 36.40 | 28.80 | 17.10 | 28.50 | 32.40 |
| METAL | Zinc | mg/kg | 47.40 | 44.20 | 119.00 | 13.80 | 19.60 | 19.40 |
| RAD | Thorium-228 | pCi/kg | 0.67 | 0.78 | 0.77 | 0.77 | 0.64 | 0.42 |
| RAD | Thorium-230 | pCi/kg | ND | ND | ND | ND | ND | 1.40 |
| RAD | Thorium-232 | pCi/kg | 0.90 | 0.75 | 0.71 | 0.92 | 0.78 | 0.46 |
| RAD | Uranium-234 | pCi/kg | 1.50 | 1.00 | 1.20 | 1.00 | 0.84 | 1.10 |
| RAD | Uranium-235 | pCi/kg | ND | 0.13 | ND | ND | ND | ND |
| RAD | Uranium-238 | pCi/kg | 1.20 | 1.60 | 1.20 | 0.92 | 1.20 | 0.72 |
| SVOA | bis(2-Ethylhexyl)phthalate | µg/kg | ND | 49.00 J | ND | ND | ND | ND |
| SVOA | Butylbenzylphthalate | µg/kg | 50.00 J | 53.00 J | ND | ND | 48.00 J | ND |
| SVOA | 2-Methylnaphthalene | µg/kg | ND | ND | ND | ND | ND | ND |
| SVOA | 4-Nitrophenol | µg/kg | ND | ND | ND | ND | ND | ND |

ND = not detected; ---- = not analyzed; J = estimated value.

**Table 3.22. Summary of Phase 2 soils analytical data at Chestnut Ridge OU 2**

| Analyte | Soil beneath the ash | | | | | | Reference Soils | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Minimum | Maximum | Mean | No. of samples | No. of detections | Detected[a] frequency | Minimum | Maximum | Mean | No. of samples | No. of detections | Detected frequency[a] |
| Aluminum | 5,670.00 | 14,000.00 | 9,273.33 | 6 | 6 | 1.00 | 4,740.00 | 10,400.00 | 6,923.00 | 3 | 3 | 1.00 |
| Arsenic | 9.40 | 50.00 | 27.62 | 6 | 6 | 1.00 | 2.70 | 7.10 | 4.73 | 3 | 3 | 1.00 |
| Barium | 46.30 | 126.00 | 79.83 | 6 | 6 | 1.00 | 57.00 | 95.40 | 74.03 | 3 | 3 | 1.00 |
| Beryllium | ND | 1.50 | 0.73 | 6 | 3 | 0.50 | ND | 0.80 | 0.57 | 3 | 2 | 0.67 |
| Calcium | 517.00 | 2,230.00 | 1,010.67 | 6 | 6 | 1.00 | 1,550.00 | 2,070.00 | 1,826.67 | 3 | 3 | 1.00 |
| Chromium | 6.90 | 16.60 | 13.05 | 6 | 6 | 1.00 | 3.70 | 5.60 | 4.67 | 3 | 3 | 1.00 |
| Cobalt | 7.40 | 52.20 | 19.20 | 6 | 6 | 1.00 | 2.80 | 7.80 | 4.70 | 3 | 3 | 1.00 |
| Copper | ND | 18.10 | 11.86 | 6 | 5 | 0.83 | ND | 6.80 | ---- | 3 | 1 | 0.33 |
| Iron | 8,950.00 | 18,000.00 | 15,175.00 | 6 | 6 | 1.00 | 3,650.00 | 5,730.00 | 4,863.33 | 3 | 3 | 1.00 |
| Lead | 10.80 | 25.00 | 18.40 | 6 | 6 | 1.00 | 9.80 | 17.10 | 14.30 | 3 | 3 | 1.00 |
| Magnesium | 1.25 | 937.00 | 528.83 | 6 | 6 | 1.00 | 370.00 | 624.00 | 464.00 | 3 | 3 | 1.00 |
| Manganese | 216.00 | 1,920.00 | 943.33 | 6 | 6 | 1.00 | 696.00 | 1,500.00 | 992.33 | 3 | 3 | 1.00 |
| Mercury | ND | 0.14 | ---- | 6 | 1 | 0.17 | 0.21 | 0.33 | 0.28 | 3 | 3 | 1.00 |
| Nickel | ND | 27.50 | 12.72 | 6 | 5 | 0.83 | ND | 8.50 | 7.35 | 3 | 2 | 0.67 |
| Potassium | ND | 1,240.00 | 1,099.67 | 6 | 3 | 0.50 | ND | 793.00 | ---- | 3 | 1 | 0.33 |
| Selenium | ND | 2.70 | 1.79 | 6 | 4 | 0.67 | ND | 1.20 | ---- | 3 | 1 | 0.33 |
| Sodium | ND | 300.00 | 122.83 | 6 | 3 | 0.50 | ND | 250.00 | 183.00 | 3 | 2 | 0.67 |
| Thallium | ND | ND | ND | 6 | 0 | 0.00 | ND | ND | ND | 3 | 0 | 0.00 |
| Vanadium | 17.10 | 36.40 | 23.72 | 6 | 6 | 1.00 | 9.40 | 13.80 | 11.37 | 3 | 3 | 1.00 |
| Zinc | 13.80 | 119.00 | 43.90 | 6 | 6 | 1.00 | 24.80 | 56.00 | 41.50 | 3 | 3 | 1.00 |
| Thorium-228 | 0.42 | 0.78 | 0.68 | 6 | 6 | 1.00 | 0.22 | 0.51 | 0.35 | 3 | 3 | 1.00 |
| Thorium-230 | ND | 1.40 | ---- | 6 | 1 | 0.17 | ND | ND | ND | 3 | 0 | 0.00 |
| Thorium-232 | 0.46 | 0.92 | 0.75 | 6 | 6 | 1.00 | 0.19 | 0.52 | 0.34 | 3 | 3 | 1.00 |
| Uranium-234 | 0.84 | 1.50 | 1.11 | 6 | 6 | 1.00 | 3.00 | 3.50 | 3.20 | 3 | 3 | 1.00 |
| Uranium-235 | ND | 0.13 | ---- | 6 | 1 | 0.17 | 0.14 | 0.34 | 0.22 | 3 | 3 | 1.00 |
| Uranium-238 | 0.72 | 1.60 | 1.14 | 6 | 6 | 1.00 | 1.50 | 2.30 | 2.00 | 3 | 3 | 1.00 |
| bis(2-Ethylhexyl)phthalate | ND | 49.00 | ---- | 6 | 1 | 0.17 | ND | ND | ND | 3 | 0 | 0.00 |
| Butylbenzylphthalate | ND | 53.00 | 50.33 | 6 | 3 | 0.50 | ND | 54.00 | 52.00 | 3 | 2 | 0.67 |
| 2-Methylnaphthalene | ND | ND | ND | 6 | 0 | 0.00 | ND | ND | ND | 3 | 0 | 0.00 |
| 4-Nitrophenol | ND | ND | ND | 6 | 0 | 0.00 | ND | 140.00 | ---- | 3 | 1 | 0.33 |

[a]Detected frequency = (detected count)/(total samples).

Table 3.23. Summary of Phase 2 geotechnical data at Chestnut Ridge OU 2

| Sample ID | Percent gravel | Percent sand | Percent silt | Percent clay | TOC percent | Percent moisture |
|---|---|---|---|---|---|---|
| **Sediments** | | | | | | |
| SD-001-01 | 8 | 25 | 45 | 22 | 2.65 | 1.12 |
| SD-002-01 | 0 | 28 | 41 | 31 | 6.68 | 5.26 |
| SD-003-01 | 0 | 21 | 44 | 35 | 3.79 | 6.67 |
| SD-003-02 | 0 | 29 | 33 | 38 | 2.84 | 1.57 |
| SD-003-03 | 32 | 22 | 11 | 35 | 0.55 | 2.73 |
| SD-004-01 | 7 | 45 | 39 | 9 | 7.35 | 1.06 |
| SD-005-01 | 4 | 49 | 27 | 20 | 3.06 | 2.08 |
| SD-006-01 | 42 | 50 | 3 | 5 | 3.12 | 0.97 |
| SD-007-01 | 22 | 67 | 4 | 7 | 4.48 | .89 |
| SD-011-01 | 59 | 36 | 3 | 2 | 3.16 | 1.23 |
| **Surface soils and ash** | | | | | | |
| SO-001-01 | 0 | 26 | 50 | 24 | ---- | 2.60 |
| SO-002-01 | 0 | 27 | 50 | 23 | ---- | 3.21 |
| SO-003-01 | 0 | 28 | 48 | 24 | ---- | 2.50 |
| SO-004-01 | 0 | 27 | 64 | 9 | ---- | 2.76 |
| SO-005-01 | 0 | 24 | 52 | 24 | ---- | 2.57 |
| SO-006-01 | 0 | 43 | 46 | 11 | ---- | 2.03 |
| SO-010-01 | 23 | 33 | 32 | 12 | ---- | 1.85 |
| SO-010-11 | 21 | 37 | 26 | 16 | ---- | 2.13 |
| SO-011-01 | 29 | 35 | 26 | 10 | ---- | 3.16 |
| SO-012-01 | 15 | 35 | 37 | 13 | ---- | 1.93 |

## Table 3.24. Phase 2 surface ash results at Chestnut Ridge OU 2

| Analysis | Analyte | Units | FCAP ash transects (AT), sycamores | | | FCAP ash transects, willows | | | Background surface soils | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | AT #1 SO-001-01 | AT #2 SO-002-01 | AT #3 SO-003-01 | AT #1 SO-004-01 | AT #2 SO-005-01 | AT #3 SO-006-01 | SO-010-01 | SO-011-01 | SO-012-01 |
| METAL | Aluminum | mg/kg | 28,800.00 | 16,900.00 | 22,600.00 | 23,000.00 | 32,900.00 | 15,000.00 | 5,630.00 | 10,400.00 | 4,740.00 |
| METAL | Arsenic | mg/kg | 140.00 | 153.00 | 158.00 | 193.00 | 161.00 | 151.00 J | 2.70 J | 4.40 J | 7.10 |
| METAL | Barium | mg/kg | 630.00 | 406.00 | 523.00 | 522.00 | 497.00 | 286.00 | 69.70 | 95.40 | 57.00 |
| METAL | Beryllium | mg/kg | 5.90 | 4.40 | 5.80 | 5.60 | 7.50 | 3.70 | ND | 0.80 | 0.33 |
| METAL | Calcium | mg/kg | 6,940.00 | 5,450.00 | 6,510.00 | 6,680.00 | 6,750.00 | 3,230.00 | 1,860.00 | 2,070.00 | 1,550.00 |
| METAL | Chromium | mg/kg | 35.30 | 18.20 | 24.20 | 26.20 | 42.10 | 17.60 | 3.70 | 5.60 | 4.70 |
| METAL | Cobalt | mg/kg | 26.10 | 14.80 | 22.10 | 23.50 | 29.80 | 14.10 | 2.80 | 7.80 | 3.50 |
| METAL | Copper | mg/kg | 90.20 | 58.00 | 79.40 | 92.70 | 101.00 | 52.60 | ND | ND | 6.80 |
| METAL | Iron | mg/kg | 37,500.00 | 15,100.00 | 19,400.00 | 30,100.00 | 41,500.00 | 23,200.00 | 3,650.00 | 5,730.00 | 5,210.00 |
| METAL | Lead | mg/kg | 20.90 | 16.60 | 22.40 | 25.80 | 27.30 | 21.40 | 9.80 J | 17.10 J | 16.00 |
| METAL | Magnesium | mg/kg | 2,120.00 | 1,320.00 | 1,740.00 | 1,950.00 | 2,400.00 | 1,080.00 | 398.00 | 624.00 | 370.00 |
| METAL | Manganese | mg/kg | 161.00 | 47.50 | 57.40 | 140.00 | 91.00 | 34.10 | 696.00 | 1,500.00 | 781.00 |
| METAL | Mercury | mg/kg | 0.94 | 1.00 | 1.30 | 0.76 | 0.84 | 0.58 | 0.29 | 0.33 | 0.21 |
| METAL | Nickel | mg/kg | 50.30 | 27.60 | 39.20 | 41.30 | 56.00 | 28.50 | ND | 8.50 | 6.20 |
| METAL | Potassium | mg/kg | 5,680.00 | 3,270.00 | 4,660.00 | 4,320.00 | 6,210.00 | 3,070.00 | ND | 793.00 | ND |
| METAL | Selenium | mg/kg | 6.50 J | 4.60 J | 13.10 J | 12.30 J | 30.80 J | 6.30 J | ND | 1.20 J | ND |
| METAL | Sodium | mg/kg | 744.00 | 555.00 | 578.00 | 843.00 | 1,080.00 | 546.00 | 161.00 | 205.00 | ND |
| METAL | Thallium | mg/kg | 2.10 | 2.50 | 3.20 | 2.40 | ND | 1.70 J | ND | ND | ND |
| METAL | Vanadium | mg/kg | 112.00 | 70.60 | 87.40 | 102.00 | 123.00 | 62.90 | 9.40 | 13.80 | 10.90 |
| METAL | Zinc | mg/kg | 64.90 | 46.50 | 60.90 | 69.90 | 83.10 | 42.60 | 24.80 | 56.00 | 43.10 |
| RAD | Thorium-228 | pCi/g | 2.00 | 1.40 | 1.40 | 2.20 | 1.20 | 1.50 | 0.51 | 0.32 | 0.22 |
| RAD | Thorium-230 | pCi/g | 4.20 | 2.90 | 2.70 | 3.60 | 2.60 | 3.10 | ND | ND | ND |
| RAD | Thorium-232 | pCi/g | 2.30 | 1.60 | 1.80 | 1.40 | 1.20 | 1.60 | 0.19 | 0.52 | 0.32 |
| RAD | Uranium-234 | pCi/g | 4.70 | 3.20 | 3.20 | 2.60 | 2.50 | 2.20 | 3.50 | 3.00 | 3.10 |
| RAD | Uranium-235 | pCi/g | 0.34 | 0.15 | 0.09 | 0.21 | 0.09 | 0.16 | 0.14 | 0.34 | 0.19 |
| RAD | Uranium-238 | pCi/g | 4.50 | 3.20 | 2.90 | 2.50 | 2.30 | 2.60 | 1.50 | 2.30 | 2.20 |
| SVOA | bis(2-Ethythexyl)Phthalate | µg/kg | ND | ND | ND | ND | 270.00 J | ND | ND | ND | ND |
| SVOA | Butylbenzylphthalate | µg/kg | ND | 83.00 J | 71.00 J | ND | ND | ND | 50.00 J | ND | 54.00 J |
| SVOA | 2-Methylnaphthalane | µg/kg | ND | ND | ND | 93.00 J | ND | ND | ND | ND | ND |
| SVOA | 4-Nitrophenol | µg/kg | ND | ND | ND | ND | ND | ND | ND | 140.00 J | ND |

ND = not detected; ---- = not analyzed; J = estimated value.

**Table 3.25. Water toxicity tests conducted**

| Site tested | Date when test was initiated | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | July 30 1990 | Apr. 11 1991 | July 11 1991 | Oct. 24 1991 | Jan. 31 1992 | Apr. 2 1992 | July 9 1992 | Sept. 17 1992 | Jan. 14 1993 | Mar. 25 1993 | Apr. 19 1993 | July 29 1993 |
| MCK 1.56 | C[a], F[b] | C | C, F | C, F | C, F | C, F | C, F | C | C, F,SF[g] | C, F | SF[c] | C, F, SF[h] |
| MCK 2.03 | C, F | C | C, F | C, F | C, F | C, F | C, F | C | C, F, SF[g] | C, F | SF | C, F, SF[h] |
| MCK 2.53 | –[f] | – | – | – | – | – | – | – | – | C, F | – | – |
| MCK 2.65 | – | C | – | – | – | – | – | – | – | – | – | – |
| FCAP | – | – | – | – | – | – | EF[d,i] | – | – | – | – | EL[e,j] |
| Sluice Channel Area | – | – | – | – | – | – | – | – | – | – | – | EL[e,j] |

[a]C = *Ceriodaphnia dubia* 7-day survival and reproduction test;
[b]F = fathead minnow larvae 7-day survival and growth test;
[c]SF = snail 4-day feeding-rate test;
[d]EF = in situ 14-day *Eisenia foetida* survival and growth test;
[e]EL = laboratory 14-day *E. foetida* survival and growth test.
[f]"–" = not tested.
[g]Snail feeding-rate test started on January 18, 1993.
[h]Snail feeding-rate test started on July 27, 1993.
[i]In situ test with earthworms started on July 1, 1993.
[j]Laboratory test with earthworms started on August 1, 1993.

Table 3.26. Results of fathead minnow and *Ceriodaphnia* tests

| Test date | Water source | Conc. (%) | *Pimephales promelas* | | *Ceriodaphnia dubia* | |
|---|---|---|---|---|---|---|
| | | | Survival (%) (mean ± SD) | Growth (mg/fish) (mean ± SD) | Survival (%) | Reproduction (mean ± SD) |
| July 30, 1990 | Control | 100 | 100.0 ± 0.0 | 0.53 ± 0.10 | 90 | 24.0 ± 3.4 |
| | MCK 1.56 | 100 | 100.0 ± 0.0 | 0.64 ± 0.04 | 90 | 18.3 ± 10.8 |
| | | 80 | 97.5 ± 5.0 | 0.69 ± 0.11 | –[a] | – |
| | MCK 2.03 | 100 | 100.0 ± 0.0 | 0.66 ± 0.09 | 90 | 11.1 ± 8.1 |
| | | 80 | – | – | 100 | 11.5 ± 9.7 |
| | | 60 | – | – | 100 | 21.0 ± 9.0 |
| | | 40 | – | – | 70 | 15.6 ± 8.6 |
| April 11, 1991 | Control | 100 | – | – | 90 | 30.1 ± 7.7 |
| | MCK 1.56 | 100 | – | – | 100 | 31.2 ± 6.4 |
| | | 80 | – | – | 100 | 33.0 ± 6.2 |
| | | 60 | – | – | 100 | 29.1 ± 6.4 |
| | | 40 | – | – | 100 | 29.2 ± 5.9 |
| | MCK 2.03 | 100 | – | – | 100 | 21.8 ± 2.8 |
| | | 80 | – | – | 100 | 19.5 ± 3.5 |
| | | 60 | – | – | 100 | 21.1 ± 2.1 |
| | | 40 | – | – | 100 | 20.3 ± 0.8 |
| | MCK 2.65 | 100 | – | – | 100 | 22.3 ± 3.5 |
| | | 80 | – | – | 100 | 25.3 ± 4.7 |
| | | 60 | – | – | 100 | 26.6 ± 4.6 |
| | | 40 | – | – | 90 | 28.8 ± 6.0 |
| July 11, 1991 | Control | 100 | 100.0 ± 0.0 | 0.73 ± 0.06 | 100 | 19.1 ± 6.0 |
| | MCK 1.56 | 100 | 95.0 ± 5.8 | 0.72 ± 0.12 | 100 | 23.8 ± 3.9 |
| | | 80 | 97.5 ± 5.0 | 0.76 ± 0.05 | 100 | 21.7 ± 6.2 |
| | | 60 | 82.5 ± 22.2 | 0.70 ± 0.06 | 100 | 23.2 ± 6.4 |
| | | 40 | 97.5 ± 5.0 | 0.68 ± 0.11 | 100 | 24.6 ± 5.3 |
| | MCK 2.03 | 60 | 97.5 ± 5.0 | 0.67 ± 0.11 | 70 | 20.0 ± 6.1 |
| | | 40 | 100.0 ± 0.0 | 0.68 ± 0.07 | 90 | 19.3 ± 6.8 |

**Table 3.26 (continued)**

| Test date | Water source | Conc. (%) | *Pimephales promelas* | | *Ceriodaphnia dubia* | |
|---|---|---|---|---|---|---|
| | | | Survival (%) (mean ± SD) | Growth (mg/fish) (mean ± SD) | Survival (%) | Reproduction (mean ± SD) |
| October 24, 1991 | Control | 100 | 95.0 ± 5.8 | 0.37 ± 0.01 | 100 | 26.8 ± 3.3 |
| | MCK 1.56 | 100 | 90.0 ± 14.1 | 0.50 ± 0.04 | 100 | 30.3 ± 4.0 |
| | | 80 | 70.0 ± 21.6 | 0.53 ± 0.04 | 100 | 20.5 ± 8.2 |
| | | 60 | 57.5 ± 20.6 | 0.67 ± 0.13 | 100 | 25.5 ± 3.6 |
| | | 40 | 80.0 ± 33.7 | 0.52 ± 0.06 | 100 | 27.4 ± 3.7 |
| | MCK 2.03 | 100 | 87.5 ± 12.6 | 0.55 ± 0.05 | 100 | 27.2 ± 2.9 |
| | | 80 | 90.0 ± 8.2 | 0.50 ± 0.03 | 100 | 26.5 ± 2.5 |
| | | 60 | 95.0 ± 5.8 | 0.52 ± 0.07 | 90 | 25.9 ± 2.5 |
| | | 40 | 90.0 ± 8.2 | 0.51 ± 0.03 | 100 | 26.0 ± 3.7 |
| January 31, 1992 | Control | 100 | 90.0 ± 14.1 | 0.51 ± 0.06 | 70 | 26.9 ± 7.2 |
| | MCK 1.56 | 100 | 87.5 ± 12.6 | 0.46 ± 0.12 | 100 | 30.4 ± 4.0 |
| | | 80 | 100.0 ± 0.0 | 0.50 ± 0.04 | 90 | 32.6 ± 4.3 |
| | | 60 | 87.5 ± 12.6 | 0.46 ± 0.05 | 80 | 29.7 ± 4.4 |
| | | 40 | – | – | 100 | 27.0 ± 4.7 |
| | MCK 2.03 | 100 | 75.0 ± 43.6 | 0.49 ± 0.11 | 70 | 23.7 ± 4.2 |
| | | 80 | 62.5 ± 45.0 | 0.42 ± 0.10 | 80 | 24.0 ± 7.5 |
| | | 60 | 65.0 ± 43.6 | 0.47 ± 0.03 | 60 | 27.6 ± 2.9 |
| | | 40 | – | – | 80 | 26.3 ± 5.7 |
| April 2, 1992 | Control | 100 | 92.5 ± 5.0 | 0.48 ± 0.08 | 90 | 24.9 ± 2.6 |
| | MCK 1.56 | 100 | 47.5 ± 18.9 | 0.60 ± 0.03 | 70 | 24.0 ± 8.6 |
| | | 80 | 42.5 ± 32.0 | 0.54 ± 0.19 | 60 | 21.8 ± 5.2 |
| | | 60 | 67.5 ± 34.0 | 0.60 ± 0.05 | 60 | 25.8 ± 4.8 |
| | | 40 | – | – | 70 | 27.4 ± 2.3 |
| | MCK 2.03 | 100 | 82.5 ± 23.6 | 0.59 ± 0.09 | 60 | 21.8 ± 4.6 |
| | | 80 | 85.0 ± 5.8 | 0.61 ± 0.08 | 40 | 23.3 ± 4.9 |
| | | 60 | 67.5 ± 27.5 | 0.69 ± 0.12 | 80 | 23.8 ± 9.9 |
| | MCK 2.03 | 40 | – | – | 40 | 20.8 ± 5.3 |

Table 3.26 (continued)

| Test date | Water source | Conc. (%) | *Pimephales promelas* | | *Ceriodaphnia dubia* | |
|---|---|---|---|---|---|---|
| | | | Survival (%) (mean ± SD) | Growth (mg/fish) (mean ± SD) | Survival (%) | Reproduction (mean ± SD) |
| July 9, 1992 | Control | 100 | 95.0 ± 10.0 | 0.39 ± 0.02 | 90 | 29.7 ± 1.7 |
| | MCK 1.56 | 100 | 95.0 ± 10.0 | 0.43 ± 0.03 | 100 | 22.0 ± 4.9 |
| | | 80 | 80.0 ± 16.3 | 0.44 ± 0.06 | 100 | 24.6 ± 6.7 |
| | | 60 | 87.5 ± 5.0 | 0.46 ± 0.06 | 90 | 18.9 ± 9.3 |
| | | 40 | 92.5 ± 9.6 | 0.40 ± 0.04 | 100 | 22.8 ± 5.8 |
| | MCK 2.03 | 100 | 82.5 ± 17.1 | 0.45 ± 0.06 | 100 | 21.0 ± 3.1 |
| | | 80 | 87.5 ± 9.6 | 0.48 ± 0.04 | 100 | 24.0 ± 2.5 |
| | | 60 | 65.0 ± 17.3 | 0.50 ± 0.06 | 100 | 26.5 ± 2.8 |
| | | 40 | 65.0 ± 25.1 | 0.53 ± 0.10 | 100 | 23.9 ± 3.2 |
| September 17, 1992 | Control | 100 | – | – | 100 | 23.7 ± 5.4 |
| | MCK 1.56 | 100 | – | – | 100 | 21.8 ± 4.8 |
| | | 75 | – | – | 100 | 24.6 ± 5.2 |
| | | 50 | – | – | 100 | 25.3 ± 5.9 |
| | MCK 2.03 | 100 | – | – | 100 | 25.8 ± 4.1 |
| | | 75 | – | – | 90 | 24.0 ± 4.2 |
| | | 50 | – | – | 90 | 22.9 ± 4.1 |
| January 14, 1993 | Control | 100 | 87.5 ± 5.0 | 0.42 ± 0.07 | 100 | 27.9 ± 3.1 |
| | MCK 1.56 | 100 | 92.5 ± 9.6 | 0.45 ± 0.06 | 90 | 27.3 ± 7.6 |
| | | 75 | 90.0 ± 8.2 | 0.43 ± 0.07 | 100 | 28.4 ± 7.0 |
| | | 50 | 85.0 ± 10.0 | 0.49 ± 0.03 | 100 | 27.8 ± 3.8 |
| | MCK 2.03 | 100 | 92.5 ± 9.6 | 0.40 ± 0.03 | 100 | 27.5 ± 3.6 |
| | | 75 | 87.5 ± 9.6 | 0.46 ± 0.06 | 90 | 28.7 ± 5.5 |
| | | 50 | 92.5 ± 9.6 | 0.49 ± 0.05 | 80 | 29.3 ± 6.8 |
| March 25, 1993 | Control | 100 | 92.5 ± 9.6 | 0.42 ± 0.04 | 100 | 7.0 ± 4.7 |
| | MCK 1.56 | 100 | 77.5 ± 18.9 | 0.57 ± 0.10 | 100 | 7.4 ± 4.3 |
| | MCK 2.03 | 100 | 75.0 ± 26.5 | 0.53 ± 0.08 | 90 | 12.9 ± 2.9 |
| | MCK 2.53 | 100 | 55.0 ± 36.9 | 0.62 ± 0.18 | 100 | 14.5 ± 6.0 |

**Table 3.26 (continued)**

| Test date | Water source | Conc. (%) | *Pimephales promelas* | | *Ceriodaphnia dubia* | |
|---|---|---|---|---|---|---|
| | | | Survival (%) (mean ± SD) | Growth (mg/fish) (mean ± SD) | Survival (%) | Reproduction (mean ± SD) |
| July 29, 1993 | Control | 100 | 95.0 ± 5.8 | 0.33 ± 0.09 | 90 | 28.0 ± 5.2 |
| | MCK 1.56 | 100 | 80.0 ± 21.6 | 0.40 ± 0.07 | 100 | 22.9 ± 4.9 |
| | MCK 1.56 | 80 | 85.0 ± 10.0 | 0.38 ± 0.03 | 100 | 21.8 ± 7.4 |
| | | 60 | 75.0 ± 20.8 | 0.39 ± 0.06 | 100 | 21.5 ± 8.0 |
| | MCK 2.03 | 100 | 82.5 ± 5.0 | 0.35 ± 0.02 | 100 | 22.8 ± 5.3 |
| | | 80 | 80.0 ± 8.2 | 0.35 ± 0.09 | 80 | 25.4 ± 4.2 |
| | | 60 | 80.0 ± 11.5 | 0.36 ± 0.08 | 90 | 21.3 ± 8.5 |

[a]— refers to not tested.

Table 3.27 Results of 3-day snail (*Elimia clavaeformis*) feeding tests

| Site/treatment | Mean wet weight (g) eaten ± s.e. |
|---|---|
| *Test #1* January 18, 1993 | |
| DMW[a] | 0.07 ± 0.026 |
| MCK 1.56-100% | 0.14 ± 0.028 |
| MCK 1.56-50% | 0.16 ± 0.026 |
| MCK 2.03-100% | 0.17 ± 0.039 |
| MCK 2.03-50% | 0.13 ± 0.021 |
| | |
| *Test #2* April 19, 1993 | |
| DMW | 0.17 ± 0.007 |
| MCK 1.56 | 0.17 ± 0.007 |
| MCK 1.56+sediment | 0.17 ± 0.007 |
| MCK 2.03 | 0.17 ± 0.011 |
| MCK 2.03+sediment | 0.18 ± 0.007 |
| | |
| *Test #3* July 27, 1993 | |
| DMW-15°C | 0.06 ± 0.004 |
| DMW-25°C | 0.15 ± 0.012 |
| MCK 1.56+sediment-15°C | 0.05 ± 0.005 |
| MCK 1.56+sediment-25°C | 0.14 ± 0.008 |
| MCK 2.03+sediment-15°C | 0.04 ± 0.005 |
| MCK 2.03+sediment-25°C | 0.14 ± 0.007 |

[a]DMW refers to diluted mineral water, which was used as a negative control and, in some of the McCoy Branch water toxicity tests, as a diluent.

Table 3.28. Results of in situ earthworm test

| Soil type | Treatment | N | Survival fraction (mean ± SD) |
|---|---|---|---|
| Peat moss | food added | 6 | 0.611 ± 0.344 |
| Peat moss | no food | 6 | 0.444 ± 0.251 |
| Fly ash | food added | 6 | 0.056 ± 0.086 |
| Fly ash | no food | 6 | 0.028 ± 0.068 |

Table 3.29. Results of soil and ash analyses for in situ earthworm test

| Soil type | Treatment | N | pH | Water content (%) (mean ± SD) | Organic content (% of dry matter) (mean ± SD) |
|---|---|---|---|---|---|
| Peat moss | food added | 6 | 4.8 ± 0.4 | 78 ± 2.2 | 96 ± 0.2 |
| Peat moss | no food | 6 | 4.6 ± 0.4 | 78 ± 1.8 | 97 ± 0.8 |
| Fly ash | food added | 6 | 6.8 ± 0.7 | 48 ± 6.2 | 34 ± 9.0 |
| Fly ash | no food | 6 | 6.6 ± 0.9 | 47 ± 5.4 | 30 ± 7.8 |

**Table 3.30. Results of 14-day laboratory earthworm tests**

| Soil or ash source | N | Treatment | Initial pH | Survival fraction (mean ± SD) | Multiple-level T-test[a] | Change in fresh mass (g; mean ± SD) |
|---|---|---|---|---|---|---|
| Reference | 4 | autoclaved | --- | 0.925 ± 0.050 | a | 0.0392 ± 0.0385 |
| Reference | 4 | none | 4.96 | 0.950 ± 0.058 | a | -0.0032 ± 0.0355 |
| Artificial | 4 | autoclaved | --- | 0.950 ± 0.100 | a | -0.0088 ± 0.0648 |
| Artificial | 4 | none | 6.74 | 0.925 ± 0.050 | a | -0.0410 ± 0.0522 |
| SLUICE-1 | 4 | autoclaved | --- | 0.950 ± 0.100 | a | -0.0558 ± 0.0311 |
| SLUICE-1 | 4 | none | 5.67 | 0.825 ± 0.150 | ab | -0.0560 ± 0.0526 |
| FCAP-1 | 4 | autoclaved | --- | 0.525 ± 0.299 | bc | -0.0818 ± 0.0373 |
| FCAP-1 | 4 | none | 5.69 | 0.325 ± 0.330 | c | -0.1820 ± 0.2027 |
| FCAP-2 | 4 | autoclaved | --- | 0.950 ± 0.058 | a | -0.0778 ± 0.0862 |
| FCAP-2 | 4 | none | 6.75 | 0.750 ± 0.379 | ab | -0.0632 ± 0.0629 |

[a]Comparisons are for fraction of animals surviving.

**Table 3.31. Summary statistics of water quality factors for MCK 1.56**

| Factor | Analysis period | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | July 1990 | Apr. 1991 | July 1991 | Oct. 1991 | Jan. 1992 | Apr. 1992 | July 1992 | Sept. 1992 | Jan. 1993 | Mar. 1993 | July 1993 |
| pH (S.U.) | 8.49[a]<br>(0.09)[b]<br>8.41-8.67[c] | 7.98<br>(0.15)<br>7.74-8.28 | 7.87<br>(0.05)<br>7.78-7.96 | 7.87<br>(0.14)<br>7.75-8.14 | 8.12<br>(0.05)<br>8.02-8.21 | 8.08<br>(0.08)<br>7.91-8.18 | 8.11<br>(0.08)<br>8.04-8.25 | 8.05<br>(0.04)<br>8.00-8.11 | 7.88<br>(0.09)<br>7.76-7.97 | 8.18<br>(0.09)<br>8.02-8.27 | 8.03<br>(0.05)<br>7.93-8.06 |
| Conductivity ($\mu$S) | 232.9<br>(3.6)<br>227-237 | 241.4<br>(13.9)<br>225-260 | 212.6<br>(12.6)<br>192-229 | 235.3<br>(19.6)<br>193-259 | 251.5<br>(21.9)<br>225-276 | 249.0<br>(22.8)<br>218-277 | 271.6<br>(9.4)<br>258-281 | 288.5<br>(5.5)<br>283-296 | 278.6<br>(13.3)<br>255-291 | 265.4<br>(12.8)<br>243-277 | 278.3<br>(11.6)<br>253-286 |
| Alkalinity (mg/L) | 82.9<br>(1.5)<br>81-84 | 95.4<br>(7.9)<br>86-113 | 92.1<br>(7.3)<br>82-101 | 105.1<br>(9.4)<br>94-118 | 110.1<br>(10.8)<br>98-125 | 107.1<br>(11.6)<br>94-119 | 123.7<br>(2.1)<br>120-126 | 125.5<br>(2.6)<br>123-130 | 123.3<br>(4.9)<br>120-134 | 118.0<br>(7.7)<br>106-130 | 125.6<br>(2.1)<br>123-129 |
| Hardness (mg/L) | 124.6<br>(6.2)<br>112-130 | 126.0<br>(12.5)<br>100-140 | 114.1<br>(8.5)<br>100-128 | 128.7<br>(12.3)<br>114-154 | 129.1<br>(12.8)<br>112-152 | 129.4<br>(13.6)<br>110-154 | 150.9<br>(10.1)<br>140-166 | 157.7<br>(10.1)<br>146-172 | 149.1<br>(10.9)<br>134-164 | 169.4<br>(46.0)<br>134-268 | 156.0<br>(15.0)<br>136-180 |

[a]7-day mean;
[b]standard deviation of the mean;
[c]range for 7-day test period.

**Table 3.32. Summary statistics of water quality factors for MCK 2.03**

| Factor | Analysis period | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | July 1990 | Apr. 1991 | July 1991 | Oct. 1991 | Jan. 1992 | Apr. 1992 | July 1992 | Sept. 1992 | Jan. 1993 | Mar. 1993 | July 1993 |
| pH (S.U.) | 8.07[a]<br>(0.07)[b]<br>7.95-8.15[c] | 7.64<br>(0.10)<br>7.49-7.78 | 7.60<br>(0.07)<br>7.52-7.72 | 7.12<br>(0.14)<br>7.50-7.87 | 8.06<br>(0.05)<br>8.00-8.17 | 7.99<br>(0.05)<br>7.91-8.05 | 8.05<br>(0.16)<br>7.75-8.26 | 8.03<br>(0.07)<br>7.92-8.09 | 7.77<br>(0.11)<br>7.61-7.93 | 8.01<br>(0.14)<br>7.74-8.16 | 8.10<br>(0.06)<br>8.01-8.17 |
| Conductivity ($\mu$S) | 336.1<br>(18.1)<br>297-352 | 274.3<br>(22.8)<br>244-307 | 314.3<br>(7.7)<br>299-321 | 345.1<br>(5.1)<br>336-350 | 284.0<br>(6.4)<br>276-293 | 268.0<br>(13.0)<br>253-286 | 328.9<br>(16.9)<br>296-350 | 360.5<br>(5.5)<br>353-365 | 259.7<br>(21.4)<br>226-284 | 196.9<br>(21.9)<br>157-226 | 338.3<br>(3.7)<br>334-343 |
| Alkalinity (mg/L) | 165.0<br>(4.3)<br>158-171 | 120.4<br>(10.5)<br>104-134 | 158.3<br>(4.3)<br>151-163 | 175.1<br>(2.4)<br>170-177 | 141.9<br>(13.0)<br>131-170 | 129.4<br>(7.0)<br>121-141 | 169.7<br>(2.7)<br>167-174 | 179.2<br>(1.9)<br>177-182 | 117.4<br>(13.1)<br>102-138 | 84.0<br>(10.0)<br>72-101 | 171.3<br>(2.1)<br>169-174 |
| Hardness (mg/L) | 189.7<br>(4.4)<br>184-194 | 143.1<br>(13.6)<br>128-164 | 177.1<br>(7.6)<br>170-192 | 194.3<br>(5.2)<br>190-204 | 151.4<br>(8.2)<br>144-166 | 146.3<br>(8.2)<br>140-162 | 185.1<br>(9.6)<br>176-202 | 204.0<br>(13.5)<br>192-230 | 149.7<br>(18.4)<br>120-176 | 122.8<br>(21.5)<br>92-164 | 185.1<br>(11.5)<br>170-196 |

[a]7-day mean;
[b]standard deviation of the mean;
[c]range for 7-day test period.

**Table 3.33. Analysis of variance results for data from MCK 1.56 and MCK 2.03**

| Site | pH | Conductivity | Alkalinity | Hardness | Alk:Cond | Alk:Hard |
|---|---|---|---|---|---|---|
| Mean[a] at MCK 1.56 | 8.06 | 262.3 | 113.7 | 143.5 | 0.433 | 0.798 |
| Mean[a] at MCK 2.03 | 7.91 | 299.7 | 146.1 | 167.6 | 0.484 | 0.867 |
| $R^{2}$[b] | 0.819 | 0.928 | 0.960 | 0.756 | 0.818 | 0.587 |
| F-ratio for: | | | | | | |
| test (df=10) | 40.08 | 61.06 | 98.80 | 11.94 | 19.52 | 10.12 |
| site (df=1) | 92.18 | 380.80 | 1089.35 | 108.42 | 259.81 | 44.08 |
| test: site (df=10) | 9.38 | 70.72 | 107.51 | 17.66 | 13.09 | 3.97 |
| Site difference (%)[c] | 1.9 | 14.3 | 28.5 | 16.8 | 11.8 | 8.7 |

[a]Each mean is computed from 76 observations.

[b]$R^2$ is the proportion of variation explained collectively by site, test, and the interaction between site and test, using ANOVA (PROC GLM, type III sum of squares). Model and error degrees of freedom were 21 and 130, respectively, for each water quality factor.

[c]Computed, for each water quality factor, as difference between means for the two sites divided by the smaller mean, times 100.

Table 3.34. Checklist and relative abundance of benthic macroinvertebrates[a]

| Taxon | Site/date | | | | | |
|---|---|---|---|---|---|---|
| | MCK 2.03 | | MCK 2.53 | | WCK 6.8 | |
| | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/92 |
| Oligochaeta | - | - | 0.3 | 6.6 | 0.2 | 0.4 |
| Crustacea | | | | | | |
| Isopoda | | | | | | |
| Asellidae | | | | | | |
| *Lirceus* | - | - | - | - | X | - |
| Amphipoda | | | | | | |
| Gammaridae | | | | | | |
| *Gammarus* | - | - | - | - | - | 0.2 |
| Decapoda | | | | | | |
| Cambaridae | - | 1.1 | - | 0.1 | X[b] | X |
| *Cambarus* | - | 0.3 | - | X | - | - |
| Insecta | | | | | | |
| Ephemeroptera | | | | | | |
| Baetidae | | | | | | |
| *Baetis* | 1.3 | 0.6 | 0.3 | 5.0 | 1.8 | 13.4 |
| *Pseudocloeon* | - | 0.3 | - | - | X | 1.3 |
| Ephemerellidae | | | | | | |
| *Ephemerella* | - | X | - | 0.4 | - | 6.0 |
| *Eurylophella* | - | 18.5 | - | 1.8 | - | - |
| Ephemeridae | | | | | | |
| *Ephemera* | - | - | - | - | - | 0.7 |
| Heptageniidae | | | | | | |
| *Stenacron* | - | - | - | - | - | 0.4 |
| Leptophlebiidae | | | | | | |
| *Habrophlebiodes* | - | 0.3 | - | 1.2 | 0.7 | - |
| Oligoneuriidae | | | | | | |
| *Isonychia* | - | - | - | - | - | X |
| Siphlonuridae | | | | | | |
| *Ameletus* | - | 0.3 | - | 0.5 | - | - |
| Odonata | | | | | | |
| Anisoptera | | | | | | |
| Aeshnidae | | | | | | |
| *Boyeria vinosa* | 0.6 | X | - | - | - | - |

Table 3.34 (continued)

| Taxon | Site/date | | | | | |
|---|---|---|---|---|---|---|
| | MCK 2.03 | | MCK 2.53 | | WCK 6.8 | |
| | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/92 |
| Cordulegastridae | | | | | | |
| *Cordulegaster* | X | 1.4 | 0.6 | 0.4 | 0.4 | - |
| Gomphidae | | | | | | |
| *Stylogomphus* | | | | | | |
| *albistylus* | X | - | - | - | X | 1.1 |
| Zygoptera | | | | | | |
| Calopterygidae | | | | | | |
| *Calopteryx* | 1.9 | 0.6 | 0.3 | - | X | 0.2 |
| Plecoptera | | | | | | |
| Chloroperlidae | | | | | | |
| *Haploperla brevis* | - | - | - | - | - | X |
| *Haploperla?* | - | - | - | 0.1 | - | 1.6 |
| *Sweltsa* | - | 0.6 | - | 0.3 | 0.7 | 1.6 |
| Leuctridae | | | | | | |
| *Leuctra* | 3.8 | 1.7 | 8.0 | 7.1 | 40.3 | 7.4 |
| Nemouridae | | | | | | |
| *Amphinemura* | - | 17.4 | - | 42.0 | - | 17.9 |
| Peltoperlidae | | | | | | |
| *Tallaperla* | - | 0.3 | - | 0.1 | 3.3 | 1.3 |
| Perlidae | | | | | | |
| *Eccoptura xanthenes* | - | - | 0.9 | 0.5 | 0.2 | 0.4 |
| *Eccoptura?* | - | - | - | X | - | - |
| *Perlesta* | - | 0.3 | 0.3 | 4.1 | X | - |
| Perlodidae | | | | | | |
| *Clioperla clio* | - | - | - | 0.1 | - | - |
| *Clioperla?* | - | - | - | 2.4 | - | - |
| *Isoperla* | - | 0.3 | - | 1.1 | - | X |
| Perlidae/Perlodidae | - | - | - | 0.3 | - | - |
| Hemiptera | | | | | | |
| Corixidae | | | | | | |
| *Sigara* | - | - | X | - | - | - |
| Megaloptera | | | | | | |
| Corydalidae | | | | | | |

Table 3.34 (continued)

| Taxon | Site/date | | | | | |
|---|---|---|---|---|---|---|
| | MCK 2.03 | | MCK 2.53 | | WCK 6.8 | |
| | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/92 |
| *Nigronia fasciatus* | X | 0.3 | - | X | 1.1 | 0.4 |
| *Nigronia* | - | 0.3 | - | - | 0.4 | - |
| Sialidae | | | | | | |
| *Sialis* | - | - | 4.6 | 0.1 | - | X |
| Trichoptera | | | | | | |
| Glossosomatidae | | | | | | |
| *Agapetus* | - | - | - | 0.7 | - | 1.1 |
| *Glossosoma* | 1.3 | 0.3 | 0.3 | - | 0.4 | - |
| Hydropsychidae | | | | | | |
| *Cheumatopsyche* | 5.1 | 1.1 | X | - | 5.8 | - |
| *Diplectrona modesta* | 0.6 | 3.6 | 2.5 | 3.0 | 2.7 | 13.6 |
| *Hydropsyche depravata* | 8.3 | 0.8 | 0.9 | 0.4 | 8.4 | X |
| Lepidostomatidae | | | | | | |
| *Lepidostoma* | X | 0.8 | X | 0.9 | X | 0.4 |
| Limniphilidae | | | | | | |
| *Ironoquia* | - | - | - | X | - | - |
| *Neophylax* | 0.6 | 2.5 | X | 1.6 | 0.2 | 2.9 |
| *Pycnopsyche gentilis* | - | - | - | - | - | X |
| *Pycnopsyche luculenta* | 0.6 | X | - | 0.1 | - | X |
| *Pycnopsyche scabripennis* | - | - | - | - | - | 0.2 |
| Molannidae | | | | | | |
| *Molanna* | 1.3 | 0.3 | 2.8 | 0.1 | 0.7 | X |
| Odontoceridae | | | | | | |
| *Psilotreta* | - | - | - | - | X | - |
| Philopotamidae | | | | | | |
| *Dolophilodes distinctus* | - | - | - | - | X | X |
| Polycentropodidae | | | | | | |
| *Polycentropus* | - | - | - | - | 1.1 | 0.7 |
| Psychomyiidae | | | | | | |
| *Lype diversa* | - | - | - | - | - | 0.2 |
| Rhyachophilidae | | | | | | |
| *Rhyacophila* | 0.6 | 0.8 | - | 0.1 | X | 2.0 |
| Coleoptera | | | | | | |

**Table 3.34 (continued)**

| Taxon | Site/date | | | | | |
|---|---|---|---|---|---|---|
| | MCK 2.03 | | MCK 2.53 | | WCK 6.8 | |
| | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/92 |
| Dytiscidae | | | | | | |
| *Hydroporus* | - | - | 1.2 | 1.2 | - | - |
| Elimidae | | | | | | |
| *Dubiraphia* | 0.6 | - | - | - | - | - |
| *Optioservus* | 15.4 | 5.0 | 0.6 | 0.5 | 2.4 | 6.7 |
| *Stenelmis* | - | 0.3 | - | X | 0.2 | X |
| Psephenidae | | | | | | |
| *Ectopria* | - | - | - | - | - | 0.2 |
| *Psephenus herricki* | - | - | - | - | - | 0.2 |
| Ptilodactylidae | | | | | | |
| *Anchytarsus bicolor* | X | 0.3 | - | - | 0.4 | 0.7 |
| Diptera | | | | | | |
| Ceratopogonidae | - | 0.3 | 0.3 | 0.9 | - | - |
| Chironomidae | 1.3 | 1.7 | 0.6 | 0.5 | 0.4 | - |
| Chironominae | | | | | | |
| Chironomini | 23.7 | - | 3.7 | 1.2 | 2.9 | 0.4 |
| Tanytarsini | 17.3 | 1.1 | 4.3 | 0.4 | 8.6 | 5.4 |
| Orthocladiinae | 9.6 | 27.3 | 49.7 | 7.7 | 2.4 | 2.7 |
| Tanypodinae | 3.2 | 3.3 | 15.6 | 3.6 | 5.3 | 1.1 |
| Dixidae | | | | | | |
| *Dixa* | 0.6 | 0.6 | 0.3 | 0.5 | 0.2 | - |
| Empididae | | | | | | |
| *Chelifera* | - | X | - | - | X | - |
| *Clinocera* | - | - | - | 0.1 | - | - |
| *Hemerodromia* | 0.6 | - | 0.3 | - | 0.7 | - |
| Simuliidae | | | | | | |
| *Simulium* | X | 3.9 | - | 0.9 | 6.4 | 1.3 |
| Stratiomyidae | | | | | | |
| *Allognosta* | - | 0.3 | 0.3 | X | 0.2 | - |
| *Caloparyphus* | - | - | - | - | - | X |
| *Odontomyia* | - | 0.3 | - | - | - | - |
| Tabanidae | | | | | | |
| *Chrysops* | - | - | - | 0.1 | - | - |

Table 3.34 (continued)

| Taxon | Site/date | | | | | |
|---|---|---|---|---|---|---|
| | MCK 2.03 | | MCK 2.53 | | WCK 6.8 | |
| | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/93 | 7/15/92 | 3/30/92 |
| *Tabanus* | - | 0.3 | - | 0.1 | - | - |
| Tupulidae | | | | | | |
| *Antocha* | - | - | - | - | 0.2 | 1.1 |
| *Hexatoma* | - | - | - | - | 0.2 | - |
| *Pseudolimnophila* | - | 0.3 | - | - | X | 0.7 |
| *Tipula abdominalis* | X | 0.8 | - | 0.9 | - | - |
| *Tipula* | 0.6 | X | X | - | X | - |
| Mollusca | | | | | | |
| Gastropoda | | | | | | |
| Lymnaeidae | | | | | | |
| *Pseudosuccinea columella* | - | - | X | - | - | - |
| Physidae | | | | | | |
| *Physella* | - | - | 1.2 | X | - | - |
| Pleuroceridae | | | | | | |
| *Elimia* | - | X | - | - | 0.9 | 3.6 |
| Bivalvia | | | | | | |
| Sphaeriidae | | | | | | |
| *Sphaerium* | 0.6 | - | - | - | - | - |

[a]Values are the relative abundance of each taxon expressed as a % of the total number of individuals in the semiquantitative portion of the sample; an "X" denotes a taxon's occurrence in the qualitative portion of the sample only; a "-" indicates the taxon was not collected or that the taxon was identified to a lower level at one or more sites; a blank indicates that a lower level of classification (e.g., family, genus, species) was possible at one or more sites.

[b]A single female specimen collected and then subsequently released; its presence is documented in field notes only.

**Table 3.35. Mean (± SD) concentrations of metals detected in leaves**

| Vegetation type | Location | n | Concentration (μg/g dry matter) | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | As | Cd | Cr | Pb | Hg | Se |
| Deciduous trees[a] | Control[b] | 14 | 1.01 ± 0.05[c] | 0.25 ± 0.01[c] | < 3.00[f] | < 1.00 | 0.11 ± 0.05[c] | < 1.00[c] |
| | FCAP-1 | 18 | 1.53 ± 0.76[d] | 0.62 ± 0.66[c] | 3.04 ± 0.19 | < 1.00 | 0.06 ± 0.02[d] | 14.8 ± 12.7[d] |
| | FCAP-2 | 12 | 1.83 ± 0.58[d] | 1.74 ± 0.96[d] | < 3.00 | 1.07 ± 0.23 | < 0.05[d] | 29.1 ± 24.5[e] |
| | SLUICE-1 | 6 | < 1.00[c] | 0.31 ± 0.14[c] | < 3.00 | < 1.00 | 0.07 ± 0.01[d] | 1.55 ± 0.87[c] |
| | | | | | | | | |
| Eastern red cedar | Control | 2 | < 1.00 | < 0.25 | < 3.00 | < 1.00 | < 0.05 | < 1.00 |
| | FCAP-1 | 3 | < 1.00 | < 0.25 | < 3.00 | <1.00 | < 0.05 | 2.43 ± 0.42 |
| | SLUICE-1 | 3 | < 1.00 | < 0.25 | < 3.00 | < 1.00 | 0.06 ± 0.01 | < 1.00 |
| | | | | | | | | |
| Ground covers | Control | 3 | 1.07 ± 0.12 | < 0.25 | < 3.00 | < 1.00 | < 0.05 | < 1.00 |
| | FCAP-1 | 6 | 1.07 ± 0.12 | < 0.25 | 10.07 ± 5.44 | 1.02 ± 0.04 | < 0.05 | 2.12 ± 0.76 |
| | SLUICE-1 | 3 | < 1.00 | < 0.25 | < 3.00 | < 1.00 | < 0.05 | 1.37 ± 0.21 |

[a]Deciduous trees included the following: red maple, sourwood, dogwood, American beech, sweet gum, sycamore, box elder, cottonwood, and willows. Ground covers were limited to a grass (*Eulalia*) and japanese honeysuckle.

[b]Walker Branch Watershed

[c,d,e]For deciduous trees, values in the same column with different superscripts are significantly different ($p < 0.05$).

[f]Less than values represent contaminant levels below the detection limit.

**Table 3.36. Small mammals collected in traps**

| Site | FCAP-1 | FCAP-2 | SLUICE-1 | Walker Branch |
|---|---|---|---|---|
| Species | White-footed mice (7) (*Peromyscus leucopus*) | White-footed mice (1)<br><br>Eastern Harvest mice (3) (*Reithrodontomys humulus*)<br><br>Fulvous Harvest mice (1) (*Reithrodontomys fulvescens*) | White-footed mice (4)<br><br>Short-tailed shrew (1) (*Blarina brevicauda*)<br><br>Pine vole (1) (*Microtus pinetorum*) | White-footed mice (4)<br><br>Short-tailed shrew (2) |
| Dates collected | June 23-29, 1993 | August 5-18, 1993 | September 9-14, 1993 | September 29 - October 13, 1993 |
| Trap-nights | 525 | 675 | 480 | 1300 |
| Animals/ trap-nights | 0.0133 | 0.0074 | 0.0125 | 0.0046 |

## Table 3.37. Concentrations of metals in small mammals

| Analyte | Chestnut Ridge OU 2 | | FCAP | | Sluice Channel Area | | Walker Branch |
|---|---|---|---|---|---|---|---|
| | Chestnut Ridge OU 2 Rodents[a] | Tolerance limit[b] | *Peromyscus* (n=8) | *Reithrodontomys* (n=4) | *Peromyscus* (n=4) | *Microtus* (n=1) | *Peromyscus* (n=4) |
| Arsenic | 0.131 ± 0.098[e] (0.050 - 0.434)[f] | 0.924 | .171 ± 0.121[c] (0.081 - 0.434) | 0.139 ± 0.064[c,d] (0.060 - 0.216) | 0.051 ± 0.001[d] (0.050 - 0.053) | 0.093 ± 0.013 | 0.054 ± 0.007[d] (<0.05 - 0.064) |
| Cadmium | 0.022 ± 0.016 (0.007 - 0.065) | 0.174 | 0.021 ± 0.014[c] (0.008 - 0.047) | 0.028 ± 0.026[c] (<0.010 - 0.065) | 0.018 ± 0.014[c] (0.007 - 0.038) | 0.022 ± 0.003 | 0.022 ± 0.007[c] (0.013 - 0.027) |
| Chromium | 1.256 ± 0.354 (0.987 - 2.253) | 2.706 | 1.389 ± 0.477[c] (0.987 - 2.253) | 1.116 ± 0.095[c] (1.004 - 1.227) | 1.083 ± 0.038[c] (1.035 - 1.117) | 1.457 ± 0.129 | 1.587 ± 1.498[c] (0.679 - 3.823) |
| Lead | 1.030 ± 2.408 (0.094 - 10.267) | 20.102 | 0.520 ± 0.469[c] (0.139 - 1.453) | 2.662 ± 5.070[c] (0.094 - 10.267) | 0.558 ± 0.234[c] (0.271 - 0.821) | 0.476 ± 0.054 | 0.334 ± 0.228[c] (0.124 - 0.648) |
| Mercury | 0.037 ± 0.025 (0.011 - 0.105) | 0.268 | 0.033 ± 0.014[c] (0.016 - 0.056) | 0.051 ± 0.045[c] (0.012 - 0.106) | 0.038 ± 0.015[c] (0.025 - 0.053) | 0.011 ± 0.002 | 0.027 ± 0.007[c] (0.018 - 0.034) |
| Selenium | 1.831 ± 1.481 (0.329 - 5.640) | 26.980 | 2.563 ± 1.580[c] (0.446 - 5.64) | 2.007 ± 1.222[c] (.644 - 3.387) | 0.506 ± 0.184[d] (0.329 - 0.764) | 0.569 ± 0.064 | 0.261 ± 0.012[d] (0.248 - 0.277) |
| Thallium | 0.203 ± 0.010 (<.200 - 0.243) | [g] | 0.205 ± 0.015[c] (< 0.200 - 0.243) | 0.200 ± 0.001[c] (<0.200 - 0.202) | <0.200[c] | <0.200 | < .200 |

[a]Chestnut Ridge OU 2 Rodents: Mean concentration found in all rodents collected within Chestnut Ridge OU 2.

[b]Tolerance limits calculated for maximum contaminant concentration that may be found in 99% of the small mammal population with a confidence level of 95%. Tolerance limit calculated only for rodents at Chestnut Ridge OU 2.

[c,d]Means with like superscripts do not differ significantly (p> 0.05) for each analyte.

[e]Mean concentration (mg/kg).

[f]Standard deviation (mg/kg).

[g]Tolerance limits could not be calculated for nondetectable values.

Table 3.38. Mean concentrations (mg/kg) of metals detected in *Blarina brevicauda*

| Analyte | Walker Branch (n=2) | Sluice Channel Area (n=1) |
|---|---|---|
| Arsenic | 0.059 ± 0.012 | 0.177 ± 0.016 |
| Cadmium | 0.066 ± 0.013 | 0.073 ± 0.014 |
| Chromium | 3.647 ± 2.522 | 0.797 ± 0.139 |
| Mercury | 0.337 ± 0.145 | 0.481 ± 0.121 |
| Lead | 1.533 ± 0.316 | 0.677 ± 0.108 |
| Selenium | 1.195 ± 0.342 | 2.330 ± 0.176 |
| Thallium | < 0.200 | < 0.200 |

Table 3.39. Number of banded sculpins introduced and recovered in Upper McCoy Branch

| Date | Number of sculpins | Disposition | Number of passes | Sample length |
|---|---|---|---|---|
| October 3, 1992 | 180 | Introduced | NA[a] | NA |
| January 6, 1993 | 55 | Recovered | 3,1 | (2) 50 m sections, 300 m |
| April 9, 1993 | 27 | Recovered | 3 | (2) 50 m sections |
| June 7, 1993 | 38 | Recovered | 4-6 | (4) 100 m sections |
| August 2, 1993 | 18 | Introduced | NA | NA |
| October 19, 1993 | 25 | Recovered | 4-6 | (4) 50 m sections |

[a]Not applicable

# 4. APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS

## 4.1 IDENTIFICATION OF ARARs

CERCLA was passed by Congress and signed into law on December 11, 1980 (Public Law 96-510). This act was intended to provide for "liability, compensation, cleanup, and emergency response for hazardous substances released into the environment and the cleanup of inactive waste disposal sites." SARA, adopted on October 17, 1986 (Public Law 99-499), did not substantially alter the original structure of CERCLA but provided extensive amendments to it.

In particular, §121 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 ARARs is the assumption that protection of human health and the environment is ensured.

The purpose of Chap. 4 is to supply a list of potential chemical- and location-specific ARARs that might be considered for the remediation of soil and, perhaps, surface or groundwater contamination at Chestnut Ridge OU 2. It is stated in Chap. XXI(F)(2) of the FFA for the ORR that the process of ARAR identification is an iterative one that is continually changing as the RI/FS progresses. Therefore, this list of ARARs represents a compilation of potential ARARs, of which subsets will be used or additional ARARs added as remedial alternatives are further refined and a final alternative is selected.

It is understood that DOE will comply with the requirements of NEPA as specified in DOE Order 5440.1D ("National Environmental Policy Act Compliance Program"). Further, DOE Order 5400.4 ("Comprehensive Environmental Response, Compensation, and Liability Act Requirements") calls for integration of NEPA and CERCLA requirements for DOE remedial actions at CERCLA sites. This issue has been reaffirmed in FFA §I(A)(3) and §III(A)(2) and the Secretary of Energy Notice (SEN) of February 5, 1990 (SEN-15-90), which was issued to ensure that DOE's NEPA activities are carried out in a centralized and uniform manner. Therefore, the regulations found in NEPA will not be addressed in this report as ARARs; however, the federal and state regulations protecting environmental resources that may be identified at Chestnut Ridge OU 2 during a NEPA assessment are discussed in Sect. 4.3.

Title I, §111(c)(6), of CERCLA mandated that OSHA promulgate standards for regulation of employee health and safety during hazardous waste operations at RCRA or CERCLA sites and during emergency response to hazardous substance releases. The final regulations for "Hazardous Waste Operations and Emergency Response" (29 CFR 1910) have appeared in 54 FR 9294 (Final Rule, March 6, 1989). These regulations are designed to protect workers involved in cleanup operations at uncontrolled hazardous waste sites and to provide worker protection during initial site characterization and analysis, monitoring activities, materials handling activities, training, and emergency response. These regulations do not apply to those workers who would not be exposed.

Federal construction activities involving no potential for hazardous substance exposure are covered by the OSHA standards in 29 CFR 1926 ("Construction Industry") and 29 CFR 1910 ("General Industry"). DOE also addresses occupational safety in DOE Orders 5480.11 ("Radiation Protection for Occupational Workers"), 5480.4 ("Environmental Protection, Safety, and Health Protection Standards"), 5483.1A ("Occupational Safety and Health Program for Contractors at Government Owned, Contractor Operated Facilities"), 5480.9 ("Construction Safety and Health Program"), and 5480.10 ("Contractor Industrial Hygiene Program"). ARARs apply to those federal and state regulations that are designed to protect the environment and do not generally apply to occupational safety regulations. EPA requires compliance with the OSHA standards in §300.150 of the NCP, not through the ARARs process. Therefore, neither the regulations promulgated by OSHA nor the DOE Orders related to occupational safety are addressed as ARARs; these regulations are addressed in the site-specific Health and Safety Plan for Chestnut Ridge OU 2.

The following is a listing of the definitions of terms used throughout this section.

**Applicable requirements** are "those cleanup standards, standards of control, and other substantive requirements, criteria, or limitations promulgated under federal environmental, state environmental, or facility siting law that specifically address a hazardous substance, pollutant, contaminant, remedial action, location, or other circumstance at a CERCLA site" (40 CFR 300.5).

**Relevant and appropriate requirements** are "those cleanup standards, standards of control, and other substantive requirements, criteria, or limitations promulgated under federal environmental, state environmental, or facility siting law that, while not applicable to a hazardous substance, pollutant, contaminant, remedial action, location, or other circumstance at a CERCLA site, address problems or situations sufficiently similar to those encountered at the CERCLA site that their use is well suited to the particular site" (40 CFR 300.5).

Requirements under federal or state law may be *either* applicable *or* relevant and appropriate to CERCLA cleanup actions, but not both. However, requirements must be *both* relevant *and* appropriate for compliance to be necessary. In the case where a federal and a state ARAR are available, or where there are two potential ARARs addressing the same issue, the more stringent regulation must be selected. However, CERCLA §121(d)(4) provides several ARAR waiver options that may be invoked, providing that the basic premise of protection of human health and the environment is not ignored. A waiver is available for state standards that have not been applied uniformly in similar circumstances across the state. In addition, CERCLA §121(d)(2)(C) forbids state standards that effectively prohibit land disposal of hazardous substances.

CERCLA on-site remedial response actions must only comply with the substantive requirements of a regulation and not the administrative requirements to obtain federal, state, or local permits [CERCLA §121(e) and FFA §XXII]. In order to ensure that CERCLA response actions proceed as rapidly as possible, EPA has reaffirmed this position in the final NCP (55 FR 8756). **Substantive requirements** pertain directly to the actions or conditions at a site; **administrative requirements** facilitate their implementation. EPA recognizes that certain of the administrative requirements, such as consultation with state agencies, reporting, etc., are accomplished through the state involvement and public participation requirements of the NCP. These administrative requirements should be observed if they are useful in determining cleanup standards at the site (55 FR 8757).

In the absence of federal- or state-promulgated regulations, there are many criteria, advisories, guidance values, and proposed standards that are not legally binding but may serve as useful guidance for setting protective cleanup levels. These are not potential ARARs but are **"to be considered"** (TBC) guidance.

As described in Chap. 3, Chestnut Ridge OU 2 was used for the disposal of coal ash from 1955 to early 1990. The FCAP was originally listed in the *RCRA Facility Investigation Plan General Document; Y-12 Plant* (Welch 1989) as a RCRA §3004(u) solid waste management unit under the Hazardous and Solid Waste Amendments general permit for the ORR since, at that time, coal ash was subject to regulation as a hazardous waste under RCRA Subtitle C. However, EPA has subsequently issued a final regulatory determination concluding that regulation under Subtitle C is inappropriate for fly ash and bottom ash (58 FR 42466, August 9, 1993). In consequence, none of the RCRA Subtitle C regulations would be ARAR for this CERCLA action. The TDEC regulations governing permitted solid waste disposal facilities, however, would be relevant and appropriate for this CERCLA action.

## 4.2 CHEMICAL-SPECIFIC ARARs

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 else indicate a safe level of discharge that may be incorporated when considering a specific remedial activity. Contaminants of potential concern (COPCs) for Chestnut Ridge OU 2 include metals and organics. Naturally occurring radionuclides are also present in the coal ash. However, available ARARs and TBC guidance for radionuclides address only man-made, not naturally occurring, radionuclides. Therefore, these ARARs would not apply to the radionuclides present in this coal ash and are not included in this document. COPCs are listed and discussed further in Sects. 5.2.4 and 6.1.3.4.

### 4.2.1 Groundwater and Surface Water

Table 4.1 lists available chemical-specific ARARs that have been promulgated under federal and state law that may be applicable to cleanup of groundwater at Chestnut Ridge OU 2. If significant groundwater and/or surface water contamination is discovered at this site, remedial activities may include cleanup of the water.

As stated in the NCP (55 FR 8666), the goal of EPA's approach to cleanup of contaminated groundwater is to return usable groundwater to its beneficial use within a given time frame that is reasonable for the particular circumstances at a CERCLA site. Groundwater at ORR has not been given an EPA classification. Although not an ARAR unless promulgated, the EPA guidance on groundwater classification should be used to determine whether groundwater at Chestnut Ridge OU 2 falls within Class I, II, or III. Classes I and IIA represent current sources of drinking water of varying value, Class IIB represents potential sources of drinking water, and Class III groundwater is not considered to be a potential source of drinking water and is of limited beneficial use.

In the NCP, EPA states the preference for Safe Drinking Water Act (SDWA) MCLs and non-zero maximum contaminant level goals (MCLGs) or other health-based standards,

criteria, or guidance for cleanup of Class I and II groundwater at CERCLA sites (55 FR 8732). Alternate concentration limits may also be used when active restoration of the groundwater to MCLs or non-zero MCLGs is not practicable (55 FR 8754). For Class III groundwaters, EPA establishes remediation levels based on specific site conditions, the beneficial use of the groundwater, and environmental receptors (55 FR 8732).

The Ground Water Management Section of the Tennessee Division of Water Supply is in the process of developing a groundwater classification system for Tennessee groundwaters (TDEC Proposed Rule 1200-4-3-.04 to .08). A draft of the proposed rule, dated May 28, 1993, was issued for public comment in June 1993; a final version is expected by the Summer of 1994. Chapter 1200-4-3-.07 of the proposed rule lists four classifications for groundwater: Class A— current or future source of water supply; Class B—current or future source of water supply, but which will not support all uses as a water supply due to contamination; Class C—not a current or future source of water supply for drinking or other beneficial use; and Class D—groundwater that flows to the surface of the ground or mixes with surface water. Groundwater in Class D can also be identified as A, B, or C. According to this classification scheme, it appears that contaminated groundwater at Chestnut Ridge OU 2 would be classified as Class B and, possibly, Class D. The water quality criteria for Class B groundwater, proposed at Chap. 1200-4-3-.08, would be the criteria for Class A groundwater (i.e., criteria established in 1200-4-3-.03, with the exception of those for fish and aquatic life) except where a groundwater use is shown to be impaired in accordance with 1200-4-3-.09(2)(c). Chapter 1200-4-3-.09 details steps to be taken to have groundwater registered as Class B, and includes a requirement for submittal of monitoring data, remediation history and plans, hydrogeologic studies, study of current and potential land uses, risk assessments, and permit applications. If the state Commissioner grants the petition for registration as Class B groundwater, water quality criteria for the contaminants of concern, as well as other permit conditions, will then be negotiated. Water quality criteria for Class D groundwater would be the surface water quality criteria (listed in TDEC 1220-4-3) for the surface water body into which the groundwater flows and would be applicable at the designated point of influence. Many comments have been received on the draft rule, necessitating revisions before final promulgation; it is expected that the classes of groundwater listed in the draft rule will remain, although the steps for remediation of groundwater may change significantly. Final determination of ARARs for any cleanup of Chestnut Ridge OU 2 groundwater, if it is deemed necessary, will depend on the chosen groundwater classification, and will be addressed further as the extent of any groundwater contamination is further characterized and remedial actions considered.

**Safe Drinking Water Act.** EPA has promulgated primary and secondary drinking water regulations applicable to public water systems that have at least 15 service connections or serve an average of at least 25 people daily at least 60 days of the year. National Primary Drinking Water Standards (NPDWS) are established in 40 CFR 141 and include MCLs and MCLGs. New drinking water standards promulgated for eight synthetic organic chemicals (52 FR 25690, July 8, 1987) added a new category of suppliers referred to as noncommunity, nontransient systems that regularly serve at least 25 people for 6 months of the year. Table 4.1 lists SDWA MCLs and MCLGs for COPCs at Chestnut Ridge OU 2. Although sulfate is also included in Table 4.1, the values listed are proposed, not final, values. EPA has deferred setting a final MCL/MCLG for sulfate pending further study (57 FR 31776, July 17, 1992).

MCLs are enforceable standards that take into consideration human health effects, available treatment technologies, and costs of treatment. MCLGs are strictly health-based standards that disregard cost or treatment feasibility and are not legally enforceable. MCLs are legally applicable to water "at the tap" but are not applicable to cleanup of groundwater or surface water. However, they may be considered as relevant and appropriate in situations where groundwater or surface water may be used for drinking water. CERCLA §121(d)(2)(A) specifically mentions that remedial actions must require a level or standard of control that at least attains MCLGs and federal ambient water quality criteria (WQC) where such goals or criteria are relevant and appropriate under the circumstances of the release. Although MCLGs and WQC are nonenforceable guidelines, Congress elevated them to a higher status by specifically mentioning them in CERCLA. Therefore, promulgated MCLGs are listed in Table 4.1. At present, EPA is planning to use the SDWA MCLs for remedial action compliance for carcinogens that have an MCLG of zero and any nonzero MCLG for remedial action compliance for systemic toxicants (55 FR 8752).

EPA has revised its drinking water standards for lead, eliminating the MCL and replacing it with a treatment technology requirement, applicable to community and nontransient, noncommunity water systems. If the "action level" of 15 $\mu$g/L for lead is exceeded at the tap, a state is required to analyze source water samples and to decide what treatment levels are necessary to minimize lead levels delivered to users from the affected distribution system. The TDEC has adopted the action level for lead (TDEC Rules Chap. 1200-5-1-.33). In the instance of contaminated groundwater at Chestnut Ridge OU 2, the action level for lead is neither legally applicable nor relevant and appropriate. However, the EPA Office of Solid Waste and Emergency Response (OSWER) has recommended that a final cleanup level of 15 $\mu$g/L for lead in groundwater usable for drinking water is protective of sensitive populations (OSWER memorandum dated June 21, 1990). This might be considered TBC guidance for remediation of lead-contaminated groundwater. The action level for lead has also been listed in the TDEC Superfund rule (discussed in Sect. 4.2.2) as a cleanup standard for groundwater to be used as a domestic water supply and would be applicable to groundwater classified as such.

Chapter 1200-5-1 of the Rules of the TDEC, as amended in January 1993, lists MCLs for public water systems that are identical to the federal MCLs. Therefore, they are not repeated here.

National Secondary Drinking Water Standards (NSDWS) regulate contaminants that affect the aesthetic qualities related to public acceptance of drinking water and are implemented in 40 CFR 143.3 as secondary MCLs. These regulations are not federally enforceable, but rather are intended to serve as guidelines for use by states in regulating water supplies. Tennessee has promulgated secondary MCLs in Chap. 1200-5-1.12 of the Rules of the TDEC. These regulations are designed to provide water to the consumer that is aesthetically pleasing, and they apply to all community water systems and to those noncommunity water systems "as may be deemed necessary" by TDEC. In that context, they would not be legally applicable to cleanup of groundwater or surface water, but may be considered as relevant and appropriate in instances where these media may provide private drinking water sources. EPA proposed NSDWS for ten additional contaminants (54 FR 22062, May 22, 1989; 55 FR 30370, July 25, 1990), and, when these are promulgated, they will be incorporated into the TDEC secondary drinking water regulations. A final NSDWS for two of these chemicals, aluminum and silver, has appeared (NPRM, 54 FR 22062; Final Rule, 56 FR 3526, January 30, 1991). SMCLs are listed in Table 4.1 and

footnoted as such, for any COPC at Chestnut Ridge OU 2 that does not have a proposed or promulgated MCL/MCLG. Although iron exceeded the SMCL, it was dropped from the list of final COCs because the risk was calculated to be below the levels required for remediation. Table 4.1 lists ARARs only for final COCs.

**Clean Water Act.** The FCAP is located in the McCoy Branch watershed. This watershed drains the south side of Chestnut Ridge southward to Rogers Quarry and ultimately discharges into the Melton Hill Reservoir on the Clinch River.

CERCLA §121(d)(2)(A) specifically states that remedial actions shall at least attain federal ambient WQC established under the Clean Water Act if they are relevant and appropriate. In determining whether any WQC are relevant and appropriate, one must consider the "designated or potential use of the surface or groundwater, the environmental media affected, the purposes for which the criteria were developed, and the latest information available" [CERCLA §121(d)(2)(B)]. Federal WQC are derived for the protection of freshwater aquatic organisms and for the protection of human health from the consumption of contaminated drinking water and/or aquatic organisms.

Chapter 1200-4-3 of the Rules of the TDEC lists seven use designation categories for Tennessee's surface waters and groundwaters. Specific water quality standards are promulgated for each use category. McCoy Branch is a tributary to the Clinch River but is not specifically listed in Tennessee's water use classification tables. The standards state that "all other streams, named and unnamed, which have not been specifically noted shall be classified for fish and aquatic life, recreation, irrigation, and livestock watering and wildlife uses." The Clinch River, which McCoy Branch drains into, is designated, in addition, as domestic and industrial water supply (Chap. 1200-4-4 of the Rules of the TDEC). Criteria for protection of recreational uses are human health criteria derived to protect the consumer from consumption of contaminated fish and are similar to the federal WQC for the protection of human health from consumption of fish alone. These criteria, or the fish and aquatic life criteria, whichever is more stringent for a particular COPC, are ARAR for the cleanup of McCoy Branch and its associated wetlands at the FCAP, and are listed in Tables 4.1 and 4.2. Federal WQC, if available, are listed in Table 4.2 for any COPC that does not have a promulgated state criterion.

### 4.2.2 Soil/Sediment

Very little legislation or guidance is available governing cleanup criteria for contaminated soils or sediments at CERCLA sites. Since the coal ash at Chestnut Ridge OU 2 is not considered RCRA waste (see Sect. 4.1), none of the RCRA regulations, including RCRA land disposal restrictions and the proposed RCRA rule addressing hazardous soils, would be ARAR.

TDEC has issued a final rule, *Inactive Hazardous Substance Site Remedial Action Program* (Rules of the TDEC, Chap. 1200-1-13), effective February 19, 1994. §1200-1-13-.08(3)(a)1.(II) and (III) of the rule are reserved for listing soil and sediment exemption criteria and preliminary remediation goals, but none have been listed in the rule as yet.

EPA has suggested cleanup values for lead in soils based on studies of blood lead levels in exposed children. The EPA OSWER Directive 9355.4-02 (dated September 7, 1989)

recommends a cleanup level for soils of 500–1000 ppm lead. These would be TBC guidance, not ARARs.

EPA Region IV has made available a summary of chemical-specific concentrations that may be used to flag contaminant levels of concern in sediments in the storm drains. The values could be used as guidance for evaluating COPCs in sediment at Chestnut Ridge OU 2, and are given in Table 4.3. These values have been calculated by NOAA and are **not intended as cleanup standards.**

## 4.3 LOCATION-SPECIFIC ARARs

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 51394). In determining the use of location-specific ARARs for selection of remedial actions at CERCLA sites, one must investigate the jurisdictional prerequisites of each of the regulations. Table 4.4 lists the major federal and state location-specific ARARs that might be pertinent to remedial actions at Chestnut Ridge OU 2.

### 4.3.1 Wetlands and Floodplains

The TDEC surveyed the site for wetlands on February 5, 1993, and TDEC concluded that over half the FCAP supports wetland vegetation and is hydrologically connected to the various waters of the state at the site (Childres 1993). A wetlands survey of Chestnut Ridge OU 2 conducted from July to September of 1991 by Larry Pounds and Rebecca Cook (Hardy et al. 1992) and a wetlands survey conducted by Barbara Rosensteel in January 1992 verified the presence of wetland areas along McCoy Branch and possibly in the FCAP area itself (Rosensteel 1992). The U.S. Army Corps of Engineers conducted a survey in March 1994 to verify and delineate any jurisdictional wetlands at Chestnut Ridge OU 2. The FCAP is located outside the 100- and 500-year floodplains of the Clinch River (Welch 1989). If any remedial alternatives are selected that would impact the 100-year floodplain of McCoy Branch, the requirements found in Executive Order 11988, 40 CFR 264.18(b), 40 CFR 6 (Appendix A), and 10 CFR 1022 would provide ARARs. If the identified wetlands are affected, consideration should be given to Executive Order 11990, 40 CFR 6 (Appendix A), 10 CFR 1022, the Clean Water Act §404, and 40 CFR 230 for applicable requirements (see Table 4.4).

### 4.3.2 Aquatic Resources

If any remedial action is taken that affects wildlife or alters McCoy Branch, which is designated by the state of Tennessee for fish and aquatic life, recreation, irrigation, and wildlife and livestock watering, then the Fish and Wildlife Coordination Act (16 USC 661 *et seq.*), the Tennessee Water Quality Control Act of 1977 (TCA 69-3-101 *et seq.*), the Clean Water Act §404, 40 CFR 230, and 33 CFR 320-330 may be applicable.

Certain portions of ORR have been designated as a DOE National Environmental Research Park (NERP), which also includes Research Areas (RSAs) located both within and outside the NERP (Parr and Pounds 1987). In addition to the NERP and its associated RSAs, DOE has also designated areas on the ORR as DOE-NERP Reference Areas (RfAs) and DOE-NERP Natural Areas (NAs). The state of Tennessee also can designate certain areas

as State Natural Areas (SNAs). Neither the NERP nor any SNAs occur at the FCAP itself (Parr and Pounds 1987). The Chestnut Ridge OU 2 and the McCoy Branch area are part of RSA No. 2 Y-12 Coal Ash Study (Parr and Pounds 1987). One reference area, RA-14, Fanny Knob white oak area, is located about 0.5 mile west of the Chestnut Ridge OU 2 site (Parr and Pounds 1987). Chestnut Ridge OU 2 is within RA-27. One natural area, NA-8, McCoy Branch embayment "barren," is located just south of the FCAP site, on the south side of Bethel Valley Road (Parr and Pounds 1987). The latter area is also registered by the state (SNA 8) and contains the largest known population of tall larkspur (*Delphinium exaltatum*), which is state listed as endangered (Parr and Pounds 1987). Because of the unique purposes and goals in establishing the NERP, the uses and restrictions that apply to these resources would be TBC guidance if remedial actions appear likely to impact the designated areas. In addition, if the McCoy Branch embayment barren is affected, the provisions of the Tennessee Natural Areas Preservation Act of 1971 (TCA 11-14-101 *et seq.*) would be applicable.

### 4.3.3 Rare, Threatened, or Endangered Species

Many of the plant and animal species known to occur on ORR are listed in the *Resources Management Plan for the U.S. Department of Energy Oak Ridge Reservation*, volumes 24 and 4 (Kroodsma 1987, Parr 1984). Although there are a number of state- and federal-listed species on ORR, there are no known federally listed threatened or endangered species at the FCAP (Kroodsma 1987, Parr 1984). A threatened and endangered plant species survey of Chestnut Ridge OU 2 done in July 1993, however, discovered the presence of a state-listed Species of Special Concern, the Lesser ladies-tresses orchid (*Spiranthes ovalis*), growing in the ash-filled area behind the dam. In addition, as noted earlier, there is a very important tall larkspur (*Delphinium exaltatum*) population just south of FCAP at the McCoy Branch NA-8. This species is state listed and is also a candidate for inclusion on the federal list.

Two new species of plants have been discovered along Chestnut Ridge and are eligible for state listing. Torrey's mountain mint (*Pycnanthemum torrei*) has been found along the south slope of Chestnut Ridge in the barrens areas and is eligible for listing as a state species of special concern (Hardy et al. 1992). Whorled Balm (*Collinsonia verticillata*), found on Chestnut Ridge, is listed by the Nature Conservancy as a globally threatened species, and is being considered for state listing (Hardy et al. 1992).

Should any remedial actions at Chestnut Ridge OU 2 impact any federally listed endangered or threatened species, the provisions found in the Endangered Species Act of 1973 (16 USC 1531 *et seq.*), 50 CFR 492, and the Fish and Wildlife Coordination Act (16 USC 661 *et seq.*) may be ARARs (see Table 4.4). If any proposed actions impact any identified state-listed endangered or threatened animal species, the Tennessee Non-Game and Endangered or Threatened Wildlife Species Conservation Act of 1974 (TCA 70-8-101 *et seq.*) may provide ARARs. The prohibitions of the Tennessee Rare Plant Protection and Conservation Act of 1985 (TCA 11-26-201 *et seq.*) do not apply to a landowner, lessee, or other person entitled to possession of the land on which the species is located (TCA 11-26-209). This also includes managers in the case of publicly owned land and those with written permission of the landowner or manager (TCA 11-26-209). These exclusions would apparently apply to ORR. However, the purpose of the statute, which is to protect and preserve rare plants, should be considered TBC guidance for any remedial actions at Chestnut Ridge OU 2.

### 4.3.4 Historic Sites and Archaeological Findings

A homestead has been discovered at the site near the Sluice Channel Area. The homestead consists of a house, shed, and outhouse. No information on the age or history of the homestead is currently available. However, there is a possibility that the structures were built before the 1960s. If the homestead meets the requirements to be eligible for listing on the National Registry of Historic Places, the appropriate state historic preservation officer should be notified for a consultation. To be eligible for listing, the homestead must meet the criteria/categories listed in 36 CFR 60.4. Age of the homestead in and of itself is not a deciding factor for eligibility; additional criteria must be met for it to be eligible for listing (36 CFR 60.4).

The ORR, as well as the surrounding region, is rich in both archaeological and historic resources. Numerous studies over the years have indicated the presence of abundant resources on and around the reservation. These surveys were summarized in Volume 3 of the *Resource Management Plan for the U. S. Department of Energy Oak Ridge Reservation* (Sanders 1984). No archaeological sites were identified at the Y-12 Plant. However, none of the surveys are recent, nor have they covered all of the specific sites where remedial action may be planned or contemplated. Several laws require that such information be obtained and documented if there is ample reason to suspect the presence of these resources [Archaeological and Historic Preservation Act (16 USC 469a-c); Archaeological Resources Recovery Act of 1979 (16 USC 470aa-ll)] (see Table 4.4).

### 4.3.5 Dams

All dams in Tennessee are subject to periodic inspection and certification under the Tennessee Safe Dams Act of 1973, as amended July 1989 (TCA, §69-12-101 *et seq.*). Although, according to the definition of "person" listed in TDEC 1200-5-7-.02(32) of the *Rules and Regulations Applied to the Safe Dam Act of 1973*, these regulations do not legally apply to the U.S. government, the substantive requirements of the regulations would be considered relevant and appropriate for CERCLA remedial actions. TDEC 1200-5-7-.06 lists standards for existing dams and addresses stability, slope protection, and emergency spillways (see Table 4.4). The earthen dam located at the FCAP is susceptible to overtopping due to its inadequate spillway capacity. The results of hydrologic analyses indicate that the dam could be overtopped even during a 1-year storm (Jones and Mishu 1986). Therefore, the dam at this time does not meet U.S. Corps of Engineers and the Tennessee Safe Dam Act requirements for the intermediate size dam category. The regulations do state, however, that if the applicant can demonstrate by engineering analysis that the dam is a safe structure and can certify that the dam is sufficient to protect against probable loss of human life downstream, the dam design can be approved by the Commissioner [TDEC 1200-5-7-.06(3)(a)].

**Table 4.1. Potential chemical-specific ARARs for groundwater and surface water at Chestnut Ridge OU 2 (μg/L)**

| Chemical | TDEC/SDWA MCL[a] | TDEC/SDWA MCLG[a] | WQC for aquatic organisms and drinking water[b] | WQC for aquatic organisms alone[b] | TDEC WQC for aquatic organisms alone[c] |
|---|---|---|---|---|---|
| Arsenic | 50 | — | 0 (0.018) | 0 (0.14) | — |
| Aluminum | 200[d] | — | — | — | — |
| Barium | 2,000 | 2,000 | — | — | — |
| Beryllium | 4 | 4 | —[e] | —[e] | 1.3 |
| Cadmium | 5 | 5 | —[e] | —[e] | — |
| Chloride | 250,000[d] | — | — | — | — |
| Chromium (VI) | 100 | 100 | —[e] | —[e] | — |
| Cobalt | — | — | — | — | — |
| Fluoride | 4,000[f] | 4,000 | — | — | — |
| Lead | TT (15)[g] | zero | —[e] | —[e] | — |
| Manganese | 50[d] | — | 30 | — | — |
| Niobium | — | — | — | — | — |
| Phthalate esters | | | | | |
| bis(2-ethylhexyl)phthalate | 6 | zero | 0 (1.8) | 0 (5.9) | 59 |
| Sulfate | 400,000/500,000[h] | 400,000/500,000[h] | — | — | — |
| Vanadium | — | — | — | — | — |
| Zinc | 5,000[d] | — | — | — | — |

[a]Federal (40 CFR 141) and state (TDEC 1200-5-1) maximum contaminant level/maximum contaminant level goal (MCL/MCLG); Tennessee state criteria are identical to the federal.

[b]The criterion value of zero for all potential carcinogens is listed in the table. Concentrations in parentheses for potential carcinogens correspond to a risk of $10^{-6}$. *Source*: EPA Region IV *Criteria Chart* (EPA 1992b).

[c]WQC for the protection of humans from consumption of aquatic organisms during recreational use (TDEC Rules, Chap. 1200-4-4). TDEC has adopted the federal WQC based on a risk of $10^{-5}$ rather than $10^{-6}$ for all carcinogens.

[d]Secondary MCL (no primary MCL/MCLG is available); this value would be TBC guidance, not ARAR.

[e]WQC withdrawn in the National Toxics Rule [57 FR 60848 (December 22, 1992)].

[f]Applies to community water systems only.

[g]TT = treatment technique; number in parentheses is an "action level" which, if measured in the 90th percentile at the consumer's tap, triggers initiation of corrosion control studies and treatment requirements.

[h]This is a **proposed** MCL/MCLG only; EPA has deferred setting a final limit pending further study.

Table 4.2. Federal and state ambient water quality criteria for COPCs in surface water at Chestnut Ridge OU 2 (μg/L)

| Chemical | TDEC WQC for protection of freshwater organisms | | TDEC WQC for recreation[c] |
|---|---|---|---|
| | Maximum[a] | 24-hour[b] | |
| Aluminum | 750[d] (federal) | 87[d] | — |
| Arsenic (III) | 360 | 190 | 0.14 (federal) |
| Barium | — | — | — |
| Beryllium | — | — | 1.3 |
| bis(2-ethylhexyl)phthalate | — | — | 59 |
| Chloride | 860,000 | 230,000 | — |
| Chromium (III) | 2499.76[e] | 297.96[e] | 670,000 |
| Copper | 18[f] | 12[f] | — |
| Fluoride | — | — | — |
| Lead | 143.83[e] | 5.60[e] | — |
| Manganese | — | — | — |
| Mercury | 2.4 | 0.012 | 0.15 |
| Nickel | 1400 | 160 | 4600 |
| Niobium | — | — | — |
| Selenium | 20 | 5 | — |
| Sulfate | — | — | — |
| pH | — | 6.5–9 | — |
| Zinc | 117[f] | 106[f] | — |

[a]One-hour average concentration not to be exceeded more than once every 3 years.
[b]Four-day average concentration not to be exceeded more than once every 3 years.
[c]WQC for the protection of humans from consumption of aquatic organisms during recreational use; TDEC has adopted the federal WQC based on a risk of $10^{-5}$ rather than $10^{-6}$ for all carcinogens.
[d]pH 6.5–9.0.
[e]Water hardness dependent criteria; based on an average water hardness of 156 mg/L(as $CaCO_3$) for McCoy Branch.
[f]Values presented are for dissolved form of this metal.
*Sources*: EPA 1992b (EPA Region IV *Criteria Chart*); Chap. 1200–4–3 of the Rules of the Tennessee Department of Environment and Conservation.

Table 4.3. NOAA[a] action levels for COPCs in sediment/soil/ash at Chestnut Ridge OU 2 (ppm)[b]

| Chemical | Low effects range | Median effects range | Overall apparent effects threshold | Degree of confidence |
|---|---|---|---|---|
| Arsenic | 33 | 85 | 50 | Low/moderate |
| Beryllium | NA[c] | NA | NA | |
| Cadmium | 5 | 9 | 5 | High/high |
| Chromium (total) | 80 | 145 | None | Moderate/moderate |
| Cobalt | NA | NA | NA | |
| Lead | 35 | 110 | 300 | Moderate/high |
| Manganese | NA | NA | NA | |
| Mercury | 0.15 | 1.3 | 1 | Moderate/high |
| Nickel | 30 | 50 | NSD[d] | Moderate/moderate |
| Phenanthrene | 0.23 | 0.14 | 0.26 | Moderate/moderate |
| Vanadium | NA | NA | NA | |

[a]National Oceanic and Atmospheric Administration.
[b]EPA Region IV memorandum from Catherine Fox, Coastal Programs Unit, dated April 15, 1991, "Suggested Guidance for Evaluating Sediment Concentration Data" (Fox 1991).
[c]Not available.
[d]Not sufficient data.

## Table 4.4. Potential location-specific ARARs for Chestnut Ridge OU 2

| Location characteristic(s) | Requirement(s) | Operating condition(s) | Citation(s) |
|---|---|---|---|
| **Wetlands** | | | |
| Presence of wetlands as defined in Executive Order 11990 §7(c) and 40 CFR 6, Appendix A §4(j) | Whenever possible, actions must avoid or minimize adverse impacts on wetlands and act to preserve and enhance their natural and beneficial values. New construction in wetlands areas should be particularly avoided unless there are no practicable alternatives. Wetlands protection considerations shall be incorporated into planning, regulating, and decision-making processes. | Agency action that involves:<br>- acquiring, managing, and disposing of lands and facilities<br>- providing federally undertaken, financed, or assisted construction and improvements<br>- conducting federal activities and programs affecting land use | Executive Order 11990<br>40 CFR 6, Appendix A<br>10 CFR 1022 |
| Presence of wetlands as defined in 40 CFR 230.3(t) and 33 CFR 328.3(b) | Action to avoid degradation or destruction of wetlands must be taken to the extent possible. Discharges for which there is a practicable alternative with less adverse impacts or those that would cause or contribute to significant degradation are prohibited.<br>If adverse impacts are unavoidable, action must be taken to enhance, restore, or create alternative wetlands. | Action involving discharge of dredge or fill material into wetlands | Clean Water Act §404<br>40 CFR 230<br>33 CFR 323 |
| **Floodplains** | | | |
| Within "lowland and relatively flat areas adjoining inland and coastal waters and other flood prone areas such as offshore islands, including at a minimum, that area subject to a one percent or greater chance of flooding in any given year." [Executive Order 11988 §6(c) and 40 CFR 6, Appendix A §4(d)] | Action shall be taken to reduce the risk of flood loss; minimize the impact of floods on human safety, health and welfare; and restore and preserve the natural and beneficial values of floodplains. The potential effects of actions in floodplains shall be evaluated and consideration of flood hazards and floodplain management ensured. If action is taken in floodplains, alternatives that avoid adverse effects and incompatible development and minimize potential harms shall be considered. | Action that involves:<br>- acquiring, managing, and disposing of lands and facilities<br>- providing federally undertaken, financed, or assisted construction and improvements<br>- conducting federal activities and programs affecting land use | Executive Order 11988<br>40 CFR 6 (Appendix A)<br>10 CFR 1022 |

**Table 4.4 (continued)**

| Location characteristic(s) | Requirement(s) | Operating condition(s) | Citation(s) |
|---|---|---|---|
| **Aquatic resources** | | | |
| Within area encompassing or affecting waters of the state of Tennessee as defined in TCA[b] 69-3-103(32) and the presence of wildlife or aquatic life | Discharge of "substances" that "will result or will likely result in harm, potential harm or detriment to the health of animals, birds, fish, or aquatic life" is prohibited. | Action involving the discharge of any pollutants into the waters of the state [see TCA 69-3-103(18) and (21) for noninclusive list] | Tennessee Water Quality Control Act of 1977 (TCA 69-3-101 *et seq.*) Stream Use Classifications (TDEC[c] Rules, Chap. 1200-4-4) |
| Within area affecting stream or river *-and-* presence of fish or wildlife resources | The effects of water-related projects on fish and wildlife resources must be considered. Action must be taken to prevent, mitigate, or compensate for project-related damages or losses to fish and wildlife resources. Consultation with the FWS[d] and/or appropriate state agency is strongly recommended for on-site actions. | Action that results in the control or structural modification of a natural stream or body of water | Fish and Wildlife Coordination Act (16 USC 661 *et seq.*) 40 CFR 6.302(g) |
| Location encompassing aquatic ecosystem with dependent fish, wildlife, other aquatic life, or habitat | Degradation or destruction of aquatic ecosystems must be avoided to the extent possible. Discharges that cause or contribute to significant degradation of the water of such ecosystems are prohibited. | Action involving the discharge of dredge or fill material into aquatic ecosystem | Clean Water Act §404 40 CFR 230 33 CFR 323 |
| Location encompassing a state-designated natural area | The scientific, scenic, recreational, and educational values of these areas must be preserved and steps taken to prevent impairment thereof. | Action which is likely to impact or adversely modify designated natural areas | TCA 11-14-101 *et seq.* |
| **Endangered, Threatened, or Rare Species** | | | |
| Presence of endangered or threatened species *-or-* critical habitat of such species as designated in 50 CFR 17, 50 CFR 226, or 50 CFR 227 | Actions that jeopardize species/habitat must be avoided or appropriate mitigation measures taken. Consultation with DOI,[e] FWS and/or state agencies, as appropriate, is recommended for on-site actions to ensure that proposed actions do not jeopardize the continued existence of the species or adversely modify or destroy critical habitat. | Action that is likely to jeopardize species or destroy or adversely modify critical habitat | Endangered Species Act of 1973 (16 USC 1531 *et seq.*) 50 CFR 402 Fish and Wildlife Coordination Act (16 USC 661 *et seq.*) |
| Presence of endangered or threatened species or critical habitat (see above citation) of same within an aquatic ecosystem as defined in 40 CFR 230.3(c) | Dredge or fill material shall not be discharged into an aquatic ecosystem if it would jeopardize such species or would likely result in the destruction or adverse modification of a critical habitat of the species. | Action involving discharge of dredge or fill material into aquatic ecosystem | Clean Water Act §404 40 CFR 230.10(b) |

**Table 4.4 (continued)**

| Location characteristic(s) | Requirement(s) | Operating condition(s) | Citation(s) |
|---|---|---|---|
| Presence of Tennessee state-listed endangered or threatened animal species as created and amended pursuant to TCA 70-8-105 | Protected species may not be taken, possessed, transported, exported, processed, sold, offered for sale, or shipped. Certain exceptions may be allowed for reasons such as education, science, etc., or where necessary to alleviate property damage or protect human health or safety. | Action impacting such species if they are identified at the site | Tennessee Nongame and Endangered or Threatened Wildlife Species Conservation Act of 1974 (TCA 70-8-101 *et seq.*) |
| Presence of Tennessee state-listed species "in need of management" | No person may knowingly destroy the habitat of such species. | Action impacting such species if they are identified at the site | Tennessee Wildlife Resources Commission Proclamation 86–29 (1991) |
| **Archaeological and Historic Resources** | | | |
| Presence of archaeological resources | The Secretary of the Interior must be advised of the presence of the data.<br>A survey of affected areas for resources and data must be conducted and steps taken to recover, protect, and preserve data therefrom or request that DOI do so. | Action involving dam construction or other alteration of terrain that might cause irreparable loss or destruction of significant scientific, prehistoric, historic, or archaeologic data | Archaeological and Historic Preservation Act (16 USC 469a-c) |
| Presence of federally owned, administered, or controlled prehistoric or historic resources *-or-* the likelihood of undiscovered resources | Cultural resources included or eligible for inclusion on the National Register of Historic Places (36 CFR 60) or National Historic Landmark Program (36 CFR 65) must be identified.<br>Action(s) that will affect such resources must be identified and alternatives to the action(s) examined and considered.<br>When alteration or destruction of the resource is unavoidable, steps must be taken to minimize or mitigate the impacts and to preserve records and data of the resource.<br>Consultation with designated officials is strongly recommended for on-site actions. | Action that will impact such resources | National Historic Preservation Act (16 USC 470a-w)<br>Executive Order 11593<br>36 CFR 800 |

**Table 4.4 (continued)**

| Location characteristic(s) | Requirement(s) | Operating condition(s) | Citation(s) |
|---|---|---|---|
| **Dams** | | | |
| Presence of dam as defined in TCA 69-12-102(3) | Construction, enlargement, repair, alteration, removal, maintenance, or operation of a dam is prohibited without first obtaining a certificate of approval and safety. Certificates are for a definite period of time, not to exceed 5 years.<br><br>Standards for existing dams:<br><br>• *Stability* - all dams shall be stable<br><br>• *Slope protection* - earth embankments shall be protected from surface erosion by appropriate vegetation or some other type of protective surface such as riprap or paving and shall be maintained; all inappropriate vegetation shall be removed from the dam<br><br>• *Emergency spillway* -<br><br>All dams shall have an emergency spillway system with capacity to pass a flow resulting from a 6-hour design storm for a hazard classification appropriate for the dam, or<br><br>if applicant can successfully demonstrate by engineering analysis that the dam is a safe structure and is sufficient to protect against probable loss of human life downstream, dam design can be approved by the Commissioner | Relevant and appropriate to remedial action involving/ impacting construction or modification of a dam at a U.S. government-owned site | Tennessee Safe Dam Act of 1973 (TCA 69-12-101 to -125); TDEC 1200-5-7-.01(1)<br><br>TDEC 1200-5-7-.06(1)<br><br>TDEC 1200-5-7-.06(2)<br><br>TDEC 1200-5-7-.06(3) |

[a]Resource Conservation and Recovery Act; definitions appear at 40 CFR 260.10.
[b]Tennessee Code Annotated.
[c]Tennessee Department of Environment and Conservation.
[d]Fish and Wildlife Service.
[e]Department of Interior.

# 5. BASELINE HUMAN HEALTH RISK ASSESSMENT REPORT

## 5.1 INTRODUCTION

The overall objective of the baseline HHRA evaluation process is to provide information necessary for making remediation decisions. The human health risk assessment methodology used in the risk evaluation is based on the *Risk Assessment Guidance for Superfund* (RAGS) (EPA 1989c). A quantitative analysis of the inorganic (metals), organic, and radionuclide analytes found in various media is used to characterize the potential risks to human health associated with exposure to these contaminants. The results of the baseline HHRA are used to (1) document and evaluate risks to human health, (2) determine the need for remedial action, (3) determine chemical concentrations protective of current and future human receptors, and (4) help select and compare various remedial alternatives (Energy Systems 1993a).

The baseline HHRA for exposure to ash, soil, sediment, surface water, and groundwater associated with the Chestnut Ridge OU 2 is presented in this report. For purposes of this baseline HHRA, Chestnut Ridge OU 2 is divided into three areas: (1) the Sluice Channel Area; (2) FCAP; and (3) the UMB. This report follows the organization of the standard risk assessment processes described in RAGS (EPA 1989c). First, the data for the different media are evaluated to determine data useability for risk assessment, then contaminants to be considered in the baseline HHRA are identified. This process is referred to as the selection of COPCs and, at this point, the representative concentrations for those contaminants included in the risk assessment are determined. Next, an assessment of the exposure potential is performed, and exposure pathways are identified. Subsequently, exposure is estimated quantitatively, and the toxicity of the COPCs are appraised. The results of the exposure and toxicity assessments are combined and summarized in the risk characterization section (Sect. 5.5).

To comprehensively evaluate the risk to human health from exposure to the various media (i.e., ash, soil, sediment, surface water, and groundwater) associated with Chestnut Ridge OU 2 and to provide risk information supportive of the project objectives, both (1) the current most reasonable and most likely exposure scenario and (2) a future reasonable maximum exposure scenario will be evaluated. Chestnut Ridge OU 2 is located on the Oak Ridge Y-12 Plant site but is not within the operational Y-12 Plant facility; access to this OU is limited/prohibited by a fence and no-trespassing signs. Because the area is currently undeveloped and is heavily vegetated, the trespasser/hunter is the most likely receptor of Chestnut Ridge OU 2 media. Another scenario would be security forces that patrol the area or field workers performing routine sampling or surveys. The assumptions used in the risk calculations (exposure duration $\times$ exposure frequency) for a trespasser/hunter or for field workers/security personnel would be similar; therefore, the lower-bound risks estimates would not be significantly different. The exposure pathway to be evaluated for the trespasser/hunter is external exposure to radionuclides in the surface ash. This trespasser evaluation represents a lower-bound estimate of risk and is not likely to overestimate risks at Chestnut Ridge OU 2 under current conditions. Therefore, if COPCs have risks exceeding the EPA target level, under this lower-bound exposure assessment, they may be considered as high priority for cleanup.

An upper-bound on the risk to human receptors will be evaluated using residential parameters. Evaluation of the residential scenario, in a Human Health Baseline Risk Assessment, is mandated/required by EPA/TDEC, and the exposure parameters used in the evaluation/calculations are provided in the *Risk Assessment Guidance for Superfund* document. An industrial worker scenario might seem more appropriate; however, EPA/TDEC believe the residential scenario is essential for them to make remedial decisions. Because Chestnut Ridge OU 2 is on the Y-12 reservation and because of the unique location of this OU (i.e., between two sanitary landfills), such a conservative scenario is unlikely. However, the intent of this evaluation is to provide managers with potential risks that are unlikely to underestimate exposure to future receptors of the Chestnut Ridge OU 2 media.

The routes/pathways of exposure to be evaluated for the residential exposure scenario include (1) dermal contact with the ash, dry sediments, and shallow surface soil beneath the ash; (2) incidental ingestion of the ash, dry sediments, and shallow surface soil beneath the ash; (3) inhalation of wind generated ash/dust, dry sediment/dust, and surface soil/dust; (4) external exposure to radionuclides in the ash, dry sediments, and the shallow surface soil beneath the ash; (5) ingestion of homegrown vegetables and fruit (garden grown in the ash); (6) ingestion of surface water; (7) dermal contact (while wading) with surface water, ash, and wet sediments; (8) ingestion of groundwater; and (9) dermal contact (while showering) with surface water and groundwater. Analytes (i.e., COPCs) with risks below the EPA target level, for this conservative residential evaluation, should require no further action in terms of remediation.

The following sections describe the methodology used in evaluating the Chestnut Ridge OU 2 analytical data, physical characteristics, potential pathways, and receptors to quantify the potential risk to human health from exposure to Chestnut Ridge OU 2 COPCs.

## 5.2 IDENTIFICATION OF CHEMICALS OF POTENTIAL CONCERN

### 5.2.1 General Data Collection Considerations

Source history is provided in Chaps. 2 and 3. Chapter 2 provides the location of Chestnut Ridge OU 2. Chapter 3 describes the OU history and current conditions.

The data presented in this baseline HHRA are from two sampling phases. The first investigation was conducted by CH2M HILL to define the nature and extent of contamination at the site, assess the potential for off-site contaminant migration, and evaluate the risks to human health from exposure to the contaminants; the results of this investigation are reported in a Phase 1 RI Report (Energy Systems 1991). Upon review of this RI by the EPA Region IV and the TDEC, it was determined that further investigation was required, which included (1) insufficient background data for all media; (2) lack of a residential scenario in the baseline HHRA; (3) ecological risk not addressed; and (4) lack of data to support the chemical characteristics and volume of ash in this OU. The results of the previous human health risk assessment, performed in the previous RI (Energy Systems 1991) report, are not included in this baseline HHRA.

To address these comments, a Phase 2 Sampling Plan was developed (Energy Systems 1993c) by CDM Federal for Energy Systems, to further characterize the site and the potential risks to human health and the environment. The data from both phases of sampling

were reviewed and are included in this baseline HHRA. Phase 1 data included surface water, groundwater, soil, and ash samples from the UMB and groundwater from the Sluice Channel Area, as well as background samples for soil. The Phase 2 data included ash, groundwater, and soil samples from the Sluice Channel Area; surface water, sediment, ash, and soil (beneath ash pond) from the FCAP; and surface water, sediment, and groundwater from the UMB area. The Phase 2 data also included background sediment, surface water, and soil samples. Note, however, that background levels for analytes detected in the Chestnut Ridge OU 2 soils will be compared with the results from the *Background Soil Characterization Project for the Oak Ridge Reservation* (Energy Systems 1993b). Refer to the Phase 1 RI Report (Energy Systems 1991) and the Phase 2 Sampling Plan (Energy Systems 1993c) for the exact sampling locations, number of samples collected, sampling and analysis guidance used, analysis and analytical techniques used, and the Quality Assurance/Quality Control methods used.

### 5.2.2 General Data Evaluation Considerations

The data considered for use in the baseline HHRA were provided to the Energy Systems Risk Analysis Section in two phases. Phase 1 data were provided by OREIS via electronic files; Phase 2 data were provided by CDM Federal.

The Chestnut Ridge OU 2 comprises three distinct geographical areas, which include the Sluice Channel Area, the FCAP, and the UMB. The results of the following data evaluation (Sect. 5.2.3) illustrate why this OU was evaluated (in terms of human health risk) as three separate areas (Sluice Channel Area, FCAP and UMB), with the exception of the Chestnut Ridge OU 2 groundwater.

### 5.2.3 Data Evaluation/Preparation

The data received from OREIS and CDM Federal were read into a Statistical Analysis System data set, and were evaluated in accordance with RAGS (EPA 1989c) for the process of selecting the COPCs to be evaluated quantitatively in this baseline HHRA. In the following sections of this baseline HHRA, the results (data) from Phase 1 and the Phase II sampling are evaluated separately and then combined into one Chestnut Ridge OU 2 data set.

The data for Chestnut Ridge OU 2 are summarized and tabulated in Sect. 5.2.3.6 of this baseline HHRA. The data included in this baseline HHRA result from the inorganic, organic, and radionuclide analyses of soil, ash, sediment, groundwater, and surface water samples from various sampling locations within this OU.

#### 5.2.3.1 Data validation considerations

Details of the validation process can be found in the *Remedial Investigation Report for Chestnut Ridge OU 2 (Filled Coal Ash Pond/McCoy Branch), at the Oak Ridge Y-12 Plant, Oak Ridge, Tennessee* (Energy Systems 1991) for the Phase 1 data and in the *Phase 2 Sampling Plan for Chestnut Ridge OU 2* (Energy Systems 1993b) for the Phase 2 data.

In summary, Phase 1 chemical data were 100% validated, and radiological data were not validated. Phase 1 QC data were not available to the Risk Analysis Section for confirmatory evaluation. However, Phase 2 data were 100% validated by C.C. Johnson & Malhotra, P.C.

(CCJM). Some data evaluation/validation was performed by CCJM which included comparison of the samples with associated blanks using two criteria: (1) if the concentration of the analyte in the sample was at least five times greater than the maximum concentration detected in the blank, then the analyte concentration was not changed from the reported value and (2) if the concentration of the analyte in the sample was greater than the concentration found in the blank but less than five times the concentration in the blank, then the analyte concentration was changed to reflect the concentration reported in the blank, and a nondetect qualifier (U) was also added to the data set for these species. For common laboratory contaminants such as acetone, 2-butanone, methylene chloride, toluene, and phthalate esters, if the concentration of the analyte in the sample was at least 10 times greater than the associated blank maximum concentration detected in the blank, then the analyte concentration was not changed from the reported value [similar to criteria (1) described previously].

The data were evaluated by the Energy Systems Risk Analysis Section for the baseline HHRA by the methods described in RAGS (EPA 1989c). Data qualifiers (U = not detected, J = estimated value, and R = rejected value) were also evaluated where: (1) J were treated as detected analytes and were included in the baseline HHRA data set; (2) U were treated as nondetected analytes, and the reported detection limit was used as the concentration value; (3) UJ were treated as data flagged with U; (4) J, DL were treated as data flagged with J; and (5) R were rejected from the baseline HHRA data set. When an analyte was not detected (U or UJ) in any of the soil samples, a separate data set (i.e., a nondetected analytes data set) was created; these data are evaluated and discussed in Sect. 5.2.3.10.

### 5.2.3.2 Original samples compared with duplicate samples

CDM Federal provided Phase 2 data (to the Risk Analysis Section) with the results of a duplicate/original sample comparison test already incorporated. For original samples with associated duplicates, a comparison of each analyte concentration (original versus duplicate) was performed by CDM Federal, and the highest concentration for each analyte was selected for use in this baseline HHRA [i.e., the original sample concentrations provided to the Energy Systems Risk Analysis Section by CDM Federal were always greater than (or equal to) the duplicate concentrations]. The Risk Analysis Section verified this evaluation and found 13 additional original/duplicate comparisons that necessitated further modifications to the data set; these changes were made prior to further data evaluation.

A similar comparison of the original versus duplicate concentrations was performed by the Risk Analysis Section for the Phase 1 data to achieve consistency with the Phase 2 data. Because this modification to the original data set (to be used in the baseline HHRA) is the most conservative in terms of analyte concentrations, no further evaluations of duplicate concentrations were considered.

**Phase 1 data.** For the Phase 1 surface water data, both wet season and dry season samples were taken at numerous sampling locations. Upon comparison of the analyte concentrations for these data, no statistical differences were found [using a t-test with a 95% confidence level (95% CL)] between the wet and dry season samples. Therefore, both dry and wet season sample results were used and combined in the Phase 1 surface water data set.

For the Phase 1 groundwater data, two of the eight wells were sampled twice. For these two wells, a comparison was made (analyte by analyte) and the higher analyte concentrations were assigned to that well.

For the Phase 1 soil (beneath ash) data, samples were taken at two sampling depths (0–6 in. and 6-12 in.). Upon comparison of the analyte concentrations for each depth, no statistical differences were found (using a t-test with a 95% CL) between the 0–6-in. and 6–12-in. depth. Therefore, data from both depths were used and combined in the Phase 1 soil (beneath ash) data set.

**Phase 2 data.** For the Phase 2 surface water data, only one location was sampled in both the wet season and the dry season; consequently, a statistical test could not be performed (on only one sample) to compare wet and dry season data. Therefore, to be conservative, a comparison was made (analyte by analyte) between the wet and dry weather analyte concentrations, and the higher value was used as the analyte concentration for this sample location. A similar comparison was performed for the data from the ecological toxicity test surface water sampling location. The results for these comparisons (i.e., the higher analyte concentration) were used in the Phase 2 surface water data set in this baseline HHRA.

For the Phase 2 sediment data, only one location was sampled in both the wet season and the dry season. Again, to be conservative, a comparison was made (analyte by analyte) between the wet and dry weather analyte concentrations, and the higher values were used as the analyte concentrations for this sediment sample location.

#### 5.2.3.3 Further Phase 1 data preparation

Radionuclide analyses for the Phase 1 data were reported as *total* radium (pCi/g or pCi/L) and *total* uranium (mg/kg or μg/L). For this baseline HHRA, the *total* radium concentration was assigned to $^{226}Ra$ (toxicity values for $^{226}Ra$ are the most conservative, that is, give the largest risk, among the radium isotopes). In addition, to be conservative, the reported *total* uranium concentration (mg/kg or μg/L) was converted to $^{238}U$ activity (pCi/g or pCi/L); the toxicity values for $^{238}U$ are the most conservative for the uranium isotopes. The only other radionuclides analyzed for in the Phase 1 data were $^{137}Cs$, $^{228}Th$, and $^{234}Th$.

#### 5.2.3.4 Further Phase 2 data preparation—determination of areal variation in contamination

To determine the similarities and differences between the three geographical areas in Chestnut Ridge OU 2, a number of statistical tests were conducted on the Phase 2 data set. Comparisons were made between the upgradient (Sluice Channel Area) groundwater analyte concentrations and those downgradient (UMB) from the FCAP; no statistical differences (using a t-test with a 95% CL) were found for the detected analytes. Therefore, all groundwater data were combined into one Phase 2 data set. Comparison of UMB surface water analyte concentrations with FCAP surface water also indicated no statistical differences. In addition, no statistical differences were evident upon comparison of FCAP and UMB sediment analyte concentrations. Based on these comparisons, no statistical differences were found to suggest separating this OU into three areas.

However, upon comparison of detected analyte concentrations found in the ash in the Sluice Channel Area with ash in the FCAP, notable statistical differences (using a t-test with a 95% CL) were identified for several analytes (Be, K, Mn, Na, $^{228}Th$, $^{232}Th$, Tl, and V). Therefore, considering these differences and those noted in Sect. 5.2.3.5, Chestnut Ridge OU 2 was divided into three areas (Sluice Channel Area, FCAP, and UMB) for the purposes of this risk assessment, with the exception of Chestnut Ridge OU 2 groundwater.

A comparison was made between the detected analyte concentrations found in the Sluice Channel Area ash versus those detected in the Sluice Channel Area soil (beneath ash). Because statistical differences (using a t-test with a 95% CL) were found for Ba, Ca, $^{230}$Th, and $^{234}$U, these two media were evaluated separately in this baseline HHRA. Note that, for the FCAP, surface ash data were combined with the "deeper" pond ash data for this baseline HHRA.

### 5.2.3.5 Phase 1 versus Phase 2 data comparisons

Comparisons were made between the Phase 1 and Phase 2 data to (1) further evaluate contamination variability across the three areas and (2) to determine the feasibility of combining both sets of data into one data set for risk assessment purposes.

Comparisons were made between the groundwater data collected in Phases 1 and 2; statistical results (using a t-test with a 95% CL) indicated that only one analyte ($^{238}$U) differed substantially between the two sampling phases. In addition, the surface water data (for both phases) from the UMB showed only one statistical difference (for zinc) for all the detected analytes. Therefore, since only two differences were found [among many detected analytes (17)], the decision was made by Energy Systems' Risk Analysis Section to combine the data from both Phase 1 and Phase 2 sampling efforts.

Comparisons were made between the soil (beneath ash) in the Sluice Channel (Phase 2 data) and in the UMB (Phase 1 data). Four analytes were determined to have statistically different (using a t-test with a 95% CL) concentrations (Hg, Ni, Na, and Zn). In addition, upon comparison of the ash from the Sluice Channel Area (Phase 2) versus ash from UMB (Phase I), one analyte ($^{228}$Th) was found to be statistically different for these two areas. The results from these comparisons also suggest that the Chestnut Ridge OU 2 should be divided into three areas to meet the risk assessment requirements.

**Final data set.** Figure 5.1 presents a map of the Chestnut Ridge OU 2 showing how this OU is divided into three areas (Sluice Channel Area, FCAP, and UMB) for risk assessment purposes.

In summary, Phase 1 data were taken in the UMB for ash, soil (beneath ash), groundwater, and surface water; groundwater samples were also taken in the Sluice Channel Area. Phase 2 data were taken from all three areas and include (1) ash, soil (beneath ash), and groundwater for the Sluice Channel Area; (2) ash, soil (beneath ash-pond), sediment, and surface water for the FCAP; and (3) sediment, surface water, and groundwater for UMB. The reader should note that few radionuclides and VOCs were analyzed in the Phase 1 data; however, many of these analytes were included in the Phase 2 data.

Based on the data evaluations described previously (Sect. 5.2.3) and the Energy Systems' Risk Analysis Section's best professional judgment, Phase 1 and Phase 2 data were combined into one Chestnut Ridge OU 2 data set (hereafter referred to as the Chestnut Ridge OU 2 data set). The Chestnut Ridge OU 2 was divided into three areas (hereafter referred to as Sluice Channel Area, FCAP, and UMB) for ash, soil (beneath ash), sediment, and surface water; the groundwater data were evaluated as a fourth area (referred to as Chestnut Ridge OU 2 groundwater), which includes all groundwater wells (both up- and downgradient of FCAP) in this OU.

### 5.2.3.6 Tentatively identified compounds

Tentatively identified compounds (TICs) are analytes not included on the Target Analyte List (TAL) or on the TCL and for which both the identity and concentration are questionable. TICs were not provided to the Risk Analysis Section for the Phase 1 data. For the Phase 2 data set, TICs were found in original and duplicate samples of ash, sediment, soil, surface water, and groundwater and are listed in Appendix J, Table J.1. According to RAGS (EPA 1989c), when only a few TICs are present, compared to TCL and TAL chemicals, and when historical and/or site-specific information about such analytes are not available, the TICs are generally not included in the risk evaluation. The results illustrated in Table J.1a show that the number of TICs are less than 10% of the number of organic target analytes. Table J.1b shows the frequency of occurrences for the TICs as well as the concentrations. No further evaluation of TICs was performed.

### 5.2.3.7 Summary statistics

Table J.2 lists the summary statistics for the Chestnut Ridge OU 2 data set. Separate tables are given for the Sluice Channel Area, FCAP, UMB, and Chestnut Ridge OU 2 groundwater analytes; these tables are divided into the various sampled media (i.e., ash, soil, sediment, surface water, and groundwater). These tables include the (1) analyte type (inorganic, radionuclide, and organic); (2) number of detected concentrations per number of observations (i.e., frequency of detection); (3) minimum, maximum, and mean concentrations; and (4) upper 95% confidence limit (UCL95) on the mean. The reader should note that for analytes with all non-detected values (i.e., chemicals that were analyzed for but were not detected at the detection limit of the analytical method used for its analysis) the UCL95 reported in this table is the maximum reported detection limit.

Essential nutrients including calcium, copper, iron, magnesium, phosphorus, potassium, silicon, and sodium (Dunne 1990) have little or no adverse effects at the detected levels reported in this study, and these nutrients are not expected to have been used in any of the Chestnut Ridge OU 2 operations. They have been included in the summary statistics for the purpose of reporting all of the available information but are not considered to be COPCs.

### 5.2.3.8 Comparison with background soil samples and reference samples

**Background soil sample comparisons.** Many naturally occurring soil constituents also occur as site-related contaminants, that is, they are detected during the sampling and analyses process along with the site contaminants. Therefore, site-related contaminants must be differentiated from background constituents to ensure that risk management decisions are based on risk posed by contaminants and not risk from background constituents. This aspect of the COPCs selection process is carried out by using the results from the *Background Soil Characterization Project for the Oak Ridge Reservation* (Energy Systems 1993b). Data collected during the site investigation of a specific hazardous waste site (e.g., Chestnut Ridge OU 2) should be compared to background data to identify COPCs. In most cases, an analyte detected in soil is assumed to be related to site activities and is therefore a COPC; however, if the detected analyte concentration is less than the background concentration, it should not be considered a COPC.

Analytes found in the Chestnut Ridge OU 2 soils and reported in the summary statistics (Table J.2) were compared with those analytes found in naturally occurring background soil

on the ORR in the Chepultepec and Copper Ridge formations (Energy Systems 1993b) (refer to Table J.3); the FCAP and the UMB are located in the Chepultepec Formation, and the Sluice Channel Area is located in the Copper Ridge Formation. The median Chestnut Ridge OU 2 analyte concentration for each analyte was compared to the upper and lower 95% confidence bound on the median of the background data (Energy Systems 1993b). Analytes with a median concentration less than the upper 95% confidence bound, for the Chepultepec or the Copper Ridge formations (for the FCAP/UMB and Sluice Channel Area, respectively), were not considered to be significantly different from ORR background soils and were not considered to be COPCs for this baseline HHRA.

**Reference sample comparisons.** Reference samples were taken specifically for this Chestnut Ridge OU 2 project. Sediment reference samples were taken from stream sediment from the Walker Branch and from the White Oak Creek headwaters; surface water reference samples were taken from stream water from Walker Branch and the White Oak Creek and from a reference pond. The sediment and surface water Chestnut Ridge OU 2 data were compared with these reference data; if the Chestnut Ridge OU 2 sediment and surface water maximum analyte concentrations were less than their respective mean reference concentrations, the analytes were not considered to be COPCs related to this OU (Table J.4).

### 5.2.3.9 Toxicity screening of COPCs

A concentration-toxicity screen was applied to the Chestnut Ridge OU 2 data set (for analytes with at least one detected value) using guidance in RAGS (EPA 1989c). The objective of this screening procedure was to identify those analytes most likely to contribute significantly to the risks and hazard indices (HIs) (i.e., those analytes with a negligible contribution to the total adverse health effects posed by site contaminants were identified and were not included on the COPC list). For the ingestion, inhalation, and external exposure toxicity values [i.e., reference doses (RfDs) and slope factors (SFs)], a toxicity score was calculated for each analyte by multiplying the analyte's maximum detected concentration by the toxicity value (i.e., conc. × SF and conc. × 1/RfD). Individual analyte toxicity scores were summed to obtain a total chemical score; a ratio of the specific analyte toxicity score to this total chemical score approximates the relative risk (or HI) for each analyte. Analytes with a relative risk or HI of less than 1% (i.e., less than 1% for all available toxicity scores) were not considered further (as COPCs) in this baseline HHRA. Note that this toxicity screen was performed for each area (Sluice Channel Area, FCAP, UMB, and Chestnut Ridge OU 2 groundwater), and radionuclides were evaluated separate from the nonradionuclides (via EPA guidance; EPA 1989c). The results of the toxicity screening can be found in Tables J.5 and J.6.

**Analytes dropped (via toxicity screening) versus preliminary remediation goals.** For the analytes eliminated from the COPC list based on the toxicity screening (Tables J.5 and J.6), comparisons were made with the appropriate preliminary remediation goals (PRGs), that is, PRGs at a cancer risk level $\geq$ 1.0E-06 and/or a noncarcinogenic (i.e., systemic) toxicity level $\geq$ 1.0 (refer to Table J.7). The maximum detected concentration for these analytes in soil, ash, and sediments were compared with the residential soil ingestion PRGs (radionuclides were compared with ingestion plus external exposure PRGs); the maximum detected concentration for analytes in surface water and groundwater were compared with the residential water ingestion PRGs. For two analytes ($^{137}Cs$ in UMB ash and $^{235}U$ in UMB sediment), the maximum detected concentration was greater than the corresponding PRG (at

a risk level of 1.0E-06); therefore, to be conservative (in terms of risk to human health), these two analytes were included on the COPC list for this baseline HHRA.

### 5.2.3.10 Summary of background comparison, toxicity screening, and essential nutrient results

Table J.8 summarizes/lists those analytes not included on the COPC list because of (1) a comparison with background soil data (Sect. 5.2.3.8), (2) a comparison with reference sediment and surface water data (Sect. 5.2.3.8), (3) the toxicity screening assessment results (Sect. 5.2.3.9), or (4) their use as essential nutrients (Dunne 1990) in the human body (Sect. 5.2.3.7).

### 5.2.3.11 Comparison of nondetected analyte concentrations with preliminary remediation goals

Summary statistics for the nondetected Chestnut Ridge OU 2 analytes are included in Table J.2; these nondetected analytes are not believed to be associated with the Chestnut Ridge OU 2 process. For the nondetected analytes that could be evaluated quantitatively (i.e., SFs and/or RfDs are available), comparisons were made with PRGs. Preliminary remediation goals are initial guidelines (protective of human health) that are based on readily available information and comply with ARARs and are used in the selection of remedial alternatives (EPA 1991a). The maximum reported concentration for each nondetected Chestnut Ridge OU 2 analyte was compared with the noncarcinogenic (HI = 1.0) and carcinogenic (risk = 1.0E-04 and risk = 1.0E-06) residential ingestion (soil or water) PRGs; residential PRGs were used to be conservative. Table J.9 shows the Chestnut Ridge OU 2 analytes that had nondetected analyte concentrations above the target PRG concentrations; therefore, the detection limits of the analytical methods used for these analytes were too high (in terms of evaluating risk to human health). Note that analytes that were always nondetected (i.e., frequency of detect was zero) will not be considered further and are not included on the COPC list for this baseline HHRA.

### 5.2.3.12 Comparison of FCAP soil (beneath ash pond) versus PRGs

Three soil samples were collected in the FCAP just beneath the ash pond (~30 ft deep); in terms of risk to human health, no realistic exposure pathways to this deep soil are anticipated. Furthermore, any potential exposure to soil contaminants will be addressed by evaluating risks to soil (beneath ash) in the Sluice Channel and in UMB (where the exposure routes are currently accessible because the ash is only a few inches deep). Therefore, the analytes detected in the FCAP soil (beneath the ash pond) samples are not included on the COPC list for this baseline HHRA. However, a comparison of the FCAP analyte concentrations (i.e., UCL95 or maximum concentration) found in the soil (beneath the ash pond) with residential PRGs (ingestion and external exposure) was performed. The results of this analysis are shown in Table J.10. Note that one analyte (beryllium— ingestion pathway) has a concentration that is equivalent to a cancer risk of 1.0E-05; all other analytes have concentrations less than PRGs (at risk = 1.0E-06 and/or HI = 1.0). These soil samples will not be considered further in this baseline HHRA.

### 5.2.4 Summary of Chemicals of Potential Concern

Table 5.1 lists the detected (i.e., at least one detected value) COPCs to be evaluated quantitatively in this baseline HHRA and their representative concentrations; complete summary statistics information can be found in Table J.2. The representative concentration was determined according to RAGS (EPA 1989c) and by comparing the UCL95 to the maximum detected value; the smaller of these two values is reported as the representative concentration. Identification of the COPCs was based on the methodology from RAGS (EPA 1989c), and the number of COPCs that could be quantitatively evaluated was limited by the availability of chemical-specific EPA-approved toxicity information (i.e., SFs and RfDs).

The COPCs for Chestnut Ridge OU 2 are specified for the four areas (Sluice Channel, FCAP, UMB, and Chestnut Ridge OU 2 groundwater) and the five media (soil, ash, sediment, surface water, and groundwater); a total of 11 inorganics, 9 radionuclides, and 1 organic are the COPCs associated with these areas and media.

With the exception of the surface water data, the analytical laboratory reported detected concentrations for *total* chromium, and although chromium may exist as species of low toxicity, the analytical data cannot substantiate this belief. Therefore, to ensure that human health is protected, chromium was assumed to be in its most toxic form (i.e., chromium VI) for the purposes of this baseline HHRA. For the surface water data (FCAP and UMB), three out of six samples collected were analyzed specifically for chromium VI as well as for *total* chromium; however, chromium VI was not detected in any of these samples (i.e., zero out of three). Therefore, for the surface water data set used in this baseline HHRA, the *total* chromium results were assumed to be in its second-most toxic form (i.e., chromium III).

Bis(2-ethylhexyl)phthalate, a common laboratory contaminant, is not believed to be associated with Chestnut Ridge OU 2 operations and therefore is not believed to be a COPC; however, no statistical tests/analyses allowed it to be eliminated from the COPC list for this baseline HHRA. These data were 100% validated and CCJM presumably evaluated the data against blanks using the 5× and 10× rule.

Table 5.2 lists the COPCs (with at least one detected value) for Chestnut Ridge OU 2 that could only be evaluated qualitatively in this baseline HHRA (i.e., no current EPA-approved SFs or RfDs exist for these analytes). These analytes (eight inorganics and two organics) will be considered qualitatively in the toxicity assessment section of this baseline HHRA.

## 5.3 EXPOSURE ASSESSMENT

An exposure assessment combines information about site characteristics and site-related data with the exposure assumptions used by the risk assessor. The objectives of the exposure assessment are to determine or estimate the magnitude, frequency, and duration of potential human exposure to site contaminants by:

- characterizing the exposure setting,
- identifying exposure pathways, and
- quantifying exposures.

### 5.3.1 Characterization of Exposure Setting

Characterization of the exposure setting involves identifying the general physical characteristics of the site (e.g., climate, vegetation, soil types) and the characteristics of the populations on or near the site. This characterization ensures that all potential contaminant migration pathways and potential receptors are evaluated in the risk assessment. Details of the physical and environmental characteristics of the Y-12 Plant and the Chestnut Ridge OU 2 site can be found in Chap. 2 and Sect. 3.1 of this RI, in the previous *RI Report for Chestnut Ridge OU 2* (Energy Systems 1991), and in the *Phase II Sampling Plan for Chestnut Ridge OU 2* (Energy Systems 1993c).

The Y-12 Plant facility is owned by DOE, consists of ~800 acres, and is located along the northeastern boundary of the ORR, adjacent to the city of Oak Ridge. Chestnut Ridge OU 2 is located on the Y-12 Site but is not within the operational Y-12 Plant facility (refer to Chap. 2 of this RI). The area is currently inactive, undeveloped, and heavily vegetated; two small ponds were observed in the northeast and northwest corners of the FCAP that are fed from small springs. The site is in a controlled access area patrolled regularly by security personnel; access to this OU is also limited by a fence and no trespassing signs.

Several businesses are located on Bethel Valley Road ~0.75 mile from this OU, and the Clark Recreational Center/Park is ~1.25 miles away. Oak Ridge residents are located ~1 mile north of Chestnut Ridge OU 2; the residential area is north of the Y-12 Plant ORR boundary. Because the Chestnut Ridge OU 2 area is currently inactive, undeveloped, heavily vegetated, and in the controlled access area, the trespasser/hunter is the most likely receptor of the Chestnut Ridge OU 2 media. The exposure pathway that will be evaluated for the trespasser/hunter is external exposure to radionuclides in the surface ash. A hypothetical future residential land use will also be evaluated, although such a conservative scenario is unlikely. In addition, children are sensitive subpopulations under the residential scenario.

### 5.3.2 Identification of Exposure Pathways

Exposure pathways of concern are identified by evaluating all of the components (source, transport medium, exposure point, potential receptors, and routes of exposure) necessary to complete the potential exposure pathway. For an exposure pathway to be considered complete, each of these components must be identified and linked to each of the other components. Routes of exposure and potential receptors are crucial in identifying the validity of an exposure pathway.

The source of contamination for Chestnut Ridge OU 2 is the coal ash. From infiltration, percolation, and runoff, contaminants associated with the ash could be transported to the soil beneath the ash, the surface water/sediments, and the groundwater. A current on-site receptor (e.g., trespasser) could be exposed to the ash, and a future hypothetical residential on-site receptor could be exposed to all of these potentially contaminated media.

In this Chestnut Ridge OU 2 baseline HHRA, potential health effects from exposure to ash and potentially contaminated soils, sediment, surface water, and groundwater associated with this OU were considered. To comprehensively evaluate the risk to human health from exposure to the media associated with the Chestnut Ridge OU 2 and to provide risk information supportive of the project objectives, both the current most reasonable (and/or most likely) exposure scenario and the future hypothetical reasonable maximum exposure

scenario have been evaluated. The trespasser represents the current exposure conditions to the Chestnut Ridge OU 2 media, in this case, ash. The exposure pathway that will be evaluated for the trespasser includes external exposure to radionuclides in the ash.

Risks determined for a hypothetical future residential receptor are conservative and unlikely to underestimate exposure to the Chestnut Ridge OU 2 media. The routes/pathways of exposure that will be evaluated include (1) dermal contact with the ash, dry sediments, and shallow surface soil beneath the ash; (2) incidental ingestion of the ash, dry sediments, and shallow surface soil beneath the ash; (3) inhalation of wind generated ash/dust, dry sediment/dust, and surface soil/dust; (4) external exposure to radionuclides in the ash, dry sediments, and the shallow surface soil beneath the ash; (5) ingestion of homegrown vegetables and fruit (garden grown in the ash); (6) ingestion of surface water and groundwater; (7) dermal contact (while wading) with surface water, ash, and wet sediments; and (8) dermal contact (while showering) with surface water and groundwater.

Note that for a resident to be exposed to soils (in the Sluice Channel Area and UMB), he/she would have to dig through the first few inches of ash. Exposure to dried sediments (in UMB and FCAP) might result from exposure near the banks of the UMB or from dredging the sediments in the UMB and/or the FCAP ponds. Exposure to the FCAP surface water (while showering) is evaluated in this baseline HHRA but is unlikely considering the small/low-flow springs feeding the small ponds. The conceptual site model for exposure to the Chestnut Ridge OU 2 media for both the trespasser and the hypothetical future resident is shown in Fig. 5.2.

### 5.3.3 Quantification of Exposure

Exposure, in the context of risk to human health, is defined as the direct contact of a person with a chemical or physical agent. To quantify exposure, one must determine exposure concentrations and calculate chemical intakes for the various exposure pathways identified for the site. The exposure concentrations and potential exposure pathways for the Chestnut Ridge OU 2 media are considered quantitatively in this section.

Exposure to contaminants can be evaluated quantitatively by developing the chronic daily intake (CDI) of a chemical (also termed "intake" or "dose" for external exposure to radionuclides). For this Chestnut Ridge OU 2 baseline HHRA, the CDI is the amount of a contaminant that an individual takes into his/her body per day via ingestion, inhalation, or dermal contact; dose is the total exposure to radionuclides. The first consideration in deriving the CDI is the methodology employed in the development of an exposure concentration, which is the amount of each chemical in the various media to which receptors are exposed. To calculate the CDI, one evaluates the exposure concentration in the context of the scenario, exposure pathway, and chemical-specific exposure variables, such as duration of exposure and intake rate. Once the CDIs for the various pathways are determined, risk and HIs can be calculated. The quantification of exposure and calculation of the CDI for the hypothetical future resident and the current trespasser are discussed in Sects. 5.3.3.1 and 5.3.3.2, respectively.

The potential direct exposure pathways associated with the hypothetical on-site residential land use scenario are (1) incidental ingestion of soil/ash/dry-sediment, (2) dermal contact with soil/ash/dry-sediment, (3) inhalation of soil/ash/dry-sediment dust, (4) external exposure to the radionuclides in the soil/ash/dry-sediment, (5) ingestion of surface water and

groundwater, (6) dermal contact with surface water and groundwater, and (7) ingestion of homegrown vegetables/fruits. The potential exposure pathway for the trespasser scenario is external exposure to radionuclides in the ash. The representative concentrations of the COPCs are the concentrations used to quantify exposures via these exposure pathways.

Tables J.11 and J.12 list the exposure variables associated with each exposure route considered for the on-site resident and the on-site trespasser, respectively; the equations used to calculate the CDIs are also given in these tables. The variables used in each exposure equation have been derived from standard intake rates, skin surface areas, and adherence factors (EPA 1989b, 1989c, 1991a, 1991c, 1992a). Estimations were made for some variables using the best professional judgment of the Risk Analysis Section. For dermal contact with ash, sediments, and surface water (while wading in the surface water) scenario, an exposure frequency of 3 hours/day for the months of April through September was estimated. In addition, for this wading scenario, the adherence factor for the ash and sediments was assumed to be one-half that for soil; guidance in RAGS (EPA 1989c) suggests that the adherence factors of some sediments are likely to be much less than for soils because the water can wash the sediment off the skin. For the trespasser external exposure scenario, a gamma shielding factor of zero was estimated and a gamma exposure time factor of 0.039, which corresponds to a 2-week exposure per 52 weeks (i.e., 2 weeks/year), was used.

#### 5.3.3.1 Derivation of CDIs for the residential scenario

The hypothetical residential exposure scenario considered in this baseline HHRA is conservative. As a result of the statistical data evaluation process described in Sect. 5.2, the concentration data set of Chestnut Ridge OU 2 COPCs for use in this baseline HHRA was compiled (Table 5.1). The upper 95% confidence limit on the mean concentration was assumed to be representative of the analyte concentration (titled "representative concentration" in the tables) and, with the exception of the vegetable/fruit ingestion pathway, was used in all calculations of the CDI (as well as dose, risk, and HI). This upper confidence limit was used to ensure that the exposure concentrations were not underestimated. Refer to Sect. 5.2 for a complete statistical evaluation of the data and the list of COPCs to be evaluated quantitatively in this baseline HHRA.

Table J.11 lists the exposure variables associated with each exposure route considered for the on-site residential scenario. For this baseline HHRA, CDIs were calculated for an adult person who was exposed for 6 years as a child and for 24 years as an adult (to give a 30-year exposure duration), and the resident was assumed to be exposed to contaminated media for 350 days/year for 30 years. A 6-year exposure duration was evaluated for the "young child" portion of the person's life, which accounts for receptors with high intake rates relative to body weight. A 24-year exposure duration was assumed for the "adult" portion of the person's life. For example, for the soil ingestion pathway, a child ingestion rate (200 mg/day) and body weight (15 kg) was assumed for 6 years, while an adult ingestion rate (100 mg/day) and body weight (70 kg) was assumed for 24 years.

Table J.13 lists the CDIs for ingestion, inhalation, and dermal contact with COPCs in the Chestnut Ridge OU 2 media; doses are listed for external exposure to radionuclides in the media. The table is divided by media (i.e., ash, soil, sediment, and surface water) and by analyte type (i.e., inorganic, radionuclide, and organic); the exposure concentration used in the CDI calculation is termed "Representative Concentration."

**Derivation of CDIs for residential ingestion of homegrown vegetables/fruits.** To calculate the CDI for ingestion of homegrown fruits and vegetables, an estimate of the contaminant concentrations in the vegetables/fruits was determined. For the three areas in Chestnut Ridge OU 2 (i.e., Sluice Channel Area, FCAP, and UMB) under residential land use, it was assumed that a garden containing leafy vegetables, root vegetables, legumes, fruiting vegetables, and fruits would be grown in the ash (with the Chestnut Ridge groundwater as the irrigation source). For the UMB, the surface water was also evaluated as the irrigation source for the UMB garden (in addition to the Chestnut Ridge OU 2 groundwater as the irrigation source).

It was assumed that plants could be contaminated by two basic means: (1) root uptake from contaminated ash and (2) contamination from water irrigation. Therefore, the plant concentration $C$ (mg/kg or pCi/kg) is then

$$C = C_r + C_{irr}$$

where $C_r$ is the concentration due to root uptake and $C_{irr}$ is the concentration due to deposition. The concentration due to irrigation is calculated assuming that the irrigation process resembles wet deposition (i.e., spray irrigation). Following the methodology used in Remedial Action Priority Systems (i.e., Multimedia Environmental Pollutant Assessment Systems), this concentration is

$$C_{irr} = \frac{(C_w I) \cdot T_f \cdot r}{Y} \frac{1 - e^{-\lambda_e t_e}}{\lambda_e}$$

where

$C_w$ = the concentration in the water used for irrigation (mg/L or pCi/L),
$I$ = the water application rate (L/m$^2$/day),
$T_f$ = the translocation factor from plant surfaces to edible parts of the plant (dimensionless),
$r$ = the fraction of deposition retained on edible parts of plant (dimensionless),
$Y$ = the plant yield (kg/m$^2$),
$\lambda_e$ = the effective weathering and decay constant,
$t_e$ = the duration of growing period for crop.

Note that the term $C_wI$ is the deposition rate of the contaminant (units of mg/m$^2$/day or pCi/m$^2$/day). The specific concentrations, parameters, and constants used in the previous equation can be found in Tables J.14–J.16.

The concentration due to root uptake is estimated by

$$C_r = B_v C_s$$

where $B_v$ is the soil-to-plant transfer coefficient which is dimensionless (refer to Table J.17), and $C_s$ is the concentration in the ash in mg/kg or pCi/kg (refer to Table J.14). The concentration in ash is the sum of the most recent measurements available, denoted here by

$C_{sm}$, and an estimate of the ash concentration due to the irrigation process itself, denoted here by $C_{si}$.

An estimate for $C_{si}$ is now derived. The following assumptions were made: (1) once deposited, the contaminant is uniformly mixed to a fixed depth $z$ (meters) (Table J.15); (2) every year, the concentration due to previous years' irrigation is assumed negligible; and (3) irrigation is assumed to occur for the duration of the growing season. Based on these assumptions, given the concentration at a given time $t$, the concentration at a later time $t+\Delta t$ is approximated by

$$C_{si}(t+\Delta t) \approx C_{si}(t) + (\textit{conc. due to deposition}) - (\textit{decreased conc. due to decay/degradation})$$

which can be approximated by

$$C_{si}(t+\Delta t) \approx C_{si}(t) + \frac{D}{\rho z}\Delta t - \lambda C(t)\Delta t$$

where $D$ is the deposition rate ($C_w$ I) and $\rho$ is the ash density (kg/m$^3$); refer to Table J.18. Letting $\Delta t$ go to zero yields

$$\frac{dC_{si}}{dt} = \frac{D}{\rho z} - \lambda C_{si}(t)$$

Solving this first order differential equation with initial condition $C_{si}(0)=0$ shows that

$$C_{si}(t) = \frac{D}{\rho z}\frac{\left(1-e^{-\lambda_e t}\right)}{\lambda_e}$$

Following Whelan et al. (1987), the concentration at harvest time is used in calculating the concentration in vegetables, in which case

$$C_{si} \equiv C_{si}(t_e) = \frac{D}{\rho z}\frac{(1-e^{-\lambda_e t_e})}{\lambda_e}$$

The property of the ash specifically addressed in these equations is its density. The density was estimated by using the measured percentages of sand, silt, and clay and the associated representative densities of these soil types. The representative values and percentages are shown in Table J.15b.

By definition, the soil-to-plant transfer coefficient ($B_v$) is the ratio of the concentration of an element in fresh vegetables to that in dry soil (Table J.17). The values for $^{137}$Cs, zinc, manganese, and barium are based on experimental results (Ng et al. 1982); in particular, the values used are 95% UCLs on the mean of values provided in Ng et al. (1982). The values for all other contaminants are from Multimedia Environmental Pollutant Assessment Systems, Version 1.0 (Strenge and Peterson 1989).

Table J.18 reports the concentrations of the COPCs in various types of vegetables/fruits as determined using the previous equations. The "Total Concentrations" reported in this table are the CVs used in the CDI calculations in Tables J.11j and J.11k. The total concentrations (CV) found in the vegetables/fruits result from contributions from (1) the unirrigated ash, (2) the irrigated ash, and (3) direct irrigation. The CDIs for the homegrown vegetables/fruits are listed in Table J.19.

### 5.3.3.2 Derivation of doses for the trespasser scenario

The most likely (and most reasonable) exposure scenario considered in this baseline HHRA was the on-site trespasser who could be exposed to the Chestnut Ridge OU 2 surface ash while walking through this area or while hunting. In this section, the Chestnut Ridge OU 2 ash radionuclide COPCs (Table 5.1) identified in Sect. 5.2, were used in the calculation of doses; the UCL95 was assumed to be representative of the analyte (radionuclide) concentrations.

Table J.12 lists the exposure variables associated with the external exposure (to radionuclides in the ash) pathway for the on-site trespasser scenario. Doses (sometimes referred to as CDIs) were calculated for a person that would be exposed to ash radionuclide COPCs, via external exposure, for 2 weeks/year for 30 years. This exposure time factor (Te) corresponds to ~1 d/month; the reader should note that hunting season on the ORR is only 2-3 weekends (6–9 days) a year.

Table J.20 lists doses (CDIs) to the trespasser from external exposure to radionuclides in the Chestnut Ridge OU 2 ash. The tables are divided by area (Sluice Channel Area, FCAP, and UMB); the exposure concentration used in the dose calculation in Table J.12 is termed "Representative Concentration" in Table J.20.

### 5.3.4 Identification of Uncertainties

Some uncertainties can be associated with the exposure pathway equations, exposure parameters, land use, and sampling and analysis of the Chestnut Ridge OU 2 media. As discussed previously, two land use scenarios were chosen to get an upper and a lower bound on the risks/HIs to human receptors. For the exposure pathways chosen for analysis in this baseline HHRA, numerous exposure parameters (Tables J.11 and J.12) were used in the calculations which each have some uncertainty (EPA 1989c) associated with them. However, most exposure parameters are conservative; therefore, the error is conservative (i.e., protecting the receptor). Some uncertainties are associated with the inhalation of dust and inhalation of VOCs (while showering) pathways, although EPA-approved default values were used in the inhalation pathway CDI calculations; modeling results were used in developing these EPA-default parameters (EPA 1989c, 1991a). For dermal exposure to water, there may be uncertainties associated with the permeability constant parameter, because many chemical-specific parameters are used in its determination. However, EPA guidance was also followed in this baseline HHRA for these types of calculations (EPA 1992a).

### 5.3.5 Summary of Exposure Assessment

Two land use scenarios were evaluated for this baseline HHRA: (1) a conservative hypothetical residential future land use and (2) a more likely current exposure scenario of a trespasser. Both land uses were evaluated to comprehensively evaluate the risk to human health from exposure to the various media associated with Chestnut Ridge OU 2 and to provide risk information supportive of the project objectives.

For the residential scenario, the primary and most direct routes/pathways of exposure that were evaluated included incidental ingestion of soil/ash/sediment, inhalation of soil/ash/dry-sediment dust, dermal contact with the soil/ash/sediment, external exposure to radionuclides in the Chestnut Ridge OU 2 soil/ash/sediment, ingestion of surface water and groundwater, dermal contact with the surface water and groundwater, and ingestion of homegrown vegetables/fruits. For the trespasser scenario, external exposure to radionuclides in the ash was evaluated.

## 5.4 TOXICITY ASSESSMENT

### 5.4.1 Toxicity Information and EPA Guidance for Noncarcinogens

Noncarcinogenic effects are evaluated by comparing an exposure experienced over a specified time period (e.g., 30 years) with an RfD [or reference concentration (RfC)] derived for a similar exposure period. The RfDs available for the COPCs present in the Chestnut Ridge OU 2 media are given in Table J.21. In this baseline HHRA, 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 $\times$ 20 $m^3$/d $\times$ (1/70 kg) = RfD) (EPA 1989c).

To evaluate the noncarcinogenic effects from exposure to COPCs in the Chestnut Ridge OU 2 media, the HI [the ratio of the exposure dose (i.e., CDI and/or dose from Tables J.13, J.19, and J.20) to the RfD] is calculated for each COPC. This noncarcinogenic HI 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 (CDI) exceeds this threshold [i.e., if CDI/RfD exceeds one (1.0)], there may be concern for potential noncarcinogenic/systemic health effects. The level of concern does not necessarily increase linearly as the HI approaches or exceeds unity; the HI is not a percentage or probability.

Chronic RfDs are developed for protection from long-term exposure to a chemical (7 years to a lifetime); subchronic RfDs are used to evaluate short-term exposure (2 weeks to 7 years) (EPA 1989c). Note that the numerical values derived for chronic and subchronic RfDs/RfCs are often identical (EPA 1993a, 1993b). For the purposes of this baseline HHRA, chronic RfDs have been used for the assessment of the on-site resident; noncarcinogenic effects (i.e., systemic toxicity) was not evaluated for the trespasser.

### 5.4.2 Toxicity Information and EPA Guidance for Carcinogens

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 (i.e., the term "incremental" refers to excess individual lifetime cancer risk). Cancer risk from exposure to contamination is expressed as excess cancer risk—cancer incurred in addition to normally

expected rates of cancer development. An excess cancer risk of 1.0E-06 indicates that each person has a one in a million chance to incur cancer from exposure to this contamination level, over a 70-year lifetime. Based on their property of emitting ionizing radiation, all radionuclides are human carcinogens.

To evaluate the carcinogenic risk from exposure to Chestnut Ridge OU 2 COPCs in the various media, the risk is calculated for each COPC [the multiplication of the exposure dose (i.e., CDI and/or dose from Tables J.13, J.19, and J.20) by the SF, which is a chemical-specific value based on carcinogenic dose-response data]. Excess cancer risks falling between 1.0E-06 and 1.0E-04 are considered to be within the EPA range of concern (also referred to as the EPA target risk range); risks within this target risk range are consistent with the cleanup goals of CERCLA and remedial action is generally not warranted (EPA 1991c). Cancer risks above 1.0E-04 are considered unacceptable by the EPA (EPA 1989c); risks that exceed the target risk range imply action is warranted and an FS is conducted (EPA 1991b). Because the SFs are the UCL95 on the probability of a carcinogenic response, the carcinogenic risk estimate represents an upper confidence bound estimate. Therefore, there is only a 5% probability 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 Chestnut Ridge OU 2 COPCs are listed in Tables J.22 and J.23.

### 5.4.3 Estimation of Toxicity Values for Dermal Exposure

Oral RfDs and SFs are often adjusted for evaluation of the dermal exposure pathway (EPA 1989c); it is conservative, in terms of risk/HI to human health, 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 time and body weight; however, dermal exposure to chemicals in soil and water are expressed as absorbed doses.

For the dermal assessments in this baseline HHRA, the oral RfD/SF for each chemical (Chestnut Ridge OU 2 COPCs) was adjusted by the percent gastrointestinal absorption efficiency (%GI) for that chemical. The %GI are 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 are often made based on data for related chemical structures. Note that (1) most organic compounds are readily absorbed (i.e., %GI =100) from the GI tract and (2) for this baseline HHRA, no adjustments were made to chemicals with %GI $\geq$ 80%.

Minor adjustments to the oral RfDs/SFs (used in the dermal assessments only), which favor conservatism, were made when %GI data were available for this baseline HHRA. 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. These toxicity values (listed in Tables J.21 and J.22) were then used in the evaluation of risk to human health from exposure to Chestnut Ridge OU 2 soils, ash, sediment, surface water, and groundwater via dermal contact.

### 5.4.4 Chemicals Without EPA Toxicity Values

SFs and RfDs are not currently available for all known chemicals because their carcinogenicity and/or noncarcinogenic effects have not yet been determined. These chemicals may contribute to carcinogenic and noncarcinogenic effects from exposure to the Chestnut Ridge OU 2 media, but their effect cannot be quantified at the present time. In addition, for several chemicals, epidemiological studies have indicated that they are not carcinogenic; consequently, these species do not have SFs. A qualitative summary of toxicity information for COPCs for Chestnut Ridge OU 2 can be found in Sect. 5.4.6.

### 5.4.5 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/RfC values are generally based on studies of the most sensitive animal species tested (unless adequate human health data are available) and the most sensitive end point measured. There are, of course, uncertainties in the experimental data set for such animal studies. 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/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/sharp safe levels of exposure for humans is not possible.

A two-part evaluation exists for assessing the carcinogenic potential of a chemical: (1) the likelihood that a chemical is a carcinogen (i.e., a weight-of-evidence assessment) and (2) 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 1989c, 1993b): (1) Group A, human carcinogen; (2) Group B, probable human carcinogen; (3) Group C, possible human carcinogen; (4) Group D, not classified as to human carcinogenicity; and (5) Group E, evidence of no carcinogenic effects to 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 upper 95% CL 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.

### 5.4.6 Summary of Toxicity Information/Profiles for Chestnut Ridge OU 2 COPCs

The purpose of any toxicity assessment is to evaluate the potential for COPCs to cause adverse health effects in exposed individuals. This usually consists of an evaluation 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.

The chemical-specific information in Sects. 5.4.6.1 and 5.4.6.2 provides general qualitative information as well as a chemical-specific discussion about health effects related to those COPCs evaluated in this baseline HHRA. Carcinogenic and noncarcinogenic health effects are considered. Data used in this section 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 *Toxicity Profiles for Contaminants of Concern on the Oak Ridge Reservation* (BEIAS 1993) report for further information regarding specific chemicals. Tables J.21–J.23 summarize the toxicity information for the Chestnut Ridge OU 2 COPCs.

### 5.4.6.1 Inorganics

**Aluminum.** Aluminum is found in abundance in the earth but in small amounts in plant and animal tissue. It can be found in tap water because aluminum sulfate is used in the water purification process and not all of the aluminum is filtered out. Aluminum is added to most table salt to prevent caking and is used in certain antacids. Aluminum is also used in foil, deodorants, baking powder, as an emulsifier in some processed cheeses, and as a bleaching agent to whiten flour (Garrison and Somer 1985). In addition, aluminum has been shown to be suitable for food containers since it is practically insoluble and is not harmful to health when dissolved to a slight degree.

Aluminum is a trace mineral in the human body, but it can be dangerous, even fatal, if consumed in excessive amounts. Aluminum has no established function in human nutrition. It weakens the living tissue of the alimentary canal (i.e., the digestive tube from the mouth to the anus). Many of aluminum's harmful effects result from its destruction of vitamins. It binds with many other substances and is never found alone in nature (Dunne 1990).

Aluminum is easily absorbed by the body and is accumulated in the arteries. Highest concentrations are found in the lungs, liver, thyroid, and brain. Usually most of the aluminum taken into the body is ultimately excreted. However, excessive amounts of aluminum can result in symptoms of poisoning including nausea, skin ailments, colic, loss of appetite, excessive perspiration, and loss of energy (Dunne 1990). In addition, aluminum toxicity has been implicated in brain disorders associated with aging, such as Alzheimer's disease (Garrison and Somer 1985); however, this theory has not been proven.

**Arsenic.** Arsenic is a metallic, steel-gray, crystalline, brittle, trivalent and pentavalent, solid, poisonous element. It is commonly used in pesticides. Trivalent compounds are generally more toxic and more likely to have systemic effects than the less soluble compounds, which are more likely to cause chronic pulmonary effects if inhaled.

Water-soluble, inorganic arsenic compounds are absorbed through the gastrointestinal tract and lungs. Symptoms of acute inorganic arsenic poisoning in humans are nausea, anorexia, vomiting, epigastric and abdominal pain, and diarrhea. In addition, dermatitis, muscle cramps, cardiac abnormalities, hepatoxicity, bone marrow suppression and hematologic abnormalities, vascular lesions, and peripheral neuropathy have also been reported. Severe

exposures can result in acute encephalopathy, congestive heart failure, stupor, convulsions, paralysis, coma, and death. Occupational exposure studies show a clear correlation between exposure to arsenic and lung cancer mortality (BEIAS 1993).

**Barium.** Barium is a divalent alkaline-earth metal found only in combination with other elements in nature. The most important of these combinations are the peroxide, chloride, sulfate, carbonate, nitrate, and chlorate. The most likely source of barium in the atmosphere is from industrial emissions. Because of the element's tendency to form salts with limited solubility in soil and water, it is expected to have a residence time of hundreds of years and is not expected to be very mobile. Trace amounts of barium have been found in more than 99% of surface waters and finished drinking water samples (average values of 43 $\mu$g/L and 28.6 $\mu$g/L, respectively) across the United States (BEIAS 1993).

The soluble salts of barium are toxic to mammalian systems. They are absorbed rapidly from the gastrointestinal tract and are deposited in the muscles, lungs, and bone. At low doses, barium acts as a muscle stimulant and at higher doses affects the nervous system, eventually leading to paralysis. Subchronic and chronic oral or inhalation exposure primarily affects the cardiovascular system resulting in elevated blood pressure. Subchronic and chronic inhalation exposure of human populations to barium-containing dust can result in a benign pneumoconiosis called baritosis, which is a condition often accompanied by an elevated blood pressure but does not usually result in a pulmonary function change. Although the effects of barium on laboratory rats have been documented and include elevated blood pressure, decreased body weight, birth defects, and increased infant mortality, these effects have not been substantiated in humans. In addition, barium has not been evaluated by the EPA for evidence of human carcinogenicity (BEIAS 1993).

**Beryllium.** Pure beryllium is a hard, grayish metal present in the earth's crust. It can be found in emissions from coal combustion; in surface water and soil; and in house dust, food, drinking water, and cigarette smoke. Industry employs beryllium in several ways, including in brake systems for airplanes, for neutron monochromatization, as window material for x-ray tubes, and in radiation detectors. Additionally, beryllium compounds are used in manufacturing ceramics and refractories, chemical reagents, and gas mantle hardeners. The highest risk for exposure to beryllium occurs among workers employed in beryllium manufacturing, fabricating, or reclaiming industries. However, people who live near these industries and are sensitive to extremely low concentrations of beryllium in the air are also at risk. In addition, smokers inhale unusually high concentrations of beryllium, depending on the source of tobacco.

A limited amount of data indicates that the oral toxicity of beryllium is low; however, the inhaled toxicity of beryllium is well documented. Humans inhaling massive doses of beryllium compounds may develop acute berylliosis. Additionally, beryllium and its compounds are presumed to have cancer-causing potential in the human lung when inhaled. The cancer-causing ability has been investigated in workers exposed to beryllium. The degree of harm depends on the amount and duration of exposure. Short-term exposure to beryllium may cause noncarcinogenic health effects, such as acute pneumonitis berylliosis, while long-term exposure may cause lung cancer (ATSDR 1988a).

**Cadmium.** Cadmium is a naturally occurring metal used to control mold and diseases that attack home lawns, golf course turfs, and other grasses. It is also used in electroplating processes, rubber tire manufacturing, and plumbing solders. However, less than 0.1% of the

annual United States consumption of 12 million pounds of cadmium is used in pesticides—most of it is used in industry. An estimated 100,000 workers in the United States are exposed to this metal, which is employed not only in the electroplating and rubber industries but also in the manufacture of nickel-cadmium batteries, brazing-soldering alloys, pigments, and chemicals that act as stabilizers in plastics. Other sources of human exposure are cigarette smoke, air pollution, drinking water, and sludge. Most cadmium gets into sewage treatment plants as part of the discharge from electroplating processes, from disintegration of rubber tires on streets, and from solder in plumbing (Winter 1979).

Cadmium in drinking water has been correlated with cancer of the pharynx, esophagus, intestine, larynx, lung, and bladder. Studies have shown that workers who had labored for at least 2 years in a cadmium smelter had significantly higher death rates from all forms of cancer, especially of the lung and prostate gland. Lung cancer occurred at more than twice the expected rate, and as with many other occupational cancers, the risk of developing the disease increased in relation to the length of elapsed time after initial exposure to the metal. Cadmium is also believed to cause mutations; this conclusion is based upon finding chromosomal and DNA damage resulting from the presence of cadmium in the blood of workers and in bacteria. In addition, exposure to home-lawn-treatment products containing cadmium may be dangerous to women of child-bearing age (Winter 1979). Cadmium is not excreted from the body and can accumulate over time to toxic levels; symptoms of cadmium toxicity include anemia, muscle wastage, hypertension, and liver and kidney damage (Somer 1992).

**Chloride.** Chloride is found in abundance in vegetable and animal foods. It is an essential anion that accounts for 0.15% of total body weight. Chloride is obtained from table salt (sodium chloride) or from salt substitutes (such as potassium chloride) (Garrison and Somer 1985). Although chloride is a compound of chlorine, it is not poisonous to the human body (Somer 1992).

Chloride is found primarily in extracellular fluids and is especially abundant in gastrointestinal secretions and cerebrospinal fluid. Small amounts of chloride are found in bone and connective tissue and lesser amounts in muscle and nerve tissue. Chloride is closely associated with sodium and water in foods, body secretions, fluids, tissues, and excretions. It is important in regulating the body's acid-base balance. Chloride is readily absorbed in the intestines. Intake is usually in excess of sodium, and amounts that are not needed are excreted through the kidneys, primarily as sodium chloride. Some losses occur through sweat and feces (Garrison and Somer 1985).

Chloride deficiency is of concern in the body. Deficiencies can result from diarrhea or vomiting, adrenal insufficiency, and acidosis. A deficiency of chloride in the body results in an acid-base imbalance, called *alkalosis* (Somer 1992). Chloride deficiency in infants can be especially harmful, causing psychomotor defects, memory loss, and growth retardation. These symptoms can be alleviated with administration of chloride (Garrison and Somer 1985). However, deficiencies are uncommon because the element is consumed in salt. No known toxic effects have been recorded for overuse or overingestion of chloride because overabundance of the element is secreted through the kidneys (Somer 1992).

**Chromium, chromium III, and chromium VI.** Elemental chromium does not occur in nature but is present in ores—primarily chromite. Chromium exhibits several oxidation states, but the most prominent of these is chromium VI and chromium III. Chromium VI in the

environment is manmade as a result of industrial emissions; in solution, chromium VI exists as hydrochromate, chromate, and dichromate ionic species and reacts over time to form chromium III. Chromium VI is much more mobile and toxic than is chromium III. Chromium is useful in glucose and cholesterol metabolism and therefore is an essential element to humans and animals. Nonoccupational exposure to the metal occurs via the ingestion of chromium-containing food and water, whereas occupational exposure occurs via inhalation. Workers are exposed to chromium during its use in the production of dichromate; the chemical, stainless steel, refractory, and chromium-plating industries; and the production and use of alloys (BEIAS 1993).

Chromium enters the body through the lungs, gastrointestinal tract, and, to a lesser extent, the skin. Inhalation is the most important route for occupational exposure. Workers exposed to chromium have developed nasal irritation, nasal ulcers, perforation of the nasal septum, and hypersensitivity reactions and "chrome holes" of the skin. Among the general population, contact dermatitis has been associated with the use of bleaches and detergents. Inhalation of chromium compounds has been associated with the development of cancer in workers in the chromate industry. Evidence also suggests an increased risk in developing nasal, pharyngeal, and gastrointestinal carcinomas. Based on sufficient evidence reporting that humans and animals are at risk of developing cancer, chromium VI has been assigned an EPA weight-of-evidence classification of A, human carcinogen (BEIAS 1993).

**Cobalt.** Cobalt is a tough, lustrous, silver-white, magnetic, metallic element that is related to and occurs with iron and nickel and is used especially in alloys. Dietary cobalt is found in some foods, such as figs, cabbage, spinach, beet greens, buckwheat, lettuce, and watercress (Garrison and Somer 1985). In addition, organ meats, such as liver and kidneys, oysters, and clams are excellent sources of cobalt, and muscle meats are good dietary sources because these foods supply the mineral as part of vitamin $B^{12}$. The only known function of cobalt in the human body is as a constituent of vitamin $B^{12}$. In addition, it is used as an anti-foaming agent in the processing of some beer (Somer 1992).

Cobalt is essential to erythropoiesis (the formation of red blood cells). The average intake of cobalt is 5 to 8 $\mu$g/day. No Recommended Daily Allowance of safe and adequate amount has been established for cobalt at this time (Garrison and Somer 1985). The human body does not require or need cobalt alone (Somer 1992).

When fed a pharmacological dose of cobalt, many animals, including humans, develop polycythemia because of increases in the hormone erythropoietin in the blood. Elevated erythrocyte and hemoglobin levels, reticulocytosis, increased red blood cell mass, and normoblastic hyperplasia in the bone marrow have also been reported. Congestive heart failure due to cardiomyopathy has been reported when beer containing 1.2 ppm of cobalt was consumed. Pericardial effusion, thyroid hyperplasia, and neurological disorders have also been noted. Large doses of cobalt may interfere with decarboxylation reactions (by binding to lipoic acid), impair pyruvate and fatty acid metabolism, and enhance iron absorption and globin synthesis (Garrison and Somer 1985).

**Fluoride.** Fluoride is the monovalent anion of fluorine, which is a nonmetallic, halogen element isolated as a pale yellowish, flammable, irritating, toxic, diatomic gas. The best dietary source of fluoride is fluoridated water. Use of fluoridated water is more effective in preventing tooth decay than is topically applied fluoride (Somer 1992).

Fluoride is essential for strong teeth and bones. It is necessary for replacing the hydroxy portion of their crystalline structure, creating a less water-soluble fluoride salt called fluorapatite. As a result, bone and tooth structure is harder, larger, more uniform, and more resistant to decay by acids and demineralization. A reduced incidence of osteoporosis is found in areas with naturally occurring or added fluoride in the drinking water. With adequate fluoride intake, some elderly patients show a reduced excretion of calcium, improved bone density, and alleviation of osteoporosis symptoms. Fluoride may also prevent hearing loss in the elderly by recalcifying the inner bone structure of the ear (Garrison and Somer 1985).

One detriment of too much fluoride intake (2 to 8 ppm in domestic water supply) is mottling, dulling, and pitting of teeth. This preliminary toxicity sign is harmless, and the teeth are strong and relatively cavity free, although it may be aesthetically unpleasing (Garrison and Somer 1985). Fluoride concentrations greater than 8 ppm can cause fluorosis of the bones that produces arthritis-like symptoms. Fatal poisoning can occur if fluoride is ingested in amounts greater than 2500 times the recommended intake. Chronic ingestion of 50 mg/day can occur with some forms of air or environmental pollution and results in bone and tooth deformities. However, the levels used in the fluoridation of water pose no harmful effects to health and greatly reduce the incidence of tooth decay and possibly periodontal disease (Somer 1992).

**Lead.** Lead is a naturally occurring, bluish-gray metal found in small quantities in the Earth's crust. Lead and its compounds can be detected in all parts of the environment (e.g., in plants and animals used for human consumption and in air, drinking water, rivers, lakes, oceans, dust, and soil). Lead used by industry comes from mined ores or from recycled scrap metal (ATSDR 1988b).

Lead has a wide range of uses. Its primary use is in the manufacture of storage batteries. Other applications include its use in the production of ammunition, miscellaneous metal products (e.g., sheet lead, solder, and pipes), and various chemicals, including gasoline additives, tetraethyl lead, and tetramethyl lead (ATSDR 1988b).

General population exposure to lead and its compounds results from breathing air, drinking water, and eating many foods that contain lead. Breathing or ingesting dust and dirt laden with lead is also a source of exposure. Skin contact with dust and dirt containing lead is also expected to occur on a daily basis, but uptake by the skin is not substantial. During the normal use of lead-containing products, little skin contact is expected to take place. Exposure to certain lead compounds that are additives can occur while pumping gasoline containing these additives and possibly during use of some do-it-yourself fuel additives. Children, especially those who are preschool age, are at particular risk to lead exposure since they commonly put their hands, toys, and other items in their mouths. These items frequently come in contact with lead-containing dust and dirt, and, as a result, children consume large amounts of lead. In some cases, children may ingest nonfood items such as paint chips and dirt. These items may contain large amounts of lead (ATSDR 1988b).

Little of the lead ingested by adults enters the blood and other parts of the body. However, a much greater proportion of the lead ingested by children enters the blood and other body parts. Exposure of a expectant mother to lead results in the transfer of lead to the fetus and may cause preterm birth, reduced birth weight, and decreased intelligence quotient (IQ) in the infant. Lead exposure may also decrease IQ scores and reduce the growth of young children. At high levels of exposure, lead can severely damage the brain and kidneys

in adults and children, cause abortion, and damage the male reproductive system. The effects of lead are the same regardless of whether it enters the body through breathing or ingestion (ATSDR 1988b).

**Manganese.** Manganese makes up about 0.10% of the earth's crust and is the 12th most abundant element. It can exist in oxidation states from -3 to +7, the most common being +4 in the chemical form of manganese dioxide. The oxides and peroxides are used in industry as oxidizers, and the metal is used for manufacturing metal alloys to increase hardness and corrosion resistance. Manganese is an essential trace element in humans, which can elicit a variety of serious toxic responses upon prolonged exposure to elevated concentrations either orally or by inhalation. The central nervous system is the primary target (BEIAS 1993).

Initial symptoms of manganese exposure are insomnia, disorientation, anxiety, lethargy, and memory loss. These symptoms will progress with prolonged exposure and will eventually include motor disturbances, tremors, and walking difficulties similar to Parkinsonism. Effects on reproduction (decreased fertility, impotence) have been observed in humans with inhalation exposure and in animals with oral exposure at the same or similar doses that initiate the central nervous system effects. Data on possible carcinogenesis following injections in mice are inconclusive; however, the EPA weight-of-evidence classification is D, not classifiable as to human carcinogenicity based on no evidence in humans and inconclusive evidence in animals (BEIAS 1993).

**Mercury and mercury salts.** Mercury is a naturally occurring element that may exist in elemental, inorganic, or organic forms and in various oxidation states. Mercury is used in a wide variety of products and processes, including pressure-sensitive devices (thermometers, barometers), electrical apparatus (wiring, switches, batteries), paints, pharmaceuticals, and in the production of various chemicals. The oxidation state and chemical form of mercury are important in determining its toxicity, with mercurous salts being less toxic than mercuric salts. Organic materials such as methyl mercury are highly toxic. In the environment, mercury may undergo transformations among the various oxidation states and chemical forms. Both environmental and occupational exposure are relevant to mercury and its compounds, although environmental exposure is unimportant for mercury vapor. Mercury intake from occupational exposure is of greater significance than that from environmental exposure. Environmental exposure to mercury may involve dietary intake (i.e., from fish) and possibly from dental amalgams, the latter being controversial and under dispute (BEIAS 1993).

Ingestion of mercury metal is usually without effect, while ingestion of inorganic salts may cause severe gastrointestinal irritation, renal failure, and death. Mercury is also known to induce hypersensitivity reactions such as contact dermatitis and acrodynia (pink disease). Inhalation of mercury vapor may cause irritation of the respiratory tract, central nervous system effects characterized by neurobehavioral changes, peripheral nervous system toxicity, renal toxicity, and death. Toxicity resulting from subchronic and chronic exposure to mercury and mercury salts usually involves the kidneys and/or the nervous system. No data are available regarding the carcinogenicity of mercury in humans or animals. The EPA has placed inorganic mercury in weight-of-evidence classification D, not classifiable as to human carcinogenicity (BEIAS 1993).

**Nickel and nickel salts.** Nickel is a naturally occurring metal existing in various mineral forms. Nickel may be found throughout the environment including rivers, lakes, oceans, soil, air, drinking water, plants, and animals. Soil and sediment are the primary receptacles for

nickel but mobilization may occur depending on physicochemical characteristics of the soil. Nickel is used in a wide variety of metallurgical processes such as electroplating and alloy production, as well as in nickel-cadmium batteries. Some evidence suggests that nickel may be an essential trace element for mammals. As for most metals, the toxicity of nickel is dependent on the route of exposure and the solubility of the nickel compound (BEIAS 1993).

Pulmonary absorption is the major route of concern for nickel-induced toxicity. Toxic effects of oral exposure to nickel usually involve the kidneys with some evidence from animal studies showing a possible development/reproductive toxicity effect. Inhalation exposure to some nickel compounds will cause toxic effects in the respiratory tract and immune system. Asthmatic conditions have also been documented for inhalation exposure to nickel. In addition, sensitivity reactions to nickel are well documented and usually involve contact dermatitis reactions resulting from contact with items such as cooking utensils, jewelry, coins, etc., containing nickel. Epidemiologic studies have shown that occupational inhalation exposure to nickel dust (primarily nickel subsulfide) at refineries has resulted in increased incidence of pulmonary and nasal cancer (BEIAS 1993).

**Niobium.** Niobium is a lustrous, platinum-gray, ductile, metallic element that resembles tantalum chemically and is used in alloys. The element was first discovered in a museum and was mistaken for columbium or columbite, which is a compound of chromium. Niobium exists in the Earth's crust in proportions of about 20 ppm. It is commercially refined as an alloy with iron and nickel and is used in atomic reactors. When alloyed with tin or aluminum, niobium is known to be superconductive (Heiserman 1992).

Niobium looks much like steel but is soft and ductile. It is a Group VB, vanadium-like element that is commonly mistaken for tantalum because of its similar physical and chemical properties. Little information is available concerning the element's toxic and carcinogenic effects; however, because of its similarity to vanadium, it is believed to exhibit the same effects. These effects include irritation of the respiratory and gastrointestinal tracts, conjunctivitis, dermatitis, cough, bronchospasm, pulmonary congestion, and bronchitis (Heiserman 1992).

**Sulfate.** The sulfate ion, $SO_4$, is one of the major anions occurring in natural waters. The majority of sulfates, with the exception of lead, barium, and strontium sulfates, are soluble in water. Therefore, dissolved sulfate is considered to be a permanent solute of water. Sulfate may be reduced to sulfide, volatilized to the air as $H_2S$, precipitated as an insoluble salt, or incorporated into living organisms. Sulfates are used for a variety of commercial purposes, including sulfuric acid for the steel and metal industries, as a reagent in manufacturing processes, and as products such as copper sulfate, which is used as a fungicide and algicide (BEIAS 1993).

The major health effect observed with sulfate ingestion is laxative action, and the cation associated with the sulfate appears to have some effect on the salt's potency as a laxative. Sulfate itself slowly penetrates mammalian cellular membranes and is rapidly eliminated through the kidneys. The mechanism by which sulfate ions induce laxative effects is complex and poorly understood. However, retention of excess fluid in the intestinal lumen and increased motor activity in the intestinal tract appear to be involved. It may be the poorly absorbed, soluble ions exert an osmotic pressure that causes retention of fluid in the intestinal lumen, and this increase in bulk indirectly stimulates intestinal transit. In addition to osmotic effects, the sulfate ion may increase the fluid volume of the intestinal tract by decreasing

water absorption in the small intestine and by stimulating increases in the secretion of pancreatic, gastric, and intestinal fluids. It is unknown if these effects are caused directly by the ion or by the ion-stimulated release of hormones such as cholecystokinin. In addition, sulfates can contribute to an undesirable taste in water. The Drinking Water Standards of the U.S. Public Health Service recommend that sulfate in water should not exceed 250 mg/L, except when no more suitable supplies are or can be made available (BEIAS 1993). Carcinogenic data are presently unavailable to determine the carcinogenic potential of sulfate.

**Thallium.** Thallium is a sparsely but widely distributed poisonous metallic element that resembles lead in physical properties and is used chiefly in the form of compounds in photoelectric cells or as a pesticide. It is an elemental metal that occurs naturally in the Earth's crust and is released into the environment from industrial sources. Thallium is most commonly found in minerals and as thallium salts. Thallium in the atmosphere can contaminate surface soil; human exposure can occur orally, dermally, or by inhalation. The most common nonoccupational sources of thallium are contaminated food crops and tobacco. Thallium is normally detected in the urine of humans; elevated urinary levels have been correlated with adverse health effects that include neurological and muscular signs and symptoms (BEIAS 1993).

The primary targets of thallium toxicity are the nervous system, the skin, and the reproductive system. In humans, acute doses produce paresthesia, retrobulbar neuritis, ataxia, delirium, tremors, and hallucination, implying central, peripheral, autonomic nervous system involvement. Chronic exposure to humans and animals results in alterations in the brain, spinal cord, and peripheral nerves. In both humans and animals, alopecia (a loss of hair, wool, or feathers; baldness) is the most common indicator of long-term thallium poisoning. An increased incidence of congenital malformations was observed in children whose parents were exposed to thallium through the consumption of home-grown vegetables and fruit, but a causal relationship between these effects and thallium could not be verified. In animal studies, thallium compounds have produced testicular effects in treated males and slight signs of fetotoxicity and significant impairment of learning ability in the offspring of treated animals (BEIAS 1993).

**Vanadium.** Vanadium is a metallic element that occurs in six oxidation states and numerous inorganic compounds. The element is used primarily as an alloying agent in steels and nonferrous metals such as copper, aluminum, and titanium. Vanadium compounds are also used as catalysts and in chemical, ceramic, or specialty applications. It may also have applications as an intermetallic compound for superconductor applications. Minor uses include applications as color modifiers in mercury-vapor lamps, as driers in paints and varnish, and as corrosion inhibitors in flue-gas scrubbers (BEIAS 1993).

Vanadium compounds are poorly absorbed through the gastrointestinal system but slightly more readily absorbed through the lungs. Absorbed vanadium is widely distributed in the body, but short-term localization occurs primarily in bone, the kidneys, and the liver. In the body, vanadium can undergo changes in oxidation state and vanadate and can also bind with blood protein (transferrin). The toxicity of vanadium depends on its physico-chemical state—particularly on its valence state and solubility. In humans, intestinal cramps and diarrhea may occur following subchronic oral exposures, thereby suggesting that, for subchronic and chronic oral exposures, the primary targets are the digestive system, kidneys, and blood. Inhalation exposures to vanadium and vanadium compounds result primarily in adverse effects to the respiratory system. In studies on workers occupationally exposed to

vanadium, the most common reported symptoms were irritation of the respiratory tract, conjunctivitis, dermatitis, cough, bronchospasm, pulmonary congestion, and bronchitis. Little evidence suggests that vanadium or vanadium compounds are carcinogenic; however, few studies have been conducted on the carcinogenicity of vanadium (BEIAS 1993).

**Zinc.** Zinc is an essential element and is used primarily in galvanized metals and metal alloys. In addition, various inorganic zinc salts have numerous commercial uses. Zinc oxide is used in the rubber industry as a vulcanization activator and accelerator and to slow down oxidation, and also as a reinforcing agent, heat conductor, pigment, UV stabilizer, supplement in animal feeds and fertilizers, catalyst, chemical intermediate, and mildew inhibitor. Zinc sulfate is used in rayon manufacture, agriculture, zinc plating, and as a chemical intermediate and mordant. Zinc chloride is used in smoke bombs, in cements for metals, in wood preservatives, in flux for soldering; in manufacture of parchment paper, artificial silk, and glues; as a mordant in printing and dye textiles; and as a deodorant, antiseptic, and astringent. Zinc chromate is used as a pigment in paints, varnishes, and oil colors. In addition, zinc phosphide is used as a rodenticide, and zinc cyanide is used in electroplating. The toxicity of the latter two compounds is caused primarily by their anion component (BEIAS 1993).

Gastrointestinal absorption of zinc is variable (20-80%) and depends on the chemical compound as well as on zinc levels in the body and dietary concentrations of other nutrients. Zinc is present in all tissues with the highest concentrations in the prostate, kidney, liver, heart, and pancreas. In humans, acutely toxic oral doses of zinc cause nausea, vomiting, diarrhea, abdominal cramps, and, in some cases, gastric bleeding. Gastrointestinal upset has also been reported in individuals taking dietary zinc supplements for up to 6 weeks. Limited evidence suggests that the human immune system may be impaired by subchronic exposures. Chronic oral exposures to zinc have resulted in hypochromic microcytic anemia associated with hypoceruloplasminemia, hypocupremia, and neutropenia in some individuals. Under occupational exposure conditions, inhalation of zinc compounds (mainly zinc oxide fumes) can result in a condition identified as "metal fume fever," which is characterized by nasal passage irritation, cough, rales, headache, altered taste, fever, weakness, hyperpnea, sweating, pains in the legs and chest, leukocytosis, reduced lung volume, and decreased diffusing capacity of carbon monoxide. "Metal fume fever" is an acute and reversible effect that is unlikely to occur under chronic exposure conditions when zinc air concentrations are less than 8–12 $mg/m^3$. No case studies or epidemiologic evidence has been presented to suggest that zinc is carcinogenic in humans by the oral or inhalation route (BEIAS 1993).

### 5.4.6.2 Radionuclides

Radionuclides are unstable atoms of chemical elements that will emit charged particles to achieve a more stable state. These charged particles are termed "alpha and beta radiation" and "neutral gamma rays." Interaction of these charged particles (and gamma rays) with matter will produce ionization events, or radiation, which may cause living cell tissue damage. Because the deposition of energy by ionizing radiation is a random process, sufficient energy may be deposited (in a critical volume) within a cell and result in cell modification or death (ICRP 1991). In addition, ionizing radiation has sufficient energy that interactions with matter will produce an ejected electron and a positively charged ion (known as free radicals) that are highly reactive and may combine with other elements, or compounds within a cell, to produce toxins or otherwise disrupt the overall chemical balance of the cell (EPA 1991d). These free radicals can also react with deoxyribonucleic acid (DNA), causing genetic damage, cancer induction, or even cell death.

Radionuclides are characterized by the type and energy level of the radiation emitted. Radiation emissions fall into two major categories: particulate (electrons, alpha particles, beta particles, and protons) or electromagnetic radiation (gamma and X rays) (ATSDR 1989a). Therefore, all radionuclides are classified by the EPA as Group A carcinogens based on their property of emitting ionizing radiation and on the extensive weight of evidence provided by epidemiological studies of humans with cancers induced by high doses of radiation. Alpha particles are emitted at a characteristic energy level for differing radionuclides. The alpha particle has a charge of +2 and a comparably large size. Alpha particles have the ability to react (and/or ionize) with other molecules, but they have very little penetrating power and lack the ability to pass through a piece of paper or human skin. However, alpha-emitting radionuclides are of concern when there is a potential for inhalation or ingestion of the radionuclide. Alpha particles are directly ionizing and deposit their energy in dense concentrations [termed high linear energy transfer (high LET)], resulting in short paths of highly localized ionization reactions. The probability of cell damage increases as a result of the increase in ionization events occurring in smaller areas; this may also be the reason for increased cancer incidence caused by inhalation of radon gas. In addition, the cancer incidence in smokers may be directly attributed to the naturally occurring alpha emitter, $^{210}Po$, in common tobacco products (Hammonds and Hoffman 1992).

Beta emissions generally refer to beta negative particle emissions. Radionuclides with an excess of neutrons achieve stability by beta decay. Beta radiation, like alpha radiation, is directly ionizing but, unlike alpha activity, beta particles deposit their energy along a longer track length (low-LET), resulting in more space between ionization events (Hammonds and Hoffman 1992). Beta-emitting radionuclides can cause injury to the skin and superficial body tissue but are most destructive when inhaled or ingested. Many beta emitters are similar chemically to naturally occurring essential nutrients and will therefore tend to accumulate in certain specific tissues. For example, $^{90}Sr$ is chemically similar to calcium and, as a result, accumulates in the bones, where it causes continuous exposure. The health effects of beta particle emissions depend upon the target organ. Those seeking the bones would cause a prolonged exposure to the bone marrow and affect blood cell formation, possibly resulting in leukemia, other blood disorders, or bone cancers. Those seeking the liver would result in liver diseases or cancer, while those seeking the thyroid would cause thyroid and metabolic disorders. In addition, beta radiation may lead to damage of genetic material (DNA), causing hereditary defects.

Gamma emissions are the energy that has been released from transformations of the atomic nucleus. Gamma emitters and X rays behave similarly but differ in their origin: gamma emissions originate in nuclear transformations, and X rays result from changes in the orbiting electron structure. Radionuclides that emit gamma radiation can induce internal and external effects. Gamma rays have high penetrating ability in living tissue and are capable of reaching all internal body organs. Without such sufficient shielding as lead, concrete, or steel, gamma radiation can penetrate the body from the outside and does not require ingestion or inhalation to penetrate sensitive organs. Gamma rays are characterized as low-LET radiation, as is beta radiation; however, the behavior of beta radiation differs from that of gamma radiation in that beta particles deposit most of their energy in the medium through which they pass, while gamma rays often escape the medium because of higher energies, thereby creating difficulties in determining actual internal exposure. For this reason, direct whole-body measurements are necessary to detect gamma radiation, while urine/fecal analyses are usually effective in detecting beta radiation (Hammonds and Hoffman 1992).

People receive gamma radiation continuously from naturally occurring radioactive decay processes going on in the earth's surface, from radiation naturally occurring inside their bodies, from the atmosphere as fallout from nuclear testing or explosions, and from space or cosmic sources. Cesium-137 (from nuclear fallout) decays to $^{137}Ba$, the highest contributor to fallout-induced gamma radiation (NCRP 1977). Beta radiation from the soil is a less penetrating form of radiation but has many contributing sources. Potassium-40, $^{137}Cs$, $^{214}Pb$, and $^{214}Bi$ are among the most common environmental beta emitters. Tritium is also a beta emitter but contributes little to the soil beta radiation because of the low energy of its emission and its low concentration in the atmosphere (NCRP 1977). Alpha radiation is also emitted by the soil but is not measurable more than a few centimeters from the ground surface. The majority of alpha emissions are attributable to $^{222}Rn$ and $^{220}Rn$ and their decay products (NCRP 1977). This contributes to what is called background exposure to radiation (ATSDR 1989a).

The general health effects of radiation can be divided into stochastic (related to dose) and nonstochastic (not related to dose) effects. The risk of development of cancer from exposure to radiation is a stochastic effect. Examples of nonstochastic effects include acute radiation syndrome and cataract formation, which occur only at high levels of exposures (Killough and Eckerman 1983).

Radiation can damage cells in different ways. It can cause damage to DNA within the cell, and the cell either may not be able to recover from this type of damage or may survive but function abnormally. If an abnormally functioning cell divides and reproduces, a tumor or mutation in the tissue may develop. The rapidly dividing cells that line the intestines and stomach and the blood cells in bone marrow are extremely sensitive to this damage. Organ damage results from the damage caused to the individual cells. This type of damage has been reported with doses of 10 to 500 rads (0.1 to 5.0 gray, in SI units). Acute radiation sickness is seen only after doses of >50 rads (0.5 gray), which is a dose rate usually achieved only in a nuclear accident (ATSDR 1989a).

When the radiation-damaged cells are reproductive cells, genetic damage can occur in the offspring of the person exposed. The developing fetus is especially sensitive to radiation. The type of malformation that may occur is related to the stage of fetal development and the cells that are differentiating at the time of exposure. Radiation damage to children exposed in the womb is related to the dose the pregnant mother receives. Mental retardation is a possible effect of fetal radiation exposure (ATSDR 1989a).

The most widely studied population that has had known exposure to radiation is the atomic bomb survivors of Hiroshima and Nagasaki, Japan. Data indicate an increase in the rate of leukemia and cancers in this population. However, the rate at which cancer incidence is significantly affected by low radiation exposures, such as results of exposure to natural background and industrially contaminated sites, is still undergoing study and is uncertain (Hammonds and Hoffman 1992). In studies conducted to determine the rate of cancer and leukemia increase, as well as genetic defects, several radionuclides must be considered.

**Cesium-137.** Cesium occurs in nature as $^{133}Cs$ in the aluminosilicates, pollucite (a hydrated silicate of aluminum and cesium) and lepidolite; in the borate, rhodizite; and in other sources (Budavari et al. 1989, Klassen et al. 1986). Cesium-137 is one of the artificial isotopes of cesium and is one of the principal radionuclides present in reactor effluent under normal operations. Cesium-137 may also be produced in nuclear and thermonuclear

explosions, through which it would be a primary contributor to human exposure through fallout radiation, assimilation through the food chain, or beta dose to the skin (Budavari et al. 1989, Klassen et al. 1986). In addition, $^{137}Cs$, along with $^{90}Sr$, is one of the most important fission products that was widely distributed in near-surface soils because of historical weapons testing. Measurable concentrations still exist in the soil today, almost exclusively in the upper 15 cm of soil; these concentrations decrease roughly exponentially with depth.

Cesium-137 may also have important roles in medical treatments (a teletherapy source or intercavitary or interstitial radiation source in treatment of malignancies) and as an encapsulated energy source (Budavari et al. 1989, Casarett 1968). Cesium-137 decays to and reaches radioactive equilibrium with its daughter product, $^{137m}Ba$ (Budavari et al. 1989, Casarett 1968). Barium-137m is a very short-lived gamma emitter that can contribute to external gamma exposure (Budavari et al. 1989).

**Radium-226.** Radium is a naturally occurring radioactive element that exists in several isotopic forms. The radium isotopes are formed from the decay of uranium and thorium. Radium-226 is formed from $^{238}U$ and $^{234}U$, and $^{226}Ra$ has the longest half-life of the radium isotopes ($^{228}Ra$, $^{224}Ra$, and $^{223}Ra$). In general, the activity concentration of $^{226}Ra$ measured in most soils and rocks is comparable to those of $^{238}U$ and $^{234}U$, suggesting that radium does not tend to migrate from either of its uranium precursors under stable conditions. Radium-226 is primarily an alpha and gamma emitter (Budavari et al. 1989, Casarett 1968).

Radium has been used as a component of luminous paints for clock and instrument dials. It has also been used in the treatment of cancer, in radiography, and in research. Radium is released into the environment in coal fly ash and in uranium mining and processing wastes. The background level of radium in industrial regions in soil is about 8.1 pCi/g. Clays and soil components generally retard the movement of radium in the environment, but acidic processing wastes can enhance its movement. Radium may bioaccumulate in plants and animals, and exposure through the food chain is possible (Budavari et al. 1989, Casarett 1968).

Many environmental problems can be directly attributed to the decay products or daughters of radium. The primary daughters are isotopes of radon—a colorless, odorless, radioactive gas. Radon gas can infiltrate basements and water systems, resulting in significant exposure via inhalation pathways (Budavari et al. 1989, Casarett 1968).

**Thorium-228, -230, -232, -234.** Thorium is a naturally occurring, radioactive metal. Small amounts of thorium are present in all rocks, soil, aboveground and underground water, plants, and animals. These small amounts of thorium contribute to the weak background radiation for such substances. Soil commonly contains an average of about 6 ppm of thorium. Rocks in some underground mines may also contain thorium in a more concentrated form. After these rocks are mined, thorium is usually concentrated and changes into thorium dioxide or other chemical forms. Thorium-bearing rock that has had most of the thorium removed from it is called "depleted" ore or tailings (ATSDR 1990b).

Thorium is a metallic element of the actinide series. It exists in several isotopic forms. The isotope $^{232}Th$ is a naturally occurring element that is radioactive. It decays through the emission of a series of alpha and beta particles, gamma radiation, and the formation of daughter products, finally yielding the stable isotope of lead, $^{208}Pb$. Isotopes $^{234}Th$ and $^{230}Th$ are produced during the decay of naturally occurring $^{238}U$, the isotope $^{228}Th$ during the decay

of $^{232}Th$, and the isotopes $^{231}Th$ and $^{227}Th$ during the decay of $^{235}U$. Of these naturally produced isotopes of thorium, only $^{232}Th$, $^{230}Th$, and $^{228}Th$ have long enough half-lives to be environmentally significant. More than 99.99% of natural thorium is $^{232}Th$; the rest is $^{230}Th$ and $^{228}Th$ (ATSDR 1990b).

Thorium is used to make ceramics, lantern mantles, and metals used in the aerospace industry and in nuclear reactions. Thorium can also be used as a fuel for generating nuclear energy. More than 30 years ago, thorium oxides were used in hospitals to make certain kinds of diagnostic X-ray photographs (ATSDR 1990b).

Because thorium is found almost everywhere, most people in the United States eat some thorium with their food every day. Normally, little of the thorium in lakes, rivers, and oceans gets into the fish or seafood used commercially. More thorium may be found near uncontrolled hazardous waste sites that contain thorium which might not have been disposed of properly. Consequently, people living near one of these sites may be exposed to slightly more thorium as a result of inhaling windblown dust containing thorium or eating food grown in soil contaminated with thorium. Larger-than-normal amounts of thorium might also enter the environment through accidental releases from thorium processing plants (ATSDR 1990b).

Breathing dust contaminated with thorium is the primary pathway for thorium exposure to the body. A large portion of this dustborne thorium will be eliminated by normal bodily functions (urine/feces); however, a small amount of thorium will be taken up by the blood and subsequently transmitted to the bones. Breathing thorium dust may cause an increased chance of developing lung disease and cancer of the lung or pancreas many years after being exposed. Changes in genetic material have also been shown to occur in workers who breathed thorium dust. Liver diseases and effects on the blood have been found in people injected to take special X rays. Many types of cancer have been shown to occur in these people many years after thorium was injected in their bodies. Since thorium is radioactive and may be stored in bone for a long time, bone cancer is also a potential concern for people exposed to thorium. Animal studies have shown that breathing in thorium may result in lung damage. Other studies in animals suggest drinking massive amounts of thorium can cause death from metal poisoning. The presence of large amounts of thorium in the environment could result in exposure to more hazardous radioactive decay products of thorium, such as radium and thoron, which is an isotope of radon. Thorium is not known to cause birth defects or to affect childbearing abilities (ATSDR 1990b).

**Uranium-234, -235, -238.** Naturally occurring uranium is a lustrous, silver-colored, radioactive, malleable, and ductile metal. Uranium is used almost exclusively as a fuel for nuclear energy; however, it can be used in pigments, ceramics, and photographic chemical processes. The three naturally occurring isotopes of uranium are $^{238}U$ (>99%), $^{235}U$ (about 0.72%), and $^{234}U$ (0.06%). The half-lives of $^{234}U$, $^{235}U$, and $^{238}U$ are 2.4E+05 years, 7.0E+08 years, and 4.5E+09 years, respectively. The average concentration of uranium in the Earth's crust is between 2 and 4 ppm. Some ores contain more concentrated uranium deposits. These deposits are mined and then the percentage of $^{235}U$ is increased by an enrichment process. The enriched $^{235}U$ is more radioactive and of great use as a nuclear reactor fuel or weapon component.

Alpha, beta, and gamma radiation are emitted during the decay of uranium. Loss of alpha particles can give rise to other radioactive elements such as $^{230}Th$, $^{226}Ra$, $^{222}Rn$, and $^{218}Po$. Radon is a significant daughter isotope because it is colorless and odorless and, as a gas, can

leak into basements or foundations. Significant exposure by inhalation of radon gas can occur unknowingly. Deposition of radon and its daughters onto the lungs exposes them to alpha radiation, increasing the probability of cancer. Radium has been known to produce bone sarcomas in radium dial painters, resulting in death. The final decay for $^{238}U$ and $^{234}U$ is stable $^{206}Pb$, and the final decay for $^{235}U$ is stable $^{207}Pb$ (ATSDR 1989b).

Uranium may be released into the air by natural processes such as volcanic eruption or resuspension of soil. The combustion of fossil fuels is a potential contamination route. Coal and shale oils have varying levels of uranium depending on ore location. Subsequent burning can concentrate the levels of uranium. Groundwater and surface water releases of uranium are primarily caused by the disposal of uranium mining waste, disposal of nuclear reactor waste, and reactor emissions. Soil naturally contains uranium, but levels may be concentrated by the addition of mining waste and nuclear facility waste. Carrots, potatoes, and other root vegetables contain higher levels of uranium than leafy vegetables. Higher levels of uranium have also been found in cattle that graze near nuclear facilities (ATSDR 1989b).

Concentrated uranium is a highly toxic element both chemically and radiologically. Kidney damage is the primary toxic effect of uranium. Chemical exposure to uranium may cause dermatitis, weight loss, acute necrotic arterial lesions, and possibly liver damage (Sanders 1986). Spontaneous ignition of finely divided uranium at room temperature is a dangerous physical property. Radiological effects are more prominent if uranium is inhaled and subsequently absorbed in the blood stream. Radioactivity in natural uranium is quite low and exists in food, water, and even in human bones. However, exposure to enriched uranium may cause damage to somatic cells resulting in tumors or cause damage to reproductive cells, affecting the next generation. Higher incidence of lung cancer have been observed in uranium mine workers. Exposure to radionuclides with greater specific-activity and shorter half-life increases the risk for developing cancer (ATSDR 1989b).

### 5.4.6.3 Polycyclic aromatic hydrocarbons

PAHs, which include Chestnut Ridge OU 2 COPCs phenanthrene and 2-methylnaphthalene, share a remarkable stability and, because of this stability, they have been found to be quite useful in industry (solvents, lubricants, dyes, etc). Combustion produces a wide variety of aromatic compounds. Ideally, when hydrocarbon (fossil) fuels are burned, carbon dioxide and water are the resulting combustion products. However, complete combustion is rare; therefore, combustion results in the production of soot and smoke. Soot and smoke contain a number of PAHs, some of which are highly toxic and most of which are toxic in large enough doses. Soot from the exhaust of diesel engines contains small PAHs, such as benzene, naphthalene, and phenanthrene, and larger PAHs, such as coronene and ovalene. Soot is believed to be an aggregate of large molecules that have many benzene rings, PAHs included (Aihara 1992).

The degree of carcinogenicity in humans exposed to PAHs directly corresponds to the size of the PAH molecule. Data prove that many PAHs are carcinogenic, as in the case of benzo(a)pyrene, which is a component in coal tar, soot, and tobacco. Researchers have proven the damage cigarette smoking can cause to the lungs over prolonged periods, and tumors have been discovered in occupational workers such as those who fuel coal-fired furnaces and chimney sweeps (Aihara 1992). However, the acute chronic and subchronic effects of PAHs on humans is not well documented, and complete data are unavailable. One reason for the lack of valuable information regarding the toxic effects of PAHs is the

difficulty in determining the source of such illnesses as lung cancer, liver cancer, skin cancer, and various other ailments. Studies of aromatic compounds show that they also exist naturally in space; carbonaceous chondrite meteorites, for example, are known to contain many kinds of aromatic compounds (Aihara 1992). Therefore, the source of such ailments as lung or liver cancer is difficult to determine because researchers are unsure how large an amount of exposure to aromatics an average person will acquire in a lifetime. However, general ailments such as headaches, dizziness, nausea, and malaise have been attributed to automobile exhaust inhalation (Aihara 1992).

## 5.5 RISK CHARACTERIZATION

The purpose of the risk characterization is to integrate and summarize the information presented in the exposure and toxicity assessments; 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 from projected intakes (and exposure) and chemical-specific dose-response data (i.e., SFs). Potential noncarcinogenic (systemic) effects are characterized by comparing projected intakes of contaminants to toxicity values (i.e., RfDs). The numerical risk/HI estimates that are presented in this section 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 (refer to Sects. 5.2, 5.3.4, 5.4.5, and 5.5.5).

### 5.5.1 General Considerations for Evaluating Risks to Human Health

The inorganic analytes listed in the tables in Chap. 5 and Appendix J include chromium III, chromium VI, nickel, and mercury. The analytical laboratory reported detected concentrations for *total* chromium, *total* mercury, and *total* nickel found, with the exception of a few surface water samples taken in the FCAP which were analyzed for chromium VI. Because the concentrations were reported in this form (i.e., no distinction between valences and speciation), it was necessary to assess all types of these analytes, which included the most toxic forms of the metals (for example, chromium VI, nickel salts, and mercury salts). For the surface water in the FCAP, *total* chromium was assumed to be chromium III, because chromium VI was specifically analyzed for in three of the surface water samples and was not detected.

For the dermal contact pathways, nickel salts and mercury salts were used in the calculations of HI, because the %GI values were available for these salts and were not available for the metals (this is the most conservative approach in terms of evaluating risks to human receptors). For all other pathways, metallic nickel and metallic mercury were evaluated. The concentrations of the nickel salts and mercury salts (used in the dermal calculations) were assumed to be the same as those for metallic nickel and metallic mercury, respectively.

#### 5.5.1.1 Methodology for evaluating carcinogens and noncarcinogens

Screening indices (SIs) (i.e., risks and HIs) for carcinogens and noncarcinogens, respectively, are calculated for the Chestnut Ridge OU 2 COPCs. These SIs are based upon EPA-approved or EPA-suggested SFs and RfDs. The SF is based upon an estimate of the incremental lifetime cancer risk which equates to incidence per unit of exposure, and the RfD

is a level of exposure (threshold) below which no noncarcinogenic adverse health effects are expected to occur.

The SI (risk estimate) for carcinogens is calculated by multiplying the EPA-approved SFs for inorganics, organics, and radionuclides by an estimate of the actual exposure to these contaminants (i.e., CDI and/or dose) via external exposure, ingestion, inhalation, or dermal contact. This SI is calculated to indicate the potential of developing excess cancer over a lifetime (EPA 1989c), above and beyond the normal (unavoidable) incidence of developing cancer.

The SI (HI estimate) for noncarcinogens is calculated by determining the ratio of the CDI for ingestion, inhalation, and dermal contact of a contaminant to the contaminant-specific RfD. This HI assumes that below a given level of exposure (i.e., the RfD) populations are unlikely to experience adverse health effects.

To estimate the potential risk to human health from all contaminants in a particular pathway, the SI are summed for all contaminants in that pathway, i.e., a cumulative risk (Total Pathway Risk or Total Pathway HI) is determined from exposure to all contaminants (EPA 1989c). In addition, an overall cumulative risk can be determined by summing the risks from all analytes from all of the pathways; summations are conducted separately for carcinogens and noncarcinogens.

#### 5.5.1.2 EPA guidance for carcinogens and noncarcinogens

The constituents detected in the Chestnut Ridge OU 2 media were evaluated within the context of EPA-approved guidelines (EPA 1989c) in which there are three regions of concern for carcinogenic risk [risk $<$ 1.0E-06, no concern; risk between 1.0E-06 and 1.0E-04, range of concern (or target risk range); and risk $\geq$ 1.0E-04, unacceptable] and two areas of concern in terms of noncarcinogenic/systemic toxicity (HI $<$ 1.0, no concern, and HI $\geq$ 1.0, concern).

### 5.5.2 Current Land Use Conditions—Trespasser Scenario

#### 5.5.2.1 Carcinogenic risk characterization for the Chestnut Ridge OU 2 COPCs

The trespasser land use scenario was evaluated for the three areas in Chestnut Ridge OU 2. The carcinogenic risks to a trespasser/hunter were determined for external exposure to radionuclides in the surface ash for a 2 weeks/year exposure duration. Such an exposure could result from, for example, walking/hunting in these areas.

Table 5.3 lists the risk estimates for the exposure of a trespasser to the Chestnut Ridge OU 2 radionuclide COPCs in the surface ash, via the external exposure pathway. The risks for individual analytes are $\leq$ 5.9E-05 for all three areas (i.e., Sluice Channel Area, FCAP, and UMB). The total pathway risks for all three areas (i.e., the sum of the risks from all analytes in the external exposure pathway) are all $\leq$ 6.0E-05 (the total pathway risks are 7.2E-06, 1.4E-05, and 6.0E-05 for the Sluice Channel Area, FCAP, and UMB, respectively).

The contaminant of concern (COC) that is the main contributor to the total cumulative risks, for all three areas, is $^{228}$Th; all other analytes have risks due to external exposure $\leq$ 3.0E-07.

### 5.5.3 Future Land Use Conditions—Residential Scenario

For the residential land use conditions, numerous exposure pathways (mentioned previously in Sects. 5.1 and 5.3.3) were evaluated in terms of risk and systemic toxicity to human health. The risk/HI table format illustrates these exposure pathways for the various COPCs associated with each area (i.e., Sluice Channel Area, FCAP, UMB, and Chestnut Ridge OU 2 groundwater). The external exposure to radionuclides pathway was evaluated for exposure to the ash, soil, and dry-sediments media; this pathway was not evaluated for exposure to surface water or groundwater.

The inhalation (dust; VOCs) pathway could only be evaluated (risk/HI) for those COPCs with an inhalation SF or RfD (refer to Tables J.21 and J.22). The inhalation exposure pathway was evaluated for both inhalation of soil, ash, and dry-sediment particulates (dust) and for inhalation of VOCs in soil, ash, dry-sediments, surface water, and groundwater; however, for these media, no VOCs were identified as COPCs. Therefore, there were no risk/HI (and CDI) calculations performed for the inhalation of VOCs pathway.

The dermal contact (while showering) pathway was evaluated for the groundwater (Chestnut Ridge OU 2 groundwater) and for surface water (in FCAP and UMB); risk/HI calculations could be performed for only those nonradionuclide COPCs with an oral SF or RfD (refer to Tables J.21 and J.22). For example, for the FCAP surface water, risk from dermal exposure (while showering) could not be calculated because none of the COPCs had an oral SF. Dermal contact (while wading) was evaluated when possible (i.e., when the COPCs had oral SFs or RfDs), for the nonradionuclide COPCs in the ash, wet-sediments, and surface water for the FCAP and UMB. Dermal contact (direct) was evaluated when possible, for the nonradionuclide COPCs in the ash, soil, and dry-sediments in all three areas (Sluice Channel Area, FCAP, and UMB).

The ingestion pathway was evaluated for all COPCs when possible [i.e., when oral SFs or RfDs were available (refer to Tables J.21 and J.22)], for all five media.

#### 5.5.3.1 Carcinogenic risk characterization for the Chestnut Ridge OU 2 COPCs

Tables 5.4 and 5.5 list the risk estimates for the exposure of a resident to the carcinogenic Chestnut Ridge OU 2 COPCs. The results shown in Table 5.4 are for the ingestion, dermal contact, inhalation, and external exposure pathways for the Chestnut Ridge OU 2 media (i.e., ash, sediment, soil, surface water, and groundwater). For the UMB and FCAP, dermal contact with the sediment, ash, and surface water was also evaluated for a wading scenario (i.e., wading in the UMB water or the FCAP ponds). A summary of the risk results will be presented in Sect. 5.5.4.

For the Sluice Channel Area (Table 5.4a), the total pathway risks (i.e., the sum of the risks from all analytes in that pathway) from exposure to the ash are all $\geq$ 4.8E-06, with the exception of the inhalation of dust pathway (risk from inhalation is 4.0E-07). The total cumulative risk (i.e., the sum of the risks from all analytes for all pathways) is 1.7E-04. The main contributors to the total cumulative risk from exposure to Sluice Channel ash are beryllium (ingestion) and $^{228}$Th (external exposure). The total pathway risk from exposure to the soil (beneath the ash) is 1.1E-04 for the external exposure pathway; all other pathways have risks $\leq$ 1.2E-07. The total cumulative risk is 1.1E-04 and the main contributor to this total cumulative risk from exposure to Sluice Channel soil is $^{228}$Th (external exposure).

For the FCAP (Table 5.4b), the total pathway risks from exposure to the ash are all $\geq$ 2.9E-07. The total cumulative risk is 3.4E-04. The main contributors to the total cumulative risk from exposure to FCAP ash are beryllium (ingestion and dermal) and $^{228}$Th (external exposure). The total pathway risk from exposure to the FCAP sediment is 1.0E-04 for the external exposure pathway; all other pathways have risks $\leq$ 1.4E-05. The total cumulative risk is 1.2E-04. The main contributors to the total cumulative risk from exposure to FCAP sediment are beryllium (ingestion) and $^{228}$Th (external exposure). The total pathway risk from exposure to the FCAP surface water is 9.8E-07 for the ingestion pathway; risks could not be determined for all other pathways, because (1) there were no available oral SFs for the nonradionuclide COPCs found in the FCAP surface water and (2) there were no VOCs on the FCAP COPC list to analyze for the inhalation pathway. The total cumulative risk is, therefore, 9.8E-07; this risk is not within or above the EPA range of concern/target risk range.

For the UMB (Table 5.4c), the total pathway risks from exposure to the ash are all $\geq$ 1.9E-07. The total cumulative risk is 1.3E-03 and the main contributor to the total cumulative risk from exposure to UMB ash is $^{228}$Th (external exposure). The total pathway risks from exposure to the UMB sediments are $\geq$ 3.2E-07. The total cumulative risk is 6.3E-04. The main contributors to the total cumulative risk from exposure to UMB sediments are beryllium (ingestion and dermal) and $^{228}$Th (external exposure). The total pathway risks from exposure to the UMB soil are $\geq$ 4.3E-07, with the exception of the inhalation pathway (risk from inhalation is 6.2E-08). The total cumulative risk for exposure to UMB soil is 9.5E-06 and the main contributor to the risk is beryllium (ingestion and dermal). The total pathway risks from exposure to UMB surface water are $\geq$ 1.6E-05. The total cumulative risk from exposure to the COPCs in the UMB surface water is 5.4E-05, and the main contributor to this risk is bis(2-ethylhexyl)phthalate (ingestion and dermal); this analyte is a common laboratory contaminant and is not believed to be associated with Chestnut Ridge OU 2 operations.

For the Chestnut Ridge OU 2 groundwater (Table 5.4d), the total pathway risks are $\geq$ 2.3E-05. The total cumulative risk is 1.6E-04, and the main contributors to the risk are beryllium (ingestion) and bis(2-ethylhexyl)phthalate (dermal).

Table 5.5 shows the risks due to ingestion of various types of homegrown vegetables/fruits. The hypothetical gardens were planted in the ash in the three areas (i.e., Sluice Channel Area, FCAP, and UMB) of Chestnut Ridge OU 2 and were irrigated with the Chestnut Ridge OU 2 groundwater; for the garden grown in the UMB, irrigation with UMB surface water was also evaluated. For all three areas, the pathway risks from ingestion of the various homegrown produce were between 3.4E-06 and 4.3E-05; therefore, no cumulative risks greater than 1.0E-04 (i.e., the EPA's unacceptable risk of $\geq$ 1.0E-04) were determined for ingestion of these homegrown vegetables. The main contributors to these pathway risks are beryllium, bis(2-ethylhexyl)phthalate, $^{238}$U, and $^{228}$Th.

### 5.5.3.2 Noncarcinogenic risk characterization for the Chestnut Ridge OU 2 COPCs

Tables 5.6 and 5.7 list the HI estimates for the exposure of a resident to the Chestnut Ridge OU 2 COPCs. The results shown in Table 5.6 are for the ingestion, dermal contact, and inhalation pathways for the Chestnut Ridge OU 2 media (i.e., ash, sediment, soil, surface water, and groundwater). For the UMB and FCAP, dermal contact with the sediment, ash,

and surface water was also evaluated for a wading (i.e., wading in the UMB water or the FCAP ponds) scenario. A summary of the HI results will be presented in Sect. 5.5.4.

For the Sluice Channel (Table 5.6a), the total pathway HI (i.e., the sum of the HI from all analytes in that pathway) from exposure to the ash are all less than one (1.0), with the exception of the ingestion pathway, which has a pathway HI of 2.0. The total cumulative HI (i.e., the sum of the HIs from all analytes for all pathways) is 2.1. The main contributor to the total cumulative HI from exposure to Sluice Channel ash is arsenic (ingestion). The total cumulative HI from exposure to the soil (beneath the ash) is 0.64, and the only contributor to this total risk is arsenic (ingestion).

For the FCAP (Table 5.6b), the total pathway HIs from exposure to the ash are all less than one (1.0), with the exception of the ingestion pathway, which has a pathway HI of 1.7. The total cumulative HI is 1.8, and the main contributor to the total cumulative HI from exposure to FCAP ash is arsenic (ingestion). The total pathway HI from exposure to the FCAP sediment is 0.98 for the ingestion pathway; all other pathways have HIs $\leq$ 4.2E-02. The total cumulative HI is 1.0, and the main contributor to the total cumulative HI from exposure to FCAP sediment is arsenic (ingestion). The total pathway HI from exposure to the FCAP surface water is 18 for the ingestion pathway; all other pathway HIs are less than one (1.0). The total cumulative HI is 18 and the main contributors are arsenic (ingestion HI=1.6) and manganese (ingestion HI=16).

For the UMB (Table 5.6c), the total pathway HIs from exposure to the ash are all less than one (1.0). The total cumulative HI is 0.92, and the main contributor to the total cumulative HI from exposure to UMB ash is arsenic (ingestion). The total pathway HIs from exposure to the UMB sediments are less than one (1.0), with the exception of the ingestion pathway, which is 28. The total cumulative HI is 29, and the main contributor to the total cumulative HI from exposure to UMB sediments is arsenic (ingestion HI=28). The total pathway HIs from exposure to the UMB soil are all less than one (1.0), and the total cumulative HI is also less than 1.0. The total pathway HIs from exposure to UMB surface water are less than 1.0, with the exception of the ingestion pathway, which is 17. The total cumulative HI is 18, and the main contributors to the HI are arsenic (ingestion HI=12) and manganese (ingestion HI=4.9).

For the Chestnut Ridge OU 2 groundwater (Table 5.6d), the total pathway HI is less than one (1.0) for the dermal (showering) pathway and greater than one (1.0) for the ingestion pathway. The total cumulative HI is 12, and the main contributor to the HI is manganese via ingestion (HI=9.5).

Table 5.7 shows the HIs due to ingestion of various types of homegrown vegetables/fruits. The hypothetical gardens were planted in the ash in the three areas (i.e., Sluice Channel Area, FCAP, and UMB) of Chestnut Ridge OU 2 and were irrigated with the Chestnut Ridge OU 2 groundwater; for the garden grown in the UMB, irrigation with UMB surface water was also evaluated. For all three areas, the pathway HIs from ingestion of the various homegrown produce were between 4.9 and 15. The main contributors to these pathway HIs are arsenic, cadmium, manganese, and mercury.

### 5.5.4 Risk Characterization Summary

The results of the carcinogenic risk (Tables 5.3, 5.4, and 5.5) and HI (i.e., systemic toxicity) (Tables 5.6 and 5.7) assessments for the Chestnut Ridge OU 2 COPCs are summarized in Tables 5.8–5.11. Tables 5.8 and 5.9 contain the Chestnut Ridge OU 2 COCs that have a risk $\geq$ 1.0E-06 and/or an HI $\geq$ 1.0 for the residential and trespasser land uses, respectively. The results summarized in Tables 5.10 and 5.11 are the cumulative risks and HIs for the residential and trespasser scenarios, respectively.

For the residential land use scenario (refer to Table 5.8), the ingestion (ash, soil, and/or sediment), dermal contact (direct and/or showering), vegetable/fruit ingestion, and external exposure (to radionuclides in ash, soil, and/or sediment) pathways contribute to significant risks/HIs for a human receptor. The results of this risk assessment also show that the inhalation of dust and/or VOCs pathway does not contribute significantly (i.e., risk $\geq$ 1.0E-06 and/or HI $\geq$ 1.0) to risks to human health for the COPCs identified for Chestnut Ridge OU 2 (with the exception of exposure to arsenic in the UMB dry-sediments via inhalation of dust; risk is 5.1E-06). Table 5.8 is divided by area (Sluice Channel, FCAP, UMB, and Chestnut Ridge OU 2 groundwater) and media; the COCs identified with an asterisk have risks $\geq$ 1.0E-04.

In summary for the Sluice Channel (Table 5.8), $^{228}$Th shows risks $\geq$ 1.0E-04 for the external exposure pathway for the ash and soil (beneath the ash) media; arsenic gives an HI $\geq$ 1.0 for incidental ingestion of Sluice Channel ash. Hazard indices $\geq$ 1.0 are found for the ingestion of vegetables/fruits pathway for arsenic and manganese.

In summary for the FCAP (Table 5.8), $^{228}$Th shows a risk $\geq$ 1.0E-04 for the external exposure to the ash pathway. Arsenic shows an HI $\geq$ 1.0 for incidental ingestion of ash; arsenic and manganese give HIs $\geq$ 1.0 for ingestion of the FCAP surface water; HIs $\geq$ 1.0 are found for the ingestion of vegetables/fruits pathway for arsenic.

In summary for the UMB (Table 5.8), $^{228}$Th shows a risk $\geq$ 1.0E-04 for external exposure to the ash and sediment in this area. Arsenic shows an HI $\geq$ 1.0 for incidental ingestion of sediment; arsenic and manganese give HIs $\geq$ 1.0 for ingestion of the UMB surface water, and HIs $\geq$1.0 are found for the ingestion of vegetables/fruits pathway for arsenic, cadmium, manganese, and mercury.

In summary for the Chestnut Ridge OU 2 groundwater (Table 5.8), a risk $\geq$ 1.0E-04 for beryllium is shown for the ingestion of Chestnut Ridge OU 2 groundwater pathway; manganese shows an HI $\geq$ 1.0 for ingestion of Chestnut Ridge OU 2 groundwater.

For the trespasser land use scenario (Table 5.9), the external exposure pathway was evaluated for exposure to Chestnut Ridge OU 2 radionuclide COPCs in the ash in the three (i.e., Sluice Channel, FCAP, and UMB) areas; risks for individual radionuclides are reported in this summary table only if they are $\geq$ 1.0E-06. Thorium-228 is the only radionuclide that shows a risk $\geq$ 1.0E-06 for the trespasser scenario; individual risks from external exposure to the radionuclide COPCs in the ash did not exceed 1.0E-04 in any of the three areas.

The cumulative risk and cumulative HI results are summarized in Tables 5.10 and 5.11 for the residential and trespasser land uses, respectively. The cumulative risk/HI values reported in Table 5.10 do not include contributions from ingestion of fruits and vegetables.

Note that leafy vegetable ingestion only is reported separately in this table; the risks and HIs for ingestion of this vegetable/fruit type were greater than the risks/HI determined for ingestion of the other vegetable/fruit types (i.e., root vegetables, legumes, fruiting vegetables, and fruits).

The cumulative risks (Table 5.10) from exposure to each media (including contributions from the ingestion, dermal, inhalation, and external exposure pathways) are all greater than 1.0E-04, with the exception of exposure to the FCAP and UMB surface water and the UMB soil. The total cumulative risk estimates (i.e., the sum of the risks from all analytes across all pathways and all media, for each area) for the carcinogenic Chestnut Ridge OU 2 COPCs, under the residential land use scenario ***and not including contributions from the vegetable/fruit ingestion pathway,*** were 2.8E-04, 4.6E-04, 2.0E-03, and 1.6E-04 for the Sluice Channel, FCAP, UMB, and Chestnut Ridge OU 2 groundwater, respectively; this information is not shown in Table 5.10.

The total cumulative pathway HI estimates (i.e., the sum of the risks from all analytes across all pathways and all media, for each area) for the noncarcinogenic Chestnut Ridge OU 2 COPCs, under the residential land use scenario *and not including contributions from the vegetable/fruits ingestion pathway*, were 2.7, 21, 48, and 12 for the Sluice Channel, FCAP, UMB, and Chestnut Ridge OU 2 groundwater, respectively; this information is not shown in Table 5.10.

The total cumulative risk estimates (i.e., the sum of the risks from all analytes for the external exposure to ash pathway) for the radionuclide Chestnut Ridge OU 2 COPCs, under the trespasser land use scenario, are 7.2E-06, 1.4E-05, and 6.0E-05 for the Sluice Channel, FCAP, and UMB areas, respectively (refer to Table 5.11).

### 5.5.5 Uncertainty Analysis and Assumptions

Risk assessment as a scientific activity is subject to uncertainty (Table J.24), although the methodology used in this baseline HHRA follows EPA guidelines. The risk evaluation in this report is subject to uncertainty pertaining to sampling and analysis, exposure assessment estimations, and availability of toxicological data.

The major assumptions used in risk assessment are (1) that contaminant concentrations detected and reported by the analytical laboratory are representative of true analyte concentrations in soils (i.e., the analyte concentration remains constant over the sampling and analysis time period), (2) that the intake rates and exposure parameters are representative of actual potentially exposed populations, and (3) that all contaminant exposures and intake are from the site-related exposure media (i.e., no other sources contribute to the receptor's health risk).

Even if these assumptions are true, other areas of uncertainty may apply. The toxicological data (SFs and RfDs) are frequently updated and revised, which can lead to over- or underestimation of risks. These values are often extrapolations from animals to humans, which also induces uncertainties in toxicity values; differences may exist in chemical absorption, metabolism, excretion, and toxic response between animals and humans. The EPA takes into account differences in body weight, surface area, and pharmacokinetic relationships between animals and humans to minimize these uncertainties. Other uncertainties associated with toxicological and exposure assessments are discussed in Sects. 5.3.4 and 5.4.5.

For each exposure pathway, assumptions were made about the parameters (e.g., exposure frequency and duration), the routes of exposure, the amount of the contaminated media an individual could be exposed to, and intake rates for different routes of exposure. In the absence of site-specific data, the assumptions used in this baseline HHRA are consistent with EPA guidance (EPA 1989c, 1989b, 1991a, 1991c, 1992a). However, the residential scenario was a conservative exposure estimate with EPA-approved parameters (and default values) that may substantially overestimate the risks. Many exposure variables/parameters recommended by EPA for the residential scenario represent 90th and 95th percentile values. When several of these upper-bound values are combined in estimating exposure for any one pathway, the resulting risks can be in excess of the 99th percentile exposure and therefore outside the range that may be reasonably expected.

The risk of increased cancer incidence from exposure to low-level radiation is estimated by application of a risk factor to either the radiation dose or the radionuclide intake. Regardless of the type of risk factor used, the same basic uncertainties remain. These uncertainties are related to the model used for determining the health effects of radiation exposure. The model most frequently used for determining risk of radiation exposure is the linear nonthreshold model which assumes there is some increased risk for any increment of radiation exposure with no threshold beneath which effects are not seen. This is the most conservative model for evaluating radiation risk; it uses data from high-dose radiation exposures (such as from the survivors of the atomic bomb) and extrapolates risk from these high exposures to the low-level environment or occupational dose range. The current EPA-recommended radiation risk factors are based on the 1980 National Academy of Sciences Biological Effects of Ionizing Radiation Committee (BEIR III) report. The BEIR III recommendations were increased slightly by EPA to reflect recent information on the health effects of exposure to ionizing radiation. In early 1990, the National Academy of Sciences published the results of the most recent studies of the health effects of ionizing radiation, the BEIR V report, which increases the estimates of cancer risk by a factor of 3 to 5 over the BEIR III report. These increases are based primarily on a re-evaluation of the doses received by the atomic bomb victims.

The quality of the analytical data used in risk assessment depends on the adequacy of the set of procedures that specify how samples were selected and handled (Energy Systems 1993b). Uncertainties associated with the data can include sampling errors, laboratory analysis errors, and data analysis errors. The quality assurance/quality control procedures (Kimbrough et al. 1988) are used to minimize these uncertainties, which are expected to have a low effect on either overestimating or underestimating the risks.

In addition, some current analytical methods are limited in their ability to achieve detection limits appropriate for use in risk assessment. Therefore, risks may be overestimated as a result of analyte concentrations being reported at the method detection limit, which may be greater than the concentration at which adverse health effects would occur.

The more extensive the data base, the more certain are the results of the assessment. The number of samples collected (for the various media) to represent the Chestnut Ridge OU 2 contamination were relatively small, and it is unknown how representative the samples are of the OU. According to EPA guidance (EPA 1992c), a minimum of 20 samples are required to provide consistent estimates of the mean. In calculating the representative concentrations for each Chestnut Ridge OU 2 COPC, the UCL95 was used as a conservative

estimate of the concentration to minimize the likelihood that exposures would be underestimated; this assumption may moderately overestimate the risks.

As mentioned previously, in the analytical analyses for metals (total metals only), risks may be overestimated because the metals that are present are conservatively assumed to be in their most toxic forms. Furthermore, not all of the Chestnut Ridge OU 2 COPCs (Table 5.2) currently have toxicity values; this can lead to an underestimation of total risk because quantitative analysis of such chemicals is currently not possible.

In the absence of information on the toxicity of specific chemical mixtures, additive (cumulative) risks and HIs are assumed (EPA 1989c). Limitations of this approach for noncarcinogens (HI) include (1) the effects of a mixture of chemicals are generally unknown; it is possible that the interactions could be *synergistic or antagonistic*; (2) the RfD/RfCs have different accuracy and precision and are not based on the same severity or effect; and (3) HI adaptivity (or CDI adaptivity) is most properly applied to compounds that induce the same effects by the same mechanism. Therefore, the potential for occurrence of noncarcinogenic effects can be overestimated for chemicals that act by different mechanisms and on different target organs.

Limitations of this additive risk approach for multiple carcinogens include: (1) the SFs represent the upper 95th percentile estimate of potency; therefore, summing individual risks can result in an excessively conservative estimate of total lifetime cancer risk and (2) the target organs of multiple carcinogens may be different, so the risks would not be additive. In the absence of data, adaptivity for risks and HIs was assumed for this baseline HHRA. However, because total risks and HIs are usually driven by a few specific chemicals, segregation of risks and HIs by target organ would not have resulted in significantly different outcomes.

## 5.6 SUMMARY AND CONCLUSIONS

### 5.6.1 Chemicals of Potential Concern

Table 5.1 lists the Chestnut Ridge OU 2 COPCs evaluated quantitatively in this baseline HHRA. Unacceptable carcinogenic risks (i.e., risks $\geq$ 1.0E-04) and/or noncarcinogenic HIs (i.e., HIs $\geq$ 1.0) were found for several of these analytes; refer to Tables 5.8 and 5.9.

Thorium-228 (external exposure) in the Sluice Channel ash, Sluice Channel soil, FCAP ash, UMB ash, and UMB sediment and beryllium (ingestion) in the Chestnut Ridge OU 2 groundwater were identified as showing the highest carcinogenic risks (i.e., individual risks $\geq$ 1.0E-04) and are, therefore, COCs.

Arsenic (ingestion) in the Sluice Channel ash, FCAP ash, FCAP surface water, UMB sediment, UMB surface water; manganese (ingestion) in the FCAP and UMB surface water and in the Chestnut Ridge OU 2 groundwater; and arsenic, cadmium, manganese, and mercury (ingestion of homegrown vegetables) show the highest HIs (i.e., individual HIs $\geq$ 1.0) and are, therefore, COCs.

### 5.6.2 Exposure Assessment

Two exposure scenarios—a trespasser scenario and a residential scenario—were evaluated in this baseline HHRA. Because of the location and current land use of Chestnut Ridge OU 2, a trespasser was evaluated as the most reasonable and most likely receptor of the Chestnut Ridge OU 2 media. This trespasser evaluation represents a lower-bound estimate of risk and is not likely to overestimate risks at Chestnut Ridge OU 2 under current land use conditions. An upper-bound on the risk to human receptors was evaluated using residential parameters to provide managers with potential risks that are conservative and unlikely to underestimate exposure to future receptors of the Chestnut Ridge OU 2 media.

The exposure routes that were evaluated for each Chestnut Ridge OU 2 media (i.e., evaluated in terms of CDI, risk, and HI for the residential land use) included ingestion, inhalation, dermal contact, and external exposure. The exposure media for Chestnut Ridge OU 2 included the ash, soil, sediment, surface water, and groundwater; ingestion of homegrown vegetables/fruits was also evaluated. External exposure to radionuclides in the surface ash was evaluated for the trespasser land use scenario.

For the two Chestnut Ridge OU 2 COCs that were identified as showing the highest carcinogenic risks (i.e., $^{228}$Th and beryllium) (refer to Sect. 5.6.1), risk from the external exposure pathway (for $^{228}$Th) and the groundwater ingestion pathway (for beryllium) contributed the most to the total cumulative carcinogenic risks.

For the Chestnut Ridge OU 2 COCs that were identified as having the highest toxic effects, arsenic (ingestion of ash, sediment, and surface water), manganese (ingestion of surface water and groundwater), and arsenic, cadmium, manganese, and mercury (ingestion of homegrown vegetables/fruits) contributed the most to the total cumulative HIs.

### 5.6.3 Toxicity Assessment

Several Chestnut Ridge OU 2 COPCs were identified as having unacceptable carcinogenic and/or noncarcinogenic/systemic effects to human health, that is, these COPCs have a risk $\geq$ 1.0E-04 and/or HI $\geq$ 1.0 (EPA 1989c) (refer to Sect. 5.6.1). Both qualitative and quantitative toxicity information for each COPC (Tables J.21–J.23) can be found in Sect. 5.4 of this baseline HHRA.

### 5.6.4 Risk Characterization

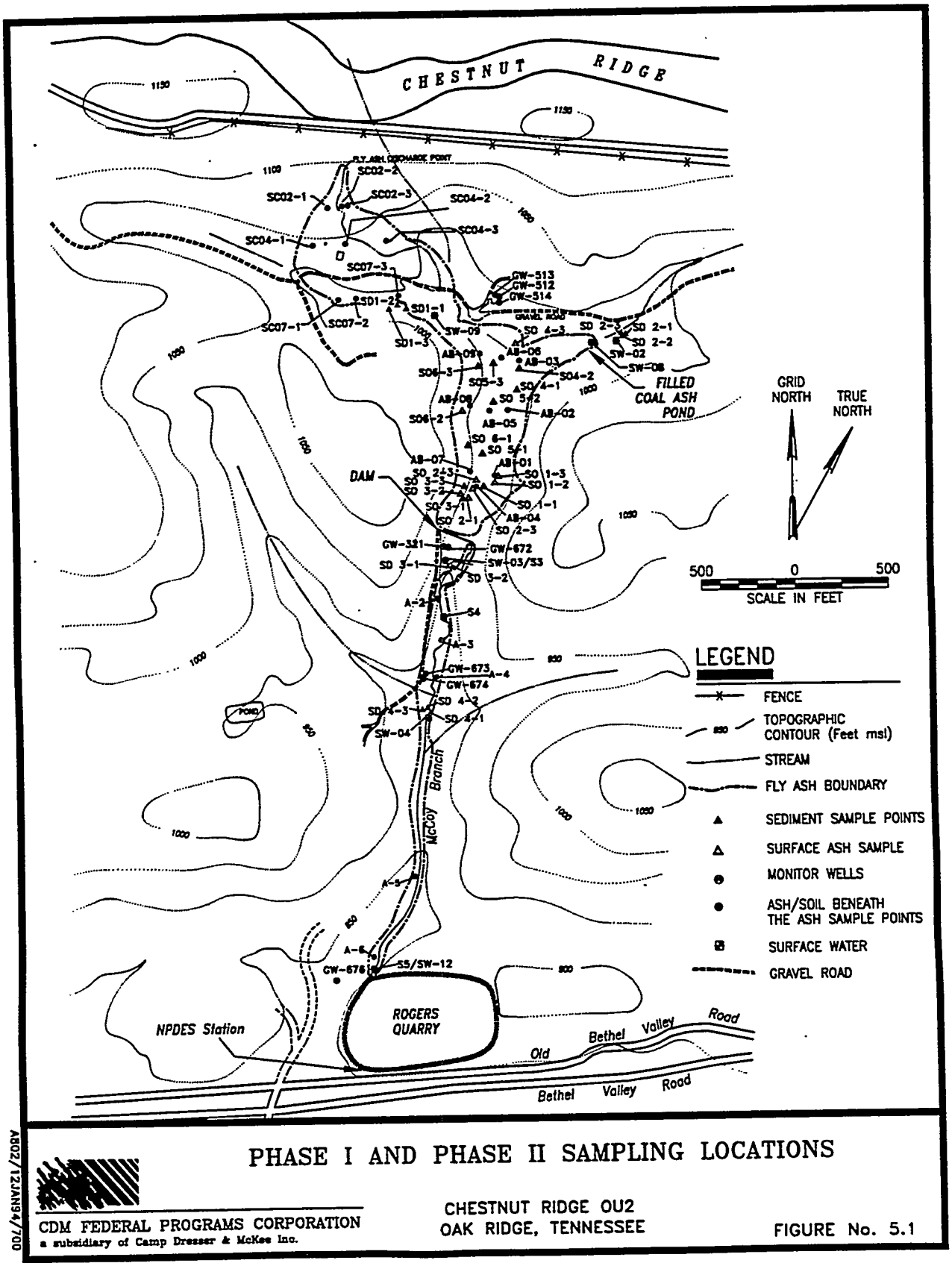
The results of the carcinogenic risk (Tables 5.3, 5.4, and 5.5) and systemic toxicity (Tables 5.6 and 5.7) assessments for the Chestnut Ridge OU 2 COPCs are summarized in Tables 5.8 to 5.11. Tables 5.8 and 5.9 show the Chestnut Ridge OU 2 COCs that have a risk $\geq$ 1.0E-06 and/or an HI $\geq$ 1.0 for the residential and trespasser land uses, respectively. The results summarized in Tables 5.10 and 5.11 are the cumulative risks and HIs for the residential and trespasser scenarios, respectively. Note that EPA guidelines (EPA 1989c) for contaminated sites consider three regions of carcinogenic risk [risk < 1.0E-06, no concern; risk between 1.0E-06 and 1.0E-04, range of concern (or target risk range); and risk $\geq$ 1.0E-04, unacceptable] and two areas of concern in terms of noncarcinogenic/systemic effects (HI < 1.0, no concern, and HI $\geq$ 1.0, concern).

As previously discussed, $^{228}$Th (external exposure) in the Sluice Channel ash, Sluice Channel soil, FCAP ash, UMB ash, and UMB sediment and beryllium (ingestion) in the Chestnut Ridge OU 2 groundwater were identified as showing the highest carcinogenic risks (i.e., the individual risks from these COCs were $\geq$ 1.0E-04 for exposure to these media in these areas).

Arsenic (ingestion) in the Sluice Channel ash, FCAP ash, FCAP surface water, UMB sediment, UMB surface water; manganese (ingestion) in the FCAP and UMB surface water and in the Chestnut Ridge OU 2 groundwater; and arsenic, cadmium, manganese, and mercury (ingestion of homegrown vegetables) show the highest HIs (i.e., the individual HIs from these COCs were $\geq$ 1.0 for exposure to these media in these areas).

Cumulative risks $\geq$ 1.0E-04 were found for exposure to (1) both media (ash and soil) in the Sluice Channel, (2) two out of three media (ash and sediment) in the FCAP, (3) two out of four media (ash and sediment) in the UMB, and (4) the Chestnut Ridge OU 2 groundwater (i.e., groundwater). Note that a cumulative risk of 1.3E-03 was found for exposure to the UMB ash. Cumulative risks between 1.0E-06 and 1.0E-04 were found for exposure to the UMB soil and surface water, and a cumulative risk of 9.8E-07 was found for exposure to the FCAP surface water.

Cumulative HIs $\geq$ 1.0 were found for exposure to all media types (ash, sediment, soil, surface water, and groundwater) in all areas (Sluice Channel, FCAP, UMB, and Chestnut Ridge OU 2 groundwater), with the exception of exposure to the UMB area soil and the Sluice Channel Area soil.




PHASE I AND PHASE II SAMPLING LOCATIONS

CDM FEDERAL PROGRAMS CORPORATION
a subsidiary of Camp Dresser & McKee Inc.

CHESTNUT RIDGE OU2
OAK RIDGE, TENNESSEE

FIGURE No. 5.1

AB02/12JAN94/700

Fig. 5.1. Phase 1 and Phase 2 sampling locations.

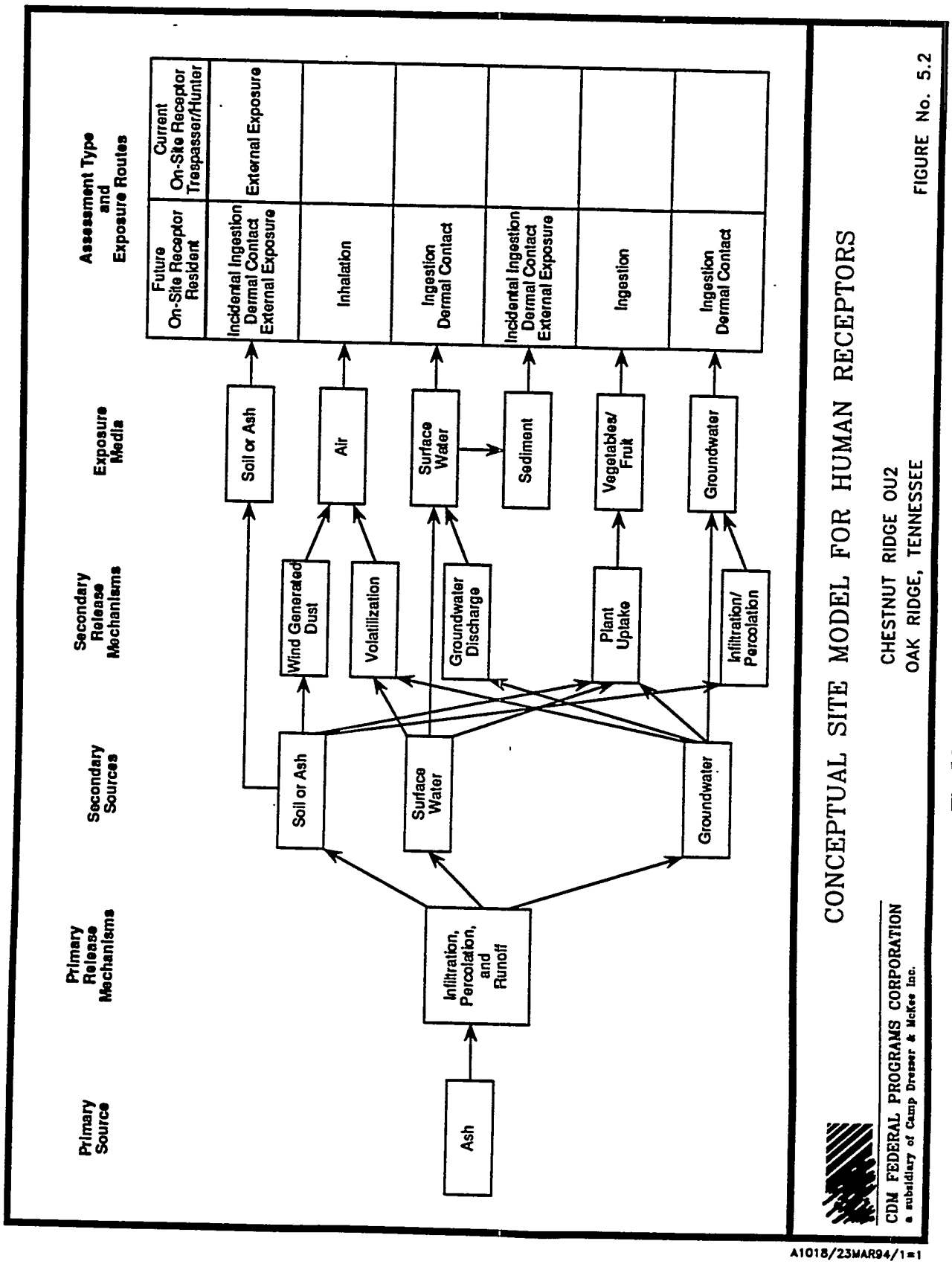



FIGURE No. 5.2

Fig. 5.2 Conceptual site model for human receptors.

**Table 5.1a. COPCs for Chestnut Ridge OU 2 Sluice Channel Area that will be evaluated quantitatively[a]**

| Analyte | Frequency of detection | Representative concentration[b] |
|---|---|---|
| **Ash** | | |
| *Inorganics (mg/kg)* | | |
| Arsenic | 3/3 | 1.59E+02 |
| Barium | 2/3 | 4.21E+02 |
| Beryllium | 3/3 | 1.60E+00 |
| Chromium VI | 3/3 | 1.71E+01 |
| Manganese | 3/3 | 3.44E+02 |
| Vanadium | 3/3 | 4.75E+01 |
| *Radionuclides (pCi/g)* | | |
| Thorium-228 | 3/3 | 1.10E+00 |
| Thorium-230 | 2/3 | 2.90E+00 |
| Thorium-232 | 3/3 | 1.30E+00 |
| Uranium-234 | 3/3 | 4.40E+00 |
| Uranium-238 | 3/3 | 3.30E+00 |
| **Soil (beneath ash)** | | |
| *Inorganic (mg/kg)* | | |
| Arsenic | 3/3 | 5.12E+01 |
| *Radionuclides (pCi/g)* | | |
| Thorium-228 | 3/3 | 8.40E-01 |
| Thorium-230 | 1/3 | 1.40E+00 |

[a]The analytes to be evaluated quantitatively in the baseline HHRA have at least one detected value; these analytes also have EPA-approved slope factors and/or reference doses.

[b]The representative concentration is the smallest of two values—the maximum detected concentration versus the upper 95% confidence limit on the mean concentration (EPA 1989c).

**Table 5.1b. COPCs for Chestnut Ridge OU 2 FCAP that will be evaluated quantitatively[a]**

| Analyte | Frequency of detection | Representative concentration[b] |
|---|---|---|
| **Ash** | | |
| *Inorganics (mg/kg)* | | |
| Arsenic | 15/15 | 1.31E+02 |
| Barium | 15/15 | 4.50E+02 |
| Beryllium | 15/15 | 4.39E+00 |
| Chromium VI | 15/15 | 2.51E+01 |
| Manganese | 15/15 | 1.52E+02 |
| Vanadium | 15/15 | 8.49E+01 |
| *Radionuclides (pCi/g)* | | |
| Thorium-228 | 15/15 | 2.17E+00 |
| Thorium-230 | 15/15 | 4.02E+00 |
| Thorium-232 | 15/15 | 2.12E+00 |
| Uranium-234 | 15/15 | 3.87E+00 |
| Uranium-235 | 15/15 | 2.35E-01 |
| Uranium-238 | 15/15 | 3.87E+00 |
| **Sediment** | | |
| *Inorganics (mg/kg)* | | |
| Arsenic | 2/2 | 7.66E+01 |
| Beryllium | 2/2 | 2.10E+00 |
| Chromium VI | 2/2 | 3.29E+01 |
| Vanadium | 2/2 | 3.55E+01 |
| *Radionuclides (pCi/g)* | | |
| Thorium-228 | 2/2 | 7.30E-01 |
| Thorium-232 | 2/2 | 1.60E+00 |
| Uranium-234 | 2/2 | 3.90E+00 |
| Uranium-235 | 2/2 | 2.90E-01 |
| Uranium-238 | 2/2 | 3.00E+00 |

**Table 5.1b (continued)**

| Analyte | Frequency of detection | Representative concentration[b] |
|---|---|---|
| **Surface water** | | |
| *Inorganics (mg/L)* | | |
| Arsenic | 1/3 | 1.00E-02 |
| Barium | 3/3 | 6.89E-02 |
| Chromium III | 1/3 | 8.80E-03 |
| Manganese | 3/3 | 1.70E+00 |
| *Radionuclides (pCi/L)* | | |
| Thorium-228 | 1/3 | 2.20E-01 |
| Thorium-230 | 1/3 | 2.00E+00 |
| Uranium-234 | 2/3 | 3.30E-01 |
| Uranium-238 | 2/3 | 1.10E-01 |

[a]The analytes to be evaluated quantitatively in the baseline HHRA have at least one detected value; these analytes also have EPA-approved slope factors and/or reference doses.

[b]The representative concentration is the smallest of two values—the maximum detected concentration versus the upper 95% confidence limit on the mean concentration (EPA 1989c).

**Table 5.1c. COPCs for Chestnut Ridge OU 2 UMB that will be evaluated quantitatively[a]**

| Analyte | Frequency of detection | Representative concentration[b] |
|---|---|---|
| **Ash** | | |
| *Inorganics (mg/kg)* | | |
| Arsenic | 6/6 | 5.92E+01 |
| Barium | 6/6 | 5.60E+02 |
| Beryllium | 6/6 | 2.90E+00 |
| Cadmium | 5/6 | 3.67E+00 |
| Chromium VI | 6/6 | 1.69E+01 |
| Manganese | 6/6 | 8.76E+02 |
| Mercury | 6/6 | 5.81E-01 |
| Vanadium | 6/6 | 7.25E+01 |
| *Radionuclides (pCi/g)* | | |
| Cesium-137 | 6/6 | 1.29E-01 |
| Thorium-228 | 1/1 | 9.20E+00 |
| Thorium-234 | 1/1 | 6.30E+00 |
| Uranium-238 | 6/6 | 3.43E+00 |
| **Sediment** | | |
| *Inorganics (mg/kg)* | | |
| Arsenic | 2/2 | 2.30E+03 |
| Barium | 2/2 | 5.96E+02 |
| Beryllium | 2/2 | 4.70E+00 |
| Chromium VI | 2/2 | 2.17E+01 |
| Manganese | 2/2 | 1.37E+04 |
| *Radionuclides (pCi/g)* | | |
| Thorium-228 | 2/2 | 4.30E+00 |
| Thorium-230 | 1/2 | 2.70E+00 |
| Thorium-232 | 2/2 | 1.20E+00 |
| Uranium-234 | 2/2 | 2.20E+00 |
| Uranium-235 | 2/2 | 2.10E-01 |
| Uranium-238 | 2/2 | 2.20E+00 |

**Table 5.1c (continued)**

| Analyte | Frequency of detection | Representative concentration[b] |
|---|---|---|
| **Soil (beneath ash)** | | |
| *Inorganics (mg/kg)* | | |
| Arsenic | 11/11 | 2.71E+01 |
| Barium | 11/11 | 1.08E+02 |
| Beryllium | 9/11 | 9.19E-01 |
| Cadmium | 11/11 | 7.48E+00 |
| Nickel | 10/11 | 2.66E+01 |
| *Radionuclide (pCi/g)* | | |
| Thorium-234 | 2/2 | 5.15E+00 |
| **Surface water** | | |
| *Inorganics (mg/L)* | | |
| Arsenic | 9/11 | 7.55E-02 |
| Barium | 11/11 | 9.08E-02 |
| Manganese | 8/11 | 5.14E-01 |
| *Radionuclides (pCi/L)* | | |
| Radium-226 | 7/8 | 5.67E-01 |
| Thorium-228 | 1/3 | 4.10E-02 |
| Thorium-230 | 2/3 | 6.20E-01 |
| Uranium-234 | 3/3 | 5.00E-01 |
| Uranium-238 | 7/11 | 3.36E-01 |
| *Organic (mg/L)* | | |
| bis(2-Ethylhexyl)phthalate | 2/3 | 4.80E-02 |

[a]The analytes to be evaluated quantitatively in the baseline HHRA have at least one detected value; these analytes also have EPA-approved slope factors and/or reference doses.

[b]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean concentration (EPA 1989c).

**Table 5.1d. COPCs for Chestnut Ridge OU 2 groundwater that will be evaluated quantitatively[a]**

| Analyte | Frequency of detection | Representative concentration[b] |
|---|---|---|
| **Groundwater** | | |
| *Inorganics (mg/L)* | | |
| Arsenic | 4/15 | 5.00E-03 |
| Barium | 16/16 | 1.88E-01 |
| Beryllium | 5/16 | 1.43E-03 |
| Cadmium | 1/16 | 4.84E-03 |
| Chromium VI | 4/16 | 2.36E-02 |
| Manganese | 14/16 | 1.00E+00 |
| Vanadium | 4/16 | 1.90E-02 |
| Zinc | 15/16 | 4.07E-01 |
| *Radionuclides (pCi/L)* | | |
| Radium-226 | 5/5 | 1.26E+00 |
| Thorium-228 | 1/8 | 1.21E-01 |
| Thorium-232 | 1/8 | 8.84E-02 |
| Uranium-234 | 8/8 | 5.50E-01 |
| Uranium-235 | 1/8 | 4.80E-02 |
| Uranium-238 | 12/16 | 4.15E-01 |
| *Organic (mg/L)* | | |
| bis(2-Ethylhexyl)phthalate | 4/8 | 2.40E-02 |

[a]The analytes to be evaluated quantitatively in the baseline HHRA have at least one detected value; these analytes also have EPA-approved slope factors and/or reference doses.

[b]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean concentration (EPA 1989c).

**Table 5.2. COPCs for Chestnut Ridge OU 2 Sluice Channel Area, FCAP, UMB, and groundwater that will be evaluated qualitatively[a]**

| Analyte | Frequency of detection | Representative concentration[b] | Concentration units |
|---|---|---|---|
| **Sluice Channel Area** | | | |
| ***Ash*** | | | |
| Aluminum | 3/3 | 1.41E+04 | mg/kg |
| Cobalt | 3/3 | 2.86E+01 | mg/kg |
| Lead | 3/3 | 1.78E+01 | mg/kg |
| **Filled Coal Ash Pond** | | | |
| ***Ash*** | | | |
| Aluminum | 15/15 | 2.19E+04 | mg/kg |
| Cobalt | 15/15 | 1.93E+01 | mg/kg |
| Lead | 15/15 | 1.88E+01 | mg/kg |
| Thallium | 4/15 | 2.21E+00 | mg/kg |
| 2-Methylnaphthalene | 1/14 | 9.30E-02 | mg/kg |
| ***Sediment*** | | | |
| Aluminum | 2/2 | 1.15E+04 | mg/kg |
| Lead | 2/2 | 2.48E+01 | mg/kg |
| ***Surface water*** | | | |
| Aluminum | 1/3 | 7.92E+00 | mg/L |
| Lead | 1/3 | 3.50E-03 | mg/L |
| **Upper McCoy Branch** | | | |
| ***Ash*** | | | |
| Aluminum | 6/6 | 1.98E+04 | mg/kg |
| Cobalt | 6/6 | 1.93E+01 | mg/kg |
| Lead | 6/6 | 1.59E+01 | mg/kg |
| ***Sediment*** | | | |
| Aluminum | 2/2 | 2.27E+04 | mg/kg |
| Cobalt | 2/2 | 8.62E+01 | mg/kg |
| Lead | 2/2 | 1.71E+01 | mg/kg |
| Phenanthrene | 1/2 | 7.00E-02 | mg/kg |

**Table 5.2 (continued)**

| Analyte | Frequency of detection | Representative concentration[b] | Concentration units |
|---|---|---|---|
| ***Soil (beneath ash)*** | | | |
| Cobalt | 10/11 | 2.09E+01 | mg/kg |
| ***Surface water*** | | | |
| Aluminum | 10/11 | 4.29E-01 | mg/L |
| Chloride | 7/8 | 2.72E+00 | mg/L |
| Fluoride | 7/8 | 1.97E-01 | mg/L |
| Lead | 1/11 | 1.63E-03 | mg/L |
| Niobium | 2/8 | 8.02E-03 | mg/L |
| Sulfate | 7/8 | 2.41E+01 | mg/L |
| **Groundwater** | | | |
| Aluminum | 9/16 | 6.11E+00 | mg/L |
| Chloride | 8/8 | 2.95E+00 | mg/L |
| Cobalt | 3/16 | 1.22E-02 | mg/L |
| Fluoride | 4/5 | 6.86E-01 | mg/L |
| Lead | 7/15 | 1.40E-02 | mg/L |
| Niobium | 2/5 | 9.27E-03 | mg/L |
| Sulfate | 8/8 | 2.34E+01 | mg/L |

[a]The analytes to be evaluated qualitatively in this baseline HHRA have at least one detected value; however, no approved toxicity values (i.e., slope factors or reference doses) are currently available.

[b]The representative concentration is the smallest of two values—the maximum detected concentration versus the upper 95% confidence limit of the mean concentration (EPA 1989c).

Table 5.3. Carcinogenic risks for Chestnut Ridge OU 2 radionuclide COPCs—trespasser scenario

| Analyte | Representative concentration[a] (pCi/g) | External exposure risk |
|---|---|---|
| **Sluice Channel Area Ash** | | |
| Thorium-228 | 1.10E+00 | 7.1E-06 |
| Thorium-230 | 2.90E+00 | 1.8E-10 |
| Thorium-232 | 1.30E+00 | 3.9E-11 |
| Uranium-234 | 4.40E+00 | 1.5E-10 |
| Uranium-238 | 3.30E+00 | 1.4E-07 |
| **Total Pathway Risk[b]** | | 7.2E-06 |
| **Filled Coal Ash Pond Ash** | | |
| Thorium-228 | 2.17E+00 | 1.4E-05 |
| Thorium-230 | 4.02E+00 | 2.5E-10 |
| Thorium-232 | 2.12E+00 | 6.4E-11 |
| Uranium-234 | 3.87E+00 | 1.3E-10 |
| Uranium-235 | 2.35E-01 | 6.5E-08 |
| Uranium-238 | 3.87E+00 | 1.6E-07 |
| **Total Pathway Risk[b]** | | 1.4E-05 |
| **Upper McCoy Branch Ash** | | |
| Cesium-137 | 1.29E-01 | 3.0E-07 |
| Thorium-228 | 9.20E+00 | 5.9E-05 |
| Thorium-234 | 6.30E+00 | 2.5E-08 |
| Uranium-238 | 3.43E+00 | 1.4E-07 |
| **Total Pathway Risk[b]** | | 6.0E-05 |

[a]The representative concentration is the smallest of two values—the maximum detected concentration versus the upper 95% confidence limit on the mean.

[b]The sum of the risks from all radionuclides for this pathway.

Table 5.4a. Carcinogenic risks from Chestnut Ridge OU 2 Sluice Channel Area COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct) risk | Inhalation of dust risk | External exposure risk | Total risk[b] |
|---|---|---|---|---|---|---|
| **Ash** | | | | | | |
| **Inorganics** | | | | | | |
| Arsenic | 1.59E+02 | | | 3.5E-07 | | 3.5E-07 |
| Barium | 4.21E+02 | | | | | |
| Beryllium | 1.60E+00 | 1.1E-05 | 4.8E-06 | 5.9E-10 | | 1.6E-05 |
| Chromium VI | 1.71E+01 | | | 3.1E-08 | | 3.1E-08 |
| Manganese | 3.44E+02 | | | | | |
| Vanadium | 4.75E+01 | | | | | |
| **Radionuclides** | | | | | | |
| Thorium-228 | 1.10E+00 | 7.6E-08 | | 3.9E-09 | 1.5E-04 | 1.5E-04 |
| Thorium-230 | 2.90E+00 | 4.8E-08 | | 3.8E-09 | 3.8E-09 | 5.5E-08 |
| Thorium-232 | 1.30E+00 | 2.0E-08 | | 1.7E-09 | 8.1E-10 | 2.2E-08 |
| Uranium-234 | 4.40E+00 | 8.9E-08 | | 5.2E-09 | 3.2E-09 | 9.7E-08 |
| Uranium-238 | 3.30E+00 | 1.2E-07 | | 7.8E-09 | 2.9E-06 | 3.0E-06 |
| **Total Pathway Risk[c]** | | 1.1E-05 | 4.8E-06 | 4.0E-07 | 1.5E-04 | 1.7E-04[d] |
| **Soil (beneath ash)** | | | | | | |
| **Inorganic** | | | | | | |
| Arsenic | 5.12E+01 | | | 1.1E-07 | | 1.1E-07 |

Table 5.4a (continued)

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct) risk | Inhalation of dust risk | External exposure risk | Total risk[b] |
|---|---|---|---|---|---|---|
| **Radionuclides** | | | | | | |
| Thorium-228 | 8.40E-01 | 5.8E-08 | | 3.0E-09 | 1.1E-04 | 1.1E-04 |
| Thorium-230 | 1.40E+00 | 2.3E-08 | | 1.8E-09 | 1.8E-09 | 2.7E-08 |
| **Total Pathway Risk[c]** | | 8.1E-08 | 0.0E+00[e] | 1.2E-07 | 1.1E-04 | 1.1E-04[d] |

[a]The representative concentration is the smallest of two values—the maximum detected concentration versus the upper 95% confidence limit on the mean; for ash and soil, units are pCi/g for radionuclides and mg/kg for all other analytes.

[b]The sum of the ingestion risk, dermal direct-contact risk, inhalation of dust risk, and external exposure risk for each analyte.

[c]The sum of the risks from all analytes for each pathway.

[d]The sum of the risks from all analytes across all pathways.

[e]Since the nonradionuclide COPC for soil does not have an oral SF, no dermal (direct) risk calculation could be performed.

Table 5.4b. Carcinogenic risks from Chestnut Ridge OU 2 FCAP COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct; shower)[b] risk | Dermal (wading) risk | Inhalation (dust; VOCs)[b] risk | External exposure risk | Total risk[c] |
|---|---|---|---|---|---|---|---|
| **Ash** | | | | | | | |
| **Inorganics** | | | | | | | |
| Arsenic | 1.31E+02 | | | | 2.9E-07 | | 2.9E-07 |
| Barium | 4.50E+02 | | | | | | |
| Beryllium | 4.39E+00 | 3.0E-05 | 1.3E-05 | 2.9E-07 | 1.6E-09 | | 4.3E-05 |
| Chromium VI | 2.51E+01 | | | | 4.5E-08 | | 4.5E-08 |
| Manganese | 1.52E+02 | | | | | | |
| Vanadium | 8.49E+01 | | | | | | |
| **Radionuclides** | | | | | | | |
| Thorium-228 | 2.17E+00 | 1.5E-07 | | | 7.7E-09 | 2.9E-04 | 2.9E-04 |
| Thorium-230 | 4.02E+00 | 6.6E-08 | | | 5.3E-09 | 5.2E-09 | 7.6E-08 |
| Thorium-232 | 2.12E+00 | 3.2E-08 | | | 2.7E-09 | 1.3E-09 | 3.6E-08 |
| Uranium-234 | 3.87E+00 | 7.8E-08 | | | 4.6E-09 | 2.8E-09 | 8.5E-08 |
| Uranium-235 | 2.35E-01 | 4.7E-09 | | | 2.7E-10 | 1.4E-06 | 1.4E-06 |
| Uranium-238 | 3.87E+00 | 1.4E-07 | | | 9.1E-09 | 3.3E-06 | 3.5E-06 |
| **Total Pathway Risk[d]** | | 3.0E-05 | 1.3E-05 | 2.9E-07 | 3.7E-07 | 3.0E-04 | 3.4E-04[e] |

Table 5.4b (continued)

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct; shower)[b] risk | Dermal (wading) risk | Inhalation (dust; VOCs)[b] risk | External exposure risk | Total risk[c] |
|---|---|---|---|---|---|---|---|
| **Sediment** | | | | | | | |
| **Inorganics** | | | | | | | |
| Arsenic | 7.66E+01 | | | | 1.7E-07 | | 1.7E-07 |
| Beryllium | 2.10E+00 | 1.4E-05 | 6.3E-06 | 1.4E-07 | 7.8E-10 | | 2.0E-05 |
| Chromium VI | 3.29E+01 | | | | 5.9E-08 | | 5.9E-08 |
| Vanadium | 3.55E+01 | | | | | | |
| **Radionuclides** | | | | | | | |
| Thorium-228 | 7.30E-01 | 5.1E-08 | | | 2.6E-09 | 9.8E-05 | 9.8E-05 |
| Thorium-232 | 1.60E+00 | 2.4E-08 | | | 2.0E-09 | 1.0E-09 | 2.7E-08 |
| Uranium-234 | 3.90E+00 | 7.9E-08 | | | 4.6E-09 | 2.8E-09 | 8.6E-08 |
| Uranium-235 | 2.90E-01 | 5.8E-09 | | | 3.3E-10 | 1.7E-06 | 1.7E-06 |
| Uranium-238 | 3.00E+00 | 1.1E-07 | | | 7.1E-09 | 2.6E-06 | 2.7E-06 |
| **Total Pathway Risk[d]** | | 1.4E-05 | 6.3E-06 | 1.4E-07 | 2.5E-07 | 1.0E-04 | 1.2E-04[e] |
| **Surface water** | | | | | | | |
| **Inorganics** | | | | | | | |
| Arsenic | 1.00E-02 | | | | | | |
| Barium | 6.89E-02 | | | | | | |

Table 5.4b (continued)

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct; shower)[b] risk | Dermal (wading) risk | Inhalation (dust; VOCs)[b] risk | External exposure risk | Total risk[c] |
|---|---|---|---|---|---|---|---|
| Chromium III | 8.80E-03 | | | | | | |
| Manganese | 1.70E+00 | | | | | | |
| **Radionuclides** | | | | | | | |
| Thorium-228 | 2.20E-01 | 2.5E-07 | | | | | 2.5E-07 |
| Thorium-230 | 2.00E+00 | 5.5E-07 | | | | | 5.5E-07 |
| Uranium-234 | 3.30E-01 | 1.1E-07 | | | | | 1.1E-07 |
| Uranium-238 | 1.10E-01 | 6.5E-08 | | | | | 6.5E-08 |
| **Total Pathway Risk[d]** | | 9.8E-07 | 0.0E+00[f] | 0.0E+00[f] | 0.0E+00[g] | | 9.8E-07[e] |

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean; for ash and sediment, units are pCi/g for radionuclides and mg/kg for all other analytes; for surface water, units are pCi/L for radionuclides and mg/L for all other analytes.

[b]The dermal direct-contact pathway is evaluated for ash and sediment; the dermal showering-contact pathway is evaluated for surface water. The inhalation of dust pathway is evaluated for ash and dry sediment; the inhalation of VOCs pathway is evaluated for surface water.

[c]The sum of the ingestion risk, dermal (direct or showering) risk, inhalation (of dust or VOCs) risk, and external exposure risk for each analyte; the risks from exposure to COPCs while wading are NOT included in these totals.

[d]The sum of the risks from all analytes for each pathway.

[e]The sum of the risks from all analytes across all pathways; the risks from exposure to COPCs while wading are NOT included in this total.

[f]Since the nonradionuclide COPCs for surface water do not have an oral SF, no dermal (showering or wading) risk calculations could be performed.

[g]Since the nonradionuclide COPCs for surface water do not have an inhalation SF and are not VOCs, no inhalation (VOCs) risk calculations could be performed.

Table 5.4c. Carcinogenic risks from Chestnut Ridge OU 2 UMB COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct; shower)[b] risk | Dermal (wading) risk | Inhalation (dust; VOCs)[b] risk | External exposure risk | Total risk[c] |
|---|---|---|---|---|---|---|---|
| **Ash** | | | | | | | |
| **Inorganics** | | | | | | | |
| Arsenic | 5.92E+01 | | | | 1.3E-07 | | 1.3E-07 |
| Barium | 5.60E+02 | | | | | | |
| Beryllium | 2.90E+00 | 1.9E-05 | 8.7E-06 | 1.9E-07 | 1.1E-09 | | 2.8E-05 |
| Cadmium | 3.67E+00 | | | | 9.8E-10 | | 9.8E-10 |
| Chromium VI | 1.69E+01 | | | | 3.0E-08 | | 3.0E-08 |
| Manganese | 8.76E+02 | | | | | | |
| Mercury | 5.81E-01 | | | | | | |
| Vanadium | 7.25E+01 | | | | | | |
| **Radionuclides** | | | | | | | |
| Cesium-137 | 1.29E-01 | 4.6E-09 | | | 1.1E-13 | 6.2E-06 | 6.2E-06 |
| Thorium-228 | 9.20E+00 | 6.4E-07 | | | 3.3E-08 | 1.2E-03 | 1.2E-03 |
| Thorium-234 | 6.30E+00 | 3.2E-08 | | | 9.1E-12 | 5.3E-07 | 5.6E-07 |
| Uranium-238 | 3.43E+00 | 1.2E-07 | | | 8.1E-09 | 3.0E-06 | 3.1E-06 |
| **Total Pathway Risk[d]** | | 2.0E-05 | 8.7E-06 | 1.9E-07 | 2.0E-07 | 1.2E-03 | 1.3E-03[e] |

Table 5.4c (continued)

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct; shower)[b] risk | Dermal (wading) risk | Inhalation (dust; VOCs)[b] risk | External exposure risk | Total risk[c] |
|---|---|---|---|---|---|---|---|
| **Sediment** | | | | | | | |
| **Inorganics** | | | | | | | |
| Arsenic | 2.30E+03 | | | | 5.1E-06 | | 5.1E-06 |
| Barium | 5.96E+02 | | | | | | |
| Beryllium | 4.70E+00 | 3.2E-05 | 1.4E-05 | 3.2E-07 | 1.7E-09 | | 4.6E-05 |
| Chromium VI | 2.17E+01 | | | | 3.9E-08 | | 3.9E-08 |
| Manganese | 1.37E+04 | | | | | | |
| **Radionuclides** | | | | | | | |
| Thorium-228 | 4.30E+00 | 3.0E-07 | | | 1.5E-08 | 5.8E-04 | 5.8E-04 |
| Thorium-230 | 2.70E+00 | 4.4E-08 | | | 3.6E-09 | 3.5E-09 | 5.1E-08 |
| Thorium-232 | 1.20E+00 | 1.8E-08 | | | 1.5E-09 | 7.5E-10 | 2.0E-08 |
| Uranium-234 | 2.20E+00 | 4.4E-08 | | | 2.6E-09 | 1.6E-09 | 4.9E-08 |
| Uranium-235 | 2.10E-01 | 4.2E-09 | | | 2.4E-10 | 1.2E-06 | 1.2E-06 |
| Uranium-238 | 2.20E+00 | 7.8E-08 | | | 5.2E-09 | 1.9E-06 | 2.0E-06 |
| **Total Pathway Risk[d]** | | 3.2E-05 | 1.4E-05 | 3.2E-07 | 5.1E-06 | 5.8E-04 | 6.3E-04[e] |

Table 5.4c (continued)

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct; shower)[b] risk | Dermal (wading) risk | Inhalation (dust; VOCs)[b] risk | External exposure risk | Total risk[c] |
|---|---|---|---|---|---|---|---|
| **Soil (beneath ash)** | | | | | | | |
| **Inorganics** | | | | | | | |
| Arsenic | 2.71E+01 | | | | 6.0E-08 | | 6.0E-08 |
| Barium | 1.08E+02 | | | | | | |
| Beryllium | 9.19E-01 | 6.2E-06 | 2.8E-06 | | 3.4E-10 | | 8.9E-06 |
| Cadmium | 7.48E+00 | | | | 2.0E-09 | | 2.0E-09 |
| Nickel | 2.66E+01 | | | | | | |
| **Radionuclide** | | | | | | | |
| Thorium-234 | 5.15E+00 | 2.6E-08 | | | 7.5E-12 | 4.3E-07 | 4.6E-07 |
| **Total Pathway Risk[d]** | | 6.2E-06 | 2.8E-06 | | 6.2E-08 | 4.3E-07 | 9.5E-06[e] |
| **Surface water** | | | | | | | |
| **Inorganics** | | | | | | | |
| Arsenic | 7.55E-02 | | | | | | |
| Barium | 9.08E-02 | | | | | | |
| Manganese | 5.14E-01 | | | | | | |

Table 5.4c (continued)

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (direct; shower)[b] risk | Dermal (wading) risk | Inhalation (dust; VOCs)[b] risk | External exposure risk | Total risk[c] |
|---|---|---|---|---|---|---|---|
| **Radionuclides** | | | | | | | |
| Radium-226 | 5.67E-01 | 1.4E-06 | | | | | 1.4E-06 |
| Thorium-228 | 4.10E-02 | 4.7E-08 | | | | | 4.7E-08 |
| Thorium-230 | 6.20E-01 | 1.7E-07 | | | | | 1.7E-07 |
| Uranium-234 | 5.00E-01 | 1.7E-07 | | | | | 1.7E-07 |
| Uranium-238 | 3.36E-01 | 2.0E-07 | | | | | 2.0E-07 |
| **Organic** | | | | | | | |
| bis(2-Ethylhexyl) -Phthalate | 4.80E-02 | 1.4E-05 | 3.8E-05 | 4.3E-05 | | | 5.2E-05 |
| **Total Pathway Risk[d]** | | 1.6E-05 | 3.8E-05 | 4.3E-05 | 0.0E+00[f] | | 5.4E-05[e] |

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean; for ash, sediment, and soil, units are pCi/g for radionuclides and mg/kg for all other analytes; for surface water, units are pCi/L for radionuclides and mg/L for all other analytes.

[b]The dermal direct-contact pathway is evaluated for ash, sediment, and soil; the dermal showering-contact pathway is evaluated for surface water. The inhalation of dust pathway is evaluated for ash, dry sediment, and soil; the inhalation of VOCs pathway is evaluated for surface water.

[c]The sum of the ingestion risk, dermal (direct or showering) risk, inhalation (of dust or VOCs) risk, and external exposure risk for each analyte; the risks from exposure to COPCs while wading are NOT included in these totals.

[d]The sum of the risks from all analytes for each pathway.

[e]The sum of the risks from all analytes across all pathways; the risks from exposure to COPCs while wading are NOT included in this total.

[f]Since the nonradionuclide COPCs for surface water do not have an inhalation SF and are not VOCs, no inhalation (VOCs) risk calculations could be performed.

Table 5.4d. Carcinogenic risks from Chestnut Ridge groundwater COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (showering) risk | Inhalation of VOCs risk | Total risk[b] |
|---|---|---|---|---|---|
| **Groundwater** | | | | | |
| **Inorganics** | | | | | |
| Arsenic | 5.00E-03 | | | | |
| Barium | 1.88E-01 | | | | |
| Beryllium | 1.43E-03 | 1.3E-04 | 4.0E-06 | | 1.3E-04 |
| Cadmium | 4.84E-03 | | | | |
| Chromium VI | 2.36E-02 | | | | |
| Manganese | 1.00E+00 | | | | |
| Vanadium | 1.90E-02 | | | | |
| Zinc | 4.07E-01 | | | | |
| **Radionuclides** | | | | | |
| Radium-226 | 1.26E+00 | 3.2E-06 | | | 3.2E-06 |
| Thorium-228 | 1.21E-01 | 1.4E-07 | | | 1.4E-07 |
| Thorium-232 | 8.84E-02 | 2.2E-08 | | | 2.2E-08 |
| Uranium-234 | 5.50E-01 | 1.8E-07 | | | 1.8E-07 |
| Uranium-235 | 4.80E-02 | 1.6E-08 | | | 1.6E-08 |
| Uranium-238 | 4.15E-01 | 2.4E-07 | | | 2.4E-07 |

**Table 5.4d (continued)**

| Analyte | Representative concentration[a] | Ingestion risk | Dermal (showering) risk | Inhalation of VOCs risk | Total risk[b] |
|---|---|---|---|---|---|
| **Organic** | | | | | |
| bis(2-Ethylhexyl)phthalate | 2.40E-02 | 6.8E-06 | 1.9E-05 | | 2.6E-05 |
| **Total Pathway Risk[c]** | | 1.4E-04 | 2.3E-05 | 0.0E+00[d] | 1.6E-04[e] |

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean; units are pCi/L for radionuclides and mg/L for all other analytes.

[b]The sum of the ingestion risk, dermal (showering) risk, and inhalation of VOCs risk for each analyte.

[c]The sum of the risks from all analytes for each pathway.

[d]Since the COPCs for groundwater are not VOCs, no inhalation of VOCs risk calculations could be performed.

[e]The sum of the risks from all analytes across all pathways.

## Table 5.5a. Carcinogenic risk for Chestnut Ridge OU 2 Sluice Channel Area COPCs—irrigation with groundwater (residential ingestion of homegrown vegetables/fruits scenario)

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 1.62E+00 | | 1.59E+00 | | 1.60E+00 | | 1.60E+00 | | 1.60E+00 | |
| Barium | 4.07E+00 | | 1.94E+00 | | 5.85E+00 | | 1.02E+00 | | 3.28E+00 | |
| Beryllium | 9.76E-03 | 3.4E-05 | 7.52E-04 | 2.6E-06 | 3.75E-03 | 1.3E-05 | 3.75E-03 | 1.3E-05 | 3.75E-03 | 6.9E-06 |
| Cadmium | 3.09E-02 | | 4.06E-04 | | 1.06E-02 | | 1.06E-02 | | 1.06E-02 | |
| Chromium VI | 1.53E-01 | | 4.28E-03 | | 5.38E-02 | | 5.38E-02 | | 5.38E-02 | |
| Manganese | 1.51E+02 | | 3.92E+01 | | 1.11E+01 | | 4.47E+00 | | 1.47E+02 | |
| Vanadium | 1.81E-01 | | 6.18E-02 | | 1.02E-01 | | 1.02E-01 | | 1.02E-01 | |
| Zinc | 2.57E+00 | | 7.86E-03 | | 8.57E-01 | | 8.56E-01 | | 8.66E-01 | |
| *Radionuclides* | | | | | | | | | | |
| Radium-226 | 7.93E-03 | 8.0E-07 | 4.94E-07 | 5.0E-11 | 2.64E-03 | 2.7E-07 | 2.64E-03 | 2.7E-07 | 2.64E-03 | 1.4E-07 |
| Thorium-228 | 5.37E-03 | 2.5E-07 | 4.62E-03 | 2.1E-07 | 4.87E-03 | 2.2E-07 | 4.87E-03 | 2.2E-07 | 4.87E-03 | 1.2E-07 |
| Thorium-230 | 1.22E-02 | 1.3E-07 | 1.22E-02 | 1.3E-07 | 1.22E-02 | 1.3E-07 | 1.22E-02 | 1.3E-07 | 1.22E-02 | 7.0E-08 |
| Thorium-232 | 6.02E-03 | 6.1E-08 | 5.46E-03 | 5.5E-08 | 5.65E-03 | 5.7E-08 | 5.65E-03 | 5.7E-08 | 5.65E-03 | 3.0E-08 |
| Uranium-234 | 1.45E-02 | 1.9E-07 | 1.10E-02 | 1.5E-07 | 1.22E-02 | 1.6E-07 | 1.22E-02 | 1.6E-07 | 1.22E-02 | 8.6E-08 |
| Uranium-235 | 3.02E-04 | 4.1E-09 | 3.36E-08 | 4.5E-13 | 1.01E-04 | 1.4E-09 | 1.01E-04 | 1.4E-09 | 1.01E-04 | 7.1E-10 |
| Uranium-238 | 1.09E-02 | 2.6E-07 | 8.25E-03 | 1.9E-07 | 9.12E-03 | 2.1E-07 | 9.12E-03 | 2.1E-07 | 9.12E-03 | 1.1E-07 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 1.51E-01 | 1.7E-06 | 1.81E-07 | 2.1E-12 | 5.04E-02 | 5.7E-07 | 5.04E-02 | 5.7E-07 | 5.04E-02 | 3.0E-07 |
| **Total Pathway Risk[b]** | | 3.8E-05 | | 3.4E-06 | | 1.5E-05 | | 1.5E-05 | | 7.8E-06 |

[a]The concentrations in vegetables/fruits are in pCi/g for radionuclides and mg/kg for all other analytes.
[b]The sum of the risks from all analytes for ingestion of each vegetable/fruit-type.

## Table 5.5b. Carcinogenic risk for Chestnut Ridge OU2 FCAP COPCs—irrigation with groundwater (residential ingestion of homegrown vegetables/fruits scenario)

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 1.34E+00 | | 1.31E+00 | | 1.32E+00 | | 1.32E+00 | | 1.32E+00 | |
| Barium | 2.81E+00 | | 1.09E+00 | | 3.46E+00 | | 7.46E-01 | | 2.02E+00 | |
| Beryllium | 1.11E-02 | 3.9E-05 | 2.06E-03 | 7.2E-06 | 5.06E-03 | 1.8E-05 | 5.06E-03 | 1.8E-05 | 5.06E-03 | 9.3E-06 |
| Cadmium | 3.09E-02 | | 4.06E-04 | | 1.06E-02 | | 1.06E-02 | | 1.06E-02 | |
| Chromium VI | 1.55E-01 | | 6.28E-03 | | 5.58E-02 | | 5.58E-02 | | 5.58E-02 | |
| Manganese | 7.04E+01 | | 1.74E+01 | | 6.06E+00 | | 3.15E+00 | | 6.62E+01 | |
| Vanadium | 2.30E-01 | | 1.10E-01 | | 1.50E-01 | | 1.50E-01 | | 1.50E-01 | |
| Zinc | 2.57E+00 | | 7.86E-03 | | 8.57E-01 | | 8.56E-01 | | 8.66E-01 | |
| *Radionuclides* | | | | | | | | | | |
| Radium-226 | 7.93E-03 | 8.0E-07 | 4.94E-07 | 5.0E-11 | 2.64E-03 | 2.7E-07 | 2.64E-03 | 2.7E-07 | 2.64E-03 | 1.4E-07 |
| Thorium-228 | 9.86E-03 | 4.6E-07 | 9.11E-03 | 4.2E-07 | 9.36E-03 | 4.3E-07 | 9.36E-03 | 4.3E-07 | 9.36E-03 | 2.3E-07 |
| Thorium-230 | 1.69E-02 | 1.8E-07 | 1.69E-02 | 1.8E-07 | 1.69E-02 | 1.8E-07 | 1.69E-02 | 1.8E-07 | 1.69E-02 | 9.7E-08 |
| Thorium-232 | 9.46E-03 | 9.5E-08 | 8.90E-03 | 9.0E-08 | 9.09E-03 | 9.2E-08 | 9.09E-03 | 9.2E-08 | 9.09E-03 | 4.8E-08 |
| Uranium-234 | 1.31E-02 | 1.8E-07 | 9.68E-03 | 1.3E-07 | 1.08E-02 | 1.5E-07 | 1.08E-02 | 1.5E-07 | 1.08E-02 | 7.6E-08 |
| Uranium-235 | 8.90E-04 | 1.2E-08 | 5.88E-04 | 7.9E-09 | 6.88E-04 | 9.2E-09 | 6.88E-04 | 9.2E-09 | 6.88E-04 | 4.9E-09 |
| Uranium-238 | 1.23E-02 | 2.9E-07 | 9.68E-03 | 2.3E-07 | 1.05E-02 | 2.5E-07 | 1.05E-02 | 2.5E-07 | 1.05E+00 | 1.3E-05 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 1.51E-01 | 1.7E-06 | 1.81E-07 | 2.1E-12 | 5.04E-02 | 5.7E-07 | 5.04E-02 | 5.7E-07 | 5.04E-02 | 3.0E-07 |
| **Total Pathway Risk[b]** | | 4.3E-05 | | 8.3E-06 | | 2.0E-05 | | 2.0E-05 | | 2.3E-05 |

[a]The concentrations in vegetables/fruits are in pCi/g for radionuclides and mg/kg for all other analytes.
[b]The sum of the risks from all analytes for ingestion of each vegetable/fruit-type.

## Table 5.5c. Carcinogenic risk for Chestnut Ridge OU2 UMB COPCs—irrigation with groundwater (residential ingestion of homegrown vegetables/fruits scenario)

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 6.23E-01 | | 5.92E-01 | | 6.03E-01 | | 6.03E-01 | | 6.03E-01 | |
| Barium | 3.21E+00 | | 1.36E+00 | | 4.21E+00 | | 8.32E-01 | | 2.42E+00 | |
| Beryllium | 1.04E-02 | 3.6E-05 | 1.36E-03 | 4.8E-06 | 4.36E-03 | 1.5E-05 | 4.36E-03 | 1.5E-05 | 4.36E-03 | 8.0E-06 |
| Cadmium | 1.13E+00 | | 1.10E+00 | | 1.11E+00 | | 1.11E+00 | | 1.11E+00 | |
| Chromium VI | 1.53E-01 | | 4.23E-03 | | 5.38E-02 | | 5.38E-02 | | 5.38E-02 | |
| Manganese | 3.75E+02 | | 9.99E+01 | | 2.49E+01 | | 8.15E+00 | | 3.71E+02 | |
| Mercury | 2.21E-01 | | 2.21E-01 | | 2.21E-01 | | 2.21E-01 | | 2.21E-01 | |
| Vanadium | 2.14E-01 | | 9.43E-02 | | 1.34E-01 | | 1.34E-01 | | 1.34E-01 | |
| Zinc | 2.57E+00 | | 7.86E-03 | | 8.57E-01 | | 8.56E-01 | | 8.66E-01 | |
| *Radionuclides* | | | | | | | | | | |
| Cesium-137 | 2.13E-03 | 5.0E-08 | 1.38E-03 | 3.2E-08 | 2.88E-03 | 6.8E-08 | 4.39E-04 | 1.0E-08 | 3.81E-04 | 4.7E-09 |
| Radium-226 | 7.93E-03 | 8.0E-07 | 4.94E-07 | 5.0E-11 | 2.64E-03 | 2.7E-07 | 2.64E-03 | 2.7E-07 | 2.64E-03 | 1.4E-07 |
| Thorium-228 | 3.94E-02 | 1.8E-06 | 3.86E-02 | 1.8E-06 | 3.89E-02 | 1.8E-06 | 3.89E-02 | 1.8E-06 | 3.89E-02 | 9.4E-07 |
| Thorium-232 | 5.57E-04 | 5.6E-09 | 1.04E-07 | 1.0E-12 | 1.86E-04 | 1.9E-09 | 1.86E-04 | 1.9E-09 | 1.86E-04 | 9.8E-10 |
| Thorium-234 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 4.7E-08 |
| Uranium-234 | 3.46E-03 | 4.7E-08 | 3.85E-07 | 5.2E-12 | 1.15E-03 | 1.5E-08 | 1.15E-03 | 1.5E-08 | 1.15E-03 | 8.1E-09 |
| Uranium-235 | 3.02E-04 | 4.1E-09 | 3.36E-08 | 4.5E-13 | 1.01E-04 | 1.4E-09 | 1.01E-04 | 1.4E-09 | 1.01E-04 | 7.1E-10 |
| Uranium-238 | 1.12E-02 | 2.6E-07 | 8.58E-03 | 2.0E-07 | 9.45E-03 | 2.2E-07 | 9.45E-03 | 2.2E-07 | 9.45E-03 | 1.2E-07 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 1.51E-01 | 1.7E-06 | 1.81E-07 | 2.1E-12 | 5.04E-02 | 5.7E-07 | 5.04E-02 | 5.7E-07 | 5.04E-02 | 3.0E-07 |
| **Total Pathway Risk[b]** | | 4.1E-05 | | 6.9E-06 | | 1.8E-05 | | 1.8E-05 | | 9.6E-06 |

[a]The concentrations in vegetables/fruits are in pCi/g for radionuclides and mg/kg for all other analytes.
[b]The sum of the risks from all analytes for ingestion of each vegetable/fruit-type.

## Table 5.5d. Carcinogenic risk for Chestnut Ridge OU2 UMB COPCs—irrigation with UMB surface water (residential ingestion of homegrown vegetables/fruits scenario)

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk | Concentration[a] (mg/kg; pCi/g) | Ingestion risk |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 1.07E+00 | | 5.92E-01 | | 7.51E-01 | | 7.51E-01 | | 7.51E-01 | |
| Barium | 2.59E+00 | | 1.36E+00 | | 4.01E+00 | | 6.28E-01 | | 2.21E+00 | |
| Beryllium | 1.36E-03 | 4.8E-06 | 1.36E-03 | 4.8E-06 | 1.36E-03 | 4.8E-06 | 1.36E-03 | 4.8E-06 | 1.36E-03 | 2.5E-06 |
| Cadmium | 1.10E+00 | | 1.10E+00 | | 1.10E+00 | | 1.10E+00 | | 1.10E+00 | |
| Chromium VI | 4.23E-03 | | 4.23E-03 | | 4.23E-03 | | 4.23E-03 | | 4.23E-03 | |
| Manganese | 3.72E+02 | | 9.99E+01 | | 2.39E+01 | | 7.12E+00 | | 3.70E+02 | |
| Mercury | 2.21E-01 | | 2.21E-01 | | 2.21E-01 | | 2.21E-01 | | 2.21E-01 | |
| Vanadium | 9.43E-02 | | 9.43E-02 | | 9.43E-02 | | 9.43E-02 | | 9.43E-02 | |
| *Radionuclides* | | | | | | | | | | |
| Cesium-137 | 2.13E-03 | 5.0E-08 | 1.38E-03 | 3.2E-08 | 2.88E-03 | 6.8E-08 | 4.39E-04 | 1.0E-08 | 3.81E-04 | 4.7E-09 |
| Radium-226 | 3.57E-03 | 3.6E-07 | 2.22E-07 | 2.2E-11 | 1.19E-03 | 1.2E-07 | 1.19E-03 | 1.2E-07 | 1.19E-03 | 6.3E-08 |
| Thorium-228 | 3.89E-02 | 1.8E-06 | 3.86E-02 | 1.8E-06 | 3.87E-02 | 1.8E-06 | 3.87E-02 | 1.8E-06 | 3.87E-02 | 9.4E-07 |
| Thorium-230 | 3.90E-03 | 4.3E-08 | 7.29E-07 | 8.0E-12 | 1.30E-03 | 1.4E-08 | 1.30E-03 | 1.4E-08 | 1.30E-03 | 7.5E-09 |
| Thorium-234 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 8.9E-08 | 2.65E-02 | 4.7E-08 |
| Uranium-234 | 3.15E-03 | 4.2E-08 | 3.50E-07 | 4.7E-12 | 1.05E-03 | 1.4E-08 | 1.05E-03 | 1.4E-08 | 1.05E-03 | 7.4E-09 |
| Uranium-238 | 1.07E-02 | 2.5E-07 | 8.58E-03 | 2.0E-07 | 9.28E-03 | 2.2E-07 | 9.28E-03 | 2.2E-07 | 9.28E-03 | 1.1E-07 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 3.02E-01 | 3.4E-06 | 3.63E-07 | 4.1E-12 | 1.01E-01 | 1.2E-06 | 1.01E-01 | 1.2E-06 | 1.01E-01 | 6.0E-07 |
| **Total Pathway Risk[b]** | | 1.1E-05 | | 6.9E-06 | | 8.2E-06 | | 8.2E-06 | | 4.3E-06 |

[a]The concentrations in vegetables/fruits are in pCi/g for radionuclides and mg/kg for all other analytes.
[b]The sum of the risks from all analytes for ingestion of each vegetable/fruit-type.

Table 5.6a. Hazard indices for Chestnut Ridge OU2 Sluice Channel COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion HI | Dermal (direct) HI | Inhalation of dust HI | Total HI[b] |
|---|---|---|---|---|---|
| **Ash** | | | | | |
| Arsenic | 1.59E+02 | 1.9E+00 | 4.3E-02 | | 2.0E+00 |
| Barium | 4.21E+02 | 2.2E-02 | 4.9E-03 | 3.0E-04 | 2.7E-02 |
| Beryllium | 1.60E+00 | 1.2E-03 | 5.2E-04 | | 1.7E-03 |
| Chromium VI | 1.71E+01 | 1.2E-02 | 2.6E-03 | | 1.5E-02 |
| Manganese | 3.44E+02 | 9.0E-03 | 4.0E-03 | 3.1E-04 | 1.3E-02 |
| Vanadium | 4.75E+01 | 2.5E-02 | 2.1E-02 | | 4.6E-02 |
| **Total Pathway HI[c]** | | 2.0E+00 | 7.6E-02 | 6.1E-04 | 2.1E+00[d] |
| **Soil (beneath ash)** | | | | | |
| Arsenic | 5.12E+01 | 6.2E-01 | 1.4E-02 | | 6.4E-01 |
| **Total Pathway HI[c]** | | 6.2E-01 | 1.4E-02 | 0.0E+00[e] | 6.4E-01[d] |

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean; units are pCi/g for radionuclides and mg/kg for all other analytes.
[b]The sum of the ingestion HI, dermal direct-contact HI, and the inhalation of dust HI for each analyte.
[c]The sum of the HIs from all analytes for each pathway.
[d]The sum of the HIs from all analytes across all pathways.
[e]Since the COPC for soil does not have an inhalation RfD, no inhalation of dust HI calculation could be performed.

Table 5.6b. Hazard indices for Chestnut Ridge OU 2 FCAP COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion HI | Dermal (direct; shower)[b] HI | Dermal (wading) HI | Inhalation (dust; VOCs)[b] HI | Total HI[c] |
|---|---|---|---|---|---|---|
| **Ash** | | | | | | |
| Arsenic | 1.31E+02 | 1.6E+00 | 3.6E-02 | 8.0E-04 | | 1.6E+00 |
| Barium | 4.50E+02 | 2.4E-02 | 5.2E-03 | 1.2E-04 | 3.2E-04 | 2.9E-02 |
| Beryllium | 4.39E+00 | 3.2E-03 | 1.4E-03 | 3.2E-05 | | 4.6E-03 |
| Chromium VI | 2.51E+01 | 1.8E-02 | 3.9E-03 | 8.6E-05 | | 2.2E-02 |
| Manganese | 1.52E+02 | 4.0E-03 | 1.8E-03 | 4.0E-05 | 1.4E-04 | 5.9E-03 |
| Vanadium | 8.49E+01 | 4.4E-02 | 3.8E-02 | 8.5E-04 | | 8.2E-02 |
| **Total Pathway HI[d]** | | 1.7E+00 | 8.6E-02 | 1.9E-03 | 4.6E-04 | 1.8E+00[e] |
| **Sediment** | | | | | | |
| Arsenic | 7.66E+01 | 9.3E-01 | 2.1E-02 | 4.6E-04 | | 9.5E-01 |
| Beryllium | 2.10E+00 | 1.5E-03 | 6.8E-04 | 1.5E-05 | | 2.2E-03 |
| Chromium VI | 3.29E+01 | 2.4E-02 | 5.0E-03 | 1.1E-04 | | 2.9E-02 |
| Vanadium | 3.55E+01 | 1.9E-02 | 1.6E-02 | 3.5E-04 | | 3.4E-02 |
| **Total Pathway HI[d]** | | 9.8E-01 | 4.2E-02 | 9.5E-04 | 0.0E+00[f] | 1.0E+00[e] |
| **Surface water** | | | | | | |
| Arsenic | 1.00E-02 | 1.6E+00 | 2.5E-03 | 2.9E-03 | | 1.6E+00 |
| Barium | 6.89E-02 | 4.7E-02 | 7.5E-04 | 8.6E-04 | | 4.7E-02 |
| Chromium III | 8.80E-03 | 4.2E-04 | 6.7E-07 | 7.7E-07 | | 4.2E-04 |

**Table 5.6b (continued)**

| Analyte | Representative concentration[a] | Ingestion HI | Dermal (direct; shower)[b] HI | Dermal (wading) HI | Inhalation (dust; VOCs)[b] HI | Total HI[c] |
|---|---|---|---|---|---|---|
| Manganese | 1.70E+00 | 1.6E+01 | 5.2E-01 | 5.9E-01 | | 1.7E+01 |
| **Total Pathway HI[d]** | | 1.8E+01 | 5.2E-01 | 6.0E-01 | 0.0E+00[g] | 1.8E+01[e] |

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean; units are mg/kg for ash and sediment; units are mg/L for surface water.

[b]The dermal direct-contact pathway is evaluated for ash and sediment; the dermal showering-contact pathway is evaluated for surface water. The inhalation of dust pathway is evaluated for ash and dry sediment; the inhalation of VOCs pathway is evaluated for surface water.

[c]The sum of the ingestion HI, dermal (direct or showering) HI, and the inhalation (of dust or VOCs) HI for each analyte; the HIs from exposure to COPCs while wading are NOT included in these totals.

[d]The sum of the HIs from all analytes for each pathway.

[e]The sum of the HIs from all analytes across all pathways; the HIs from exposure to COPCs while wading are NOT included in this total.

[f]Since the COPCs for sediment do not have an inhalation RfD, no inhalation (dust) HI calculations could be performed.

[g]Since the COPCs for surface water do not have an inhalation RfD and are not VOCs, no inhalation (VOCs) HI calculations could be performed.

Table 5.6c. Hazard indices for Chestnut Ridge OU 2 UMB COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion HI | Dermal (direct; shower)[b] HI | Dermal (wading) HI | Inhalation (dust; VOCs)[b] HI | Total HI[c] |
|---|---|---|---|---|---|---|
| **Ash** | | | | | | |
| Arsenic | 5.92E+01 | 7.2E-01 | 1.6E-02 | 3.6E-04 | | 7.4E-01 |
| Barium | 5.60E+02 | 2.9E-02 | 6.5E-03 | 1.5E-04 | 4.0E-04 | 3.6E-02 |
| Beryllium | 2.90E+00 | 2.1E-03 | 9.4E-04 | 2.1E-05 | | 3.1E-03 |
| Cadmium | 3.67E+00 | 1.3E-02 | 5.0E-03 | 1.1E-04 | | 1.8E-02 |
| Chromium VI | 1.69E+01 | 1.2E-02 | 2.6E-03 | 5.8E-05 | | 1.5E-02 |
| Manganese | 8.76E+02 | 2.3E-02 | 1.0E-02 | 2.3E-04 | 7.9E-04 | 3.4E-02 |
| Mercury* | 5.81E-01 | 7.1E-03 | 1.1E-03 | 2.4E-05 | 7.0E-07 | 8.1E-03 |
| Vanadium | 7.25E+01 | 3.8E-02 | 3.2E-02 | 7.2E-04 | | 7.0E-02 |
| **Total Pathway HI[d]** | | 8.5E-01 | 7.5E-02 | 1.7E-03 | 1.2E-03 | 9.2E-01[e] |
| **Sediment** | | | | | | |
| Arsenic | 2.30E+03 | 2.8E+01 | 6.2E-01 | 1.4E-02 | | 2.9E+01 |
| Barium | 5.96E+02 | 3.1E-02 | 6.9E-03 | 1.5E-04 | 4.3E-04 | 3.8E-02 |
| Beryllium | 4.70E+00 | 3.4E-03 | 1.5E-03 | 3.4E-05 | | 5.0E-03 |
| Chromium VI | 2.17E+01 | 1.6E-02 | 3.3E-03 | 7.5E-05 | | 1.9E-02 |
| Manganese | 1.37E+04 | 3.6E-01 | 1.6E-01 | 3.6E-03 | 1.2E-02 | 5.3E-01 |
| **Total Pathway HI[d]** | | 2.8E+01 | 7.9E-01 | 1.8E-02 | 1.3E-02 | 2.9E+01[e] |
| **Soil (beneath ash)** | | | | | | |
| Arsenic | 2.71E+01 | 3.3E-01 | 7.3E-03 | | | 3.4E-01 |
| Barium | 1.08E+02 | 5.6E-03 | 1.3E-03 | | 7.7E-05 | 7.0E-03 |
| Beryllium | 9.19E-01 | 6.7E-04 | 3.0E-04 | | | 9.7E-04 |

**Table 5.6c (continued)**

| Analyte | Representative concentration[a] | Ingestion HI | Dermal (direct; shower)[b] HI | Dermal (wading) HI | Inhalation (dust; VOCs)[b] HI | Total HI[c] |
|---|---|---|---|---|---|---|
| Cadmium | 7.48E+00 | 2.7E-02 | 1.0E-02 | | | 3.7E-02 |
| Nickel* | 2.66E+01 | 4.9E-03 | 2.2E-03 | | | 7.0E-03 |
| **Total Pathway HI[d]** | | 3.7E-01 | 2.1E-02 | | 7.7E-05 | 3.9E-01[e] |
| **Surface water** | | | | | | |
| Arsenic | 7.55E-02 | 1.2E+01 | 1.9E-02 | 2.2E-02 | | 1.2E+01 |
| Barium | 9.08E-02 | 6.2E-02 | 9.9E-04 | 1.1E-03 | | 6.3E-02 |
| Manganese | 5.14E-01 | 4.9E+00 | 1.6E-01 | 1.8E-01 | | 5.0E+00 |
| bis(2-Ethylhexyl)Phthalate | 4.80E-02 | 1.1E-01 | 3.2E-01 | 3.6E-01 | | 4.3E-01 |
| **Total Pathway HI[d]** | | 1.7E+01 | 4.9E-01 | 5.6E-01 | 0.0E+00[f] | 1.8E+01[e] |

*For the dermal pathways, the oral RfD absorbed for mercury *salts* and nickel *salts* was used because the %GI values were available for these salts and were not available for these metals.

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean; for ash, sediment, and soil, units are pCi/g for radionuclides and mg/kg for all other analytes; for surface water, units are pCi/L for radionuclides and mg/L for all other analytes.

[b]The dermal direct-contact pathway is evaluated for ash, sediment, and soil; the dermal showering-contact pathway is evaluated for surface water. The inhalation of dust pathway is evaluated for ash, dry sediment, and soil; the inhalation of VOCs pathway is evaluated for surface water.

[c]The sum of the ingestion HI, dermal (direct or showering) HI, and the inhalation (of dust or VOCs) HI for each analyte; the HIs from exposure to COPCs while wading are NOT included in these totals.

[d]The sum of the HIs from all analytes for each pathway.

[e]The sum of the HIs from all analytes across all pathways; the HIs from exposure to COPCs while wading are NOT included in this total.

[f]Since the COPCs for surface water do not have an inhalation RfD and are not VOCs, no inhalation (VOCs) HI calculations could be performed.

Table 5.6d. Hazard indices for Chestnut Ridge OU 2 groundwater COPCs—residential scenario

| Analyte | Representative concentration[a] | Ingestion HI | Dermal (showering) HI | Inhalation of VOCs HI | Total HI[b] |
|---|---|---|---|---|---|
| **Groundwater** | | | | | |
| Arsenic | 5.00E-03 | 7.9E-01 | 1.3E-03 | | 7.9E-01 |
| Barium | 1.88E-01 | 1.3E-01 | 2.1E-03 | | 1.3E-01 |
| Beryllium | 1.43E-03 | 1.4E-02 | 4.4E-04 | | 1.4E-02 |
| Cadmium | 4.84E-03 | 4.6E-01 | 1.2E-02 | | 4.7E-01 |
| Chromium VI | 2.36E-02 | 2.2E-01 | 3.4E-03 | | 2.3E-01 |
| Manganese | 1.00E+00 | 9.5E+00 | 3.1E-01 | | 9.8E+00 |
| Vanadium | 1.90E-02 | 1.3E-01 | 8.0E-03 | | 1.4E-01 |
| Zinc | 4.07E-01 | 6.4E-02 | 2.1E-04 | | 6.5E-02 |
| bis(2-Ethylhexyl)Phthalate | 2.40E-02 | 5.7E-02 | 1.6E-01 | | 2.1E-01 |
| **Total Pathway HI[c]** | | 1.1E+01 | 4.9E-01 | 0.0E+00[d] | 1.2E+01[e] |

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean; units are pCi/L for radionuclides and mg/L for all other analytes.
[b]The sum of the ingestion HI, dermal (showering) HI, and the inhalation of VOCs HI for each analyte.
[c]The sum of the HIs from all analytes for each pathway.
[d]Since the COPCs for groundwater do not have an inhalation RfD and are not VOCs, no inhalation of VOCs HI calculations could be performed.
[e]The sum of the HIs from all analytes across all pathways.

## Table 5.7a. Hazard indices for Chestnut Ridge OU 2 Sluice Channel Area COPCs—irrigation with groundwater (residential ingestion of homegrown vegetables/fruits scenario)

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 1.62E+00 | 1.0E+01 | 1.59E+00 | 1.0E+01 | 1.60E+00 | 1.0E+01 | 1.60E+00 | 1.0E+01 | 1.60E+00 | 5.3E+00 |
| Barium | 4.07E+00 | 1.1E-01 | 1.94E+00 | 5.3E-02 | 5.85E+00 | 1.6E-01 | 1.02E+00 | 2.8E-02 | 3.28E+00 | 4.7E-02 |
| Beryllium | 9.76E-03 | 3.7E-03 | 7.52E-04 | 2.9E-04 | 3.75E-03 | 1.4E-03 | 3.75E-03 | 1.4E-03 | 3.75E-03 | 7.5E-04 |
| Cadmium | 3.09E-02 | 5.9E-02 | 4.06E-04 | 7.7E-04 | 1.06E-02 | 2.0E-02 | 1.06E-02 | 2.0E-02 | 1.06E-02 | 1.1E-02 |
| Chromium VI | 1.53E-01 | 5.8E-02 | 4.28E-03 | 1.6E-03 | 5.38E-02 | 2.0E-02 | 5.38E-02 | 2.0E-02 | 5.38E-02 | 1.1E-02 |
| Manganese | 1.51E+02 | 2.0E+00 | 3.92E+01 | 5.3E-01 | 1.11E+01 | 1.5E-01 | 4.47E+00 | 6.1E-02 | 1.47E+02 | 1.0E+00 |
| Vanadium | 1.81E-01 | 4.9E-02 | 6.18E-02 | 1.7E-02 | 1.02E-01 | 2.8E-02 | 1.02E-01 | 2.8E-02 | 1.02E-01 | 1.5E-02 |
| Zinc | 2.57E+00 | 1.6E-02 | 7.86E-03 | 5.0E-05 | 8.57E-01 | 5.4E-03 | 8.56E-01 | 5.4E-03 | 8.66E-01 | 2.9E-03 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 1.51E-01 | 1.4E-02 | 1.81E-07 | 1.7E-08 | 5.04E-02 | 4.8E-03 | 5.04E-02 | 4.8E-03 | 5.04E-02 | 2.5E-03 |
| **Total Pathway HI[a]** | | 1.3E+01 | | 1.1E+01 | | 1.1E+01 | | 1.0E+01 | | 6.5E+00 |

[a]The sum of the hazard indices from all analytes for ingestion of each vegetable/fruit-type.

**Table 5.7b. Hazard indices for Chestnut Ridge OU 2 FCAP COPCs—irrigation with groundwater (residential ingestion of homegrown vegetables/fruits scenario)**

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 1.34E+00 | 8.5E+00 | 1.31E+00 | 8.3E+00 | 1.32E+00 | 8.4E+00 | 1.32E+00 | 8.4E+00 | 1.32E+00 | 4.4E+00 |
| Barium | 2.81E+00 | 7.6E-02 | 1.09E+00 | 3.0E-02 | 3.46E+00 | 9.4E-02 | 7.46E-01 | 2.0E-02 | 2.02E+00 | 2.9E-02 |
| Beryllium | 1.11E-02 | 4.2E-03 | 2.06E-03 | 7.8E-04 | 5.06E-03 | 1.9E-03 | 5.06E-03 | 1.9E-03 | 5.06E-03 | 1.0E-03 |
| Cadmium | 3.09E-02 | 5.9E-02 | 4.06E-04 | 7.7E-04 | 1.06E-02 | 2.0E-02 | 1.06E-02 | 2.0E-02 | 1.06E-02 | 1.1E-02 |
| Chromium VI | 1.55E-01 | 5.9E-02 | 6.28E-03 | 2.4E-03 | 5.58E-02 | 2.1E-02 | 5.58E-02 | 2.1E-02 | 5.58E-02 | 1.1E-02 |
| Manganese | 7.04E+01 | 9.6E-01 | 1.74E+01 | 2.4E-01 | 6.06E+00 | 8.2E-02 | 3.15E+00 | 4.3E-02 | 6.62E+01 | 4.7E-01 |
| Vanadium | 2.30E-01 | 6.2E-02 | 1.10E-01 | 3.0E-02 | 1.50E-01 | 4.1E-02 | 1.50E-01 | 4.1E-02 | 1.50E-01 | 2.1E-02 |
| Zinc | 2.57E+00 | 1.6E-02 | 7.86E-03 | 5.0E-05 | 8.57E-01 | 5.4E-03 | 8.56E-01 | 5.4E-03 | 8.66E-01 | 2.9E-03 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 1.51E-01 | 1.4E-02 | 1.81E-07 | 1.7E-08 | 5.04E-02 | 4.8E-03 | 5.04E-02 | 4.8E-03 | 5.04E-02 | 2.5E-03 |
| **Total Pathway HI[a]** | | 9.7E+00 | | 8.6E+00 | | 8.6E+00 | | 8.5E+00 | | 4.9E+00 |

[a]The sum of the hazard indices from all analytes for ingestion of each vegetable/fruit-type.

## Table 5.7c. Hazard indices for Chestnut Ridge OU2 UMB COPCs—irrigation with groundwater (residential ingestion of homegrown vegetables/fruits scenario)

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 6.23E-01 | 3.9E+00 | 5.92E-01 | 3.7E+00 | 6.03E-01 | 3.8E+00 | 6.03E-01 | 3.8E+00 | 6.03E-01 | 2.0E+00 |
| Barium | 3.21E+00 | 8.7E-02 | 1.36E+00 | 3.7E-02 | 4.21E+00 | 1.1E-01 | 8.32E-01 | 2.3E-02 | 2.42E+00 | 3.4E-02 |
| Beryllium | 1.04E-02 | 4.0E-03 | 1.36E-03 | 5.2E-04 | 4.36E-03 | 1.7E-03 | 4.36E-03 | 1.7E-03 | 4.36E-03 | 8.7E-04 |
| Cadmium | 1.13E+00 | 2.1E+00 | 1.10E+00 | 2.1E+00 | 1.11E+00 | 2.1E+00 | 1.11E+00 | 2.1E+00 | 1.11E+00 | 1.1E+00 |
| Chromium VI | 1.53E-01 | 5.8E-02 | 4.23E-03 | 1.6E-03 | 5.38E-02 | 2.0E-02 | 5.38E-02 | 2.0E-02 | 5.38E-02 | 1.1E-02 |
| Manganese | 3.75E+02 | 5.1E+00 | 9.99E+01 | 1.4E+00 | 2.49E+01 | 3.4E-01 | 8.15E+00 | 1.1E-01 | 3.71E+02 | 2.6E+00 |
| Mercury | 2.21E-01 | 1.4E+00 | 2.21E-01 | 1.4E+00 | 2.21E-01 | 1.4E+00 | 2.21E-01 | 1.4E+00 | 2.21E-01 | 7.3E-01 |
| Vanadium | 2.14E-01 | 5.8E-02 | 9.43E-02 | 2.6E-02 | 1.34E-01 | 3.6E-02 | 1.34E-01 | 3.6E-02 | 1.34E-01 | 1.9E-02 |
| Zinc | 2.57E+00 | 1.6E-02 | 7.86E-03 | 5.0E-05 | 8.57E-01 | 5.4E-03 | 8.56E-01 | 5.4E-03 | 8.66E-01 | 2.9E-03 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 1.51E-01 | 1.4E-02 | 1.81E-07 | 1.7E-08 | 5.04E-02 | 4.8E-03 | 5.04E-02 | 4.8E-03 | 5.04E-02 | 2.5E-03 |
| **Total Pathway HI**[a] | | 1.3E+01 | | 8.7E+00 | | 7.8E+00 | | 7.5E+00 | | 6.6E+00 |

[a]The sum of the hazard indices from all analytes for ingestion of each vegetable/fruit-type.

**Table 5.7d. Hazard indices for Chestnut Ridge OU 2 UMB COPCs—irrigation with UMB surface water (residential ingestion of homegrown vegetables/fruits scenario)**

| Analyte | Leafy vegetables | | Root vegetables | | Legume vegetables | | Fruiting vegetables | | Fruits | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI | Concentration (mg/kg) | Ingestion HI |
| *Inorganics* | | | | | | | | | | |
| Arsenic | 1.07E+00 | 6.8E+00 | 5.92E-01 | 3.7E+00 | 7.51E-01 | 4.8E+00 | 7.51E-01 | 4.8E+00 | 7.51E-01 | 2.5E+00 |
| Barium | 2.59E+00 | 7.0E-02 | 1.36E+00 | 3.7E-02 | 4.01E+00 | 1.1E-01 | 6.28E-01 | 1.7E-02 | 2.21E+00 | 3.1E-02 |
| Beryllium | 1.36E-03 | 5.2E-04 | 1.36E-03 | 5.2E-04 | 1.36E-03 | 5.2E-04 | 1.36E-03 | 5.2E-04 | 1.36E-03 | 2.7E-04 |
| Cadmium | 1.10E+00 | 2.1E+00 | 1.10E+00 | 2.1E+00 | 1.10E+00 | 2.1E+00 | 1.10E+00 | 2.1E+00 | 1.10E+00 | 1.1E+00 |
| Chromium VI | 4.23E-03 | 1.6E-03 | 4.23E-03 | 1.6E-03 | 4.23E-03 | 1.6E-03 | 4.23E-03 | 1.6E-03 | 4.23E-03 | 8.4E-04 |
| Manganese | 3.72E+02 | 5.0E+00 | 9.99E+01 | 1.4E+00 | 2.39E+01 | 3.2E-01 | 7.12E+00 | 9.7E-02 | 3.70E+02 | 2.6E+00 |
| Mercury | 2.21E-01 | 1.4E+00 | 2.21E-01 | 1.4E+00 | 2.21E-01 | 1.4E+00 | 2.21E-01 | 1.4E+00 | 2.21E-01 | 7.3E-01 |
| Vanadium | 9.43E-02 | 2.6E-02 | 9.43E-02 | 2.6E-02 | 9.43E-02 | 2.6E-02 | 9.43E-02 | 2.6E-02 | 9.43E-02 | 1.3E-02 |
| *Organic* | | | | | | | | | | |
| bis(2-Ethylhexyl)-Phthalate | 3.02E-01 | 2.9E-02 | 3.63E-07 | 3.4E-08 | 1.01E-01 | 9.6E-03 | 1.01E-01 | 9.6E-03 | 1.01E-01 | 5.0E-03 |
| **Total Pathway HI[a]** | | 1.5E+01 | | 8.7E+00 | | 8.7E+00 | | 8.4E+00 | | 7.0E+00 |

[a]The sum of the hazard indices from all analytes for ingestion of each vegetable/fruit-type.

**Table 5.8. Summary of significant risks and hazard indices from Chestnut Ridge OU 2 COPCs[a]—residential scenario**

| Media | Analyte | HI | Risk | Pathway[b] |
|---|---|---|---|---|
| | | ***Sluice Channel Area*** | | |
| Ash | Arsenic | 1.9E+00 | | Ingestion |
| | Beryllium | | 1.1E-05 | Ingestion |
| | | | 4.8E-06 | Dermal |
| | Thorium-228[*] | | 1.5E-04 | External Exposure |
| | Uranium-238 | | 2.9E-06 | External Exposure |
| Soil (beneath ash) | Thorium-228[*] | | 1.1E-04 | External Exposure |
| Vegetables/fruits: | | | | |
| *Leafy vegetables* | Arsenic | 1.0E+01 | | Ingestion |
| | Beryllium | | 3.4E-05 | Ingestion |
| | Manganese | 2.0E+00 | | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 1.7E-06 | Ingestion |
| *Root vegetables* | Arsenic | 1.0E+01 | | Ingestion |
| | Beryllium | | 2.6E-06 | Ingestion |
| *Legumes* | Arsenic | 1.0E+01 | | Ingestion |
| | Beryllium | | 1.3E-05 | Ingestion |
| *Fruiting vegetables* | Arsenic | 1.0E+01 | | Ingestion |
| | Beryllium | | 1.3E-05 | Ingestion |
| *Fruits* | Arsenic | 5.3E+00 | | Ingestion |
| | Beryllium | | 6.9E-06 | Ingestion |
| | Manganese | 1.0E+00 | | Ingestion |
| | | ***Filled Coal Ash Pond*** | | |
| Ash | Arsenic | 1.6E+00 | | Ingestion |
| | Beryllium | | 3.0E-05 | Ingestion |
| | | | 1.3E-05 | Dermal |
| | Thorium-228[*] | | 2.9E-04 | External Exposure |
| | Uranium-235 | | 1.4E-06 | External Exposure |
| | Uranium-238 | | 3.3E-06 | External Exposure |

**Table 5.8 (continued)**

| Media | Analyte | HI | Risk | Pathway[b] |
|---|---|---|---|---|
| Sediment | Beryllium | | 1.4E-05 | Ingestion |
| | | | 6.3E-06 | Dermal |
| | Thorium-228 | | 9.8E-05 | External Exposure |
| | Uranium-235 | | 1.7E-06 | External Exposure |
| | Uranium-238 | | 2.6E-06 | External Exposure |
| Surface water | Arsenic | 1.6E+00 | | Ingestion |
| | Manganese | 1.6E+01 | | Ingestion |
| Vegetables/fruits: | | | | |
| *Leafy vegetables* | Arsenic | 8.5E+00 | | Ingestion |
| | Beryllium | | 3.9E-05 | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 1.7E-06 | Ingestion |
| *Root vegetables* | Arsenic | 8.3E+00 | | Ingestion |
| | Beryllium | | 7.2E-06 | Ingestion |
| *Legumes* | Arsenic | 8.4E+00 | | Ingestion |
| | Beryllium | | 1.8E-05 | Ingestion |
| *Fruiting vegetables* | Arsenic | 8.4E+00 | | Ingestion |
| | Beryllium | | 1.8E-05 | Ingestion |
| *Fruits* | Arsenic | 4.4E+00 | | Ingestion |
| | Beryllium | | 9.3E-06 | Ingestion |
| | Uranium-238 | | 1.3E-05 | Ingestion |
| ***Upper McCoy Branch*** | | | | |
| Ash | Beryllium | | 1.9E-05 | Ingestion |
| | | | 8.7E-06 | Dermal |
| | Cesium-137 | | 6.2E-06 | External Exposure |
| | Thorium-228[•] | | 1.2E-03 | External Exposure |
| | Uranium-238 | | 3.0E-06 | External Exposure |

Table 5.8 (continued)

| Media | Analyte | HI | Risk | Pathway[b] |
|---|---|---|---|---|
| Sediment | Arsenic | 2.8E+01 | | Ingestion |
| | | | 5.1E-06 | Inhalation |
| | Beryllium | | 3.2E-05 | Ingestion |
| | | | 1.4E-05 | Dermal |
| | Thorium-228[a] | | 5.8E-04 | External Exposure |
| | Uranium-235 | | 1.2E-06 | External Exposure |
| | Uranium-238 | | 1.9E-06 | External Exposure |
| Soil (beneath ash) | Beryllium | | 6.2E-06 | Ingestion |
| | | | 2.8E-06 | Dermal |
| Surface water | Arsenic | 1.2E+01 | | Ingestion |
| | Manganese | 4.9E+00 | | Ingestion |
| | Radium-226 | | 1.4E-06 | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 1.4E-05 | Ingestion |
| | | | 3.8E-05 | Dermal (showering) |
| | | | 4.3E-05 | Dermal (wading) |
| Vegetables/fruits: (irrigation with Chestnut Ridge OU 2 groundwater) | | | | |
| *Leafy vegetables* | Arsenic | 3.9E+00 | | Ingestion |
| | Beryllium | | 3.6E-05 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Manganese | 5.1E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 1.7E-06 | Ingestion |

Table 5.8 (continued)

| Media | Analyte | HI | Risk | Pathway[b] |
|---|---|---|---|---|
| *Root vegetables* | Arsenic | 3.7E+00 | | Ingestion |
| | Beryllium | | 4.8E-06 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Manganese | 1.4E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| *Legumes* | Arsenic | 3.8E+00 | | Ingestion |
| | Beryllium | | 1.5E-05 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| *Fruiting vegetables* | Arsenic | 3.8E+00 | | Ingestion |
| | Beryllium | | 1.5E-05 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| *Fruits* | Arsenic | 2.0E+00 | | Ingestion |
| | Beryllium | | 8.0E-06 | Ingestion |
| | Cadmium | 1.1E+00 | | Ingestion |
| | Manganese | 2.6E+00 | | Ingestion |
| Vegetables/fruits: (irrigation with UMB surface water) | | | | |
| *Leafy vegetables* | Arsenic | 6.8E+00 | | Ingestion |
| | Beryllium | | 4.8E-06 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Manganese | 5.0E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 3.4E-06 | Ingestion |

**Table 5.8 (continued)**

| Media | Analyte | HI | Risk | Pathway[b] |
|---|---|---|---|---|
| *Root vegetables* | Arsenic | 3.7E+00 | | Ingestion |
| | Beryllium | | 4.8E-06 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Manganese | 1.4E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| *Legumes* | Arsenic | 4.8E+00 | | Ingestion |
| | Beryllium | | 4.8E-06 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 1.2E-06 | Ingestion |
| *Fruiting vegetables* | Arsenic | 4.8E+00 | | Ingestion |
| | Beryllium | | 4.8E-06 | Ingestion |
| | Cadmium | 2.1E+00 | | Ingestion |
| | Mercury | 1.4E+00 | | Ingestion |
| | Thorium-228 | | 1.8E-06 | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 1.2E-06 | Ingestion |
| *Fruits* | Arsenic | 2.5E+00 | | Ingestion |
| | Beryllium | | 2.5E-06 | Ingestion |
| | Cadmium | 1.1E+00 | | Ingestion |
| | Manganese | 2.6E+00 | | Ingestion |

**Table 5.8 (continued)**

| Media | Analyte | HI | Risk | Pathway[b] |
|---|---|---|---|---|
| *Chestnut Ridge OU 2 Groundwater* | | | | |
| Groundwater | Beryllium* | | 1.3E-04<br>4.0E-06 | Ingestion<br>Dermal (showering) |
| | Manganese | 9.5E+00 | | Ingestion |
| | Radium-226 | | 3.2E-06 | Ingestion |
| | bis(2-Ethylhexyl)Phthalate | | 6.8E-06<br>1.9E-05 | Ingestion<br>Dermal (showering) |

*Analytes with an asterisk have a risk $\geq$ 1.0E-04, under the residential land use scenario.

[a]The COPCs reported in this table have a risk greater than 1.0E-06 and/or a hazard index greater than 1.0, under the residential land use scenario.

[b]This is the pathway for which the reported COPC has a risk $\geq$ 1.0E-06 or an HI $\geq$ 1.0.

**Table 5.9. Summary of significant risks from Chestnut Ridge OU 2 COPCs[a]—trespasser scenario**

| Media | Analyte | Risk | Pathway[b] |
|---|---|---|---|
| ***Sluice Channel Area*** | | | |
| Ash | Thorium-228 | 7.1E-06 | External Exposure |
| ***Filled Coal Ash Pond*** | | | |
| Ash | Thorium-228 | 1.4E-05 | External Exposure |
| ***Upper McCoy Branch*** | | | |
| Ash | Thorium-228 | 5.9E-05 | External Exposure |

[a]The COPC reported in this table has a risk greater than 1.0E-06; under this trespasser land use scenario, external exposure to radionuclides in the ash was the only pathway evaluated in this baseline HHRA.

[b]This is the pathway for which the reported COPC has a risk $\geq$ 1.0E-06.

**Table 5.10. Cumulative risk and HI summary for Chestnut Ridge OU 2 COPCs—residential scenario**

| | Ingestion | Dermal | Inhalation | External exposure | Cumulative risk or HI* | Ingestion of leafy vegetables |
|---|---|---|---|---|---|---|
| **Sluice Channel Area** | | | | | | |
| *Ash* | | | | | | |
| Pathway Risk | 1.1E-05 | 4.8E-06 | 4.0E-07 | 1.5E-04 | 1.7E-04 | 3.8E-05 |
| Pathway HI | 2.0E+00 | 7.6E-02 | 6.1E-04 | | 2.1E+00 | 1.3E+01 |
| *Soil* | | | | | | |
| Pathway Risk | 8.1E-08 | | 1.2E-07 | 1.1E-04 | 1.1E-04 | |
| Pathway HI | 6.2E-01 | 1.4E-02 | | | 6.4E-01 | |
| **Filled Coal Ash Pond** | | | | | | |
| *Ash* | | | | | | |
| Pathway Risk | 3.0E-05 | 1.3E-05 | 3.7E-07 | 3.0E-04 | 3.4E-04 | 4.3E-05 |
| Pathway HI | 1.7E+00 | 8.6E-02 | 4.6E-04 | | 1.8E+00 | 9.7E+00 |
| *Sediment* | | | | | | |
| Pathway Risk | 1.4E-05 | 6.3E-06 | 2.5E-07 | 1.0E-04 | 1.2E-04 | |
| Pathway HI | 9.8E-01 | 4.2E-02 | | | 1.0E+00 | |
| *Surface water* | | | | | | |
| Pathway Risk | 9.8E-07 | | | | 9.8E-07 | |
| Pathway HI | 1.8E+01 | 5.2E-01 | | | 1.8E+01 | |
| **Upper McCoy Branch** | | | | | | |
| *Ash* | | | | | | |
| Pathway Risk | 2.0E-05 | 8.7E-06 | 2.0E-07 | 1.2E-03 | 1.3E-03 | 4.1E-05[a] or 1.1E-05[b] |
| Pathway HI | 8.5E-01 | 7.5E-02 | 1.2E-03 | | 9.2E-01 | 1.3E+01[a] or 1.5E+01[b] |
| *Sediment* | | | | | | |
| Pathway Risk | 3.2E-05 | 1.4E-05 | 5.1E-06 | 5.8E-04 | 6.3E-04 | |
| Pathway HI | 2.8E+01 | 7.9E-01 | 1.3E-02 | | 2.9E+01 | |

**Table 5.10 (continued)**

| | Ingestion | Dermal | Inhalation | External exposure | Cumulative risk or HI* | Ingestion of leafy vegetables |
|---|---|---|---|---|---|---|
| *Soil* | | | | | | |
| Pathway Risk | 6.2E-06 | 2.8E-06 | 6.2E-08 | 4.3E-07 | 9.5E-06 | |
| Pathway HI | 3.7E-01 | 2.1E-02 | 7.7E-05 | | 3.9E-01 | |
| *Surface water* | | | | | | |
| Pathway Risk | 1.6E-05 | 3.8E-05 | | | 5.4E-05 | |
| Pathway HI | 1.7E+01 | 4.9E-01 | | | 1.8E+01 | |
| **Chestnut Ridge OU 2 Groundwater** | | | | | | |
| *Groundwater* | | | | | | |
| Pathway Risk | 1.4E-04 | 2.3E-05 | | | 1.6E-04 | |
| Pathway HI | 1.1E+01 | 4.9E-01 | | | 1.2E+01 | |

*This cumulative risk/HI does not include ingestion of leafy vegetables.
[a]Risk/HI based on irrigation with Chestnut Ridge OU 2 groundwater.
[b]Risk/HI based on irrigation with UMB surface water.

**Table 5.11. Cumulative risk summary for Chestnut Ridge OU 2 COPCs—trespasser scenario**

| | Sluice Channel Area ash | Filled Coal Ash Pond ash | Upper McCoy Branch ash |
|---|---|---|---|
| External Exposure Pathway Risk | 7.2E-06 | 1.4E-05 | 6.0E-05 |
| Cumulative Risk | 7.2E-06 | 1.4E-05 | 6.0E-05 |

# 6. BASELINE ECOLOGICAL RISK ASSESSMENT REPORT

The general objective of the baseline ERA is to provide a basis for decisions concerning the need for remediation based on risks to nonhuman organisms. ERA is a process to evaluate the likelihood that adverse ecological effects may occur or are occurring due to exposure to one or more stressors (Risk Assessment Forum 1992). The purpose of the ERA is to define potential ecological effects resulting from chemicals in environmental media. The ERA consists of the following elements:

- Ecological Hazard Identification (Sect. 6.1),
- Exposure Assessment (Sect. 6.2),
- Effects Assessment (Sect. 6.3), and
- Risk Characterization (Sect. 6.4).

## 6.1 ECOLOGICAL HAZARD IDENTIFICATION

This section discusses the attributes and selection of appropriate ecological endpoints, describes the ecological setting at Chestnut Ridge OU 2 sites, provides information on the sources and hazards to which organisms may be exposed, and integrates this information into a conceptual model that portrays the interaction among sources and endpoints at the sites. The information provided here sets the stage for the exposure assessment section (Sect. 6.2) that follows.

### 6.1.1 Ecological Assessment Endpoints

Ecological assessment endpoints are actual environmental attributes that are to be protected. If these attributes are found to be significantly affected by contamination, remediation may be necessary (Suter 1989, Risk Assessment Forum 1992). While assessment endpoints represent the actual valued environmental attributes, measurement endpoints are measurable ecological parameters that are related to assessment endpoints for which effects of the hazard have been observed or measured. Assessment and measurement endpoints may, in some instances, be identical. The EPA's field and laboratory manual for ecological assessment of hazardous waste sites and the EPA's framework for ecological risk assessment identify characteristics of good assessment endpoints. They should be ecologically relevant, should include policy goals and societal values, and should be susceptible to hazards at the site (Suter 1989, Risk Assessment Forum 1992). As instructed by EPA Region IV, we use only the ecological relevance and susceptibility criteria.

Five endpoints that satisfy one or both of the requirements for good assessment endpoints were selected for Chestnut Ridge OU 2 (Table 6.1). These endpoints were the fish, benthic macroinvertebrate, terrestrial plant, soil heterotroph, and small mammal communities. Four endpoints are biologically relevant: fish and benthic macroinvertebrate communities and plant and soil heterotroph communities are integral to energy flow and nutrient cycling in aquatic and terrestrial ecosystems, respectively. Food chain transfer is of particular concern for those compounds, such as selenium, that bioaccumulate.

The second endpoint criterion, susceptibility, is the tendency of an organism to respond to a chemical (Suter 1993). For an organism to be susceptible to a chemical, it must be both exposed to and sensitive to that chemical. All the endpoints chosen for Chestnut Ridge OU 2 are exposed to and most are believed to be sensitive to elements in or characteristics of coal ash. For example, fish and benthic macroinvertebrates are exposed to contaminants in water and sediments and their abundance can be reduced by exposure to the high pH and metals associated with ash (Cairns et al. 1972, Cumbie and Van Horn 1978). Terrestrial plants and soil heterotrophs are directly exposed to contaminants in soil. Among plants, in addition to bioaccumulating metals, growth may be reduced among individuals grown in coal ash (Adriano et al. 1980). Small mammals experience contaminant exposure through food, water, and incidental soil ingestion. Mammals also bioaccumulate metals from ash (Fleming et al. 1979), some of which (selenium and arsenic) have known mammalian toxicity.

Toxicity to individuals of any threatened or endangered species resulting in reduced viability or fecundity was also considered as an additional assessment endpoint. While threatened and endangered species are, in general, too rare to be ecologically significant and may not be particularly susceptible or clearly associated with the site, EPA Region IV indicated during the March 4, 1993, meeting in Chattanooga that these species may be used as assessment endpoints because they are ARARs. Because of the particular societal value placed on these organisms, as reflected in the Endangered Species Act, the characteristics of this endpoint are any toxic effect that would reduce survivorship or fecundity of individuals.

The complete definition of an assessment endpoint requires a subject (listed above as fish, plants, small mammals, etc.) and a level of effect that will be used to determine if an impact has occurred (Suter 1993). Assessment endpoints were considered to be affected if characteristics of the endpoint measured at Chestnut Ridge OU 2 (second column, Table 6.1) represented 80% or less of those characteristics observed at the reference site.

The 20% reduction level was chosen for two reasons (Suter et al. 1992a). First, this level of effect is approximately the limit of detection of field measurement techniques and is probably below the detection limits of the public (i.e., fishermen would probably not notice a 20% reduction in the abundance of fish). For example, at least 20 years worth of data would be necessary to detect a 50% reduction in mean year-class strength of white perch in the Hudson River, even allowing a Type-II error of 50% (Vaughan and Van Winkle 1982). The minimum detectable difference varies with species, habitat, and sampling method, but for mobile species, differences of less than 20% can seldom be reliably detected.

The second reason for choosing a 20% level is that it is approximately the level of effect that corresponds to benchmarks used by the EPA and others, including the benchmark used most often for ecological assessments of hazardous waste sites, the chronic National Ambient Water Quality Criteria (NAWQC) for protection of aquatic life (Office of Policy Analysis 1989). NAWQC are based on thresholds for statistically significant effects on individual responses of fish and aquatic invertebrates. Those thresholds correspond to ~25% reductions in the parameters of fish chronic tests (Suter et al. 1987) and 20% reductions in the parameters of fish "subchronic" tests (Anderson and Norberg-King 1991). Because of the compounding of individual responses across life stages, the NAWQC frequently correspond to >20% effects on populations (Barnthouse et al. 1990).

The 20% level is also consistent with practice in ecological assessments of terrestrial effects because the lowest-observed-effect concentration (LOEC) for dietary tests of avian

reproduction (the most important chronic test endpoint for ecological assessment of terrestrial effects of pesticides and arguably the most applicable test for hazardous waste sites) corresponds to ~ a 20% effect on individual response parameters (Office of Pesticide Programs 1982). Hence, a population effects level for endpoints lower than 20% would result in benchmarks lower than the current EPA practice and could not be reliably confirmed by field studies.

Measurement endpoints selected to evaluate effects on the assessment endpoints are listed in Table 6.1.

Ecological assessment of hazardous waste sites is limited to those biota that can be definitively associated with the site. This approach does not address contaminant exposure that may be experienced by wide-ranging species (e.g., white-tailed deer, wild turkey, red fox, hawks, etc.) that may periodically use the site. To thoroughly evaluate risks to these species, a reservation-wide context is necessary in which the cumulative effects of exposure to multiple sites is considered. Data from individual waste sites are still required to provide estimates of the contaminant exposure wide-ranging species experience at each site. At Chestnut Ridge OU 2, potential reservation-wide endpoints include white-tailed deer and avian and mammalian predators of small mammals (e.g., hawks, owls, fox, etc.). White-tailed deer are exposed to contaminants in vegetation and through consumption of soil and ash. Predators are exposed to contaminants retained in tissues of small mammals resident at the site. These exposure estimates will be incorporated into the reservation-wide assessment to evaluate the contribution of Chestnut Ridge OU 2 to the risk experienced by populations of these species at the reservation level.

### 6.1.2 Environmental Description

Chestnut Ridge OU 2 has been divided into three areas based on topographic and hydrologic characteristics. These areas are the Sluice Channel Area, FCAP, and UMB. Characteristics of these areas determine the transport and deposition of coal ash, the media affected by ash-borne contaminants, and the assessment endpoints to be considered.

#### Sluice channel

The 6-acre (2.43-ha) area closest to the source of the coal ash has been designated the "Sluice Channel Area." This is the hill (15% slope) down which the ash flowed in a slurry over the well-drained, acidic, highly weathered, residual soils. Coal ash depth on the slope varies between zero and 2.5 ft (0.75 m), 25% of the area with essentially no ash and 55% with less than 6 in. (15 cm). Mature, secondary mixed pine and hardwood forest shelters a hardwood understory and sparse ground cover.

The soil of the Sluice Channel Area is expected to be a biologically active zone due to the limited depth of ash. The growth and nutrient cycling functions of soil microbial populations may be negatively affected by contaminants in surface ash, and in surface soil into which contaminants are leaching, or have been leached, from the ash. Soil and litter organisms active in the degradation of organic matter, feeding on living tissue, and other litter-dwellers may be adversely affected by contaminants in vegetation derived from the contaminated growth substrate. Plant growth and community species composition may be limited by toxic or other effects of contaminants taken up from the coal ash. Local and

transient birds and mammals may be exposed to toxic levels of metals and organics by ingestion of contaminated vegetation, invertebrates, and ash.

The mature secondary mixed pine and hardwood forest of the Sluice Channel Area should remain stable if left undisturbed. Little erosion of the ash should occur as long as the vegetative cover remains intact. Continued addition of organic matter to the ash may reduce the mobility and biological availability of the metal and organic contaminants, while providing microhabitats for soil heterotrophs active in humus formation and biodegradation of the natural and contaminant organic materials.

### FCAP

The second area of the Chestnut Ridge OU 2 is the flat, approximately 9-acre (3.6-ha) FCAP located between the side slopes of the original valley of UMB. The 64-ft (19-m) dam at the southern end of the FCAP has restricted water flow from the ash pond (Energy Systems 1991). The difference in water level in the ash has led to the division of the FCAP into two vegetative zones (Fig. 2.8). The lower one-third of the area starting at the dam has a lower water table and is vegetated by hardwood trees (Sycamore Area) approximately 20 years old with a dense understory. The higher water table of the upper portion of the FCAP has resulted in vegetation dominated by willows and grasses with small wetland areas in the northern corners (Willow Area). UMB usually carries little water as it flows over the ash surface on the east side of the FCAP. During wet periods, however, the FCAP surface is crossed by many small streams draining into UMB, increasing in volume as it falls over the dam into the UMB portion of the site.

The extreme depth of the ash, approximately 3 ft (1 m) at the sides to 60 ft (18 m) at the dam, precludes the soil beneath from being a biologically active zone over most of the FCAP. The growth of vegetation and soil organisms is dependent on, and limited by, the physical and chemical characteristics and water content of the ash. The surface water in the stream and wetlands, and the pore water of the ash layer bring soil organisms, mammals, and birds into direct contact with ash-derived contaminants, especially in the wetter portion of the FCAP. Local and transient birds and mammals may be exposed to toxic levels of metals and organics by ingestion of contaminated vegetation, invertebrates, water, and ash.

The hardwood forest of the lower FCAP area should continue to develop if left undisturbed. Little erosion of the ash will occur over the major portion of the area as long as the dam remains intact. Concern has been expressed, however, about continuing incision of UMB into the ash deposit (Energy Systems 1991). The instability of the channel may cause high ash loads in UMB during periods of high flow. Continued addition of organic matter to the ash may reduce the mobility and biological availability of the metal and organic contaminants, while providing microhabitats for soil heterotrophs. As the vegetation of the FCAP and surrounding slopes develops, a gradual lowering of the water table of the FCAP may be expected. Drying of the surface ash may affect vegetative community characteristics, ash weathering, contaminant chemistry and bioavailability, and decrease direct contact of animals with contaminated water.

### Upper McCoy Branch

The third area of Chestnut Ridge OU 2 consists of UMB from the FCAP dam to Rogers Quarry. The stream is fed by springs at the foot of the dam representing seepage from the

FCAP through and below the dam, UMB flowing over the dam, and small tributary streams from surrounding slopes. This is a narrow (2 to 6.5 ft; 0.5 to 2 m), fast-flowing stream that travels approximately 1 km over a 7% slope. Under normal flow conditions it is shallow (<12 in.; <30 cm) and consists of a series of runs and riffles (Energy Systems 1991, Appendix B). The flat area at the confluence of UMB and Rogers Quarry is a wetland. Riparian vegetation along the length of the stream includes mixed hardwood forest and areas of grass cover.

Ash deposited on the stream's floodplain varies in depth from zero to ~ 3 ft (1 m). The ash was generally mixed with soil during or after deposition, and in some reaches, the ash has been covered by alluvial soil deposits. The greatest accumulation of ash lies within ~ 800 ft (240 m) downstream of the dam where it extends the width of the valley (~ 200 ft; 60 m). Further downstream, the deposits are limited to a few meters horizontally from the edge of the stream (Energy Systems 1991).

Soil heterotrophs, vegetation, and local and transient birds and mammals may be exposed as discussed for the Sluice Channel Area and the FCAP. UMB contains aquatic habitats that may be degraded by contaminants in water that is derived primarily from seepage and overland flow from the FCAP, and by the presence of ash in the stream channel. Contaminants in these media may impact benthic macroinvertebrate population species richness and species richness and viability of the fish population.

Downstream movement of ash in the stream bed and banks may occur during downward and lateral cutting of UMB. This may eventually rid the stream and its floodplain of ash and improve the terrestrial habitat. However, without an improvement in the quality of the water entering UMB, the aquatic habitat may remain negatively affected by the ash-derived contaminants.

### 6.1.3 Sources

#### 6.1.3.1 Source history

The location of Chestnut Ridge OU 2 is described in Chap. 2. A detailed discussion of the OU history and current conditions is presented in Chap. 3.

#### 6.1.3.2 Source data

Data for this assessment were collected in two phases. The first investigation was conducted by CH2M HILL to define the nature and extent of contamination at the site, to assess the potential for off-site migration of contaminants, and to evaluate the risks to human health from exposure to the contaminants. The Phase 1 RI Report (Energy Systems 1991) was reviewed by EPA Region IV and the TDEC and was determined to require further investigation. Comments included the lack of an Ecological Risk Assessment and insufficient background data for all media. The Phase 2 Sampling Plan (Energy Systems 1993c) was developed by CDM Federal to address these issues.

Phase 1 data included contaminant concentrations in ash, deep and shallow groundwater, soil, and surface water samples from UMB; groundwater samples from the Sluice Channel Area; and background soil samples from the ridge immediately west of UMB. Phase 2 data included: ash, deep groundwater, and soil samples from the Sluice Channel Area; ash, deep

groundwater, sediment, soil, and surface water samples from the FCAP; deep and shallow groundwater and surface water samples from UMB; background soil samples from the ridge west of UMB; background sediment and surface water samples from Walker Branch and White Oak Creek; and terrestrial bioaccumulation samples (vegetation and small mammals) from the Sluice Channel Area and the FCAP. Refer to the Phase 1 RI Report (Energy Systems 1991), the Phase 2 Sampling Plan (Energy Systems 1993c), and Sect. 3.2.6 for sampling and analysis details.

The ash, groundwater, soil, sediment, surface water and abiotic reference data included in this assessment were evaluated (i.e., comparison of Phase 1 and Phase 2 samples and an assessment of validation results) and prepared (i.e., grouped by area and statistically summarized) by the Energy Systems Risk Analysis Section (see Sect. 5.2.3). The resulting summary statistics for the abiotic site and reference data are presented in Tables J.2 and 6.2, respectively. The terrestrial bioaccumulation data were evaluated and prepared by the Energy Systems Environmental Sciences Division and are discussed in Energy Systems 1991 and Sect. 3.2.6 of this RI.

There are four data preparation issues to note for the ERA. First, results from the *Background Soil Characterization Project for the Oak Ridge Reservation* (Energy Systems 1993b) were used (Table 6.3) in addition to, rather than instead of, the reference soil data collected for this RI. Second, only shallow groundwater was assessed (see Sect. 6.2.2.2, and Table 6.4). Third, all surface water results are for unfiltered samples. Fourth, the surface water uranium activities presented in Tables J.2 and 6.2 were converted to total inorganic uranium concentrations. For each isotope the specific activity (Ci/g) was used to convert the pCi/L to mg/L (DHEW 1970). The total uranium concentrations are presented in Table 6.5.

### 6.1.3.3 Sources of exposure

Ash may be a source of exposure to the plant, soil heterotroph, and wildlife communities in the Sluice Channel Area, FCAP, and UMB. Contaminants in surface ash may be taken up by plants and soil microbes. Soil heterotrophs and wildlife may be directly exposed to, or ingest, contaminants in the ash.

Soil beneath the ash may be a source of exposure to the plant and soil heterotroph communities in the Sluice Channel Area and UMB. Soil beneath the ash is not a source of exposure at the FCAP because this soil occurs at depths of 3 to 60 ft (1 to 20 m) (see Sect. 6.1.2). Contaminants in soil exposed to ash leachate may be taken up by plants and soil microbes. Soil heterotrophs may be directly exposed to, or ingest, contaminants in soil.

Shallow groundwater may be a source of exposure for the plant community. Deep rooting vegetation may take up contaminants from shallow groundwater.

Surface water may be a source of exposure to the fish, benthic macroinvertebrate, and terrestrial wildlife communities at the FCAP and UMB. Contaminated water may be respired by the fish and benthic macroinvertebrates, or ingested by local and transient mammals and birds.

Ash-contaminated sediment may be a source of exposure for the fish and benthic macroinvertebrate communities at the FCAP and UMB. Sediments, which may contain ash or contaminants leached from the ash, may be ingested by fish and invertebrates. Interstitial

water, which may be in equilibrium with sediment contaminants, may be respired by benthic macroinvertebrates.

Vegetation, which accumulates some contaminants, may be a source for detritivores and herbivorous wildlife at the Sluice Channel Area, the FCAP, and UMB. Plant material may be consumed by soil and litter organisms that feed on living tissue or are active in the degradation of organic matter. Contaminated plant material may be ingested by local and transient birds and mammals.

Invertebrates and small mammals may be a source of exposure to vermivores/insectivores and carnivores, respectively. Annelids and other invertebrates consuming ash and contaminated soil and plant material may accumulate contaminants and then be consumed by shrews or other vermivores. Small mammals, which may accumulate contaminants through the ingestion of water, soil, ash, plant material and prey, may be consumed by predatory birds and mammals at the site.

#### 6.1.3.4 Screening results

Screening ERAs are conducted, using new and existing data, in the early phases of an RI. One objective of a screening assessment is to identify and prioritize contaminants of potential ecological concern. A biological monitoring assessment of McCoy Branch was conducted in support of the Phase 1 RI (Energy Systems 1991). However, a screening ERA of Chestnut Ridge OU 2, as required in the current regulatory environment, was not performed (see Sect. 1.4). Therefore, contaminants analyzed for in the RI can not be removed from the list of contaminants of potential ecological concern prior to the completion of this assessment. Consequently, all contaminants that were measured in the potential exposure sources were included in this assessment.

### 6.1.4 Description and Properties of Coal Ash

Coal ash is an unusual CERCLA waste in that it is exempted from treatment as a hazardous waste under RCRA Subtitle C. In a recent report to Congress, the EPA (1988a) justified this status by contending that coal ash does not possess any of the four RCRA hazardous properties (ignitability, corrosivity, reactivity, and toxicity). However, as the authors themselves admit, the EPA's study (1988c) of coal ash's potential hazards presented no original data, did not include a complete literature review, and devoted relatively little attention to ecological risks of coal ash. Therefore, a review of the ecologically hazardous properties of coal ash is a necessary component of the source characterization for this site.

Use of coal combustion as a means of energy production has resulted in environmental contamination that has been shown to be hazardous and adversely affects aquatic and terrestrial ecosystems. Hazardous characteristics arise from the physical and chemical properties, pH, trace element composition and content, and organic chemical content of coal ash. This section addresses the potential hazards of coal ash from the Y-12 steam plant by reviewing studies of effects of coal ash from various sources.

The physical, chemical, and mineralogical properties of coal ash depend on the parent coal composition, conditions during combustion, efficiency of emission control devices, storage and handling of the byproduct, and climate (Adriano et al. 1980). Geographical origin of the parent coal and sampling position within the power plant may also contribute to the variability

of coal ash properties (El-Mogazi et al. 1988). Studies have shown that ashes from different geographical regions vary in elemental concentrations and pH (Klein et al. 1975, Furr et al. 1977). For example, ashes derived from coals of the western United States tend to have higher concentrations of elemental boron, sodium, and calcium; lower concentrations of sulfur; and higher pH values than coal ashes from the eastern United States (Shannon and Fine 1974, James et al. 1982). Chemically, coal contains every naturally occurring compound. High concentrations of selenium, boron, copper, chromium, lead, molybdenum, and arsenic in oxidized states are typical (El-Mogazi et al. 1988). The elevated trace element concentration present in coal ashes makes ash products potentially hazardous (Adriano et al. 1980).

The toxicity of fly ash to the aquatic environment has largely been studied through accidental discharge of massive quantities of fly ash into aquatic systems. The Clinch River Spill in 1967 is one such example. As a result of failure of an ash pond dike, ash and highly caustic sluice water (pH 12) were released to Dump's Creek, which enters the Clinch River 0.5 mile (0.8 km) downstream (Cairns et al. 1972). The release killed more than 200,000 fish over 90 river miles (140 km), eliminated all benthic invertebrates for 3-4 river miles (5-6 km), and drastically reduced benthic invertebrate abundance and diversity for 77 miles (120 km) (Anonymous 1967). The effects were attributed to high pH, but metals were not characterized. Two years after the spill, the fish and invertebrate communities had not fully recovered (Cairns et al. 1972).

In the aquatic environment, coal ash contamination through surface deposition can be a major source of water pollution (Rohrman 1971). Heavy ash intrusion and its subsequent siltation reduces aquatic invertebrate density and limits macrobenthic organisms (Cherry et al. 1979). Body burdens of hazardous elements in aquatic biota gradually increase with chronic exposure to a fly ash-contaminated environment, resulting in decreased diversity in affected communities (Adriano et al. 1980). Turbidity has been shown to inhibit the growth of benthic algae. The subsequent filling in of rocky crevices by sediment reduces the microhabitat available to certain aquatic species (Roy et al. 1981). Besides direct toxicity to aquatic animal species, fly ash may also affect aquatic ecosystems by altering nutrient cycling. Ash may serve as a phosphorous sink, reducing phosphorous levels in lakes, or as a silicon source (Roy et al. 1981). In addition, stratified sedimentation of fly ash deposits in ponds forms hard layers that may impede root penetration of aquatic plant species (Adriano et al. 1980).

Fly ash leachates may adversely affect aquatic ecosystems by altering aquatic pH. The high pH of ash leachates may affect metal speciation and water chemistry (Cherry et al. 1979). As a result, elements may get converted to more or less toxic forms (Webster et al. 1986). Available data further suggest that certain species of fish may be adversely affected more by the alkaline pH (characteristic of many fly ash leachates) and physical presence of particles (as turbidity) than by leached metals (Roy et al. 1981). Wasserman et al. (1974) reported that runoff from alkaline ash settling ponds was lethal to catfish because the runoff raised the pH, causing precipitation of ferric hydroxide that clogged the gill mucus of the fish, resulting in asphyxiation. High alkalinity in fly ash has also been shown to cause a decrease in both microbial nitrification activity and respiration leading to increased instability in bacterial communities.

Field studies have shown that increased drift of stream macroinvertebrates has been induced by stream acidification (Hall et al. 1980) and sediment addition due to coal ash intrusion (Rosenberg and Wiens 1978). The scouring and smothering of benthic biota from

inefficient settling basin operations, uptake of chemical elements in aquatic biota, and acidic pH changes from fly ash input and coal seepages into streams can result in significant environmental impact (Cherry et al. 1979).

The release of trace elements in water is greatly influenced by the equilibrium pH of the fly ash suspension. Release rates of soluble trace elements from fly ash may be higher in groundwater than in surface water, due to the groundwater's lower pH and higher $CO_2$ content (Phung et al. 1979). Highly volatile elements that get leached into surface and groundwater sources form anionic species upon hydrolysis that remain soluble in alkaline environments while the more metallic cations tend to form hydroxide precipitates. Heavy metals leached from coal ash are accumulated at considerably high levels in aquatic plants. Accumulations generally include higher levels of titanium, manganese, arsenic, and mercury (Adriano et al. 1980). Macroinvertebrates usually accumulate cobalt, mercury, copper, chromium, cadmium, and arsenic in their tissues (Adriano et al. 1980).

Coal ashes also contain organic compounds, such as PAHs, that were not completely combusted during the firing process of the parent coal (El-Mogazi et al. 1988). These biologically active substances may induce gene mutations in a variety of species (Chrisp et al. 1978). Total hydrocarbon content in the ash is estimated to be less than 10 mg/kg and total PAH concentrations are less than 0.2 mg/kg (Wei et al. 1982). Mutagenicity may also be influenced by the method of ash collection used at the individual plant. Conclusive studies have not been done on the effects of organics in the aquatic environment. To date the effects of organic compounds present in fly ash on aquatic and terrestrial ecosystems have been inadequately characterized (Roy et al. 1981).

In the terrestrial environment, coal ash may be hazardous to human, animal, and ecological health (Roy et al. 1981). Elements that have been found to be hazardous to human and animal health preferentially concentrate on the surface of the smaller particles (Page et al. 1979). Dependence of heavy metal concentration on particle size of fly ash has been observed by Davison et al. (1974) and Lee and von Lehmden (1973). Although fly ash is emitted into the atmosphere in relatively small quantities, the micron- and submicron-size particles that make up fly ash are readily respired, can have long residence times in the lung, and serve as a source of trace elements to both aquatic and terrestrial ecosystems (Natusch and Wallace 1974, Linton et al. 1976, Rohrman 1971). Among terrestrial animals fly ash may be hazardous by direct external reactions (skin, eyes, etc.) or by reactions in the alimentary canal (Roy et al. 1981). Small mammal studies suggest respiratory toxicity of fly ash aerosols is minimal (Roy et al. 1981).

Although boron is the only element to have been associated with any significant reduction in crop production due to cytotoxicity, selenium and molybdenum tend to accumulate in both plant and animal tissues (Adriano et al. 1980). High accumulations of selenium, boron, nickel, and molybdenum have been found in a number of woody plants (Scanlon and Duggan 1979), and elevated concentrations of selenium have been found in the meat, milk, and eggs of animals trapped on a fly ash-contaminated site (El-Mogazi et al. 1988). Selenium accumulation is greater in legumes than grasses, and although molybdenum and selenium are generally nontoxic to plants, uncontrolled accumulation could be a potential hazard to livestock animals consuming forage with elevated levels of these two elements (Arthur et al. 1992a, Adriano et al. 1980). At the Chestnut Ridge OU 2 site, concentration levels that exceed benchmarks for numerous plant and animal species have been found for arsenic. A review of available literature has indicated the presence of elevated

concentrations of arsenic at other geographical sites but not as a major contaminant with the detrimental concentration levels associated with this site. The presence of high levels of arsenic at this site can be attributed to the composition of the parent coal used to fuel the Y-12 steam plant.

The major adverse effects of coal ash contamination in both terrestrial and aquatic environments stem from an accumulation of toxic metals and a reduction in pH. Long-term studies are needed to adequately characterize the properties of fly ash and to aid in the unification and organization of data.

### 6.1.5 Conceptual Model

The ecological conceptual model graphically represents the relationships between the contaminant sources and the endpoint receptors. It integrates the information in the other subsections of the hazard identification and presents them graphically. It is not intended to show all of the possible sources, routes of transport, modes of exposure, or effects. Rather, it includes the only identified CERCLA source (deposited coal ash), the receptors that are designated as assessment endpoint species or communities, and the major routes that result in exposure to either the ash or contaminants leached from the ash.

Figure 6.1 depicts the direct exposure of the terrestrial biota to the ash deposited in the OU and the food-chain transfer of contaminants taken up from the ash. Direct exposure results in uptake of contaminants by plants and soil organisms and ingestion of ash by animals. Soil becomes contaminated by leaching of contaminants from ash, and receptors may be directly exposed to the contaminated soil by the same mechanisms as ash. Secondary exposure results from consumption of plants by herbivorous mice and of soil invertebrates by shrews. Leachate from ash deposited on soil also moves to the surface water resulting in the exposures shown in Fig. 6.2. It mixes with leachate from the ash deposited in UMB and the two ponds on the FCAP (in situ ash), resulting in direct exposure of the aquatic biota (benthic invertebrates in all waters and fish in UMB). Fish are secondarily exposed by consuming invertebrates.

Figure 6.3 depicts the connections between this OU and the receptors that are not clearly associated with the site and must be assessed on a reservation-wide basis. Ash contaminates plants and small mammals by the on-site processes depicted in Fig. 6.1. When foraging on the OU, deer consume contaminated plants and ash, red foxes consume small mammals and ash, and red-tailed hawks consume small mammals. This risk assessment addresses the exposure received by those receptors when using the OU but does not estimate effects on their populations. Effects on those populations of exposures to all OUs will be estimated in the reservation-wide risk assessments.

## 6.2 EXPOSURE ASSESSMENT

This section describes the modes of exposure that occur on the OU, the ways in which exposure is estimated, and the available exposure data for ecological risk assessment. It is organized in terms of the endpoints for which exposure must be estimated (wildlife, fish, benthic invertebrates, soil invertebrates, and plants) and the types of exposures associated with the different lines of evidence (media analyses, body burden analyses, media toxicity

tests, and biological surveys). This introductory section addresses issues that are common to all endpoints.

Estimation of exposure to measured contaminant concentrations requires that the data from multiple measurements distributed over space and time be reduced to a single exposure value to be used in the screening calculations. There is no guidance from regulatory agencies on how this should be done. The guidance for HHRA at CERCLA sites calls for assessment of reasonable maximally exposed individuals. Therefore, we have devised procedures to generate single exposure values for each endpoint for this EA, as explained below.

**Fish and Water-Exposed Invertebrates.** Fish and invertebrates in small stream segments like UMB or small bodies of standing water like the FCAP ponds do not spatially integrate the exposure environment like wildlife. However, unlike sedentary organisms in soils or sediments, they are not exposed to a relatively constant concentration. Rather, they average their exposure over time as the concentrations in water change with changes in leaching or in dilution. Therefore, we use the upper 95th percentile on the estimate of the mean (i.e., the upper 95% confidence bound), which can be thought of as a reasonably conservative estimate of the time-averaged exposure.

**Plants, Soil Invertebrates, and Sediment-Exposed Invertebrates.** These relatively immobile organisms do not integrate their exposures over space like wildlife or humans. Rather, they are exposed to contaminant concentrations at a single location. Therefore, the reasonable maximally exposed individual is the one completing its life cycle in the most contaminated location. That exposure is estimated by the maximum observed concentration.

**Wildlife.** Because wildlife are exposed to a variety of sources on a site including food, surface water, and soil; because they are mobile and use various portions of the site like humans; and because they are taxonomically similar to humans, we assume that the exposure summarization method for humans is applicable. Therefore, we use the upper 95% confidence bound or the maximum value, whichever is lowest. The logic of this rule is unclear, but it is adopted as the nearest thing to applicable regulatory guidance.

Note that these discussions of exposure values for individuals do not negate the definitions of endpoints in terms of populations and communities (Sect. 6.1.1). These values are used in the calculation of hazard quotients in the screening portion of the assessments. The actual characterization of risks in Sect. 6.4 will consider the complete distributions of exposures when estimating risks to the endpoint populations and communities.

### 6.2.1 Characterization of the Exposure Environment and Description of Exposure Routes

Chestnut Ridge OU 2 may be divided into three areas of environmental contamination: the FCAP, the Sluice Channel Area, and UMB. The FCAP, which serves as the settling basin for the coal byproducts from the Y-12 steam plant, is the largest area of contamination within Chestnut Ridge OU 2. The pond is a 9-acre storage area located in the McCoy Branch Watershed, on the southern slope of Chestnut Ridge. The area designated as the "Sluice Channel Area" served as the area over which coal ash is sluiced and allowed to flow overland by gravity into the pond. Although the FCAP is impounded by a 64-ft earthen dam across UMB, it flows over the emergency spillway on the east side of the pond and into UMB. UMB then enters Rogers Quarry.

Plant and animal species present at the site include a secondary mixed pine and hardwood forest with a dense understory,round cover, and willow and grasses with small wetland areas. A stream that is fed by a spring at the foot of the dam and its floodplain comprise the site.

The contamination at the site includes elevated concentrations of trace metals and suspended solids. Sources of exposure are ash, surface water, soils and sediments, groundwater, vegetation, and invertebrates and small mammals. Ecological endpoint communities include fish, benthic macroinvertebrates, plants, soil heterotrophs, small mammals, and terrestrial wildlife. Routes of exposure are through ingestion of surface water and groundwater, ingestion of soils and sediments, and ingestion of vegetation, invertebrates, and small animals by predators.

Fish and benthic macroinvertebrate populations are at risk at the FCAP and UMB from exposure to contaminated sediments, sediment-associated food, and associated pore water, which may be ingested by aquatic species. Exposure may also occur from contaminated surface water, which may be respired by fish populations, and from interstitial water, which may be respired by benthic macroinvertebrates. In addition, the increase in pH associated with ash intrusion may cause gill abrasion and clog gills with mucus (in fish), and decrease microbial nitrification activity. Ash (as turbidity) in the aquatic environment may reduce invertebrate density, limit macrobenthic organisms, inhibit the growth of benthic algae, or alter nutrient cycling.

Terrestrial plants and shrubs are exposed directly to contaminants in soil. Tree and ground cover species in the Sluice Channel Area and the FCAP accumulate heavy metals in leaves and stems. Site data have shown that some metals in the ash, particularly selenium and arsenic, are taken up by plants and retained in foliage. Effects of selenium and arsenic on plant species include leaf chlorosis, reduced yield, retardation of root and top growth, and leaf necrosis.

Soil microbes, earthworms, and site vegetation at the Sluice Channel Area, the FCAP, and UMB are directly exposed to ash through external contact with contaminated soil and pore water, and ingestion of soil. Contaminants present in the soil, either beneath ash or in localized areas with no surface ash present, and contaminants present in shallow groundwater in the form of soil/ash pore water serve as a source of exposure for plants and soil-dwelling organisms.

Small mammals and other terrestrial wildlife are exposed to contaminants in ash through incidental consumption of ash when foraging. Species that may forage at the site include the white-footed mouse, the short-tailed shrew, the red fox, and the white-tailed deer. Mechanisms of exposure include ingestion of ash or soil present on foliage or adhering to plant roots, soil adherence to food sources such as earthworms, soil ingested while burrowing, and soil ingested while grooming paws or fur.

Other wildlife, such as transient birds and mammals, are exposed to contamination through incidental consumption of contaminated vegetation, invertebrates, surface water, and ash. Vertebrate predators, vermivores, and insectivores are exposed to coal ash contaminants retained in tissues of prey species resident at the site, and through ingestion of surface water. Ash contaminants such as heavy metals are further introduced into the food web through

consumption of live plants by soil/litter-dwelling organisms, litter-decomposing invertebrates, and herbivorous wildlife.

In the future, it is estimated that some source levels will decrease while others remain constant. Over time, it is expected that the aquatic and terrestrial environment will recover. It is not expected that the habitat will change or be altered in such a way as to increase ecological contamination. In the terrestrial environment, slow rates of natural recolonization and the limited number of species during the first few years will reflect the adverse plant growth conditions in ash deposits (Adriano et al. 1980). However, as the weathering of soil continues over time, higher plant species will display some growth and recovery toward normalcy (Adriano et al. 1980). As the vegetation of the FCAP and surrounding hill slopes develops, a gradual lowering of the water table of the FCAP may be expected. As a direct result, heavy metals and toxic element concentrations may be increased along with the initiation of a succession of related and cause-dependent events in the ecological environment (Cherry and Guthrie 1979).

Breakage of the dam below the FCAP could result in further exposure of fish and benthic macroinvertebrates to contaminated surface water, sediments, and ash. However, disbarring dam breakage, no further contaminants are likely to be introduced into the Chestnut Ridge OU 2 site.

The routes of exposure at Chestnut Ridge OU 2 include ingestion of surface water by fish and benthic macroinvertebrates, small mammals, and other terrestrial wildlife; ingestion of soils and sediments by fish and benthic macroinvertebrates, soil microbes, earthworms, and incidental soil ingestion by small mammals and other terrestrial wildlife while foraging; ingestion of vegetation by soil/litter-dwelling organisms, litter-decomposing invertebrates, and herbivorous wildlife; and ingestion of invertebrates and small animals by predators.

Based on site-specific information provided by biomonitoring data and toxicological benchmarks developed for the site, exposure routes have been established. Source data have been collected for the risk assessment and compared with background concentrations. Descriptions for routes of exposure for all assessment endpoints along with justifications of screening procedures, results of screening procedures and a discussion of uncertainties in screening procedures can be found in Sects. 6.2.2, 6.3, and 6.4.3.

### 6.2.2 Quantification of Exposure

#### 6.2.2.1 Quantification of exposure for aquatic organisms

**Surface Water**

Fish and benthic macroinvertebrates of the FCAP and UMB may be exposed to contaminants in the surface water. Although fish communities were not found in UMB above and below the earthen dam (Ryon et al. 1992, Appendix G), introduction or future invasion from Rogers Quarry could result in exposure of fish to contaminated surface water. A benthic macroinvertebrate survey of surface water in the FCAP was not performed. Benthic macroinvertebrate communities were found in UMB (see Sect. 3.2.6), and could invade the FCAP surface water via reproductive flight, resulting in exposure to contaminated surface water. However, the surface water in FCAP flows over the dam and down a steep bedrock slope, thus preventing the upstream migration of individuals. Therefore, it was assumed that

fish and benthic macroinvertebrates received 100% of their exposure to contaminated surface water at either the FCAP or UMB.

Conventional toxicity data are the result of controlled tests using individual contaminants dissolved in laboratory water, and standard test organisms. This high degree of control results in better evidence of causality, but less ecological relevance. Contaminant concentrations in surface water were analyzed for both the FCAP and UMB (see Sect. 6.1.3). For the comparison of water concentrations to toxicity data, the UCL95 was used instead of the maximum detected concentration. This is because the maximally exposed aquatic organism would be exposed to spatially and temporally varying contaminant concentrations.

Only total recoverable concentrations, which are required for the HHRA, were determined. Assessing the exposure and effects of surface water contaminants based on total recoverable concentrations is conservative because the concentrations of bio-available contaminants is generally less than the total recoverable concentrations. Metals, for example, typically include 30-95% particle bound material (HECD 1992). Conventional toxicity data on the effects of surface water contaminants are based on laboratory studies using contaminants dissolved in filtered water.

The toxicities of arsenic, chromium, and mercury are dependent upon their chemical species and the test organisms. Arsenic was not speciated for this RI, but an independent study of the biogeochemistry of arsenic in the Clinch River (Ford et al. 1993) analyzed surface water samples from MCK 2, Rogers Quarry, and MCK 1.6 weekly from February 1992 to March 1993. During this period, total arsenic concentrations at the inlet to Rogers Quarry (MCK 2) were less than 5 $\mu$g/L. Arsenic (V) represented almost all of the arsenic in the surface water samples. However, arsenic was conservatively assumed to be in its most toxic form, which is arsenic (V) for six of the seven benchmarks with both arsenic (III) and arsenic (V) values. To support the HHRA, total chromium and chromium VI concentrations were determined in three of the six surface water samples from Chestnut Ridge OU 2. Chromium VI was not detected in any of these samples. Therefore, the total chromium concentrations were assumed to represent the trivalent form of chromium. Like arsenic, mercury was not speciated in the analyses for this RI and was conservatively assumed to be in its most toxic form for each benchmark.

Media toxicity data are the result of controlled tests using contaminated site media and standard test organisms. Aquatic toxicity tests, both laboratory and *in situ*, were performed for UMB (Ryon et al. 1992, Sect. 6.1.3 this RI).

These test data summarize the effects of exposures to all contaminants in the water at the times the tests were conducted. However, water was analyzed only for basic water quality parameters (pH, conductivity, alkalinity, hardness, and, in one or two Phase 1 samples, orthophosphate, sulfate, and nitrate) to ensure that apparent toxicity was not due to fundamental water characteristics. Sampling was conducted on a planned schedule and did not include storm events that may have elevated dissolved and suspended contaminant concentrations. Water for 4-day and 7-day tests was collected daily for renewal. Water for renewal of the life cycle *C. dubia* test was collected on Tuesdays and Fridays and stored at 7°C. Dilutions were made with mineral water. Water was not filtered or aerated. Water for testing was collected from the inlet to Rogers Quarry and the spring at the base of the dam, thereby bracketing any gradient in toxicity within UMB. Tests began in January 1989, following the termination of ash disposal in the FCAP, and continued irregularly through the

spring of 1993 (Ryon et al. 1992, Sect. 3.1.3 of this RI). Therefore, they do not include the period of exposure to sluice water, but do include exposure to relatively fresh and unweathered ash.

Biological surveys result in demographic data for the biota naturally occurring at the site. These data summarize the effects of the actual exposures to all contaminants of the aquatic communities surveyed. However, because exposures are not controlled they cannot be quantitatively specified. One can only state that the surveyed communities consist of organisms that have been exposed for their entire life cycles to the physical and chemical conditions on the site including in situ ash, suspended ash, and ash leachate. Exposures included exposures to episodic events that could not be included in the aqueous sampling program. Benthic invertebrate surveys were conducted 0.03 mi (0.06 km) above Rogers Quarry (MCK 2.03) in 1989, immediately following the termination of ash disposal in the FCAP, and continued irregularly through the spring of 1993 (Ryon et al. 1992, Sect. 3.2.6 of this RI). Therefore, they do not include the period of exposure to sluice water, but do include exposure to relatively fresh and unweathered ash.

**Sediment**

Sediment-associated biota, including fish and benthic macroinvertebrates, in the FCAP and UMB may be exposed to contaminated sediment and the associated pore water. Sediment-associated biota can absorb or respire interstitial water and can ingest sediment and sediment-associated food. Although fish communities were not found in UMB above and below the earthen dam (Ryon et al. 1992) introduction or future invasion from Rogers Quarry could result in exposure of fish to contaminated sediment. A benthic macroinvertebrate survey of surface water in the FCAP area was not performed. Benthic macroinvertebrate communities were found in UMB (see Sect. 3.2.6), and could invade the FCAP surface water via reproductive flight, resulting in exposure to contaminated sediment. However, the surface water in the FCAP flows over the dam and down a steep bedrock slope, thus preventing the upstream migration of individuals. Therefore, it was assumed that sediment-associated biota received 100% of their exposure to contaminated sediment at either the FCAP or UMB.

Conventional toxicity data, which are the result of one or more methods of assessment, are expressed as sediment concentrations of contaminants causing adverse biological effects. Contaminant concentrations in sediment were available for both the FCAP and UMB, and are discussed in Sect. 6.1.3. Unlike surface water, sediments integrate contaminant concentrations over time, resulting in reduced temporal variability. Also, the maximally exposed sediment-associated organism, which is likely to be less mobile than a water column-associated organism, could receive 100% of its exposure from the most contaminated sediment. Therefore, the maximum detected concentration, instead of the upper UCL95, was used to compare the sediment concentrations to the conventional toxicity data.

The chemical analyses for this assessment were for total sediment-associated contaminants, and did not differentiate between sediment-bound concentrations and pore water concentrations. Although the toxicities of arsenic, chromium, and mercury are dependant upon their chemical species, most of the available toxicity data, and the resulting benchmarks used in this assessment (Hull and Suter 1993), are expressed as the total inorganic concentrations in sediment.

The bio-availability of contaminants in sediment can be affected by physical conditions, including the redox potential, cation exchange capacity, and organic carbon content. Total organic carbon content (Chap. 3) was measured at both areas and used in the equilibrium partitioning formula presented in Hull and Suter (1993) to assess the effects of nonionic organic compounds.

Media toxicity data are the result of controlled tests using contaminated site media. Use of this more ecologically relevant test medium results in weaker evidence of causality. A 4-day, static-renewal, snail (*Elimia clavaeformis*) feeding-rate test was conducted for two sites in UMB. One test used sediment collected at MCK 2.53 and surface water collected at MCK 2.03. Another test used sediment and surface water collected at MCK 2.03. Renewal water was collected daily and diluted with mineral water. Sediment was collected once at the beginning of the test and used to replenish the test chambers daily. Sample water was analyzed for pH, conductivity, alkalinity, and hardness (Sect. 3.2.6).

Biological surveys result in demographic data for the biota naturally occurring at the site. These are the most ecologically relevant data, but the lack of control makes inferences of causality difficult. Benthic macroinvertebrate surveys were conducted at the UMB. Fish community surveys were conducted at both the FCAP and UMB (Ryon et al. 1992, Sect. 3.2.6 this RI).

During Phase 1 quantitative and qualitative benthic macroinvertebrate surveys were conducted at MCK 2.03. One qualitative sampling was conducted in the spring of 1989, and seasonal quantitative samplings were performed from the spring of 1989 to the winter of 1990, resulting in measures of density, biomass, richness, and diversity. Qualitative fish surveys were made of UMB, from the quarry (MCK 1.92) to the earthen dam (MCK 2.65), and of McCoy Branch in the FCAP, to determine the presence or absence of fish communities. Phase 2 surveys consisted of two more quantitative benthic macroinvertebrate surveys at MCK 2.03, one in the summer of 1992 and the other in the spring of 1993.

#### 6.2.2.2 Quantification of exposure of plants, earthworms, other invertebrates, and soil microbes

**Ash**

Vegetation, earthworms, and soil microbes of the Sluice Channel Area, the FCAP, and UMB are exposed to contaminants in the ash. Summary statistics for double acid-extractable (nitric and hydrochloric) concentrations of analytes in the ash given in Sect. 3.2.1 represent the total potential reservoir of elements and compounds, not that to which the organisms are exposed at any one time. Some elements are more readily available for uptake in dissolved form from pore water by plants and soil microbes than others. This depends on the chemical equilibrium established between the ash matrix and pore water. The reported concentrations will be used for assessing soil toxicity to plants, earthworms, and microorganisms because the screening benchmarks are derived from literature values of essentially total (or added) soil metal concentrations.

In the Sluice Channel Area, ash samples were taken from a depth of 0 - 12 in. (0 to 30 cm) along three transects. All samples from a transect were composited and subsamples were used for analyses (Energy Systems 1993c). The reported concentrations from this area are means of values from sampling sites of varying ash depth over soil. In the UMB area, ash

samples were taken from 5 locations at a depth of 0–12 in. (0–30 cm) (Energy Systems 1991). Here also, the depth of ash over soil is unknown. It is not possible to estimate what proportion of the total contaminant exposure of "soil" inhabiting organisms was derived from ash, as opposed to the underlying soil. Therefore, it is assumed that the plants are rooted entirely in the ash and that the total reported amounts of the contaminants are available for uptake by the plants. Likewise, it is assumed that the soil microbial population is functioning entirely in the ash and that the total reported amounts of the contaminants are available for uptake and are effective in relation to microbial enzyme processes. Earthworms may be exposed to contaminants by dermal contact with ash and pore water and by ingestion of ash. It is not possible here to distinguish and quantify the exposure to earthworms by way of these two routes. It is assumed that the earthworm population is living entirely in the ash and that the total reported amounts of the contaminants are effective in relation to potential toxicity by way of the ingestion and dermal contact routes of exposure. These assumptions are entirely applicable to organisms living in the deep ash of the FCAP.

Media toxicity testing gives an indication of the overall effect of coal ash on growth and survival of earthworms. It take into account the physical properties of the ash as well as potentially complex chemical interactions influencing bioavailability and effects of individual contaminants. The use of samples from different locations within the OU can yield information concerning spatial variability at the site with regard to factors affecting these organisms.Toxicity of the coal ash to earthworms was assessed using 14-day in situ and 14-day laboratory tests. In these tests, the earthworm *Eisenia foetida* was exposed to undiluted ash from the Sluice Channel Area (laboratory) and the FCAP (in situ and laboratory). Survival and growth were assessed and compared to that in reference growth media. Test protocols and results are given in Sect. 3.2.6.1. These results will be applied as a line of evidence in the determination of the risk to earthworm populations posed by the ash (Sect. 6.4.1).

The biosurvey of earthworms in ash at Chestnut Ridge OU 2 gives an indication of the long-term effects of the physical and chemical properties of the ash on earthworm populations. Information is also gained concerning spatial variability at the site with regard to factors affecting presence of these organisms. A preliminary survey of soil-dwelling invertebrates in the ash of the FCAP and the Sluice Channel Area is described in Sect. 3.2.6.3. An objective of the survey conducted in July 1993 was to determine the abundance of earthworms present at the site. Five 1-$ft^3$ (28-L) samples taken from the Willow area of the FCAP were hand sorted. Two additional samples of ash from the Sluice Channel Area were also examined. Although no earthworms were found in any sample, the limited number and spatial distribution of the sampling does not allow us to conclude that earthworms are not present in ash or soil at the site. Results of the media toxicity tests suggest that any earthworms present may be negatively impacted by toxicants in the ash.

**Soil**

Vegetation, earthworms, and soil microbes of the Sluice Channel Area and UMB are exposed to contaminants in soil, either beneath ash or in localized areas with no surface ash present. Summary statistics for double acid-extractable (nitric and hydrochloric) concentrations of analytes in the soil given in Sect. 3.2.1 represent the total potential reservoir of elements and compounds, not that to which the organisms are exposed at any one time. Some elements are more readily available for uptake than others. This depends on the chemical equilibrium established between the soil matrix and pore water. The reported concentrations will be used for assessing soil toxicity to plants, earthworms, and microorganisms because the screening

benchmarks are derived from literature values of essentially total (or added) soil metal concentrations.

In the Sluice Channel Area, soil samples were taken from a depth of 0 - 12 in. (0 to 30 cm) below the ash along three transects. All samples from a transect were composited and were used for analyses (Energy Systems 1993c) The reported concentrations from this area are means of values from sampling sites of varying ash depth over soil. In UMB, soil samples were taken from five locations at a depth of 0 - 12 in. (0 - 30 cm) below the ash (Energy Systems 1991). Here also, the depth of ash over soil is unknown.

It is not possible to estimate what proportion of the total contaminant exposure of "soil" inhabiting organisms was derived from soil, as opposed to overlying ash. Therefore, it is assumed plants are rooted entirely in the soil and the total reported amounts of the contaminants are available for uptake by plants. Likewise, it is assumed that the soil microbial population is functioning entirely in soil and the total reported amounts of the contaminants are available for uptake and are effective in relation to microbial enzyme processes.

Earthworms may be exposed to contaminants by dermal contact with soil and pore water, and by ingestion of soil. It is not possible here to distinguish and quantify the exposure to earthworms by way of these two routes. It is assumed that the earthworm population is living entirely in soil and the total reported amounts of the contaminants are effective in relation to potential toxicity by way of the ingestion and dermal contact routes of exposure.

**Vegetation**

The survey of plants growing at Chestnut Ridge OU 2 gives an indication of the long-term effects of the physical and chemical properties of ash on contaminant uptake by vegetation. Plants were sampled in the Sluice Channel Area and in two distinct vegetation zones on the FCAP (Willow and Sycamore areas) to determine total concentrations of As, Cd, Cr, Pb, Hg, Se, and Tl in tree foliage and ground cover species (Sect. 3.2.6.4). Leaves of trees and leaves and stems of grasses were collected. Analyses were conducted on unwashed samples to approximate the contaminant load experienced by consumers. Results of the analyses are given in Table 3.35 of Sect. 3.2.6.4.

Contaminants become available to litter-decomposing invertebrates upon death and abscission of plant parts, and to soil/litter-dwelling organisms feeding on live plants. For this reason, contaminant concentrations in vegetation will be screened against dietary concentrations known to be toxic to invertebrates (e.g., arthropods). It is assumed that the total concentration of a contaminant in the vegetation sample is available to organisms and that their entire food supply comes from this vegetation.

There is no vegetation data from the Sluice Channel Area or UMB. The extremely variable depth of ash in these areas precludes our making the assumption that contaminant concentrations measured in the FCAP vegetation are representative of those in the Sluice Channel Area and UMB. Therefore, no estimation of exposure of soil/litter-dwelling invertebrates in the Sluice Channel Area and UMB will be made.

### Groundwater

Groundwater was sampled in the designated Sluice Channel Area and UMB areas of the OU. Wells in the former are not located in the area of past or present ash deposition but were drilled to assess the possibility of upgradient off-OU contamination entering the OU. Wells in the UMB area have been drilled in three areas: at the toe of the FCAP dam, downstream ~ 800 ft (243 m), and near the point where UMB enters Rogers Quarry (see Sect. 3.2.4). Of the UMB wells, one at each location is drilled to the depth at which samples drawn may represent groundwater from the unconsolidated overburden (Energy Systems 1991). The wells are designated GW-672, GW-674, and GW-676 (see Sect. 3.2.4 for locations). The shallow groundwater system of the UMB was considered to be continuous as these wells are located along a downgradient path between the FCAP and Rogers Quarry. For this reason the data from the three wells were composited for statistical analysis.

The analyses of groundwater samples from these wells may be used as an indication of the contaminant content of soil/ash pore water to which plants and soil-dwelling organisms may be exposed in the FCAP and UMB portions of the site. Although all the wells to be considered were located in UMB, there are actively flowing springs at the toe of the dam indicating that water in the ash behind the dam is entering the shallow groundwater in UMB. Results of the comparison of contaminant concentrations from these wells with solution concentrations known to be toxic to plants will be applied to the FCAP and UMB. It is assumed that the total concentration of a contaminant in the groundwater sample is available to plants and that their entire water supply comes from this groundwater. There are no benchmarks available for assessing the effects of solution contaminants to the other soil endpoints.

## 6.2.2.3 Quantification of exposure to terrestrial wildlife

### Introduction

The primary routes of exposure for terrestrial wildlife species that use hazardous waste sites are through ingestion of food (either plant or animal) and surface water and inhalation. In addition, some species may ingest soil (or ash) incidentally while foraging or grooming or purposefully to meet nutrient needs. The total exposure experienced by terrestrial wildlife is represented by the sum of the exposures from each individual source. In its simplest form, total exposure may be represented by the following generalized equation:

$$E_{total} = E_{food} + E_{water} + E_{air} + E_{soil}$$

where

$E_{total}$ = exposure from all sources,

$E_{food}$ = exposure from food consumption,

$E_{water}$ = exposure from water consumption,

$E_{air}$ = exposure through inhalation,

$E_{soil}$ = exposure through consumption of soil or ash (either incidental or deliberate).

Due to the fine particulate nature of coal ash, particles could be suspended by wind currents and then be inhaled by wildlife. However at Chestnut Ridge OU 2, during dry, summer conditions, suspension of ash is limited because most ash is covered by vegetation or leaf litter. In winter, when vegetative cover is reduced, increased rain and moisture reduces the likelihood of ash being suspended by winds. Because the aerial suspension of ash at the site is expected to be minimal and a review of the literature does not suggest that inhalation of ash poses a significant hazard (Roy et al. 1981), the exposure equation was simplified by excluding $E_{air}$ to read:

$$E_{total} = E_{food} + E_{water} + E_{soil}.$$

Because wildlife are mobile and many species have varied diets, it is unlikely that all food, water, or soil ingested by an individual will be obtained from a contaminated source. Species with large home ranges may spend only a portion of their time on a contaminated site, so their exposure should be represented by the proportion of food, water, or soil obtained from contaminated sources. For these reasons, the exposure parameters in the previous equation should be proportional to the amount of the media consumed from contaminated sources.

For Chestnut Ridge OU 2, contaminant concentrations were available for the following media: ash, surface water, vegetation, and small mammals. These data are described and summarized in Sect. 5.2.3.7., Table J.2 (ash and surface water), Sect. 3.2.6.4. (vegetation), and Sect. 3.2.6.5. (small mammals). With these data, contaminant exposures were estimated for the following terrestrial wildlife species: white-footed mice (*Peromyscus leucopus*), short-tailed shrew (*Blarina brevicauda*), white-tailed deer (*Odocoileus virginianus*), red fox (*Vulpes vulpes*), and red-tailed hawk (*Buteo jamaicensis*).

The area comprised by the components that make up Chestnut Ridge OU 2 (e.g., FCAP, Sluice Channel Area, or UMB) is ~ 15 acres (6 ha) (see Sect. 6.1.2). The reported home range for white-footed mice is 0.2 acre (0.1 ha) (Lackey et al. 1985) and that for short-tailed shrews is 6.2 acres (2.5 ha) (George et al. 1986). Because the home ranges for both small mammal species are smaller than the size of Chestnut Ridge OU 2, only one exposure estimate was calculated for each small mammal species, based on the assumption that 100% of their time was spent on Chestnut Ridge OU 2.

In contrast to mice and shrews, red fox, white-tailed deer, and red-tailed hawks all have large home ranges. Red fox home ranges may vary from 252 acres to 2400 acres (102 ha to 960 ha) (Trewhella et al. 1988); Smith (1991) reports that the home range of white-tailed deer ranges from 150 acres to 1300 acres (59 ha to 520 ha); and Janes (1984) reports the mean territory size for red-tailed hawks to be 576 acres (233 ha). Because the area encompassed by Chestnut Ridge OU 2 is far smaller than the minimum reported home range for these species, it is unlikely that any individuals will reside exclusively within Chestnut Ridge OU 2.

Two exposure estimates were calculated for these wider ranging species, an absolute maximum exposure and a maximum proportional exposure. For the absolute maximum exposure, it was assumed that 100% of each species time (and therefore 100% of their food, water, and ash consumption) was obtained from Chestnut Ridge OU 2.

For the maximum proportional exposure, it was assumed that amount of time (and therefore resources) obtained from Chestnut Ridge OU 2 was proportional to the size of the

site in relation to the minimum reported home range for each species. Since the size of the site is 15 acres (6 ha) and the minimum home range for red fox is 252 acres (102 ha), the proportion of resources obtained from Chestnut Ridge OU 2 was assumed to be: 6 ha/102 ha × 100 = 5.9%. The minimum home range for white-tailed deer is 150 acres (59 ha), therefore, the proportion of resources obtained from Chestnut Ridge OU 2 was assumed to be: 6 ha/59 ha × 100 = 10.2%. For red-tailed hawks with a 576 acres (233 ha) territory, the proportion of resources obtained from Chestnut Ridge OU 2 was assumed to be: 6 ha/233 ha × 100 = 2.3%.

To determine if exposure estimates differed among the different components of Chestnut Ridge OU 2 (e.g., FCAP, Sluice Channel Area, UMB), estimates were calculated separately for each area as data permitted. In addition, exposure estimates were calculated using available reference or background data for comparison. Because it was not possible to associate an area to reference or background data, it was assumed that all species received 100% of their exposure within reference or background areas.

Exposure parameters for each medium were calculated according to the following equation:

$$E_{media} = \frac{\text{Medium Consumption Rate (kg or L/day)} \times \text{Analyte Concentration in Medium (mg/kg or mg/L)}}{\text{Body Weight (kg)}}$$

where $E_{media}$ = estimated exposure (mg analyte/kg body weight/day) for each medium (e.g., food, water, ash, etc.). Body weights for white-footed mice (0.02 kg), short-tailed shrew (0.015 kg), red fox (6 kg), and white-tailed deer (60 kg) were obtained from Opresko et al. (1993). The body weight for red-tailed hawk (1.126 kg) was obtained from Dunning (1984).

As outlined by the EPA (1992), all exposure estimates for terrestrial wildlife were calculated using the upper 95% confidence interval on the mean or the maximum detected value, whichever was the lowest. This is consistent with EPA guidance for human health risk assessment (EPA 1992).

**Exposure to Terrestrial Wildlife Through Consumption of Ash and Soil**

Terrestrial wildlife that use Chestnut Ridge OU 2 may be exposed to contaminants in ash through incidental consumption of ash while foraging or grooming. Routes of exposure for incidental ash consumption are the same as those for soil ingestion. They include consuming ash or soil deposited on foliage or adhering to plant roots, soil adhering to or contained in prey items (i.e., earthworms), soil ingested while burrowing, and soil consumed while grooming paws or fur.

To determine the magnitude of contaminant exposure experienced by terrestrial wildlife through incidental ash consumption, exposure estimates were calculated using published soil ingestion rates and the analyte concentrations in ash from the FCAP, Sluice Channel Area, and UMB. For comparison, exposure estimates were calculated for metals observed in soils from Walker Branch. Exposure estimates were calculated only for those analytes whose concentrations in ash (or soil) exceeded detection limits. All estimates were calculated using the lowest of either the upper 95% confidence interval or the maximum detected value for analyte concentrations in ash (or soil). These values are listed in Table 6.6. Incidental ash

exposure estimates were calculated for the following species: short-tailed shrew, white-footed mouse, red fox, and white-tailed deer. Soil ingestion by red-tailed hawks was considered negligible, and no exposure estimate for ash (or soil) was calculated for this species.

Both short-tailed shrews and white-footed mice have home ranges smaller than the size of Chestnut Ridge OU 2; therefore, it was assumed that 100% of their time was spent on site. Conservative soil ingestion rates for short-tailed shrews and white-footed mice were reported by Talmage and Walton (1993) to be 13% and 1% of daily food consumption, respectively. The daily food consumption rate for white-footed mice was assumed to be equivalent to that for deer mice, *Peromyscus maniculatus*, or 3.36 g food/day (Green and Miller 1987). The food consumption rate for short-tailed shrews has been reported to be 7.95 g food/day (Barrett and Stueck 1976) and 10 g food/day (Buckner 1964). The mean of these values, or 8.975 g food/day, was assumed to be the food consumption rate for shrews. If soil consumption by white-footed mice is 1% of 3.36 g food/day, then mice consume 0.0336 g soil/day. For shrews, if soil consumption is 13% of 8.975 g food/day, then shrews consume 1.17 g soil/day. Exposure estimates for incidental consumption of ash and soil by mice and shrews are displayed in Table 6.7.

Two incidental ash (soil) exposure estimates were calculated for white-tailed deer and red fox: an absolute maximum exposure estimate (Table 6.8) and a maximum proportional exposure estimate (Table 6.9). For the absolute maximum exposure, it was assumed that 100% of the deer or fox diet (and therefore soil or ash ingestion) was obtained from Chestnut Ridge OU 2. The daily food consumption rate for white-tailed deer is 1.74 kg food/day (Mautz et al. 1976) and for fox 0.796 kg food/day (Vogtsberger and Barrett 1973). Beyer et al. (in press) reports that incidental soil ingestion accounts for 2% of white-tailed deer diet and 2.8% of red fox diet. Therefore, under the absolute maximum exposure scenario, soil consumption by white-tailed deer is 2% of 1.74 kg food/day or 0.035 kg soil /day. For red fox, soil consumption is 2.8% of 0.796 kg food/day or 0.0022 kg soil/day.

For the maximum proportional exposure, it was assumed that amount of food (and ash) obtained from Chestnut Ridge OU 2 was proportional to the size of the site in relation to the minimum reported home range for each species. As shown in Sect. 6.2.2.3 (Introduction), Chestnut Ridge OU 2 represents 10.2% of the home range for a white-tailed deer and 5.9% of the range of a fox. Therefore, under the maximum proportional exposure scenario, soil consumption by white-tailed deer is 10.2% of 0.035 kg soil/day or 0.00357 kg soil/day. For red fox, soil consumption is 5.9% of 0.0022 kg soil/day or 0.00013 kg soil/day.

In addition to being exposed to contaminants in ash through incidental consumption, white-tailed deer may be exposed by consuming ash to meet mineral needs. Because most plant material contains very little sodium (Robbins 1993) the diet of many herbivores, including white-tailed deer, is sodium deficient. To compensate for this sodium deficiency, many ungulates locate and consume soils rich in sodium (Weeks 1978). Coal ash in Chestnut Ridge OU 2 is rich in sodium relative to soil in Walker Branch (Table 6.6), and areas where white-tailed deer appear to have consumed ash have been observed on the FCAP (T. Ashwood, personal communication.

Pletscher (1987) estimates that white-tailed deer require 301 mg Na/day of which only 23 mg Na/day is provided in their diet. This leaves a sodium deficit of 278 mg Na/day to be made up from other sources. An exposure estimate was calculated in which it was estimated that a deer would consume the amount of ash sufficient to eliminate the sodium deficit. Since

ash consumption has been reported only for the FCAP, exposure estimates were calculated for this area only. To estimate the amount of ash consumed, it was assumed that deer would locate and consume ash from the most sodium-rich areas; therefore, the maximum observed concentration of sodium (1080 mg/kg) was used to represent sodium concentrations in ash. Ash consumption was estimated to be: (278 mg Na/day)/(1080 mg Na/kg ash) = 0.257 kg ash/day. Exposure estimates for consumption of ash by deer to meet sodium requirements are displayed in Table 6.10.

**Exposure to Terrestrial Wildlife Through Consumption of Surface Water**

All terrestrial wildlife that use Chestnut Ridge OU 2 may be exposed to contaminants in surface water through water consumption. To determine the magnitude of contaminant exposure experienced by terrestrial wildlife through consumption of surface water, exposure estimates were calculated using calculated water consumption rates and the contaminant concentrations in surface water from the FCAP, UMB, and background locations reported in Sect. 5.2.3.7. Exposure estimates were calculated only for those analytes whose concentrations in surface water exceeded detection limits. All estimates were calculated using the lowest of either the upper 95% confidence interval or the maximum detected value for analyte concentrations in surface water. These values are presented in Table 6.11. Surface water exposure estimates were calculated for the following species: short-tailed shrew, white-footed mouse, red fox, white-tailed deer, and red-tailed hawk.

Water consumption rates for all species were calculated according to the allometric equations of Calder and Braun (1983). For mammals the equation was:

$$W = 0.099\ (bw)^{0.90}$$

where W is the water consumed in liters per day and bw is the body weight in kilograms. A similar model was used for birds:

$$W = 0.059\ (bw)^{0.67}$$

Using these equations, the following water consumption rates were calculated:

| | | | |
|---|---|---|---|
| white-footed mouse | 0.00293 L/day | white-tailed deer | 3.944 L/day |
| short-tailed shrew | 0.00226 L/day | red-tailed hawk | 0.064 L/day |
| red fox | 0.497 L/day | | |

Both short-tailed shrews and white-footed mice have home ranges smaller than the size of Chestnut Ridge OU 2; therefore, it was assumed that 100% of their time was spent on-site. Surface water exposure estimates for mice and shrew are presented in Table 6.12.

Two water consumption exposure estimates were calculated for red fox, white-tailed deer, and red-tailed hawks: an absolute maximum exposure estimate (Table 6.13) and a maximum proportional exposure estimate (Table 6.14). For the absolute maximum exposure, it was assumed that 100% of each species water requirement was obtained from Chestnut Ridge OU 2. Therefore, under the absolute maximum exposure scenario, water consumption by red fox, white-tailed deer, and red-tailed hawk was 0.497 L/day, 3.944 L/day, and 0.064 L/day, respectively.

For the maximum proportional exposure, it was assumed that amount of water obtained from Chestnut Ridge OU 2 was proportional to the size of the site in relation to the minimum reported home range for each species. As shown in Sect. 6.2.2.3 (Introduction), Chestnut Ridge OU 2 represents 10.2% of the home range for a white-tailed deer, 5.9% of the range of a fox, and 2.3% of the range of a red-tailed hawk. Therefore, under the maximum proportional exposure scenario, water consumption by red fox is 5.9% of 0.497 L/day or 0.029 L/day; water consumption by white-tailed deer is 10.2% of 3.944 L/day or 0.402 L/day; and water consumption by red-tailed hawks is 2.3% of 0.064 L/day or 0.0015 L/day.

**Exposure of Herbivorous Wildlife Through Consumption of Vegetation**

To determine if elements in coal ash bioaccumulate in plants and would therefore present a risk to herbivorous wildlife, vegetation at Chestnut Ridge OU 2 was sampled and metal content was determined. These data are reported in Table 3.35, Sect. 3.2.6.4. Foliage samples from mixed deciduous trees (species listed in Sect. 3.2.6.4.), eastern red cedar (*Juniperus virginiana*), and groundcover species (*Eulalia viminea* and *Lonicera* spp.) were collected and concentrations of As, Cd, Cr, Pb, Hg, Se, and Tl in foliage were determined. Vegetation samples were collected from two areas on the FCAP (the sycamore or FCAP-1 area and the willow or FCAP-2 area) and the Sluice Channel Area (SLUICE-1). Samples from the Walker Branch watershed were used as a reference. A more detailed discussion of methods and results may be found in Sect. 3.2.6.4.

Vegetation exposure estimates were calculated for the white-footed mouse and white-tailed deer. While as much as 10% of the diet of red fox may consist of plant material, these are usually fruits (persimmon, apple, cherry, and grape; Hockman and Chapman 1983). Because these plants are not known to be found at Chestnut Ridge OU 2 (see Sect. 2.7.1.) it was assumed that red foxes obtained no plant food from the site.

The diet of white-tailed deer consists entirely of plant material, which includes foliage from deciduous trees and groundcovers (Smith 1991). Eastern red cedar is not a preferred food and is rarely consumed by deer (J. Story, personal communication to B. Sample, Jan. 31, 1994), and metal levels were generally not elevated in this species (see Table 3.35, Sect. 3.2.6.4.); therefore, deer are assumed not to be exposed to metals in eastern red cedar foliage.

Two exposure estimates were calculated for white-tailed deer foraging on plants within Chestnut Ridge OU 2, an absolute maximum exposure and a maximum proportional exposure (Table 6.15). The upper 95% confidence interval for metal concentrations in deciduous tree and groundcover foliage was lower than the maximum observed concentration in foliage, therefore both estimates were calculated using the upper 95% confidence interval. Because deer consume both tree and groundcover foliage, these data were combined and confidence intervals were based on the pooled sample. Samples from FCAP-1 and FCAP-2 were also pooled to produce one value for the FCAP. For analytes whose concentrations did not exceed detection limits (i.e., thallium), values for the detection limits were used. For the absolute maximum exposure, it was assumed that 100% of the deer's diet was obtained from Chestnut Ridge OU 2. The daily food consumption rate for white-tailed deer is 1.74 kg food/day (Mautz et al. 1976). For the maximum proportional exposure, it was assumed that amount of food obtained from Chestnut Ridge OU 2 was proportional to the size of the site in relation to the minimum reported white-tailed deer home range. As shown in Sect. 6.2.2.3 (Introduction), Chestnut Ridge OU 2 represents 10.2% of the home range for a white-tailed

deer, therefore, the amount of food consumed by deer in the maximum proportional exposure estimate is 10.2% of 1.74 kg food/day or 0.177 kg food/day. For comparison, exposure estimates were calculated using the 95% confidence interval for data from Walker Branch (Table 6.15) and the assumption that 100% of the diet (1.74 kg food/day) was obtained from the reference site.

Note that the exposure estimates produced by the Absolute Maximum Exposure scenario represent a worst case; due to the relationship between deer home range and the size of Chestnut Ridge OU 2 (discussed above) it is unlikely that individuals will forage exclusively within Chestnut Ridge OU 2 and receive this level of exposure. Exposures estimated using the Maximum Proportional Exposure scenario may provide a more realistic estimate of exposure. However, because white-tailed deer prefer to forage along forest edges and on foliage in forest clearings, and the FCAP provides edge and acts as a forest clearing, actual use of Chestnut Ridge OU 2 by white-tailed deer will be higher than suggested by the proportional size of Chestnut Ridge OU 2 to home range alone. Therefore, actual exposure to white-tailed deer could be higher than that estimated by the Maximum Proportional Exposure scenario.

White-footed mice forage opportunistically, consuming a wide variety of foods. In Virginia, 57% of the diet consisted of arthropods and 34% was plant material (fruit, seeds, and green vegetation; Wolff et al. 1985). In Indiana, diets were ~ 30% arthropods and 67% plant material (Whitaker 1966). The diet of white-footed mice in Illinois was found to be 50% arthropods and 48% seeds, fruit, and other plant material (Batzli 1977). The mean diet composition from these three studies was assumed to represent the diet of mice at Chestnut Ridge OU 2. Therefore, arthropods were assumed to represent 46% and plant material (including seeds, fruit, and foliage) 50% of the diet of mice. The home range for white-footed mice is reported to be ~ 0.2 acre (0.1 ha) (Lackey et al. 1985); therefore, it is very likely that individual mice resident in Chestnut Ridge OU 2 will forage exclusively at the site and obtain 100% of their diet from the site. The daily food consumption rate for white-footed mice was assumed to be equivalent to that for deer mice, *Peromyscus maniculatus*, or 3.36 g food/day (Green and Miller 1987).

Because the vegetation residue data for Chestnut Ridge OU 2 is restricted to metal concentrations in plant foliage, concentrations in other plant material consumed by white-footed mice was estimated. In a study of metal uptake by woody plants grown in fly ash, Scanlon and Duggan (1979) found that the concentrations of As, Hg, Pb, and Se were lower in fruits of cherry olive (*Elaeagnus multiflora ovata*) than in foliage; concentrations of cadmium and chromium were slightly elevated in fruit relative to foliage. Selenium concentrations in fruits of oats (*Avena sativa*) grown in a fly ash landfill were 17% greater than that in the oat straw (Arthur et al. 1992a). In a related study, selenium concentration in corn kernels was approximately twice that in stalks but only two-thirds of that in foliage (Arthur et al. 1992b). These studies suggest that, in general, metal concentrations in seeds and fruits are equal to or less than those in foliage. For this reason, metal concentrations in seeds and fruits from Chestnut Ridge OU 2 were assumed to be equal to those observed in foliage from Chestnut Ridge OU 2.

Exposure estimates for white-footed mice consumption of plant material were calculated as follows. Metal concentrations in seeds and fruit from ground cover and trees was assumed to be equal to those observed in ground cover and tree foliage. Because mice consume seeds, fruit, and foliage from both trees and ground cover, these data were combined and 95%

confidence intervals were based on the pooled sample. Samples from FCAP-1 and FCAP-2 were also pooled to produce one value for the FCAP. For analytes whose concentrations did not exceed detection limits (i.e., thallium), values for the detection limit were used. Plant material (seeds, fruit, and foliage) were assumed to account for 50% of the mouse diet. Therefore, 50% of 3.36 g food/day = 1.68 g plant material/day. Vegetation exposure estimates for white-footed mice are presented in Table 6.16.

In general, the greatest exposure of white-footed mice to metals in their plant foods is predicted for mice at the FCAP (Table 6.16). The only exception is for mercury; the greatest exposure is predicted at Walker Branch.

The primary weakness of these exposure estimates for white-footed mice is the lack of data on the metal concentrations in seeds and fruits. While it was assumed that metal concentrations in seeds and fruits were equal to those in foliage because of minimal information available, it is unknown if this pattern is true for all metal and plant species combinations. It is possible that for some plant-metal combinations, metal concentrations in fruits and seeds may be greater or less than those in the foliage. Our calculations therefore provide the best estimate given the available information.

**Exposure to Vertebrate Predators through Consumption of Small Mammals**

Vertebrate predators may consume small mammals resident on Chestnut Ridge OU 2 and therefore may be exposed to contaminants accumulated by these small mammals. A summary of small mammals collected and metals detected are reported in Sect. 3.2.6.5. Red fox and red-tailed hawks were chosen as representative predators resident on the ORR and likely to forage on Chestnut Ridge OU 2. To determine the magnitude of contaminant exposure experienced by predators through consuming small mammals, exposure estimates were calculated using the upper 95% confidence interval for analyte concentrations in small mammals from the FCAP and the Sluice Channel Area. Data from Walker Branch were used as references. It was assumed that all small mammals are equally likely to be consumed by predators; therefore, data from all small mammal species were pooled for use in the exposure estimates.

Two exposure estimates were calculated for red fox (Table 6.17) and red-tailed hawks (Table 6.18): an absolute maximum exposure and a maximum proportional exposure. For the absolute maximum exposure, it was assumed that 100% of the foxes' or hawks' diet was obtained from Chestnut Ridge OU 2. The daily food consumption rate for fox is 0.796 kg food/day (Vogtsberger and Barrett 1973), of which 68.8% consists of mammals (Hockman and Chapman 1983). Therefore, under the absolute maximum exposure scenario, small mammal consumption by red fox is 68.8% of 0.796 kg food/day or 0.548 kg small mammal/day. Specific data for food consumption by red-tailed hawks was unavailable; therefore, it was assumed to be comparable to that reported for ferruginous hawks (*Buteo regalis,* 0.91 kg food/day; Wakeley 1978), a congeneric species of equivalent body weight ($Mean_{red\text{-}tail}$ = 1.126 kg, $Mean_{ferruginous}$ = 1.145 kg; Dunning 1984). Therefore, it was assumed that red-tailed hawks consume 0.91 kg food/day, of which 78.5% is small mammals (Janes 1984). Thus, for red-tailed hawks, small mammal consumption is 78.5% of 0.91 kg food/day or 0.71 kg small mammal/day. The remaining 21.5% of the hawks diet was assumed to consist of uncontaminated prey.

For the maximum proportional exposure, it was assumed that amount of food obtained from Chestnut Ridge OU 2 was proportional to the size of the site in relation to the minimum reported home range for each species. As shown in Sect. 6.2.2.3 (Introduction), Chestnut Ridge OU 2 represents 5.9% of the range of a fox and 2.3% of the range of a red-tailed hawk. Therefore, under the maximum proportional exposure scenario, small mammal consumption by red fox is 5.9% of 0.548 kg small mammal/day or 0.032 kg small mammal/day. For red-tailed hawks, small mammal consumption is 2.3% of 0.71 kg small mammal/day or 0.016 kg small mammal/day.

**Total Exposure to Terrestrial Wildlife from all Sources**

As stated in Sect. 6.2.2.3 (Introduction), the total exposure experienced by terrestrial wildlife is the sum of the exposure from each source (e.g., food, water, soil). These values were summed for all endpoint species and exposure scenarios and are reported in Tables 6.19-6.26.

These exposure values represent our best estimates given the data available. However, because some information was unavailable, most estimates may be lower than exposures that may be actually experienced by individuals in the field. For example, no data were available on residues in arthropods, which account for ~ 46% of the white-footed mouse diet. Because data were not available to estimate the metal concentrations in the arthropod foods of mice, they were assumed to be uncontaminated. In addition, because there were no water data for the Sluice Channel Area and no vegetation data for UMB, values for mice in these areas must be viewed as partial exposure estimates (Table 6.19).

Because no earthworms or soil invertebrates were collected, exposures calculated for short-tailed shrews (Table 6.20) only represent exposure through incidental soil ingestion and water (except for the Sluice Channel Area, where there were no water data).

Exposure estimates for red fox and white-tailed deer (Tables 6.21-6.24) are complete for the FCAP and Walker Branch (i.e., they use food, water, and soil data). However, water and food (small mammal or vegetation) data were not available for the Sluice Channel Area and UMB estimates, respectively.

Total exposure was tabulated for red-tailed hawks (Tables 6.25 and 6.26). Because soil ingestion was considered to be negligible, exposure from soil was not included for any site. In addition, food and water data were not available from UMB and the Sluice Channel Area, respectively.

### 6.2.3 Occurrence of Contaminants in Coal Ash and Natural Soils

COPCs in the ash and soil at Chestnut Ridge OU 2 have been reported in the literature as occurring in coal ash and natural soils. An understanding of the forms and quantities reported may help place the analytical results for the OU media in perspective and thereby aid in the risk assessment process.

**Ash**

COPCs at Chestnut Ridge OU 2 have been reported in the literature as occurring in coal ash from many locations. The particular characteristics of the coal, combustion process, and

the age of the ash all influence the quantities and forms in which the contaminants are found. Nevertheless, some general statements about their occurrence can be made and may be useful in assessment of the risk to soil-dwelling organisms posed by the ash at Chestnut Ridge OU 2.

Aluminum and iron are major matrix elements as part of amorphous ferro-alumino silicate minerals (El-Mogazi et al. 1988). Manganese is also a major element in coal ash and occurs in an oxidized state. The other COPCs are minor components of ash and occur in a highly water-soluble form associated with the surfaces of small particles [e.g., antimony, arsenic, cadmium, chromium (VI), selenium, and zinc] or associated with the particle core [e.g., chromium (III), and nickel] (Adriano et al. 1980, Babcock et al. 1983). Under aerobic conditions, arsenic will be primarily in the relatively non-toxic arsenic (V) state, although anaerobic conditions can cause conversion to the arsenic (III) state. Barium, cobalt, copper, mercury, and thallium are also noted as minor components of coal ash (Adriano et al. 1980). Copper may be strongly sorbed by iron and manganese oxides. Silver is not noted as a component of fly ash.

Soil

COPCs at Chestnut Ridge OU 2 exist in natural soils in varying quantities and forms. These factors affect the bioavailability of the contaminants and the likelihood that some organisms may become adapted to their presence in the growth medium. An understanding of the natural occurrence of the contaminants will be useful in assessing the risk to soil-dwelling organisms posed by the quantities and forms in which these elements are found in soil and ash at the OU.

**Aluminum.** Aluminum exists in soils in aluminosilicate clays and other minerals, as well as on the exchange complex of clays, and bound in organic complexes. Toxicity is generally a problem only in acid soils and soils very high in Aluminum (Barnhisel and Bertsch 1982).

**Antimony.** Antimony in most soils exists in the Sb(III) form and has low mobility under normal soil conditions (Fergusson 1990).

**Arsenic.** Arsenic occurs in soils mainly as arsenates [arsenic (V)] under toxic conditions. It is strongly sorbed onto clays, iron and aluminum oxides and hydroxides, and organic matter (Fergusson 1990). Arsenic may become more available at more alkaline conditions as iron and aluminum arsenates are replaced by more soluble calcium arsenates. Under moderately reducing conditions, the more available and more toxic arsenite [arsenic (III)] may dominate.

**Barium.** Barium has limited mobility in soils as it is easily precipitated as sulfates and carbonates, concentrated in manganese and iron concretions, and is strongly sorbed to clays (Kabata-Pendias and Pendias 1984). Its availability is increased under acid conditions.

**Cadmium.** Cadmium occurs in oxidized soils in several compounds, and in reduced soils as CdS. It is also adsorbed by aluminum and iron hydrous oxides, clay, and organic matter. The mobile form under acid conditions is $Cd^{2+}$, and under alkaline conditions, an oxyanion (Fergusson 1990). It is relatively immobile in alkaline soils as most is precipitated as CdO and $Cd_3(PO4)_2$.

**Chromium.** Chromium occurs in soils in two trivalent states ($Cr^{3+}$ and $CrO^{2-}$) and two hexavalent states ($Cr_2O_7^{2-}$ and $CrO_4^{2-}$). The trivalent form is relatively benign while the

hexavalent form is relatively toxic. Chromium (III) is dominant under reducing conditions and is sorbed on clays, iron oxides, and organic matter. In neutral, well-aerated soils, chromium (VI) may equal the chromium (III) form. Chromium (VI) anions are less adsorbed at high pH and become very mobile (Kabata-Pendias and Pendias 1984).

**Cobalt.** Cobalt in soils is strongly sorbed on manganese and aluminum oxides. Sorption to manganese oxides increases with increasing pH (Kabata-Pendias and Pendias 1984).

**Copper.** Copper in soil is relatively immobile due its the high affinity for organic matter and inorganic exchange sites (Kabata-Pendias and Pendias 1984). Copper precipitates with sulfide, carbonate, and hydroxide anions.

**Iron.** Iron exists in soils mainly as oxides and hydroxides, as small particles, or associated with particle surfaces. It is also chelated by organic matter. Soluble iron is very low compared with total iron and its availability decreases with increasing pH (Kabata-Pendias and Pendias 1984). Iron is found in association with trace elements and governs their availability.

**Manganese.** Manganese in soils occurs as oxides and hydroxides, as nodules, and as coatings on other soil particles. It is found in association with other trace elements (especially cobalt) and governs their availability (Kabata-Pendias and Pendias 1984). Manganese is more mobile under acid conditions although it may be mobilized under alkaline, poorly aerated conditions.

**Mercury.** Mercury in soil is strongly associated with organic matter and, to a lesser extent, with clay and iron and manganese oxides. Volatile losses may occur, especially under alkaline conditions. Inorganic mercury can be methylated by bacteria to a form with greater availability and toxicity (Kabata-Pendias and Pendias 1984).

**Nickel.** Nickel in soil is sorbed by iron and manganese oxides but relatively easily solubilized. It is also associated with clay and organic matter. Solubility decreases with increasing pH (Kabata-Pendias and Pendias 1984).

**Selenium.** In well-drained mineral soils, selenium occurs as relatively insoluble selenite [selenium (IV)] compounds. Selenites are rapidly and nearly completely fixed by iron hydroxides and oxides. Under more alkaline conditions, selenate [selenium (VI)] is formed that does not form insoluble compounds and, therefore, selenium is available for uptake (Fergusson 1990).

**Silver.** Silver can occur in anionic and cationic forms in soils; however, it is immobile above pH 4. Humic substances absorb and complex silver in soil (Kabata-Pendias and Pendias 1984).

**Thallium.** Thallium in soils is fixed by clays and amorphous iron and aluminum oxides in a manner similar to potassium. It is not very mobile (Kabata-Pendias and Pendias 1984).

**Vanadium.** Vanadium occurs in soils in association with iron oxides, clay, and organic matter. All of these forms are considered to be available for plant uptake. The anionic form is considered more toxic than the cationic form, and it is more prevalent under neutral to alkaline conditions (Kabata-Pendias and Pendias 1984). It is a minor component of fly ash (Adriano et al. 1980).

**Zinc.** Much of the zinc in soils is associated with iron and aluminum oxides and is available to plants. It is also adsorbed by clays and organic soil components. Solubility generally decreases with increasing pH (Kabata-Pendias and Pendias 1984).

## 6.3 EFFECTS ASSESSMENT

Ecological effects assessment involves the identification of known effects of contaminants on receptors using biological survey data, ambient media toxicity tests, and conventional toxicity data. This section discusses the three sources of evidence that will be used in the risk characterization section (Sect. 6.4) to evaluate risks to surface water and sediment-associated biota, terrestrial plants, soil invertebrates, soil microbes, and terrestrial wildlife.

In the conventional toxicity section (Sect. 6.3.1) the types, development, and interpretation of appropriate toxicological benchmarks are discussed and available toxicity data relevant to the endpoint organisms is summarized in toxicity profiles for contaminants of potential concern. Conventional toxicity data consists of published values for toxicity of contaminants to test species. These data are generally not readily useful for ERA. They are used in development of toxicological benchmarks applied in the initial phases of risk assessment to determine if biological effects are likely. By comparing contaminant concentrations measured in media at a site to the benchmarks, the likelihood that contaminants pose a risk can be estimated. Contaminant concentrations or exposure doses are compared to benchmarks to compute hazard quotients used in risk evaluation. These comparisons are contaminant specific.

The media toxicity section (Sect. 6.3.2) describes the results of aquatic and soil toxicity tests performed on ash and water from Chestnut Ridge OU 2 sites, including 7-day static renewal fathead minnow and *Ceriodaphnia* laboratory tests and 4-day feeding rate tests with the snail, *Elimia clavaeformis*, laboratory and in situ earthworm toxicity tests, and a test of the effects of fly ash on the biomass of bacteria involved in symbiotic relationships with leguminous plants. These tests are not contaminant specific, but their results provide direct evidence regarding the toxicity of a combination of contaminants in various media to the test organisms.

The biological survey section (Sect. 6.3.3) discusses the results of benthic macroinvertebrate and soil invertebrate surveys, studies of metal bioaccumulation in plants and small mammals, and a study of the ability of the UMB to support fish populations. The surveys and fish study provide direct evidence of the impact of contaminants at the site on fish and invertebrates. The bioaccumulation data indicate the potential for transfer of contaminants through the food chain and possible risks to higher level consumers.

### 6.3.1 Conventional Toxicity Data

#### 6.3.1.1 Toxicity data for aquatic biota

**Toxicological Benchmarks for Aquatic Biota**

Contaminant concentrations in surface water are compared to the toxicological benchmarks in order to assess the ecological risks posed by the contaminated water. A series of benchmarks of differing conservatism are used. If an upper screening benchmark is

exceeded, this suggests a severe hazard and the need for urgent action; no hazard is suggested if a lower screening benchmark is not exceeded. If an increasing number of benchmarks are exceeded, this constitutes increasing evidence of the need for measurement and assessment.

## 1. Types of Aquatic Benchmarks

The simplest screening benchmarks are toxicity test endpoints. A test endpoint is a statistically-derived numeric summary of the results of a toxicity test. Test endpoints can be calculated in two ways. First, a level of effect can be estimated by fitting a function such as the probit or logit to the concentration-response data to derive a concentration-response model. Then by inverse regression, a concentration that causes a particular level of effect such as the median lethal concentration ($LC_{50}$) can be estimated. Second, hypothesis testing can be used to determine whether each of the tested concentrations caused an effect statistically significantly different from the controls. The lowest concentration causing such an effect is termed the LOEC; the highest concentration for which there were no such effects is termed the no-observed-effect concentration (NOEC). The geometric mean of the LOEC and NOEC is termed the chronic value (CV), and was formerly termed the maximum acceptable toxicant concentration.

Toxicity tests are conventionally divided into acute and chronic tests. Standard acute aquatic toxicity tests are 48 or 96 hours in duration, use juvenile or adult organisms, and the test endpoints are the median lethal concentration ($LC_{50}$) or median effective concentration ($EC_{50}$) for death or some equivalent effect (e.g., immobilization). Standard chronic tests include all or most of the life cycle of the test organisms, and they include observations of growth, deformities, and reproductive success as well as lethality. The standard endpoint for chronic tests is the CV.

Another important distinction is between response-specific and integrative endpoints. Conventionally, NOECs and LOECs are calculated for each response parameter and the results for the most statistically sensitive parameter are reported. Because effects on populations and ecosystems are a result of the integrated effects of the toxicant on all life stages, it makes more sense to integrate the responses in the test when calculating the test endpoint. Integrative endpoints may be simple arithmetic combinations of effects such as proportional mortality across all tested life stages or population parameters derived from simple models such as the intrinsic rate of natural increase, r.

Benchmarks may be combinations of multiple test endpoints. An example is the chronic NAWQC, which are derived from at least eight $LC_{50}$s and three CVs.

Finally, benchmarks may be derived by using mathematical models to simulate an assessment endpoint, a specific environmental characteristic that is valued and is at risk due to the contamination or disturbance that is being assessed (Suter 1989). For example, the population $EC_{20}$ values are the concentrations estimated to correspond to a 20% reduction in recruit abundance for largemouth bass (*Micropterus salmoides*), because production of fish, particularly game fish, is an assessment endpoint for ORR ERAs (Suter et al. 1992).

## 2. Derivation of Aquatic Benchmarks

**Water Quality Criteria.** The NAWQC are ARARs and therefore they provide the basis for the screening benchmarks for contaminants in water. The acute NAWQC are calculated

by the EPA as half the final acute value, which is the fifth percentile of the distribution of 48- to 96-hour $LC_{50}$ values or equivalent $EC_{50}$ values for each criterion chemical (Stephan et al. 1985). The acute NAWQC are intended to correspond to concentrations that would cause less than 50% mortality in 5% of exposed populations in a brief exposure. They may be used as a reasonable upper screening benchmark because waste site assessments are concerned with sublethal effects and continuous exposures, rather than the lethal effects and episodic exposures to which the acute NAWQC is applied. The chronic NAWQC are the final acute values divided by the final acute-chronic ratio (FACR), which is the geometric mean of quotients of at least three $LC_{50}$/CV ratios from tests of different families of aquatic organisms (Stephan et al. 1985). It is intended to prevent significant toxic effects in chronic exposures and is used in this assessment as one possible lower screening benchmark.

NAWQC for several metals are functions of water hardness; the criteria are lower for lower hardness levels. We used the criteria for 100 mg/L hardness reported by the EPA, which is near the lower end of the range of hardness values reported for the ORR. Thus, it is moderately conservative.

EPA's compilations of NAWQC contain values for many chemicals that have no NAWQC listed here (EPA 1986b). EPA reports lowest CVs for those chemicals for which there is not enough data to calculate a criterion but for which there is at least one chronic value. Lowest CVs are treated as a separate category of benchmarks in our compilation. However, if sufficient data were available to calculate a final acute value, and if some chronic values were available but not the three or more needed to calculate a chronic criterion, the chronic criterion was approximated using the final acute value and the geometric mean of the available chronic values.

For particular chemicals, the lower screening benchmark could be lower than the chronic NAWQC for any of the following five reasons. First, the chronic NAWQC are based on a threshold for statistical significance rather than biological significance. In some chronic tests, because of highly variable results, the statistical threshold corresponds to greater than 50% effect on a response parameter (Stephan and Rogers 1985, Suter et al. 1987). Second, not all important responses are included in the subchronic toxicity tests that are used to calculate many chronic NAWQC. In particular, effects on fecundity, which is the most sensitive response parameter on average in fish toxicity tests (Suter et al. 1987), are often not included. Third, the chronic NAWQC are based on the most statistically sensitive of the measured response parameters in each chronic or subchronic test. Therefore, cumulative effects over the life cycle of fish and invertebrates are not considered. Fourth, the NAWQC are set at a level that protects "most species most of the time." A lower screening benchmark should protect all species nearly all of the time. Finally, many of the NAWQC have not been revised since 1980, so they do not incorporate recent data that are included in the calculation of other benchmarks. These concerns are confirmed by the recent finding that nickel concentrations on the ORR below chronic NAWQC nonetheless are toxic to daphnids (Kszos et al. 1992).

**Lowest Chronic Values.** The lowest CVs for fish and invertebrates reported in the literature are potential lower benchmarks. CVs are used to calculate the chronic NAWQC, but the lowest CV may be lower than the chronic NAWQC.

**Estimated Lowest Chronic Values.** Estimated lowest CVs for fish and invertebrates are another set of potential lower benchmarks. Estimated CVs were extrapolated from 96-hour

$LC_{50}$s using equations from Suter et al. (1987) and Suter (1993). The equations are as follows where $LC_{50}$ equals the lowest species mean 96-hour $LC_{50}$ for fish and 48-hour $EC_{50}$ for daphnids, and CV equals the estimated chronic value for that taxon. The 95% prediction interval at the mean is log CV $\pm$ the PI value (95% prediction intervals contain 95% of observations versus 95% confidence intervals which contain the mean with 95% confidence).

Fish CV for a metallic contaminant:

$$\log CV = 0.73 \log LC_{50} - 0.70$$
$$PI = 1.2$$

Fish CV for a nonmetallic contaminant:

$$\log CV = 1.07 \log LC_{50} - 1.51$$
$$PI = 1.5$$

Daphnid CV for a metallic contaminant:

$$\log CV = 0.96 \log LC_{50} - 1.08$$
$$PI = 1.56$$

Daphnid CV for a nonmetallic contaminant:

$$\log CV = 1.11 \log LC_{50} - 1.30$$
$$PI = 1.35$$

**Test $EC_{20}$s.** Another potential lower benchmark is the test $EC_{20}$ for fish, which is defined as the highest tested concentration causing less than 20% reduction in (1) the weight of young fish per initial female fish in a life cycle or partial life cycle test, or (2) the weight of young per egg in an early life-stage test. A similar potential lower benchmark is the test $EC_{20}$ for daphnids, which is the highest tested concentration causing less than 20% reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. (Daphnids include members of the genera *Daphnia, Ceriodaphnia,* and *Simocephalus.*) These benchmarks are intended to be indices of population production. They are equivalent to chronic values in that they are simply a summary of the results of chronic toxicity tests, and in most cases the same test supplied the lowest chronic value and the lowest test $EC_{20}$. However, the test $EC_{20}$s are based on a level of biological effect rather than a level of statistical significance, and they integrate all of the stages of the toxicity test rather than treating each response independently. The 20% figure was chosen as approximately the mean level of effect observed at CVs and as a minimum detectable difference in population characteristics in the field (Suter et al. 1987, Suter 1992).

**Estimated Test $EC_{20}$s.** The estimated test $EC_{20}$ is another potential benchmark. The estimated values were extrapolated from 96-hour $LC_{50}$ values using equations from Suter (1993). The equation for the lowest fish test $EC_{20}$ is as follows where $LC_{50}$ equals the lowest species mean 96-hour $LC_{50}$ for fish and the $EC_{25}$ for weight of juveniles per egg is used as an estimate of the test $EC_{20}$ value. (The difference between 20% and 25% effect is

trivial given the uncertainties in these estimates and the steepness of the concentration-response curves.) The log-scaled 95% prediction interval at the mean is log $EC_{25}$ ± the PI value:

$$\log EC_{25} = 0.90 \log LC_{50} - 0.86$$
$$PI = 1.6$$

**Sensitive Species Test $EC_{20}$s.** The sixth potential benchmark is $EC_{20}$s, adjusted to approximate the fifth percentile of the species sensitivity distribution. It is calculated in the same way as the chronic NAWQC except that test $EC_{20}$s are used in place of CVs, and salt water species were not included. The final acute value for each of the criterion chemicals was divided by the geometric mean of ratios of $LC_{50}$s to $EC_{20}$s. They are referred to as sensitive species test $EC_{20}$s.

**Population $EC_{20}$s.** The last potential benchmark is an estimate of the continuous concentration that would cause a 20% reduction in the recruit abundance of largemouth bass. The method used was described by Barnthouse et al. (1990) and is briefly summarized here. The recruit abundance estimates are generated by a matrix model of a reservoir largemouth bass population (Bartell 1990). The fecundity, hatching success, larval survival, and post-larval survival of the model population are each decremented by a value generated from statistical extrapolation models. For each life stage for which a concentration-response relationship could be calculated, that relationship was adjusted for the relative sensitivity of the test species and the bass. For those life stages with no concentration-response relationship, the relationship was estimated using life stage to life stage extrapolation models, and the taxonomic adjustment was made. However, if the authors of the study reported that life stage was unaffected, the decrement for that life stage was set to zero. If no chronic test data were available, extrapolations from $LC_{50}$s to chronic responses of each life stage were performed. Uncertainties in all of these extrapolations were propagated through the models to generate estimates of uncertainty. For each chemical, each available freshwater fish chronic test was used to parameterize a model run. If no chronic test data were available, each available freshwater fish $LC_{50}$ was used to parameterize a model run.

### 3. Interpretation of Aquatic Benchmarks

The use of total concentrations results in a conservative screening procedure. This level of conservatism eliminates the need for screening against NAWQC advisory criteria. Therefore, the advisory benchmarks were not used for this assessment. The benchmarks used in this assessment are presented in Table 6.27.

Each of the alternative benchmarks has a different interpretation. Because they are ARARs, when NAWQC are exceeded, a regulatory imperative for action is created under CERCLA because they are ARARS. If a CV is exceeded, this indicates that the observed concentration is greater than a concentration dividing statistically insignificant from significant effects in a chronic toxicity test. If a test $EC_{20}$ is exceeded, this indicates that biologically significant effects levels were exceeded in a chronic toxicity test. Exceedence of the sensitive species test $EC_{20}$ indicates that a biologically significant effect level may be exceeded in a sensitive species. Exceedence of a population $EC_{20}$ indicates that a significant reduction in a largemouth bass population could occur. Hence, exceedence of either the acute or chronic NAWQC indicates a need for action. Exceedence of a chronic advisory level implies a low

risk. Exceedence of any of the other benchmarks indicates a risk of real effects which should lead to additional assessment.

As discussed in the introduction, the chronic benchmarks are to be used as lower screening benchmarks. The acute NAWQC and advisory values are to be used as upper screening benchmarks. However, because of their conservatism, exceedence of the acute advisory values cannot be taken to indicate that severe effects are likely to be occurring. If an acute advisory value is exceeded, the assessor should examine the acute values used to generate the advisory value and judge whether in fact severe effects are likely.

All of these benchmarks are based on toxicity tests conducted in the laboratory. They should be compared to concentrations dissolved in ambient water (i.e., filtered through a 0.1-0.45 $\mu$m filter before being extracted with acid or organic solvents) (HECD 1992). Total recoverable concentrations are often reported, rather than dissolved concentrations, because they are required for HHRAs. However, for metals total recoverable concentrations typically include 30-95% particle bound material (HECD 1992). Therefore, total recoverable concentrations should be used for aquatic ERAs only if dissolved concentrations are not available.

Background water concentrations should be used as a check for these benchmarks. Because some of these benchmarks are quite conservative and the measured concentrations in ambient water may include forms not bioavailable, benchmark concentrations may be lower than background water concentrations. If the background concentrations are valid and represent an uncontaminated state, and if the background sites support normal biotic communities, then screening benchmarks lower than the background concentration should not be used.

These benchmarks may also be applied to sediments if pore water concentrations can be measured or estimated. The EPA (OWRS 1989) has recommended that sediment concentrations be converted to pore water concentrations with equilibrium partitioning models so that the NAWQC can be used as sediment quality criteria.

## Toxicity Profiles for COPCs to Aquatic Biota

**Aluminum.** Aluminum can be highly toxic to aquatic biota under some circumstances, but because of its complex environmental chemistry its toxicity is strongly dependent on pH, hardness, and organic matter content. Aluminum is amphoteric, that is, it is more soluble in both acidic and basic solutions than in circumneutral solutions. EPA's water quality criteria for aluminum (EPA 1988a) assume a pH range of 6.5 to 9.0, within which aluminum occurs primarily in the form of monomeric, dimeric, and polymeric hydroxides. Within this range acute toxicities of aluminum reported by EPA range between 1.9 mg/L (*Ceriodaphnia dubia*) and >80 mg/L (midge larvae). Acute toxicities for fish ranged from 3.6 mg/L (juvenile brook trout) to >40 mg/L (juvenile chinook salmon). Chronic toxicities reported by EPA ranged from 0.7 mg/L (*Daphnia magna*) to 3.3 mg/L (fathead minnow). Toxicities to aquatic plants fall within roughly the same range. Toxicities of aluminum in the field may be substantially lower than indicated by total aluminum analysis values because of complexation with humic and fulvic acids. At pH values below 6.5, however, aluminum may be substantially more toxic than indicated by the EPA criteria because low pH favors the formation and solubilization of cationic aluminum ($Al^{+++}$).

**Anthracene.** Anthracene is a PAH. Although substantial mammalian data exist for PAHs, comparatively little aquatic toxicity data are available (EPA 1980d). PAHs toxicity to aquatic organisms is often induced ultraviolet radiation (Holst and Giesy 1989). Anthracene was acutely toxic to a protozoan at 0.0001 mg/L (EPA 1980d) and adversely affected *Daphnia magna* reproduction at 0.0082 mg/L (Holst and giesy 1989). Bioconcentration factors for anthracene range from 200 for a cladoceran to 9200 for a mayfly (Eisler 1987b).

**Arsenic.** Inorganic arsenic occurs as two forms in surface water, arsenic (III) and arsenic (V) (EPA 1985a). The form is dependent on environmental conditions, including Eh, pH, organic content, suspended solids and sediment. Acute toxicity of arsenic (III) to 16 species ranged from 812 μg/L for a cladoceran to 97,000 ug/L for a midge. Although a similar range was observed for arsenic (V), the relative toxicities of each form appears to vary by species: the rainbow trout was slightly more sensitive to arsenic (V) than to arsenic (III), whereas the fathead minnow was nearly twice as sensitive to arsenic (III). Three chronic toxicity values were calculated for arsenic (III) ranging from 914 μg/L to 3026 μg/L. One chronic value (914 μg/L) for arsenic (V) was determined for the fathead minnow. Aquatic plants appear to differ greatly as to their sensitivity to arsenic (III) and arsenic (V). Plant values for arsenic (III) were often comparable to the chronic values for aquatic animals; several plant values for arsenic (V) were lower than the one chronic value for fish. Although arsenic is not readily bioconcentrated by fish, lower forms of aquatic life may accumulate higher arsenic residues than fish.

**Barium.** There is relatively little information of the aquatic toxicity of barium, and none has been found for fish. Barium sulfate and chloride were lethal to yellow crabs (*Cancer anthonyi*) in 7-day exposures at 1000 and 100 mg/L, respectively (MacDonald et al. 1988). Barium inhibited growth of duckweed by 50% at 102 to 400 mg/L in various ambient waters (Wang 1988). Barium inhibited calcification of the freshwater green alga *goeotaenium* at 50 mg/L (Prasad 1984). The state of Illinois has a water quality standard for indigenous aquatic life of 5 mg/L (Wang 1988). Barium is bioconcentrated from water by algae and vascular plants, but is not bioaccumulated through food chains (Moore 1991).

**Bis(2-ethylhexyl)phthalate.** Phthalate esters are a family of chemicals used as plasticizers. Bis(2-ethylhexyl)phthalate, also known as di-2-ethylhexyl phthalate or DEHP, is the most widely used phthalate ester (EPA 1980c). Of the available acute toxicity tests, only the *Daphnia magna* data resulted in an acute range (1.0 - 5.0 mg/L). Calculated $LC_{50}$ values exceeded the highest test concentrations for the midge, scud, and bluegill were >18.0 mg/L, >32.0 mg/L, and >770 mg/L, respectively. Data are available from two chronic studies of bis(2-ethylhexyl)phthalate: rainbow trout and *Daphnia magna* had chronic values of 0.0084 mg/L and < 0.003 mg/L, respectively. Although acute toxicity tests with plants were not conducted using bis(2-ethylhexyl)phthalate, the lowest acute values for the three other phthalate esters ranged from 0.110 mg/L to 85.6 mg/L. The bioconcentration factors for fish and invertebrate species ranged from 54 to 2680.

**Benzo(a)anthracene.** Benzo(a)anthracene is a PAH. Although substantial mammalian data exist for PAHs, comparatively little aquatic toxicity data are available (EPA 1980d). PAHs toxicity to aquatic organisms is often induced ultraviolet radiation (Holst and giesy 1989). Chronic toxicity of benzo(a)anthracene to bluegill occurred at 1.0 mg/L. A bioconcentration factor for a cladoceran of 10,109 was reported in Eisler 1987b.

**Benzo(a)pyrene.** Benzo(a)pyrene is a PAH. Although substantial mammalian data exist for PAHs, comparatively little aquatic toxicity data are available (EPA 1980d). PAHs toxicity to aquatic organisms is often induced ultraviolet radiation (Holst and Giesy 1989). The $LC_{50}$ of benzo(a)pyrene to a sandworm species was >1.0 mg/L. Bioconcentration factors ranged from 0.02 for a teleost to 134,248 for a cladoceran (Eisler 1987b).

**Cobalt.** Comparatively little is known about the toxicity of cobalt (Kimball n.d.). Acute and chronic toxicity tests using fathead minnows and *Daphnia magna* were conducted in relatively hard water. Acute toxicity values for cobalt ranged from 3.61 mg/L for fathead minnow to 6.68 mg/L for *D. magna*. Chronic toxicity values ranged from 0.0011 mg/L for fathead minnow to 0.0051 mg/L for *D. magna*. Cobalt toxicity was decreased by the presence of food during the *D. magna* acute toxicity tests.

**Copper.** The aquatic toxicity of copper is relatively well defined (EPA 1985c). Acute toxicity data are available for 41 genera of freshwater animals. The acute toxicity of copper has been shown in several studies to be a function of water hardness with greater hardness resulting in reduced toxicity. At a hardness of 50 mg/L, the acute median lethal concentration of copper to animals ranges from 0.0167 mg/L to 10.24 mg/L. Chronic values are available for 15 freshwater animal species. They range from 3.87 µg/L fir brook trout to 60.36 µg/L for northern pike. Freshwater fish and invertebrates appear to be approximately equally sensitive to the chronic effects of copper, and aquatic plants appear to be approximately equal in sensitivity to aquatic animals. Copper is a well-regulated essential nutrient and is not appreciably bioaccumulated. Because the toxicity of copper is primarily due to the cupric ion that readily complexes with a many organic and inorganic constituents, copper toxicity may be considerably lower than is indicated by total copper analyses of ambient waters.

**Chromium.** Inorganic chromium occurs as two forms in ambient water, chromium (III) and chromium (VI) (EPA 1985b). Toxicity appears to be quite different for each state and has not been shown to be additive. For this reason trivalent and hexavalent chromium are treated as separate materials. Acute values for chromium (III) are available for 18 animal genera ranging from 2.2 mg/L for a mayfly to 71 mg/L for a caddis fly. Chronic values ranged from <0.044 mg/L for a cladoceran to 1.025 mg/L for the fathead minnow. Acute values for chromium (VI) are available for 27 genera and range from 0.23 mg/L for a cladoceran to 1870 mg/L for a stonefly. Daphnids appear to be especially sensitive to chromium (VI). Chronic values for rainbow trout and brook trout were both 0.264 mg/L, as compared to the chronic value of 1.987 mg/L for the fathead minnow. Both valences of chromium appear to increase in toxicity with decreasing hardness. Green algae are quite sensitive to chromium (VI) and were affected by a concentration of 0.397 mg/L of chromium (III) in soft water. The bioconcentration factor for chromium (VI) was less than three and a bioconcentration factor was not measured for chromium (III).

**Iron.** Iron is the fourth most abundant element in the earth's crust and its concentration in water is dependent upon redox conditions and pH. The most common dissolved inorganic form of iron is iron (II) hydroxide [$Fe(OH)_2^+$] (Dave 1984). Iron is acutely toxic to freshwater fish and invertebrates at concentrations ranging from 0.3 mg/L to 2.0 mg/L (Train 1979 cited in Dave 1984). Reproduction of *Daphnia magna* was stimulated at concentrations up to 0.001 mg/L, and reproduction and survival were inhibited by iron at 0.128 mg/L and 0.256 mg/L, respectively (Dave 1984). Survival of brown trout alevin (swim-up) and eyed eggs was reduced at an average iron concentration of 5.17 mg/L (Geertz-Hansen and Mortensen 1983).

**Lead.** Although lead occurs in a variety of forms in the aquatic environment, the relative toxicities of these forms are not well defined (EPA 1985d). Aquatic criteria are expressed as acid-soluble (dissolved) lead because criteria expressed as total recoverable lead are probably too rigorous. Acute and chronic lead toxicity increases as hardness decreases. Acute toxicity ranged from 0.142 mg/L for an amphipod to 235.9 mg/L for a midge at a hardness of 50 mg/L. Cladocerans have the lowest chronic values (0.012 mg/L in soft water) of the four species tested. Freshwater algae were affected by concentrations above 0.5 mg/L. Bioconcentration was observed in four invertebrate and two fish species with concentration factors ranging from 42 to 1700.

**Manganese.** Comparatively little is known about the toxicity of manganese (Kimball n.d.). Acute and chronic toxicity tests using fathead minnows and *Daphnia magna* were conducted in relatively hard water. Acute toxicity values for manganese ranged from 17.48 mg/L for *D. magna* to 33.8 mg/L for fathead minnow. Chronic toxicity values ranged from <1.1 mg/L for *D. magna* to 1.77 mg/L for fathead minnow. Manganese toxicity was increased by the presence of food during the *D. magna* acute toxicity tests.

**Mercury.** Mercury in the environment commonly occurs as mercury (II) and methylmercury (EPA 1985e). Data are available on the acute toxicity of mercury (II) to 28 genera of freshwater animals and ranged from 0.0022 µg/L for a cladoceran to 2.0 mg/L for a mayfly, stonefly, and caddisfly species. Fish had a similar range of 0.030 mg/L for the guppy to 1.0 mg/L for the mozambique tilapia. Although little data are available for methylmercury, the range of acute toxicity values for rainbow trout and brook trout range from 0.024 mg/L to 0.084 mg/L. Methylmercury is also the most chronically toxic form that was tested: cladoceran and brook trout values were < 0.00007 mg/L. The chronic value for mercury (II) toxicity to *Daphnia magna* was 0.0011 mg/L. Plants are generally less sensitive to both forms of mercury than freshwater animals. Bioconcentration factors for methylmercury range from 4,000 to 85,000. The one available bioconcentration factor for mercury (II) is 4.994.

**Nickel.** Nickel is one of the most common metals in surface water (EPA 1986a). Although nickel can exist in several oxidation states, the divalent cation state predominates and is generally considered the most toxic form. As with many metals the toxicity of nickel increases as hardness decreases. Fish and invertebrates have approximately the same range of sensitivity. Acute values are available for 18 genera and range from 1.1 mg/L for a cladoceran to 43.24 mg/L for a fish. Chronic toxicity data are available for two invertebrates and two fishes: chronic values ranged from 0.0147 mg/L with *Daphnia magna* in soft water to 0.526 mg/L with the fathead minnow in hard water. Concentrations as low as 0.050 mg/L produce significant inhibition in freshwater algae. Nickel bioconcentration factors range from 0.8 for fish muscle to 193 for a cladoceran.

**Pentachlorophenol.** Pentachlorophenol (PCP) is one of the most widely used pesticides (EPA 1980b). PCP toxicity increases considerably with decreasing pH. Fish species, especially the salmonids, appear to be more sensitive than invertebrates. Acute toxicity values for PCP range from 0.034 mg/L for the coho salmon to 2.0 mg/L for a cladoceran. Chronic toxicity to salmonids occurs at concentrations as low as 0.0032 mg/L, whereas chronic values for the fathead minnow and *Daphnia magna* were 0.057 mg/L and 0.24 mg/L, respectively. Acute values for plants ranged from 0.0075 mg/L for algae and 0.80 mg/L for duckweed. Fish rapidly absorb PCP but bioconcentration is relatively low because PCP is rapidly conjugated and excreted.

**Zinc.** Zinc toxicity, which is relatively well defined, increases as either hardness decreases or temperature increases (EPA 1987a). Acute toxicity values are available for 43 species of freshwater animals, ranging from 0.05 mg/L for a cladoceran to 88.96 mg/L for a damselfly, when adjusted to a hardness of 50 mg/L. Chronic values ranged from 0.0467 mg/L for a cladoceran to 0.855 mg/L for a brook trout. Although freshwater plants are often more sensitive to zinc than freshwater animals, the range of toxicity values was much greater. Zinc bioaccumulated in freshwater animal tissue from 51 to 1130 times the water concentration.

### 6.3.1.2 Toxicity data for sediment-associated biota

#### Toxicological Benchmarks for Sediment-Associated Biota

To decide whether a chemical or biological measurement of sediment quality indicates impairment, site-specific data may be compared with benchmarks that indicate whether sediment quality is acceptable. Existing criteria and standards are considered a type of benchmark.

There are three distinct approaches to the development of sediment quality benchmarks (SQBs). These approaches are based on analytical chemistry, toxicity test results, and field survey data. A fourth integrative approach incorporates all three types of data. Regardless of the method, a numeric benchmark results.

Scientific and regulatory guidance concerning development of sediment-quality guidelines is unclear. Hull and Suter (1993) evaluated eight methods for developing sediment quality objectives. Based on the evaluation criteria outlined in their report, three methods were recommended for deriving sediment quality objectives. The methods, uncertainties, and limitations and the recommendations that are relevant to this assessment are presented below.

#### 1. Methods for Benchmark Development

**Inorganics.** The NOAA values for inorganics will be used as benchmarks at the ORR. The majority of data compiled in the NOAA report (Long and Morgan 1991) are from marine and estuarine locations. Where freshwater data were available, they were merged with the marine data and treated equally. The use of the NOAA values for freshwaters is appropriate for two reasons. First, in a California-based study, a statistical test of medians was applied to freshwater and marine acute toxicity data for As, Cd, Cr, Cu, Pb, Hg, Ni, Ag, and Zn and in only one case (Cd) was there a statistically significant difference in the median response of marine and freshwater organisms (Klapow and Lewis 1979). Second, the NOAA values were developed using data from several investigations throughout the United States, and these studies used different approaches to evaluate sediment quality [e.g., toxicity tests, equilibrium partitioning (EqP), Apparent Effects Threshold (AET)]. It is assumed that the use of numerous data and the calculation of percentiles help eliminate the influence of a single (possibly outlier) data point, thereby making the NOAA sediment quality values [Effects Range-Low (ER-L), Effects Range-Median (ER-M), AET] more credible (Long and Morgan 1991).

The ER-L, ER-M, and AET values may be used to help identify sites with the potential to cause adverse biological effects. These are not NOAA criteria or standards and are not intended for use in regulatory decisions or any other similar applications (Long and Morgan 1991). However, EPA Region IV has issued a memorandum (Fox 1991) quoting

these values as suggested guidance for evaluating sediment contamination data. The memorandum stressed that the purpose of these values was to flag contamination levels of concern and that the contaminated sediments of concern should then undergo biological-effects testing. This is one purpose of benchmarks listed in this report.

Note that the NOAA report identifies the chemical concentration above which adverse effects may be first expected and the concentration above which adverse effects always or almost always may be expected. The intent was not to identify only the lowest concentration of contaminants at which an adverse effect had been observed or predicted for any organism (Long and Morgan 1991). The NOAA ER-L, ER-M, and AET values for selected inorganic chemicals are presented in Hull and Suter (1993).

A compilation of other selected sediment criteria values for inorganic chemicals is given in Hull and Suter (1993). Although not a comprehensive list of criteria, those selected were considered some of the most applicable as benchmarks for COPCs in sediments at ORR.

Except for the state of Washington, states have not yet adopted chemical-specific standards for sediments. Washington used the sediment quality triad approach to develop AETs to derive the sediment quality criteria. Note that these standards are overridden if laboratory toxicity tests do not confirm toxic effects (Southerland et al. 1992).

The Ontario Ministry of the Environment (MOE) has prepared provincial sediment quality guidelines using the screening level concentration (SLC) approach. These values are based on Ontario sediments and benthic species from a wide range of geographical areas within the province (Persaud et al. 1990). The lowest effect level (low) is the level at which actual ecotoxic effects become apparent. The severe effect level (severe) represents contaminant levels that could potentially eliminate most of the benthic organisms (Persaud et al. 1990). These "low" and "severe" effect values are potential lower and upper benchmarks, respectively.

Beak Consultants in Canada developed potential sediment guidelines for Ontario MOE using a variety of approaches. For the majority of metals, the background value was used as the guideline value. The SLC method was used for arsenic and nickel, and toxicity test methods were used for chromium and copper (Hart et al. 1988 cited in Geisy and Hoke 1990).

The Wisconsin Department of Natural Resources (WI DNR) has developed sediment quality criteria (Sullivan et al. 1985 cited in Geisy and Hoke 1990). These values were derived using the background approach and were based on dredge material suitability for in-water disposal (WADOE 1991).

The EPA Region V Sediment Classification guidelines are for the classification of sediments of Great Lakes Harbors. The "polluted" values listed in Hull and Suter (1993) are qualified as being "moderately polluted" in the guidelines. Any value greater than the "moderately polluted" value was classified as "heavily polluted." These values were extracted from Anon. (1977) cited in Geisy and Hoke (1990). The EPA Region V values are considered arbitrary and not well founded scientifically (WADOE 1991). They were considered adequate only for determining the suitability of dredged material for open water disposal. However, the mercury guideline is strictly adhered to, and sediment with concentrations greater than

the guideline are considered severely polluted and have to be disposed of by means other than open water disposal (WADOE 1991).

**Nonionic Organics.** EPA has chosen the EqP approach for developing sediment quality criteria for nonionic organics (EPA 1989a). This is also a methodology that ORNL supports for developing SQBs when bulk sediment concentrations and water quality benchmarks (WQBs) are available.

The EqP approach requires certain information for the development of the benchmark. The WQB is paramount to the calculation. Because many chemicals do not have NAWQC, sets of WQBs of varying conservatism have been developed at ORNL (Suter et al. 1992b).

An SQB using this method is calculated as follows (EPA 1989a): the sediment quality benchmark, SQB ($\mu$g/kg sediment), is computed using the partition coefficient $K_p$ (L/kg sediment) between sediment and water and the WQB ($\mu$g/L) for the chemical of interest (Suter et al. 1992b):

$$SQB = K_p \times WQB .$$

The partitioning of nonionic chemicals between particles and water depends on the partition coefficient $K_{oc}$ for the particles' organic carbon and the mass fraction of organic carbon $f_{oc}$ (kilograms organic carbon per kilogram sediment) of the particles:

$$K_p = f_{oc} \times K_{oc} .$$

Where the $K_{oc}$ is unavailable, it is estimated by the octanol-water partition coefficient, $K_{ow}$, of the chemical for sediments (Di Toro 1985). Therefore,

$$SQB = f_{oc} \times K_{ow} \times WQB .$$

EqP can be used only if $f_{oc} > 0.5\%$. At $f_{oc} < 0.5\%$, the factors controlling second-order effects on partitioning (e.g., particle size, sorption to nonorganic mineral fractions) become relatively more important (EPA 1989a).

The EqP approach makes three assumptions: pore water is the toxic component of the sediment to benthic organisms, benthic organisms respond to pore water contaminant concentrations similarly to the response of water column biota to water column contaminant concentrations, and there is an equilibrium between the dissolved bioavailable and particulate-bound forms of the contaminant (Maughan 1993).

For both the direct and indirect approaches for estimating pore water concentrations, it is assumed that the WQBs, when applied to the interstitial water of sediments, would protect infaunal organisms. EPA (1989) has concluded that the sensitivities of benthic species are sufficiently similar to those of water column species to tentatively permit the use of WQBs for the derivation of SQBs.

The EqP approach is favored by the EPA over the direct measuring of pore water approach (EPA 1989a). The free chemical concentration in pore water can be estimated directly from the organic carbon normalized sediment concentration, and the estimate is independent of the dissolved organic carbon (DOC) concentration. Using the pore water

chemical concentration to estimate the free pore water chemical concentration requires that the DOC concentration and the DOC partition coefficient be known because the proportion of a chemical in pore water that is complexed to DOC can be substantial. However, it is the free, uncomplexed component that is bioavailable and in equilibrium with the OC normalized sediment concentration. Therefore, for highly hydrophobic chemicals and where there is significant DOC complexing, the solid-phase chemical concentration gives a more direct estimate of the bioavailable pore water contaminant concentration than do the pore water concentrations (EPA 1989a).

After the sediment benchmarks have been calculated, comparisons can be made to site sediment contaminant concentrations. The data required for this comparison include the bulk sediment contaminant concentration and the fraction organic carbon ($f_{oc}$) of the sediment. No other measurements are required.

**Polar Ionic Organic Compounds.** Polar organic compounds include methyl and thiocarbamates, triazines, amines and analines, and organic acid pesticides (aliphatic and aromatic acids and esters, phenoxy compounds, and ureas). Unlike nonpolar and nonionic organic contaminants, both polar and ionic organic compounds may adsorb onto sediments by a variety of mechanisms, including hydrophobic interaction, nonspecific ion association, ion exchange, ion-dipole interactions, hydrogen bonding, and complex formation by surface metals (Shea 1988). It is possible that a multiple-term model might account for polar organic partitioning between sediment and aqueous phases, but currently such a model does not exist (Shea 1988).

The behavior of ionic organic pollutants has not been extensively studied. As with the nonionic organic chemicals, OC appears to be a critical factor in the partitioning behavior in sediments (Jafvert 1990). The critical micelle concentration (Di Toro et al. 1990) and pH (Jafvert 1990) also appear to be dominating factors.

The state of Washington has developed sediment quality standards for some ionic organic compounds. These are listed in Table 6.28. NOAA did not calculate ER-L or ER-M values for polar ionic organic chemicals (Long and Morgan 1991).

### 2. Limitations of Sediment Benchmarks

The EqP methodology relies on an empirical model to compute the pore water concentration from the solid phase measurements. There is an uncertainty associated with the use of the model. Also, there is uncertainty with respect to the $K_{ow}$ associated with the specific chemical because it is an experimentally determined quantity (EPA 1989a). Various types of organic matter present in sediments can have significantly different binding capacities for organic contaminants; the affinity depends in large part on the source and nature of the carbon. For example, organics associated with sediments contaminated with petroleum hydrocarbons would tend to be much less toxic than those associated with sediments whose OC is natural OC (Lee and Jones-Lee 1993).

Uncertainty associated with the assumption that benthic organisms have similar sensitivities to water column species (EPA 1989a) may be of particular concern for tube-dwelling amphipods. The tubes tend to isolate them from the interstitial water, causing speculation that their exposure is at the sediment/water interface. Also, the amphipod tubes are matrices of organics and inorganics; the tube walls could sorb appreciable amounts of

organic contaminants, which could alter the availability of sediment-associated contaminants to those organisms (Lee and Jones-Lee 1993).

The EqP approach is known not to work for all nonpolar organics. For example, Nebeker et al. (1989) reported that the organic carbon content of endrin-spiked sediment had little apparent effect on toxicity. It is well known that many pesticides that are sorbed onto soils and sediments are in the form of "bound" pesticide residues that do not participate in equilibrium reactions with water (Lee and Jones-Lee 1993).

The aquatic benchmarks for PCBs and several PAHs are class criteria based on the cumulative concentration of all members of the class. In the derivation of sediment benchmarks using the EqP approach and the aquatic benchmarks, it is necessary to apply the class level to each member of the class individually because each has a unique $K_{oc}$ (Lake et al. 1990). In environments where one class member comprises the majority of the sediment burden of the class, this approach should be adequate. However, if numerous class constituents are significantly enriched, a safe threshold for the class as a whole may be exceeded even though no individual constituent violates its predicted safe level (based on the aquatic benchmark) (Pavlou 1987).

The Washington state AET and NOAA ER-L, ER-M, and AET values have several limitations. Primarily, all or most of the data used in their derivation were based on marine and estuarine systems. These values are being applied to freshwater systems at the ORR. Differences include physico-chemical characteristics of the system as well as possible differences in sensitivity of biota. Washington state and NOAA values are for single chemicals, although sediments containing chemical mixtures were used for their derivation. The Ontario MOE values were derived to be applicable to sediment types throughout the province of Ontario. The differences between Ontario and East Tennessee sediments and biota introduce a level of uncertainty.

### 3. Recommended Sediment Benchmarks

**Metals.** In accordance with a memorandum from EPA Region IV (Fox 1991), NOAA ER-L values will be used as lower screening benchmarks to flag contaminant levels of concern unless there is evidence that these ER-Ls may be too high. The ER-M is an upper benchmark. Chemical concentrations exceeding the ER-M (median of the effects data) will negatively impact some organisms. There is a high degree of confidence in the accuracy of the ER-L and ER-M data when the NOAA AET is within the ER-L to ER-M range (Long and Morgan 1991). The AET values are upper benchmarks; concentrations above the AET always would be expected to cause adverse biological effects. The lower and upper benchmarks used for this assessment are presented in Table 6.29.

The Ontario MOE "low" values are very low compared with the other benchmarks for As, Cd, Cr, and Cu. It would be advantageous to compare these values with background concentrations and detection limits. For Pb, Hg, Ni, and Zn, the MOE "low" values approximate the ER-L. Therefore, these lend support to the use of the NOAA ER-Ls as lower benchmarks. The "low" value for cadmium is an important benchmark because freshwater organisms are, in general, more sensitive to cadmium than marine organisms (Klapow and Lewis 1979).

The Ontario MOE "severe" values generally agree with AET and ER-M values except for arsenic and copper, where the "severe" value is closer to the ER-L. In both of these cases, only a factor of 2 separates the ER-L and ER-M. Thus, the ER-L should act as a flag for potential concern for these chemicals.

Beak guidelines are another set of lower benchmarks. These were developed for use in freshwater systems as broad-based guidelines for Ontario. Because Beak primarily used background concentrations as their guidelines, a comparison with local site-specific background values could provide information regarding the validity of using both the Beak sediment quality guidelines and Ontario MOE values.

Recommended benchmark values for individual metals are discussed in the following paragraphs.

**Antimony.** Data are available from only two geographic regions (Puget Sound/Commencement Bay and San Francisco Bay), and hence the degree of confidence in the ER-L and ER-M values is moderate (Long and Morgan 1991). However, no criteria are available for comparison from Beak, Ontario MOE, WI DNR, or EPA Region V. Therefore, the ER-L and ER-M will be used as lower and upper benchmarks, respectively.

**Arsenic.** Confidence in the ER-L is relatively poor; confidence in the ER-M is moderate (Long and Morgan 1991). Therefore, the other benchmarks are important, especially because Ontario MOE's "severe" level is equal to the ER-L. The ER-L may be more appropriate as an upper benchmark. The Ontario MOE, EPA Region V, and Beak values will be used as lower benchmarks and the ER-L as an upper benchmark.

**Barium.** Very little information is available for barium; therefore, the 20-mg/kg benchmark should be used with caution.

**Cadmium.** A relatively large amount of data exists for cadmium. The degree of confidence in the ER-L and ER-M values is high (Long and Morgan 1991). However, the ER-L is equal to the NOAA AET. Also, Klapow and Lewis (1979) calculated a statistically significant difference in the medians of acute toxicity data from saltwater and freshwater organisms. This supports the findings of Eisler (1985a), who found resistance to cadmium higher among marine than freshwater species. Therefore, use the freshwater values of Ontario MOE, Beak, and WI DNR as lower benchmarks.

**Chromium.** There are some inconsistencies in the Long and Morgan (1991) data available for chromium, possibly due to lack of speciation information. All data were reported as total chromium, whereas the hexavalent form is more toxic than the trivalent form. There also are no supporting data from single-chemical spiked-sediment toxicity tests or from the EqP approach (Long and Morgan 1991). The Ontario "low" and EPA Region V "nonpolluted" values are less than the ER-L. The EPA Region V "polluted," WI DNR, Beak, and ER-L values are all approximately equal. Therefore, it is unclear which concentration represents a safe level. Use the ER-L as the lower benchmark for screening.

**Copper.** Considerable data exist for copper in sediments. A high degree of confidence exists for the ER-L and ER-M values (Long and Morgan 1991). However, concentrations of about 20 mg/kg induce sublethal behavioral effects when copper is not tightly chelated or bound to sediments. In one study analyzed by Long and Morgan (1991), the minimum

concentration associated with toxic effects was 20 mg/kg. However, EPA classifies copper in sediments below 25 mg/kg as nonpolluted. Because of these inconsistencies, concentrations exceeding the ER-L would cause copper to be retained as a contaminant of concern.

**Cyanide.** Very little information exists relating sediment concentrations of cyanide to toxic effects. Because WQC and other benchmarks (Suter et al. 1992b) are available for cyanide, sediment pore water concentrations should be analyzed and compared to those benchmarks.

**Iron.** There is inconsistency between the various benchmarks for iron. EPA Region V lists 1.7 to 2.5% concentration as being polluted, Ontario MOE lists 3% as the "low" effect level and 4% as a "severe" effect level, and Beak lists 5.9% as background. Because Ontario MOE derived its guidelines for freshwater sediments, the 3% level will act as the lower benchmark.

**Lead.** A relatively large amount of data exist for lead. Confidence in the ER-L and ER-M values is moderate and high, respectively. Effects were usually seen at concentrations >110 mg/kg, but the AET value is above the ER-L to ER-M range. This may be because the chemical data were not speciated to indicate the proportion of organic or inorganic lead, and there were no spiked-sediment toxicity test data to confirm the toxic concentrations (Long and Morgan 1991). The EPA Region V "nonpolluted," Ontario "low," Beak, and WI DNR values are similar to the ER-L. This adds confidence to the use of 35 mg/kg as the lower screening benchmark. Concentrations greater than that would result in the need for further investigation.

**Manganese.** There is no information for manganese in Long and Morgan (1991). The Ontario "low" value will be used as the lower screening criterion.

**Mercury.** A moderate amount of data exist for mercury. The confidence in the ER-L and ER-M values should be moderate and high, respectively. Chronic effects are predicted by EqP principles to occur at 0.032 mg/kg, which is considerably lower than the ER-L (Long and Morgan 1991). The WI DNR and Ontario "low" values are similar to the ER-L and add confidence to the use of the ER-L as the lower screening benchmark.

**Nickel.** Toxicity of nickel is greatly influenced by water hardness and salinity. The degree of confidence in the ER-L and ER-M is moderate. Data were only from field studies; no spiked-sediment toxicity tests or EqP approaches were used (Long and Morgan 1991). No overall effects threshold was apparent in the data compiled by NOAA. Concentrations greater than the ER-L would require further investigation; this value surpasses the Beak, Ontario "low," and EPA Region V "nonpolluted" values.

**Silver.** Only a small amount of data is available for silver in sediments. The ER-L and ER-M values hold moderate confidence. There are no data from spiked-sediment toxicity tests or from EqP approaches (Long and Morgan 1991). No criteria are available from Beak, Ontario MOE, WI DNR, or EPA Region V. Use the ER-L as the lower benchmark.

**Zinc.** Biological effects have not been observed in association with zinc concentrations of 50 mg/kg or less in sediments (Long and Morgan 1991). The data strongly suggest that sublethal and other sensitive measures of effects may occur at zinc concentrations of between 50 and 125 mg/kg. Confidence in the ER-L and ER-M values is high (Long and

Morgan 1991). The ER-L value is consistent with the Ontario "low," WI DNR, and EPA Region V values; therefore, the ER-L is the lower benchmark.

**Nonionic Organics.** The EqP approach will be used to determine benchmarks for nonionic organics and requires the use of WQBs that were developed by Suter et al. (1992b). Consult this publication for a complete discussion of the aquatic benchmarks and their uses. The benchmarks calculated for FCAP and UMB are presented in Table 6.30.

Sediment benchmarks should be calculated using the EqP approach for every water benchmark available. Each of the benchmarks has a different interpretation (Suter et al. 1992b). Concentrations that exceed the sediment benchmark calculated using the chronic advisory level indicate a low risk. If concentrations exceed benchmarks that used the NAWQC, the chemicals must be contaminants of concern because the NAWQC are ARARs. Concentrations that exceed any of the other benchmarks indicate a risk of real effects. Contaminants should be selected on the basis of the number of benchmarks exceeded and the conservatism of the particular benchmark values, as discussed in Suter et al. (1992b).

The ER-Ls presented in Long and Morgan (1991) were also used to screen the non-polar organics. These values represent a lower screening criteria and are presented in Table 6.31.

**Ionic Organics.** Because very little information is available for ionic organics, these contaminants should not be eliminated in a screening risk assessment. Preliminary comparisons can be made to the Washington state sediment quality standards (Table 6.31) to give an indication of the magnitude of the contamination.

**Recommended Application of Benchmarks**

If a chemical concentration or the reported detection limit exceeds a proposed lower benchmark, more analysis is needed to determine the hazards posed by that chemical. If, however, the chemical concentration falls below the lower benchmark value, the chemical may be eliminated from further study.

Comparison of site contaminant levels with background levels is a simple screening method. The assumption is that concentrations that are lower than background are not hazardous. Appropriate background samples must be obtained for waste site samples. The American Society for Testing and Materials (ASTM) is currently developing guidelines for selection of sediment and soil background sampling locations (ASTM Sect. E47.13.01, Task E).

This approach has two major disadvantages: it has no biological effects basis, and it cannot be used for synthetic organic compounds, which should not be present in background sediments. Therefore, this approach should not be used as the only screening method. It is appropriate to use the background concentrations to screen the other sediment benchmarks. Sediment benchmarks that are below background contaminant concentrations are not retained as benchmarks when screening site sediments.

To evaluate ecological effects of contaminated sediments for a baseline ERA, it is recommended that sediment be collected for toxicity testing. This is important because chemical concentrations are not accurate predictors of biological and ecological effects. This

is because the percentage of the chemical that is bioavailable may range from 0 to 100% (Burton and Scott 1992). Benchmarks may be used to determine which chemicals present in the sediment are most likely causing the toxicity. Use of a weight-of-evidence approach enables a more accurate evaluation of adverse ecological impact.

ASTM has approved standard methods for conducting whole-sediment toxicity tests with the freshwater invertebrates *Hyalella azteca* (an amphipod) and *Chironomus tentans* and *C. riparius* (both midges) (Ingersoll and Nelson 1990) and with zooplankton *Daphnia* sp. and *Ceriodaphnia* sp. (both cladocerans). A draft method exists for the burrowing mayfly *Hexagenia limbata* (Ingersoll 1991 cited in Burton and Scott 1992). Based on a number of sediment contamination studies, the most consistently efficient indicators of acute and short-term chronic toxicity in whole sediments are these aforementioned organisms (Burton and Scott 1992).

### Toxicity Profiles for COPCs to Sediment-associated Biota

**Antimony.** Data available for antimony from Long and Morgan (1991) are from only two marine systems: San Francisco Bay and Puget Sound. The AET value for Puget Sound ranged from 3.2 mg/kg, based on benthic community composition, to 200 mg/kg, based on a R. abronium amphipod bioassay.

**Arsenic.** Sediments of Waukegan Harbor, Wisconsin, which had <47 mg/kg arsenic, caused 34% mortality in H. azteca. Sediment from the Sheboygan River, Wisconsin, which had 2.7 mg/kg arsenic, caused significant mortality to the freshwater prawn, Macrobrachium rosenbergii (Long and Morgan 1991). However, this sediment was also contaminated with metals and PCBs. At two rivers in Illinois, a low number of benthic macroinvertebrate taxa was associated with an arsenic concentration of 7.4 and 3.7 mg/kg. However, a high number of taxa were associated with 5.9 and 5.0 mg/kg arsenic in the same two rivers (Long and Morgan 1991).

**Cadmium.** Fertilized eggs of the goldfish, leopard frog, and largemouth bass were placed in natural stream sediment spiked with cadmium (Francis et al. 1984) and maintained until 4 days posthatching, which resulted in a total exposure duration of 6-7 days. Sediments with 1000 mg/kg cadmium were not toxic to goldfish or leopard frog, but resulted in 25% mortality 4 days posthatching for largemouth bass. A sediment concentration of 100 mg/kg cadmium was not toxic to the largemouth bass. Organic matter content of the sediment was 2.3%.

Spiked sediment toxicity tests were performed using sediment from Steele's Run (organic carbon content of 2.3%), a stream in the Kentucky River drainage system. Treatment was of early life stage rainbow trout, in which the test was initiated at the early eyed-egg stage and continued through 10 days posthatching, giving an exposure period of 20 days. Sediment concentrations of between 2 and 13 mg/kg resulted in between 47 and 63% mortality relative to controls. A concentration of 121 mg/kg resulted in 89% mortality (Birge et al. 1979).

An Illinois harbor containing 2.5 mg/kg cadmium resulted in 34% mortality of H. azteca (Long and Morgan 1991). A concentration of 1.98 mg/kg caused 95% mortality to H. azteca in a Washington lake (Long and Morgan 1991). Significant mortality to the prawn M. rosenbergii resulted from 2.8 mg/kg cadmium in a Wisconsin river.

Confidence in the Long and Morgan (1991) ER-L and ER-M is very high, because data are available from many approaches, from multiple methods for some approaches, and they are relatively consistent (Long and Morgan 1991). However, resistance to cadmium is higher among marine species than among freshwater species (Eisler 1985a).

**Mercury.** Spiked sediment toxicity tests were conducted using sediment from Steele's Run (organic carbon content of 2.3%), a stream in the Kentucky River drainage system. Treatment was of early life stage rainbow trout in which the test was initiated at the early eyed-egg stage and continued through 10 days posthatching, giving an exposure period of 20 days. Sediment concentrations of between 1 and 12 mg/kg resulted in between 55 and 77% mortality relative to controls. A concentration of 107 mg/kg resulted in 100% mortality (Birge et al. 1979).

Sediments from the Little Grizzly Creek system in California which had 0.8 to 3.5 (mean 1.5) mg/kg mercury and an organic carbon content of approximately 1% caused >90% mortality in D. magna and H. limbata (Maleug et al. 1984). A Michigan lake sediment in the same study, with an organic carbon content of 2.2% and mercury concentration of 0.29 mg/kg caused 100% mortality in D. magna and 40% mortality in H. limbata. However, sediments from the Phillips Chain of Lakes in Wisconsin, with mercury concentrations from 0.8 to 9.4 mg/kg resulted in <20% mortality in both species. These sediments had a high organic carbon content of approximately 15%.

Sediments from Waukegan Harbor, Illinois, with a mercury concentration of 0.1 mg/kg resulted in 34% mortality to H. azteca (Long and Morgan 1991). A study of benthic macroinvertebrate communities in Illinois rivers conducted by the Illinois EPA (Long and Morgan 1991) found that the lowest number of benthic macroinvertebrate taxa occurred at a site with 1.6 mg/kg mercury, while the highest number of taxa occurred at a site with 0.3 mg/kg mercury, indicating that factors other than the mercury concentration along were affecting the benthic community.

**Nickel.** There was no information from spiked sediment toxicity tests for nickel; data are only from field studies (Long and Morgan 1991). In sediments of Lake Union, Washington, 88 mg/kg resulted in 95% mortality of H. azteca (Long and Morgan 1991). Sediments with between 22 and 63 (mean of 40) mg/kg nickel from the Little Grizzly Creek system in California caused >90% mortality in D. magna and H. limbata (Maleug et al. 1984). This lake had an organic carbon content of 1%. A Michigan lake sediment in the same study, with an organic carbon content of 2.2% and nickel concentration of 150 mg/kg caused 100% mortality in D. magna and 40% mortality in H. limbata. However, sediments from the Phillips Chain of Lakes in Wisconsin, with a high organic carbon content of approximately 15% and nickel concentrations from 140 to 350 mg/kg resulted in <20% mortality in both species.

**Silver.** Effects were not observed in association with silver concentrations less then approximately 0.6 mg/kg (Long and Morgan 1991). No data were from freshwater systems. At concentrations of 2.1 to 2.6 mg/kg in marine systems, there was an absence of feral Macoma balthica in Fraser River sediments, low arthropod abundance and low species richness in southern California benthos, and increased burrowing time of M. balthica exposed to Strait of Georgia sediments (studies cited in Long and Morgan 1991).

**Selenium.** Although insoluble elemental selenium and metal selenides [selenium (II)] are poorly assimilated by aquatic organisms, selenate [selenium (VI)] and selenite [selenium (IV)] are readily taken up. Of the two, selenite is more toxic. Organic selenium is approximately

10 times more toxic than inorganic forms. Chronic toxicity studies on invertebrates are lacking (Masscheleyn and Patrick 1993). Hyalella is the most sensitive genus of those for which there were data for the calculation of the NAWQC.

**Acenaphthene.** In the studies used to calculate the ER-L, effects were usually observed in association with concentrations of 0.15 mg/kg or greater (Long and Morgan 1991). The AET (concentration at which adverse effects are always expected to occur) for Puget Sound sediments, derived from the Microtox bioassay, the benthic community composition, oyster larvae bioassay, and R. abronius amphipod bioassay were 0.5, 0.5, 0.5, and 0.63 mg/kg, respectively, in a 1986 study and 0.5, 0.73, 0.5, and 2.0 mg/kg, respectively in a 1988 study (Long and Morgan 1991). Confidence in the ER-L is low (Long and Morgan 1991). The EPA sediment quality criterion assuming 1% organic carbon is 1.3 mg/kg (EPA 1993c).

**Anthracene.** In the studies used to calculate the ER-L, effects are almost always observed with concentrations exceeding 0.3 mg/kg (Long and Morgan 1991). Confidence in the ER-L is low because there is no consistency in the effects observed at lower concentrations (Long and Morgan 1991) .

**Benzo(a)Anthracene.** In bioassays of the lower Columbia River sediments, no toxicity to H. azteca was observed at concentrations of 2.2 mg/kg. However, these data were not used in the calculation of the ER-L (Long and Morgan 1991). In the studies used to calculate the ER-L, no effects were observed at concentrations below 0.06 mg/kg. The overall AET from these data was 0.55 mg/kg, although AET values in the data base ranged from 1.3 to 5.1 mg/kg for Puget Sound to 0.06 to 1.1 mg/kg for San Francisco Bay. Confidence in the ER-L is low.

**Chrysene.** In freshwater systems, a concentration of 4.1 mg/kg was not toxic to H. azteca in the Columbia River but a concentration of 170 mg/kg in Lake Union, Washington, resulted in 95% mortality to H. azteca (Long and Morgan 1991). However, these data were not used to calculate the ER-L (Long and Morgan 1991). In the studies used to calculate the ER-L, effects were almost always observed at concentrations exceeding 0.9 mg/kg, although the lowest Puget Sound and San Francisco Bay AET values were 1.4 and 1.7 mg/kg, respectively. Confidence in the ER-L is moderate.

**Dibenz(a,h)Anthracene.** In studies used to calculate the ER-L, effects in sediments were observed at concentrations as low as 0.042 mg/kg in marine systems (Long and Morgan 1991). Confidence in the ER-L is moderate. The safe sediment level based upon equilibrium partitioning and the acute NAWQC is 240 mg/kg (Long and Morgan 1991).

**Fluoranthene.** Lake Union, Washington, sediment with a concentration of 570 mg/kg caused 95% mortality to H. azteca. Columbia River, Washington, sediment with 2.1 mg/kg fluoranthene was not toxic to H. azteca. The lowest AET for Puget Sound and San Francisco Bay was 1.7 and 2.0, respectively. The confidence in the ER-L is high (Long and Morgan 1991). The EPA freshwater sediment quality criterion assuming 1% organic carbon is 6.2 mg/kg (EPA 1993d).

A freshwater sediment with approximately 0.5% organic carbon was spiked with fluoranthene in order to generate an $EC_{50}$ for H. azteca and Chironomus tentans, using organism immobility as the endpoint (Suedel et al. 1993). The $EC_{50}$ values were between 2.3 and 7.4 mg/kg for H. azteca and 3.0 and 8.7 mg/kg for C. tentans.

**Fluorene.** Lake Union, Washington, sediment with a concentration of 40 mg/kg fluorene caused 95% mortality in H. azteca. A sediment quality criterion calculated using equilibrium partitioning and the acute NAWQC and 1% organic carbon yields a value of 7.0 mg/kg. Confidence in the ER-L is low (Long and Morgan 1991).

**2-methylnaphthalene.** No data were from freshwater systems, and the marine data were from only a few geographic regions. Therefore, the confidence in the ER-L is low (Long and Morgan 1991). The lowest AET for Puget Sound and San Francisco Bay were 0.67 and 0.027 mg/kg, respectively.

**Naphthalene.** Lake Union, Washington, sediments with 40 mg/kg naphthalene caused 95% mortality in H. azteca. Effects were almost always observed at concentrations greater than 0.50 mg/kg in marine systems (Long and Morgan 1991). The Puget Sound AET was 2.1 mg/kg. Confidence in the ER-L is moderate (Long and Morgan 1991).

**Phenanthrene.** Lake Union, Washington, sediments with 410 mg/kg phenanthrene caused 95% mortality in H. azteca. Columbia River, Washington, sediments with 0.58 mg/kg were not toxic to the same species. Confidence in the ER-L is moderate. The EPA sediment quality criterion is 1.8 mg/kg (EPA 1993e).

**Pyrene.** Lake Union, Washington, sediments with 750 mg/kg pyrene caused 95% mortality in H. azteca. A concentration of 2.5 mg/kg in Columbia River, Washington, sediments was not toxic to the same species (Long and Morgan 1991). The EPA interim sediment quality criterion based on equilibrium partitioning and 1% organic carbon is 13.1 mg/kg (Long and Morgan 1991). Confidence in the ER-L is moderate.

### 6.3.1.3 Terrestrial plant toxicity data

#### Toxicological Benchmarks for Screening COPCs for Effects on Terrestrial Plants

Tests of the toxicity of chemicals in the rooting medium of plants are conducted using a variety of rooting media. Tests conducted in natural soils (even when brought into the laboratory, dried, sieved, fertilized, etc.) are assumed to be representative of the exposure of plants to contaminants measured in field soils. Tests conducted in nutrient solutions are assumed to be representative of exposures of plants to contaminants measured in soil solutions or in very shallow groundwater.

Soil benchmarks are based on data provided only by toxicity studies in field, greenhouse, or laboratory settings. Most of the soil concentrations of metals reported from waste sites are from extractions with mineral acids that are intended to provide total concentrations. Similarly, concentrations of organic contaminants in waste site soils are total concentrations derived from rigorous solvent extractions. Toxicity tests report concentrations extracted from contaminated soil by various extractants or, more commonly, nominal concentrations of the form of the chemical added to soil. For most metals of concern, the nominal concentration closely approximates the total soil concentration of that metal.

Solution benchmarks include data from toxicity tests conducted using whole plants rooted in aqueous nutrient solutions. Tests are commonly conducted in this manner because plants are assumed to be exposed to contaminants in the solution phase of soil and the presence of

soil in test systems reduces the experimenter's degree of control over exposure. Groundwater samples from waste sites are typically acidified before analysis to obtain total concentrations.

These benchmarks are for contaminant screening only (Table 6.32). Plant toxicity may be affected by many variables: pH, Eh, cation exchange capacity, moisture content, interactions with other elements, and organic matter and clay content of the soil. In addition, different plant species react to different contaminants with varying degrees of toxicity, and the sensitivity of a plant may be affected by its physiological condition. We know of no systematic tests that thoroughly examine the effects of these variables on plant toxicity. An assessor must realize that these soil characteristics play a large part in plant toxicity and incorporate these site-specific considerations in the evaluation of the potential hazards of a chemical. If chemical concentrations reported in field soils that support vigorous and diverse plant communities exceed one or more of the benchmarks, or if a benchmark is less than background soil concentrations, it is generally safe to assume that the benchmark is a poor measure of risk at that site.

References on the toxicity of selected chemicals to terrestrial plants were obtained from searches of bibliographic data bases (BIOSIS, POL TOX I), a numeric data base (PHYTOTOX), review articles, and conventional literature searching. The target was reports of toxicity tests of individual chemicals in laboratory, greenhouse, or field settings.

Data on which benchmarks are based were derived mainly from primary sources. Secondary sources were used if the primary source cited in the secondary source was unavailable, if only a little data for a particular chemical were available, and if secondary sources suggested that a benchmark derived from limited primary source material was too high.

The majority of the responses reported in the literature were oven-dry weights of whole plants or their parts. Others included root length, plant height, relative growth rate, grain yield, seeds per plant, percent seed germination, and fresh and air-dry weights. Responses other than these growth and yield parameters were included only if growth or yield parameters were unavailable for a chemical. Transpiration rate, $CO_2$ uptake, and chlorophyll content of needles were recorded for methyl mercury; chlorophyll content of needles for mercury also was recorded.

Growth and yield parameters were used for two reasons. First, as the most common class of response parameters reported from phytotoxicity studies, they allowed for derivation of reasonably consistent benchmarks for a large number of contaminants. Second, growth and yield are ecologically significant responses both in terms of the plant populations and the ability of the biota to support higher trophic levels.

The LOEC is defined here as the lowest applied concentration of the chemical of interest that gave a greater than 20% reduction in a measured response. In some cases, the LOEC for the test was the lowest concentration tested (LCT) or the only concentration tested when the $EC_{50}$ was reported.

Twenty percent reduction in growth or yield was used as the threshold for significant effects to be consistent with other screening benchmarks for ERA and with current regulatory practice (Suter et al. 1993a). Most regulatory criteria are based on concentrations that cause effects that are statistically significantly different from controls, which on average correspond

to greater than 20% effects. In addition, regulatory actions may be based on comparisons of biological parameters measured on contaminated sites to those from reference sites. Differences between sites generally must be greater than 20% to be reliably detected in such studies. Therefore, the 20% effects level is treated as a conservative approximation of the threshold for regulatory concern.

**Derivation of Benchmarks.** Because of the diversity of soils, plant species, chemical forms, and test procedures, it is not possible to estimate concentrations that would constitute a threshold for toxic effects on the plant communities at particular sites from published toxicity data. The method used for deriving soil benchmarks is based on the NOAA'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 10th percentile of the distribution of various toxic effects thresholds for various organisms in sediments.

This approach is justified by assuming that the toxicity of a chemical in soil is a random variate, that the toxicity of contaminated soil at a particular site is drawn from the same distribution, and that the assessor should be 90% certain of protecting plants growing in the site soil. Any bias in the data set would mitigate against that assumption. In this approach, the most likely bias is the use of soluble salts of metals, which are likely to be more toxic than the mixture of forms encountered in field soils, in the toxicity tests. That bias would result in conservative benchmark values. Other possible sources of bias include the use of predominantly domestic plant species that may not be representative of plant species in general, use of predominantly agricultural soils that may not be representative of soils in general, and the laboratory test conditions that may not be representative of field conditions. The direction and magnitude of these potential biases is unknown.

The phytotoxicity benchmarks were derived by rank ordering the LOEC values and then picking the number that approximated the 10th percentile. 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. If there were 10 or fewer values for a chemical, the lowest LOEC was used. If there were more than 10 values, the 10th percentile LOEC value was used. If the 10th percentile fell between LOEC values, a value was chosen by interpolation. In all cases, benchmark values were rounded to one significant figure.

Values recommended in published reviews of the phytotoxicity literature are another possible source of benchmark values. When primary literature is unavailable for a particular contaminant, concentrations identified in reviews as thresholds for phytotoxicity are used as benchmarks. In addition, when fewer than three LOEC values were found for a chemical in soil or solution, and a toxicity threshold from a review is lower than the lowest LOEC, the toxicity threshold is used as the benchmark for that chemical.

### Toxicity Profiles for COPCs to Plants

Plants are affected by contaminants in the soil in which they are anchored and from which they derive nutrients. Uptake and toxicity depend on factors such as media concentration, chemical speciation, plant growth stage, and tolerance of the specific plant type. A general understanding of the phytotoxicity of contaminants of concern at Chestnut Ridge OU 2 will aid in evaluating the risk posed by their measured concentrations in the specific media at the OU.

**Aluminum.** Aluminum has been studied in relation to its increased availability at low pH. Effects on root and shoot weight have been observed at <1 ppm in solution at pH 4.7 (Wheeler and Follett 1991), and 50 ppm (white clover) in soil at pH 4.5 (MacKay et al. 1990). Douglas fir seedlings grown in a pH 7.5 solution exhibited reduced root weight upon exposure to 8 ppm aluminum (Keltjens 1990). Physiologically, aluminum interferes with cell division in roots, decreases root respiration, fixes phosphorus in unavailable forms in roots, interferes with uptake, transport, and use of Ca, Mg, P, K, and water, and interferes with enzyme activities (Foy et al. 1978). Symptoms of toxicity include stubby, brittle roots, stunting, late maturity, and collapse of growing points. Seedlings are more susceptible to damage from aluminum toxicity than are older plants.

**Antimony.** Antimony is not an essential element in plants (Kabata-Pendias and Pendias 1984). There is very little information on phytotoxic effects of this element.

**Arsenic.** Arsenic is not essential for plant growth. In experiments with toxic levels of arsenic, rice and legumes appear to be more sensitive than other plants. It has been found to have undefined phytotoxic effects at 0.02 ppm in solution (Scharrer 1955) and to reduce plant weight of cotton and soybeans at 10 ppm in soil (Deuel and Swoboda 1972). Symptoms include retardation of root and top growth, and leaf necrosis (Aller et al. 1990).

**Barium.** Barium is not an essential element for plant growth (Vanselow 1966). Undefined phytotoxic effects occurred at 500 ppm in solution (Chapman 1966), and barley plant weight was reduced at 500 ppm applied to soil (Chaudhry et al. 1977). Because of its similar chemical properties, barium can replace cadmium in plants to some extent.

**Cadmium.** Cadmium is not an essential element for plants. Several wild grasses and dicots have shown reduced seed germination and plant weight with 100 ppm cadmium added to soil (Miles and Parker 1979b). Agricultural crops exhibited reduced growth at 1 ppm added to soil (Miller et al. 1976). Trees show a range of tolerance to cadmium added to soil. Red oak plant weight was reduced by 50 ppm (Dixon 1988) while sycamore growth was reduced by 5 ppm cadmium added to soil (Carlson and Bazzaz 1977).The most important biochemical characteristic of cadmium is its affinity for sulfhydryl groups, other side chains of proteins, and phosphate groups (Kabata-Pendias and Pendias 1984). Symptoms of plant toxicity include growth retardation, root damage, and leaf chlorosis. Cadmium interferes with metabolism of some micronutrients, inhibits photosynthesis, transpiration, and $CO_2$ fixation, and alters permeability of cell membranes.

**Chromium.** Chromium is not an essential element in plants. The +6 form is more soluble, thus available to plants, and is considered the more toxic form (Smith et al. 1989). Chromium (VI) added to soil at <2 ppm resulted in a 50% reduction in lettuce plant weight (Adema and Henzen 1989). Less than 1 ppm chromium (VI) in solution reduced the weight of several plants (Adema and Henzen 1989, Wallace et al. 1977c). Symptoms of toxicity include stunted growth, poorly developed roots, and leaf curling. Chromium may interfere with C, N, P, Fe, and Mo metabolism and with enzyme reactions (Kabata-Pendias and Pendias 1984).

**Cobalt.** Cobalt is not known to be essential to plants except legumes with symbiotic nitrogen-fixing microorganisms. Less than 0.1 ppm in solution reduced growth of tomatoes and beans (Wallace et al. 1977c, Patel et al. 1976), and 25 ppm in soil had an undefined

phytotoxic effect (Linzon 1978). Toxicity symptoms include death of leaf margins and tips, inhibition of mitosis, root tip damage, and chromosome damage (Aller et al. 1990).

**Copper.** Copper is an essential micronutrient that has a role in a large number of enzymes. Copper at 100 ppm added to soil reduced plant growth of bluestem grass (Miles and Parker 1979a). Less than 0.1 ppm copper in solution inhibited growth of rye grass (Wong and Bradshaw 1982). Toxicity symptoms include root growth inhibition. Copper inhibits photosynthesis through its effect on electron transport (Fernandes and Henriques 1991).

**Iron.** Iron is essential in plants. It occurs in proteins and is involved in photosynthesis electron transport. As little as 10 ppm in solution produced an undefined phytotoxic effect (Chapman 1966). Toxicity symptoms include stunted top and root growth (Foy et al. 1978).

**Manganese.** Manganese is essential for plant growth. It is involved in nitrogen assimilation, as a catalyst in plant metabolism, and functions with iron in the synthesis of chlorophyll (Labanauskas 1966). Growth of beans was reduced by the addition of 500 ppm manganese to soil (Wallace et al. 1977a). Rye grass growth in soil was reduced by <1 ppm (Wong and Bradshaw 1982) and spruce by 44 ppm added to soil (Langheinrich et al. 1992). Toxicity symptoms include marginal chlorosis and necrosis of leaves and root browning. Excess manganese interferes with enzymes, decreases respiration, and is involved in the destruction of auxin (Foy et al. 1978).

**Mercury.** Mercury is not essential for plant growth. An addition of 0.3 ppm to soil was phytotoxic (Kloke 1979). Concentrations of $\leq$0.1 ppm in solution were toxic to wild plants (Mhatre and Chaphekar 1982). Inorganic mercury at 0.02 ppm in solution negatively affected spruce, and methyl mercury at 0.002 ppm in solution reduced the transpiration rate (Schlegel et al. 1987). Symptoms of toxicity include stunting of root and shoot growth. Mercury has a strong affinity for the sulfhydryl or thiol groups involved in enzyme reactions. It affects metabolic activity by producing structural changes in proteins. It also inhibits photosynthesis (Kabata-Pendias and Pendias 1984).

**Nickel.** Nickel is not generally considered to be an essential element for plants. However, it may be required by nodulated legumes for internal nitrogen transport as part of the urease enzyme (Aller et al. 1990). Red oak trees displayed reduced growth at 50 ppm nickel added to soil (Dixon 1988). Other plants have shown reduced growth at as low as 25 ppm nickel added to soil (Wallace et al. 1977a). Symptoms of nickel toxicity are generally iron chlorosis. Excess nickel affects nutrient absorption by roots, root development, and metabolism, and it inhibits photosynthesis and transpiration.

**Selenium.** Selenium is not known to be essential for plant growth. Of the two forms present in soil, selenate ($SeO_4^{2-}$) is considered more toxic to plants than selenite ($SeO_3^{2-}$) (Aller et al. 1990), perhaps due to its greater bioavailability. Unlike most metal contaminants, bioavailability of selenium increases with soil pH. As little as 1 ppm selenium as selenate reduced grass growth (Carlson et al. 1991). Toxicity symptoms include leaf chlorosis and reduced yield. Selenium can replace sulfur in amino acids and biological processes and disrupts normal protein structure and function (Aller et al. 1990).

**Silver.** Silver is not an essential element for plant growth. Two ppm silver in surface soil had an undefined phytotoxic effect (Linzon 1978). Less than 1 ppm in solution reduced

bean growth (Wallace 1979). By substituting for $K^+$ in membranes, silver can inhibit absorption of other cations by roots (Kabata-Pendias and Pendias 1984).

**Thallium.** There is little available information on specific phytotoxic effects of thallium. One ppm in solution has been shown to negatively effect photosynthesis (Carlson et al. 1975). One ppm in soil resulted in an undefined phytotoxic effect (Kloke 1979).

**Vanadium.** Vanadium is not known to be essential for plant growth although it may be involved in nitrogen fixation in nodules of legume roots. A concentration of 2.5 ppm in soil had an undefined phytotoxic effect (EPA 1975). Less than 1 ppm reduced root growth in beans (Wallace 1979). Toxicity symptoms include chlorosis, dwarfing, and inhibited root growth (Pratt 1966).

**Zinc.** Zinc is an essential element for plant growth. It has a part in many enzymes and is involved in disease protection and metabolism of carbohydrates and proteins. Plant growth has been reduced by the addition of as little as 25 ppm to soil (Aery and Sakar 1991). Zinc at <0.5 ppm in solution inhibited growth of several legumes (Carroll and Loneragan 1968). Toxicity symptoms include chlorosis and depressed plant growth (Chapman 1966).

### 6.3.1.4 Soil invertebrate toxicity data

#### Toxicological Benchmarks for Screening COPCs for Effects on Earthworms

Tests of the toxicity of chemicals in soil to earthworms are conducted using a variety of growth media. Tests conducted in natural soils and artificial soils are assumed to be representative of the exposure of earthworms to contaminants measured in field soils.

Most of the soil concentrations of metals and organics reported from waste sites are from extractions intended to provide total concentrations. Toxicity tests report concentrations extracted from contaminated soil by various extractants or, more commonly, nominal concentrations of the form of the chemical added to soil. For most metals of concern, the nominal concentration closely approximates the total soil concentration of that metal.

The benchmarks are for contaminant screening only (Table 6.33). Exposure of earthworms to contaminants, either externally or internally, may be affected by many variables: pH, Eh, cation exchange capacity, moisture content, interactions with other elements, and organic matter and clay content of the soil. In addition, different species react to different contaminants with varying degrees of toxicity. We know of no systematic tests that thoroughly examine the effects of these variables on earthworm toxicity. An assessor must realize that these factors play a large part in earthworm toxicity and incorporate these site-specific considerations in the evaluation of the potential hazards of a chemical. If chemical concentrations reported in field soils that support vigorous earthworm populations exceed one or more of the benchmarks presented in this report or if a benchmark is less than background soil concentrations, it is generally safe to assume that the benchmark is a poor measure of risk at that site.

References on the toxicity of selected chemicals to earthworms were obtained from searches of bibliographic data bases, review articles, and conventional literature searching. The target was reports of toxicity tests of individual chemicals in laboratory, greenhouse, or field settings. Data on which benchmarks are based were derived mainly from primary sources.

Secondary sources were used if the primary source cited in the secondary source was unavailable, if only a little data for a particular chemical were available, and if secondary sources suggested that a benchmark derived from limited primary source material was too high.

Responses reported in the literature were measures of reproductive success, survival, and growth of organisms. Measures of reproductive success are the most ecologically significant responses, both in terms of earthworm populations and their ability to support higher trophic levels.

The LOEC is defined here as the lowest applied concentration of the chemical of interest which gave a greater than 20% reduction in a measured response. In some cases, the LOEC for the test was the LCT or the only concentration tested when the $EC_{50}$ was reported. The justification for using a 20% reduction as a threshold for significant effects is the same as noted for plant toxicity benchmarks.

**Derivation of Benchmarks.** The same limitations apply to the estimation of a toxic effects threshold concentration for earthworms as was noted for plants in soil (Sect. 6.3.1.3.) For this reason, the method used for deriving the benchmarks was the same as for phytotoxicity.

**Toxicity Profiles for COPCs to Earthworms**

Earthworms are affected by soil contaminants through uptake from contaminated food sources and contact with the body surface. Uptake and toxicity depend on a large number of factors, some of the most important of which are media concentration, chemical speciation, and tolerance of the organism (Ma 1982). A general understanding of the toxicity to earthworms of contaminants of concern at Chestnut Ridge OU 2 will aid in assessing the risk to those organisms posed by the contaminants.

**Cadmium.** Cadmium added to various artificial and natural soils has been found to reduce survival of earthworms of the *Eisenia* genus at ~ 550 ppm (van Gestel and van Dis 1988), number of cocoons produced per worm at 18 ppm (van Gestel et al. 1992), and growth at 100 ppm (Neuhauser and Malecki 1984). Cadmium is known to interfere with copper metabolism (Beeby 1991).

**Chromium.** The more toxic +6 form of chromium has been shown to reduce survival of earthworms of the *Octochaetus* genus at a concentration of 2 ppm (Abbasi and Soni 1983). The number of cocoons per worm and percent fertile cocoons of earthworms of the genus *Eisenia* were negatively impacted at 100 ppm (van Gestel et al. 1992).

**Copper.** Copper concentrations of ~ 100 ppm resulted in reduced reproduction and second generation survival of earthworms of the *Allolobophora* and *Dendrobaena* genera (van Rhee 1975, Bengtsson et al. 1986). Survival was generally not as sensitive to copper. $LC_{50}$ concentrations ranged between 180 and 2500 ppm for various earthworms (Streit and Jaggy 1983, Neuhauser et al. 1985, Malecki et al. 1982).

**Mercury.** Inorganic mercury was found to reduce survival and cocoon production in the earthworm *Octolasium* at concentrations as low as 0.5 ppm (Abbasi and Soni 1983). Methylmercury reduced survival and segment regeneration of *Eisenia* at ~ 10 ppm (Beyer et al. 1985a). Mercury is known to interfere with copper metabolism (Beeby 1991).

**Toxicological Benchmarks for Screening COPCs for Effects on Soil Invertebrates Other than Earthworms**

Tests of the toxicity of chemicals to soil invertebrates other than earthworms are conducted by exposing native soil populations or single introduced species of microarthropods or other soil/litter-inhabiting invertebrates to individual contaminants in foodstuffs. These foodstuffs range from natural soil organic matter and litter to green algae. Data from one experiment conducted in nutrient solution was included due to the paucity of information on toxicity of metals (in this case, cadmium) to soil invertebrates.

Experiments examining the effects of contaminants on soil invertebrates other than earthworms are conducted by exposing the organisms to the contaminant in foodstuffs rather than in the ambient medium to which they would have direct, continuous, bodily contact. Experiments to evaluate the effects of contaminants on earthworms are generally carried out with the latter, qualitatively and quantitatively different, method of exposure. Benchmarks for invertebrates other than earthworms are based on concentrations of contaminants in foodstuffs and will be used for screening concentrations of metals in vegetation from the site.

Most of the vegetation concentrations of metals and organics reported from waste sites are from extractions intended to provide total concentrations. Toxicity tests generally report nominal concentrations of the form of the chemical added to food that, for most metals of concern, closely approximates the total concentration of that metal.

These benchmarks are for contaminant screening only (Table 6.34). If chemical concentrations reported in field soils that support vigorous, functioning, soil invertebrate populations exceed one or more of the benchmarks presented in this report or if a benchmark is less than background soil concentrations, it is generally safe to assume that the benchmark is a poor measure of risk at that site.

References on the toxicity of selected chemicals to soil invertebrates other than earthworms were obtained from searches of bibliographic data bases and conventional literature searching. The target was reports of toxicity tests of individual chemicals in laboratory, greenhouse, or field settings. Data on which benchmarks are based were derived from primary sources.

Responses reported in the literature were mainly measures of survival. Reproduction, population growth rate, growth of organisms, and nitrogen mineralization were also reported. The measures generally indicate the viability of the population exposed to the contaminant, and indirectly the potential effects on litter decomposition as a step in nutrient cycling.

The LOEC is defined here as the lowest applied concentration of the chemical of interest resulting in a greater than 20% reduction in a measured response. In some cases, the LOEC for the test was the LCT or the only concentration tested when the $EC_{50}$ was reported. The justification for using a 20% reduction as a threshold for significant effects is the same as noted for plant toxicity benchmarks.

**Derivation of Benchmarks.** The same limitations generally apply to the estimation of a toxic effects threshold concentration for soil invertebrate populations as was noted for plants in soil (Sect. 6.3.1.3.) For this reason, the method used for deriving the benchmarks was the same as for phytotoxicity.

### 6.3.1.5 Soil microbe and microbially mediated soil processes toxicity data

**Toxicological Benchmarks for Screening COPCs for Effects on Microorganisms and Microbially Mediated Soil Processes**

Tests of the toxicity of chemicals in soil to microbes and microbially mediated processes are conducted using a variety of growth media. Tests conducted in natural soil and soil/litter systems are assumed to be representative of the exposure of microbial populations to contaminants measured in field soils.

Most of the soil concentrations of metals and organics reported from waste sites are from extractions intended to provide total concentrations. Toxicity tests generally report nominal concentrations of the form of the chemical added to soil. For most metals of concern, the nominal concentration closely approximates the total soil concentration of that metal.

These benchmarks are for contaminant screening only (Table 6.35). Toxicity to soil microbes and microbially mediated processes may be affected by the same biotic and abiotic variables as noted for plants and earthworms. A lack of understanding of the effects of these variables requires that the assessor evaluate the applicability of the benchmarks to the conditions of each site. If chemical concentrations reported in field soils that support vigorous, functioning microbial populations exceed one or more of the benchmarks presented in this report or if a benchmark is less than background soil concentrations, it is generally safe to assume that the benchmark is a poor measure of risk at that site.

References on the toxicity of selected chemicals to soil microbes and microbially mediated processes were obtained from searches of bibliographic data bases and conventional literature searching. The target was reports of toxicity tests of individual chemicals in laboratory, greenhouse, or field settings. Data on which benchmarks are based were derived from primary sources.

Responses reported in the literature were measures of respiration, nitrogen transformations, enzyme synthesis, enzyme activity, sporulation, and growth of microbial cultures. Test populations are either native soil microbial populations or axenic cultures of bacteria, fungi, or yeasts introduced into sterilized growth media. Measures of respiration, sporulation, and growth indicate the viability of the population exposed to the contaminant. Enzyme synthesis, nitrogen transformations, and measures of other enzyme activities are ecologically significant responses as they elucidate the effects of contaminants on the functioning of the population in the soil system.

The LOEC is defined here as the lowest applied concentration of the chemical of interest which gave a greater than 20% reduction in a measured response. In some cases, the LOEC for the test was the LCT or the only concentration tested when the $EC_{50}$ was reported. The justification for using a 20% reduction as a threshold for significant effects is the same as noted for plant toxicity benchmarks.

**Derivation of Benchmarks.** The same limitations apply to the estimation of a toxic effects threshold concentration for microbial populations and microbially mediated processes as was noted for plants in soil (Sect. 6.3.1.3) For this reason, the method used for deriving the benchmarks was the same as for phytotoxicity.

## Toxicity Profiles for COPCs to Microbes and Microbial Processes

Microorganisms are affected by contaminants in soil and surface organic matter. Uptake and toxicity depends on factors such as media concentration, chemical speciation, and tolerance of the organisms and process enzymes. A general understanding of the toxicity to microorganisms and microbially mediated soil processes of contaminants of concern at Chestnut Ridge OU 2 will aid in assessing the potential risk to those organisms.

**Aluminum.** There is very little information on toxic effects of aluminum. A concentration of 135 ppm aluminum added to a clay loam soil reduced nitrogen mineralization by native soil microflora (Liang and Tabatabai 1977).

**Cadmium.** Carbon mineralization has been found to be negatively impacted by 6 ppm cadmium added to soil/litter microcosms (Lighthart and Bond 1976). Bollag and Barabasz (1979) measured a reduction in denitrification by native microflora in loam soil in the presence of 10 ppm added cadmium. Nitrogen mineralization was less sensitive and was not reduced until ~ 500 ppm cadmium was added to soils (Liang and Tabatabai 1977). Soil enzyme activities (phosphatase, arylsulfatase, and urease) were evaluated for sensitivity to added cadmium. Arylsulfatase activity was reduced by 50% in the presence of 1 ppm cadmium added to soil (Haanstra and Doelman 1991).

**Chromium.** Metabolic toxic effects of chromium in microbial cells often involve changes in electron transport, and malfunctioning of proteins and DNA synthesis. Chromium (VI) is consistently found to be more toxic than chromium (III) (Coleman 1988). Addition of 10 ppm chromium (VI) to a sandy soil reduced respiration of native soil microflora (Ross et al. 1981). Liang and Tabatabai (1977) measured a reduction in nitrogen mineralization in the presence of 260 ppm chromium (VI), and urease and arylsulfatase activities were reduced 50% by <1 ppm chromium added to soil (Haanstra and Doelman 1991).

**Copper.** Nitrogen mineralization and nitrification by native soil microflora were reduced in a sewage sludge-amended soil containing 200 ppm copper (Chang and Broadbent 1982). Doelman and Haanstra (1989) measured a 50% reduction in phosphatase activity of native soil microflora in response to the presence of 3 ppm chromium.

**Iron.** There is very little information concerning toxicity of iron to microorganisms. Liang and Tabatabai (1977) measured a reduction in nitrogen mineralization by native soil microflora in soil to which 280 ppm iron had been added.

**Manganese.** There is very little information concerning toxicity of manganese to microorganisms. Nitrification by native soil microflora was found to be sensitive to 100 ppm manganese added to soil (Premi and Cornfield 1969). Liang and Tabatabai (1977) measured a reduction in nitrogen mineralization in soil in the presence of 275 ppm manganese.

**Mercury.** The effects of inorganic mercury on soil microorganisms has been examined. Landa and Fang (1978) found that <1 ppm mercury added to soil reduced carbon mineralization by native soil microflora. Approximately 1000 ppm mercury were found to reduce nitrogen mineralization in experiments conducted by Liang and Tabatabai (1977).

**Vanadium.** There is very little information concerning toxicity of vanadium to microorganisms. In work by Lighthart et al. (1977), the addition of ~ 20 ppm of vanadium to a soil/litter microcosm reduced respiration of native soil microflora.

**Zinc.** In an examination of the effects of zinc on the activities of phosphatase, arylsulfatase, and urease, Doelman and Haanstra (1989) found the addition of 3 ppm zinc to soil decreased phosphatase activity. Respiration was diminished in the presence of ~ 480 ppm zinc in a soil/litter microcosm in studies by Chaney et al. (1978). Nitrogen transformations by native soil microflora varied in sensitivity from an inhibitory concentration of 100 ppm zinc for nitrification (Premi and Cornfield 1969) and 250 ppm for denitrification (Bollag and Barabasz 1979), to 1000 ppm for nitrogen mineralization in experiments conducted by Bhuiya and Cornfield (1976).

### 6.3.1.6 Wildlife toxicity data

#### Toxicological Benchmarks for Wildlife

To determine if the contaminant exposure experienced by wildlife that use Chestnut Ridge OU 2 could produce adverse effects, exposure estimates from Sect. 6.2.2.3 were compared to NOAELs derived according to the methods outlined by Opresko et al. (1993) and EPA (1993f).

Toxicological studies of the effects of contaminants observed at Chestnut Ridge OU 2 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 long-term, chronic oral exposures, whether in food, water, or by oral intubation, were used. 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. If the only available data consist of a NOAEL for a subchronic exposure (<3-6 months for mammals, <10 weeks birds), then the equivalent NOAEL for a chronic exposure was estimated as follows:

$$chronic\ NOAEL = \frac{subchronic\ NOAEL}{10}$$

In cases where a NOAEL for a specific chemical was not available, but a LOAEL had been determined experimentally, the NOAEL was estimated as follows:

$$NOAEL = \frac{LOAEL}{10}$$

EPA (1993f) suggests the use of uncertainty factors of 1 to 10 for subchronic to chronic NOAEL and LOAEL to NOAEL estimation. To be conservative and because we have no data to suggest the use of lower uncertainty factors, a factor of 10 was chosen for both estimates.

Studies have shown that numerous physiological functions such as metabolic rates, as well as responses to toxic chemicals, are a function of body size. 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 the best measure of differences in body size are those based on 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 1980e). If the dose (d) itself has been calculated in terms of unit body weight (i.e., mg/kg); then the dose per unit surface area (D) equates to

$$D = \frac{d \times bw}{bw^{2/3}} = d \times bw^{2/3}$$

The assumption is that the dose per body surface area for species "a" and "b" would be equivalent:

$$d_a \times bw_a^{1/3} = d_b \times bw_b^{1/3}$$

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:

$$d_a = d_b \times \frac{bw_b^{1/3}}{bw_a^{1/3}} \quad d_b \times (bw_b/bw_a)^{1/3}$$

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:

$$NOAEL_w \times NOAEL_t \times (bw_t/bw_w)^{1/3}$$

This methodology is similar to that the EPA uses in their carcinogenicity assessments and Reportable Quantity documents for adjusting from animal data to an equivalent human dose (EPA 1985f, 1988b).

NOAELs were derived for short-tailed shrews, white-footed mice, red fox, white-tailed deer, and red-tailed hawks. Of the 45 contaminants for which wildlife exposure estimates were calculated, 21 mammalian and 13 avian NOAELs have been derived. Mammalian and avian NOAELs and experimental information used to estimate wildlife NOAELs (e.g., test species, test endpoints, citation, etc.) are listed in Tables 6.36 and 6.37.

Mammalian NOAELs have not yet been derived for the following contaminants from Chestnut Ridge OU 2:

| | | | | |
|---|---|---|---|---|
| Calcium | Phosphorus | Radium-226 | Uranium-234 | Butylbenzylphthalate |
| Chloride | Potassium | Thorium-228 | Uranium-235 | alpha-BHC |
| Cobalt | Sodium | Thorium-230 | Uranium-238 | di-n-octylphthalate |
| Iron | Sulphate | Thorium-232 | 2-Methylnaphthalene | Phenol |
| Magnesium | Cesium-137 | Thorium-234 | 4-Nitrophenol | |

Avian NOAELs have not yet been derived for the following contaminants from Chestnut Ridge OU 2:

| | | | | |
|---|---|---|---|---|
| Barium | Magnesium | Sulphate | Thorium-234 | Chloroform |
| Beryllium | Manganese | Thallium | Uranium-234 | alpha-BHC |
| Calcium | Niobium | Cesium-137 | Uranium-235 | Di-n-octylphthalate |
| Chloride | Nitrate | Radium-226 | Uranium-238 | Phenol |
| Cobalt | Phosphorus | Thorium-228 | 2-Methylnaphthalene | |
| Fluoride | Potassium | Thorium-230 | 4-Nitrophenol | |
| Iron | Sodium | Thorium-232 | Butylbenzylphthalate | |

**Effects of Metal Body Burdens on Small Mammals**

An extensive literature review was conducted to identify body burdens of As, Cd, Cr, Pb, Hg, Se, and Tl in small mammals and effects associated with those body burdens. These data were used for comparison with body burdens observed in small mammals captured at Chestnut Ridge OU 2 sites and at the Walker Branch reference site. This comparison provides evidence concerning the possible effects of observed body burdens, the adequacy of the reference area, and the level of contamination at the site. Results are described below for contaminants in rodents and shrews. Only studies reporting whole body (or, in some cases, carcass) concentrations were included in the summary provided in Table 6.38.

**Arsenic: Rodents.** Whole body concentrations of arsenic in four *Peromyscus* from the Walker Branch reference site averaged 0.054 mg/kg, wet weight with a range from <0.05 to 0.064 (Sect. 3.2.6: Table 3.37). These values were only slightly higher than those reported at reference sites in other studies (Table 6.38). Elfving et al. (1979) reported arsenic concentrations in pine voles and white-footed mice ranging from 0 to 0.028 and from 0.006 to 0.021 μg/g fresh weight, respectively, at orchards in New York with soil concentrations of 2.4 μg/g. White-footed mice at a proposed mine site in Wisconsin had whole body arsenic concentrations of <0.01 μg/g fresh weight (Smith and Rongstad 1982).

Whole body arsenic concentrations for rodents as a group at all Chestnut Ridge sites averaged 0.131 mg/kg wet weight (Sect. 3.2.6: Table 3.37). *Peromyscus*, the most frequently trapped small mammal, averaged 0.171 mg/kg with a range from 0.081 to 0.434 at the FCAP and 0.051 (0.05-0.053) in the Sluice Channel Area. Concentrations at the Sluice Channel Area were not different from the reference area, but concentrations in white-footed mice at the FCAP were significantly elevated over those at the reference area. Concentrations in *Peromyscus* at orchards in New York treated with inorganic arsenic compounds were similar to those found at the FCAP; body burdens ranged from 0.079 to 0.589 μg/g (Elfving et al. 1979). Soil arsenic concentrations in the New York orchards were 31-44 ppm.

The pine vole from the Sluice Channel Area had an arsenic concentration of 0.093 mg/kg (Sect. 3.2.6: Table 3.37), less than the 0.16-0.956 μg/g pine voles from orchards with 34-94 ppm arsenic in the soil averaged (Elfving et al. 1979), but still greater than the average for white-footed mice at the Walker Branch reference area and the average for pine voles at reference orchards. *Reithrodontomys* at the FCAP also had elevated arsenic body burdens (Sect. 3.2.6: Table 3.37).

**Arsenic: Shrews.** Short-tailed shrews at Walker Branch averaged 0.059 mg arsenic/kg (Sect. 3.2.6: Table 3.38), similar to the concentration detected in rodents (Sect. 3.2.6: Table 3.37), but somewhat higher than the concentration of <0.01 μg/g reported for short-tailed shrews at a proposed mine site in Wisconsin (Smith and Rongstad 1982).

Only one short-tailed shrew was captured from the Chestnut Ridge OU 2 Sluice Channel Area. Its whole body arsenic concentration was 0.177 mg/kg (Sect. 3.2.6: Table 3.38). This is nearly 3 times the mean concentration of shrews from the reference area, but is substantially less than the 0.62 μg/g reported for shrews at a zinc-copper mine in Ontario (Smith and Rongstad 1982).

**Cadmium: Rodents.** Cadmium does not appear to be a concern for rodents at Chestnut Ridge OU 2 sites. Whole body burdens in all rodent species at the FCAP and Sluice Channel Area sites were similar to those at the Walker Branch reference area (Sect. 3.2.6: Table 3.37). Concentrations in mice at the Walker Branch reference area (0.013-0.027 mg/kg, wet weight) were generally less than concentrations in several species of voles and mice captured at reference areas in Virginia (Scanlon 1987), Wales (Johnson et al. 1978), and Wisconsin (Smith and Rongstad 1982). The maximum concentration detected in a rodent at a Chestnut Ridge OU 2 site was 0.434 mg/kg wet weight for a white-footed mouse at the FCAP. Blus et al. (1987) observed no adverse effects in white-footed mice with body burdens of 0.42-0.81 μg/g fresh weight at a smelter complex in Idaho, and Beyer et al. (1985b) characterized mice with carcass concentrations of 2.6 μg/g dry weight living near a zinc smelter in Pennsylvania as healthy. Cadmium concentrations in soil near the zinc smelter averaged 249.5 ppm with concentrations in vegetation of 1.2-8.1 ppm.

**Cadmium: Shrews.** Cadmium does not appear to be a concern for shrews at Chestnut Ridge OU 2 sites. Whole body concentrations in shrews from the Sluice Channel Area were similar to those from Walker Branch (Sect. 3.2.6: Table 3.38). Concentrations in shrews from the Walker Branch reference site were similar to those reported for reference sites in other studies (Smith and Rongstad 1982, Beyer et al. 1985b, Scanlon 1987). No adverse effects attributable to cadmium were detected in short-tailed shrews with carcass body burdens of 4.8 μg/g dry weight from a reference area in Pennsylvania, nor shrews with 7.3 μg/g dry weight from a zinc smelter (Beyer et al. 1985b). The maximum cadmium concentration detected in a small mammal at the Chestnut Ridge OU 2 sites was 0.073 ppm whole body wet weight (0.224 ppm dry weight) for a short-tailed shrew at the Sluice Channel Area.

**Chromium: Rodents.** Average whole body chromium body burdens in rodents at Chestnut Ridge OU 2 sites were slightly less than at the Walker Branch reference area (Sect. 3.2.6: Table 3.37). Beardsley et al. (1978) reported chromium body burdens >6.9 μg/g dry weight for voles at a sewage farm and a reference site in England. Body burdens in cotton rats (Sigmodon hispidus) 100-130 m from a cooling tower and a reference site were <0.19 mg/kg fresh weight (Taylor and Parr 1978). None of these studies reported effects associated

with observed body burdens. Chromium is not expected to be of concern at Chestnut Ridge OU 2 sites.

**Chromium: Shrews.** Chromium concentrations in the short-tailed shrew from the Sluice Channel Area were less than in the shrews from the reference area. No references reporting chromium concentrations in shrews were found for comparison.

**Lead: Rodents.** Mean lead body burdens in rodents from Chestnut Ridge OU 2 sites were not significantly different from those at the Walker Branch reference area (Sect. 3.2.6: Table 3.37). Body burdens of <0.648 mg/kg wet weight at Walker Branch were well within the range reported for reference sites in other studies (Table 6.38). Although mean body burdens indicate lead is not likely to be a problem for rodents at Chestnut Ridge OU 2 sites, one *Reithrodontomys* from the FCAP had a burden of 10.267 mg/kg wet weight (28.107 ppm dry weight). This is much higher than the maximum reported for the reference site. Getz et al. (1977) observed whole body concentrations of 3.1-10.8 μg/g dry weight in western harvest mice (*Reithrodontomys megalotis*) near roads but concluded that no impacts on small mammal populations were likely (Table 6.38). Beyer et al. (1985b) described white-footed mice with carcass concentrations of 17 mg/kg dry weight as healthy. Blus et al. (1987) failed to observe adverse effects in deer mice (*Peromyscus maniculatus*) with body burdens of >50 μg/g fresh weight (Table 6.38). Although they could not clearly associate effects with lead concentrations, Blus et al. (1987) reported one case of embryo absorption and one case of disease in tail and feet of voles with whole body concentrations between 23.5 and 173 μg/g fresh weight. Intranuclear renal inclusions associated with lead toxicosis were observed in field voles (*Microtus agrestis*) with body burdens averaging 42.8 and 45.3 μg/g fresh weight, but not in voles with 2.76 μg/g (Roberts et al. 1978). The high body burden in the one individual from the FCAP seems to fall between concentrations resulting in observed effects and those with no observed effects. Although not certain, lead could be a concern for rodents at Chestnut Ridge OU 2 sites.

**Lead: Shrews.** Lead body burden was lower in the shrew from the Sluice Channel Area than in the shrews from the reference area (Sect. 3.2.6: Table 3.38). The concentration in Walker Branch shrews was much less than that reported for Pennsylvania short-tailed shrews at a reference site (Beyer et al. 1985b) and similar to concentrations reported by Getz et al. (1977) and Clark (1979). Getz et al. (1977) concluded that body burdens of 3.3-17.2 μg/g dry weight would not result in major impacts on small mammal populations. Adverse effects reported by Beyer et al. (1985b) occurred at a heavily contaminated zinc smelter (soil concentrations averaging 989 ppm) in shrews with body burdens exceeding 100 μg/g dry weight (Table 6.38). Although the number of shrews examined from Chestnut Ridge OU 2 sites is low, lead does not appear to be a significant problem.

**Mercury: Rodents.** Concentrations of mercury in rodents from Chestnut Ridge OU 2 sites were not significantly different from those at the Walker Branch reference area (Sect. 3.2.6: Table 3.37). Field mice (*Apodemus sylvaticus*) inhabiting reference fields contained whole body burdens of 0.1 μg/g dry weight (Jefferies and French 1976, Jefferies et al. 1973), similar to the mean body burden observed in white-footed mice at Walker Branch. Field mice in fields with mercury-soaked wheat had body burdens of 1.3-2.7 (Jefferies and French 1976, Jefferies et al. 1973) and 0.1-2.6 (Westlake et al. 1980) μg/g dry weight. Concentrations in vegetation at these sites averaged 16 ppm and 3-9 ppm, respectively.

**Mercury: Shrews.** The mercury body burden in the short-tailed shrew from the Sluice Channel Area was slightly higher than in the shrews from the reference area (Sect. 3.2.6: Table 3.38). Body burdens from shrews at these sites were higher than burdens of <0.1 μg/g reported by Smith and Rongstad (1982) for short-tailed shrews from a zinc-copper mine and a reference area (Table 6.38).

**Selenium: Rodents.** Rodents at the FCAP had significantly greater selenium body burdens than those at the Sluice Channel Area and the Walker Branch reference area (Sect. 3.2.6: Table 3.37). Rodents inhabiting an area irrigated with selenium-laden drainwater in California had much higher body burdens, ranging from 0-33 ppm dry weight, but no adverse effects were documented (Clark 1987).

**Selenium: Shrews.** The selenium body burden in the shrew from the Sluice Channel Area was roughly twice that of the shrews from the reference area (Sect. 3.2.6: Table 3.38). However, the Sluice Channel Area shrew's body burden was much lower than the 10-100 ppm reported for ornate shrews (*Sorex ornatus*) in an area irrigated with selenium-laden drainwater (Clark 1987). Clark (1987) was unable to document any adverse effects associated with the high body burdens observed in ornate shrews.

**Thallium.** Thallium was detected in only two small mammals from Chestnut Ridge OU 2 sites; thallium was not detected in any of the small mammals captured at the Walker Branch reference area. When it was detected, concentrations were only slightly above detection limits (Sect. 3.2.6: Table 3.37). Thallium is not likely to be a concern for small mammals at Chestnut Ridge OU 2 sites. No references reporting whole body burdens of thallium in small mammals were found.

### Toxicity Profiles for COPCs to Wildlife

**Aluminum.** Aluminum is a ubiquitous metal, being the third most abundant element in the earth's crust (Krueger et al. 1984). Relative to other metals, the toxicity of aluminum is low (Sorensen et al. 1974). The oral $LD_{50}$ for mice ranges from 770 to 980 mg aluminum/kg body weight (Ondreicka et al. 1966). The principal effect of aluminum is to interfere with phosphorus metabolism. Aluminum forms insoluble compounds with phosphorus in the alimentary canal resulting in an imbalance of calcium and phosphorus (Carriere et al. 1986). Other effects of aluminum include neurotoxicity. Rats exposed to aluminum display behavioral abnormalities and have reduced acetylcholinesterase activity (Krueger et al. 1984). Mice consuming diets containing 500 to 1000 ppm aluminum displayed ataxia and paralysis of the hind limbs (Golub et al. 1987). In humans, aluminum has been associated with several degenerative diseases of the nervous system, including Alzheimer's disease, Parkinson's disease, and amyotrophic lateral sclerosis (Ganrot 1986).

Ondreicka et al. (1966) evaluated the effects of aluminum on mammalian reproduction. Mice received 19.3 mg aluminum/kg body weight per day (as $AlCl_3$) in drinking water for three generations. While the number of litters and offspring per litter was not reduced, growth was significantly reduced among all offspring in the second and third generations. In a similar study, rats received daily intragastric doses of 0, 180, 360, or 720 mg aluminum/kg body weight per day (Domingo et al. 1987) for one generation. Growth and survival of young was reduced among the groups that received 360 and 720 mg aluminum/kg per day. Other studies also report that while aluminum does not appear to affect the number

of litters or number of offspring/litter, growth and survival of offspring of aluminum exposed parents is reduced (Golub et al. 1987, Paternain et al. 1988).

Due to its interference with phosphorus and calcium metabolism, it has been suggested that aluminum may impair eggshell formation by birds, resulting in eggshell thinning (Nyholm 1981). To test this hypothesis, Carriere et al. (1986) fed breeding ring doves (*Streptopelia risoria*) a diet containing 1000 ppm aluminum sulfate (and adequate but reduced calcium and phosphorus) and observed reproduction. While no reproductive effects or embryonic malformations were observed at this dosage level, significant reproductive effects resulted when birds were fed a diet deficient in calcium and phosphorus that contained 750 ppm aluminum. Therefore, it appears that the manifestation of toxic effects of aluminum among birds is dependent upon the nutritional quality of their diet.

**Arsenic.** Arsenic is present in the earth's crust at ~ 2 ppm, but tissues of animals generally contain an average of <0.5 ppm (Venugopal and Luckey 1978). Arsenic may be a required micronutrient; growth, survival, and reproduction of goats is poor if the diet contains <0.05 ppm arsenic (NAS 1977).

Arsenic is a carcinogen and teratogen. Other effects include reduced growth, hearing/sight loss, liver/kidney damage, and death (Eisler 1988a). Inorganic arsenic is usually more toxic than organic arsenic compounds. Wildlife mortality and malformations have been observed for chronic doses of 1-10 mg arsenic/kg body weight and dietary concentrations of 5-50 ppm (Eisler 1988a). Acute $LD_{50}$s for mammals of 35-100 mg calcium arsenate/kg body weight and 10-50 mg lead arsenate/kg body weight have been reported (NRCC 1978).

Schroeder and Mitchener (1971) exposed mice to 5 ppm sodium arsenite in drinking water for three generations. While mice fed arsenic survived well, litter size decreased in subsequent generations. A dose of 0.38 mg arsenic/kg over a lifetime was sufficient to cause a slight decrease in the median lifespan of laboratory mice (Schroeder and Balassa 1967), but it had no effect on growth. As little as 3 mg arsenic trioxide/kg body weight or 1 mg sodium arsenite/kg body weight can be lethal (NAS 1977). Because metabolism of arsenic in rats is unlike that in other animals, results of toxicity studies using rats generally should not be extrapolated to other species (Eisler 1988a).

Among birds, $LD_{50}$s for arsenic compounds range from 17.4 to 3300 mg/kg body weight (Eisler 1988a). While no mortality was observed among mallard ducks fed a diet containing 100 ppm sodium arsenite for 128 days, 12% to 92% mortality was observed for ducks fed diets containing 250-1000 ppm arsenite (USFWS 1964). Camardese et al. (1990) and Whitworth et al. (1991) fed mallards diets containing 30, 100, or 300 ppm sodium arsenate. While no effects were observed on behavior, growth was reduced for male ducks consuming 300 ppm arsenic and for female ducks at all exposure levels.

**Barium.** The soluble salts of barium, an alkaline earth metal, are toxic in mammalian systems. At low doses, barium acts as a muscle stimulant and at higher doses affects the nervous system, eventually leading to paralysis. The $LD_{50}$ for rats is listed as 630 mg/kg for barium carbonate; 118 mg/kg for barium chloride, and 921 mg/kg for barium acetate (Lewis and Sweet 1984).

Schroeder and Mitchener (1975a, 1975b) exposed rats and mice to 5 mg barium/L in drinking water for their lifetime. There was a slight but significant reduction in longevity of

treated male mice when measured as the mean age at death of the last surviving 10% of animals. The overall average life span of the group, however, was about the same as the control group. In another study, Perry et al. (1983) exposed rats to 0, 1, 10 or 100 ppm barium for up to 16 months. A significant increase in average blood pressure was observed in the highest dose group; a slight but statistically significant increase was seen in the 10 ppm dose group. Information on developmental and reproductive toxicity of barium to mammals is not available.

The $LD_{50}$ of barium to chickens is 623 mg/kg (Johnson et al. 1960). Johnson et al. (1960) report that while chickens will tolerate 1000 ppm barium in their diet without adverse effects, 2000 ppm reduces growth, 8000 ppm produces 50% mortality in 4 weeks, and diets containing 16,000 or 32,000 ppm barium are 100% lethal.

**Cadmium.** While there is little information to indicate that this relatively rare metal is biologically essential or beneficial, cadmium has been suggested as the cause of various deleterious effects to wildlife (Eisler 1985a). Mammals and birds are comparatively resistant to the biocidal properties of cadmium, which include growth retardation, anemia, and testicular damage. Cadmium tends to bioaccumulate in the liver and kidney, eventually acting as a cumulative toxin. Cadmium residues of 2 ppm whole body fresh weight are evidence of cadmium contamination, and residues of >5 ppm whole animal fresh weight may be life threatening (Eisler 1985a).

The lowest oral dose resulting in death for rats was 250 mg cadmium/kg body weight (EPA 1980a). Weigel et al. (1987) fed rats 0.24, 0.85, or 2.25 μg/g cadmium in diet for 8 weeks. Concentrations ≥0.85 μg/g resulted in reduced food intake, reduced body weights, and reduced enzyme activity, but no hematological effects were noted. Ma et al. (1991) determined that an average cadmium intake of 15 μg/g per day corresponded with critical renal metal loads of 120 μg/g, a level indicative of adverse health effects. Rats on a diet with 5 ppm cadmium suffered shortened life spans (Schroeder et al. 1965). Cadmium at 50 ppm in the diet depleted iron from rat livers (Whanger 1973). Rats eating diets with 7.15 ppm cadmium (as CdO) exhibited growth reductions, but those consuming a diet with 2.80 ppm cadmium did not (Weigel et al. 1984). In a three-generation reproductive study, the population of mice exposed to 1 ppm $CdCl_2$ in their drinking water died out after the second generation (Schroeder and Mitchener 1971). Rats receiving >6 mg cadmium/kg body weight daily during pregnancy gave birth to malformed fetuses (Ferm and Layton 1981).

No mortality was observed among adult mallard ducks fed diets containing 0, 2, 20, and 200 ppm cadmium; however, egg production was significantly reduced in the group consuming 200 ppm cadmium (White and Finley 1978). In addition, the testes of males in the 200 ppm cadmium group atrophied and the spermatogenic process was disrupted (White et al. 1978). Among mallard ducklings, 20 ppm cadmium in the diet produced mild to severe kidney lesions, and reduced packed cell volume and hemoglobin concentrations in the blood (Cain et al. 1983). Avoidance behavior of black ducklings is impaired by diets containing 40 ppm cadmium (Heinz and Haseltine 1983).

**Chromium.** Chromium occurs as either $Cr^{+3}$ or $Cr^{+6}$. Trivalent chromium is an essential metal in man and wildlife, playing an important role in insulin metabolism (Langard and Norseth 1979). Hexavalent chromium is more toxic than $Cr^{+3}$ because of its high oxidation potential and the ease with which it penetrates biological membranes (Steven et al. 1976, Taylor and Parr 1978). However, it is unlikely that all chromium in soil would be $Cr^{+6}$ because

$Cr^{+6}$ is a highly oxidizing chemical species that is usually reduced by soil organic matter to $Cr^{+3}$. Trivalent chromium solubility decreases with increasing pH, and it is completely precipitated at pH above 5.5. In most soils chromium is primarily present as precipitated $Cr^{+3}$ and is not bioavailable. Most chromium in soil and sediments is unavailable to living organisms, and there is little evidence of chromium biomagnifying through food chains in its inorganic form (Eisler 1986). Concentrations of total chromium >4.0 mg/kg dry weight should be viewed as presumptive evidence of chromium contamination (Eisler 1986).

At high concentrations, chromium is a mutagen, teratogen, and carcinogen (Eisler 1986). The $LD_{50}$ for $Cr^{+3}$ in mice is 260 mg/kg body weight and 5 mg/kg body weight for $Cr^{+6}$ (Steven et al. 1976). Rats fed $Cr^{+6}$ reached a toxic threshold at 1000 ppm (Steven et al. 1976). Pregnant hamsters injected with 5 to 15 mg $CrO_3$ ($Cr^{+6}$)/kg body weight displayed a dose dependent increase in the number of resorbed and malformed fetuses (Gale 1978). Guinea pigs fed 50 ppm $Cr^{+3}$ for 21 weeks showed no adverse effects (Preston et al. 1976). Similar results were observed among rats consuming water containing 25 ppm $Cr^{+6}$ for 1 year (Mackenzie et al. 1958).

Injection of 0.002 to 0.05 mg $CrO_3$ ($Cr^{+6}$)/ chicken egg produced a dose dependent decrease in egg viability and increased frequency of malformed embryos (Gilani and Marano 1979). In contrast, adult black ducks fed a diet containing 0, 10, or 50 ppm $Cr^{+3}$ for 10 months displayed normal growth and reproduction (Haseltine et al. unpublished data). While no malformations were observed among ducklings from treated birds, growth and survivorship was reduced. Heinz and Haseltine (1981) observed no effects on avoidance behavior of black ducklings fed a diet containing 20 or 200 ppm $Cr^{+3}$.

**Lead.** Lead is a comparatively rare metal, averaging 16 ppm in the earth's crust, that is neither essential nor beneficial in living organisms (Eisler 1988b). Lead has adverse effects on survival, growth, reproduction, development, behavior, learning, and metabolism. In general, organic lead compounds are more toxic than inorganic compounds, biomagnification of lead is minimal, and younger organisms are more susceptible to lead toxicity (Eisler 1988b).

Acute oral doses of 5-108 mg lead/kg body weight reduced rat survival (Eisler 1988b), and rats fed diets with 5 ppm lead had shortened life spans (Schroeder et al. 1965). An acute $LD_{50}$ based on a single oral dose of 12 mg tetraethyllead/kg body weight was reported by Branica and Konrad (1980). Rats fed 0.5, 5, 25, or 250 ppm inorganic lead in diets over two generations exhibited no substantial developmental effects (Kimmel et al. 1980). In another study, Azar et al. (1973) fed rats a diet containing 0, 10, 50, 100, 1000, or 2000 ppm lead acetate for three generations. While the number of litters and young per litter was not affected by any dose level, growth was reduced and kidney histopathologies were observed among offspring in the 1000 and 2000 ppm treatments. Frequency of pregnancy was reduced in mice ingesting 3 mg/kg body weight tetraethyllead daily, and daily ingestion of 1.5 mg/kg tetraethyllead chloride resulted in a reduction in the success of implanted ova (Clark 1979).

Anemia and other hematological effects were induced among pigeons orally dosed with 6.25 mg lead/kg body weight per day (Anders et al. 1982). Kendall and Scanlon (1981) exposed ring doves to drinking water containing 0 or 100 ppm lead and observed no effects on time to produce eggs, egg production, or fertility. However, testes weight and sperm count was decreased among lead-exposed males. Grandjean (1976) correlated eggshell thickness and eggshell lead levels in European kestrels (*Falco tinnunculus*), suggesting that lead may cause

eggshell thinning. Among American kestrels (*Falco sparverius*) fed diets containing 0, 10, or 50 ppm lead, no adverse effects on survival, egg laying, initiation of incubation, egg fertility, or eggshell thickness were observed (Pattee 1984).

**Mercury.** Mercury has no known biological function and is potentially toxic to fish and wildlife. Mercury is a mutagen, teratogen, and carcinogen that adversely affects the central nervous, renal, and reproductive systems of wildlife (Eisler 1987b). Inorganic mercury compounds in aquatic systems are readily converted to organomercury by microbial action (Berlin 1979), with organomercury compounds being more toxic than inorganic mercury compounds. Biota bioconcentrate mercury compounds, which can be further biomagnified through food chains (Wren 1986).

Daily doses of 0.1-0.5 mg/kg body weight per day and dietary concentrations of 1.0-5.0 ppm are lethal to sensitive mammals (Eisler 1987b). Central nervous system toxicity, weight loss, and mortality were observed among rats fed a diet containing 250 ppm methylmercury (MeHg) for two weeks (Verschuuren et al. 1976a). Rats consuming 2.5 ppm MeHg in the diet for 2 years displayed reduced growth, increased kidney weight, and altered kidney histochemistry (Verschuuren et al. 1976b). To study effects on reproduction, Verschuuren et al. (1976c) fed rats a diet containing 0, 0.1, 0.5, and 2.5 ppm MeHg for three generations. While no effects were observed among rats fed 0.1 or 0.5 ppm MeHg, offspring viability was reduced among rats in the 2.5 ppm treatment. Among mink, 93 days consumption of diets containing 1.8 to 15.0 ppm MeHg produced mortality, ataxia, anorexia, and paralysis (Wobeser et al. 1976), with the highest exposures showing the greatest effects.

The $LD_{50}$ for MeHg for *Coturnix* quail ranges from 14.4 to 33.7 mg/kg body weight (Eisler 1987b). Growth was decreased and mortality increased among leghorn cockerels fed diets containing 6 to 18 ppm MeHg (Fimreite 1970). Ring-necked pheasants fed diets of MeHg-treated grains displayed reduced egg production and hatchability and laid more shell-less eggs than controls (Fimreite 1971). Heinz (1979) fed mallard ducks a diet containing 0.5 ppm MeHg for three generations. While MeHg consumption did not affect adult weights or weight change during the reproductive season, MeHg-exposed females laid fewer eggs (with more eggs outside the nest box), produced fewer young, and displayed slightly thinner eggshells. Young of MeHg-treated adults were less responsive to maternal calls and hyper-responsive to fright stimuli.

**Selenium.** While selenium is an essential nutrient that interacts with Vitamin E and maintains muscle integrity, it has a very narrow tolerance range; in humans, while 0.04-0.1 ppm is required in diet, 4 ppm may produce toxic effects (Eisler 1985b). In mammals, chronic selenium poisoning is induced by diets containing 1-44 ppm selenium (Harr 1978). Symptoms include liver cirrhosis, lameness, loss of hair, emaciation, reduced conception and increased fetal resorption. Plants convert inorganic selenium to organic selenium compounds, thus increasing their biological availability (Lo and Sandi 1980).

To evaluate the effects of selenium on reproduction, Schroeder and Mitchener (1971) exposed mice to 3 ppm selenate in drinking water for three generations. This dosage level increased juvenile mortality and number of runts and resulted in reproductive failure by the third generation. In another study, exposure to 3 ppm selenate or selenite in water for a lifetime had no effect on mouse longevity, and no tumorigenicity was observed (Schroeder and Mitchener 1972).

Selenium is both embryotoxic and teratogenic to birds, with organic selenium (selenomethionine) being more toxic than inorganic selenium (Hoffman and Heinz 1988). Mallard ducks were fed diets containing 1, 5, 10, 25, or 100 ppm selenite (Heinz et al. 1987) or 1, 2, 4, 8, or 16 ppm selenomethionine (Heinz et al. 1989) for about 10 weeks. Exposure to 1, 5, or 10 ppm selenite or 1, 2, or 4 ppm selenomethionine in the diet had no effect on survival, growth, or reproductive success of adults. The diet containing 100 ppm selenite killed 11 of 12 adults. While only one adult receiving the 25 ppm diet died, time to laying and interval between eggs was increased, and duckling survivorship was reduced in this treatment (Heinz et al. 1987). Diets containing 8 and 16 ppm selenomethionine resulted in 6.8% and 67.9% malformed embryos, respectively. In addition, duckling survival was significantly reduced (Heinz et al. 1989).

The most visible incident of environmental selenium toxicity occurred at the Kesterson National Wildlife Refuge in California. Agricultural wastewater containing ~ 0.3 ppm selenium was used for marsh management at the refuge (Ohlendorf et al. 1986). Mean selenium concentrations in plants, invertebrates, and fish at the site were 22-175 ppm (dry weight). As a result, reproductive success among water birds was poor, and the incidence of embryo mortality and developmental abnormalities was dramatically increased. Raccoons on the refuge were found to bioaccumulate selenium (Clark et al. 1989). While peak births at the refuge occurred 2 months later than reported at other locations, no adverse effects on raccoon reproduction were observed.

Metabolism of selenium may be significantly modified through interactions with heavy metals, and selenium may provide some protection from adverse effects associated with various metals, including cadmium and mercury (Eisler 1985b). Arsenite inhibits methylation of selenium but increases fecal excretion of selenite (Venugopal and Luckey 1978).

**Thallium.** Thallium is a widely distributed metal, occurring at concentrations of ~ 1 ppm in the earth's crust (Kazantzis 1979). Principal systems affected by thallium exposure include nervous and digestive; renal damage and hair loss have also been observed. Thallium sulfate, which has been widely used as a rodenticide, has an acute oral $LD_{50}$ of 16 mg/kg (Ware 1978). In chronic studies, rats tolerated a dose of 10 mg thallium acetate/kg, while 30 mg/kg was lethal to males by 15 weeks. All rats fed a daily dose of 0.45 mg thallium/kg died after 4 months (Kazantzis 1979). Rats exposed to 10 ppm thallium in drinking water for 2 months accumulated thallium in testes and exhibited signs of testicular toxicity including reduced sperm motility (Formigli et al. 1986).

When Bean and Hudson (1976) orally dosed three golden eagles with 60 and 120 mg thallium sulfate ($Tl_2SO_4$)/kg body weight, the bird receiving 60 mg $Tl_2SO_4$/kg survived while the two dosed with 120 mg $Tl_2SO_4$/kg died, suggesting an $LD_{50}$ between the doses. Oral $LD_{50}$s for quail, geese, and ducks are 12, 15, and 30 mg/kg, respectively (Shaw 1933). No long-term studies of thallium toxicity to birds are currently available.

**Vanadium.** Vanadium is a metallic element that occurs in six oxidation states and numerous inorganic compounds. The toxicity of vanadium depends on its physicochemical state, particularly on its valence state and solubility. Based on acute toxicity, pentavalent ammonium vanadate ($NH_4VO_3$) has been reported to be more than twice as toxic as trivalent vanadium trichloride ($VCl_3$) and more than 6 times as toxic as divalent vanadium iodide ($VI_2$). Pentavalent vanadium pentoxide ($V_2O_5$) has been reported to be more than 5 times as toxic as trivalent vanadium (III) trioxide ($V_2O_3$) (Roschin 1967). In animals, acutely toxic oral

doses cause vasoconstriction, diffuse desquamative enteritis, congestion and fatty degeneration of the liver, congestion and focal hemorrhages in the lungs and adrenal cortex (Gosselin et al. 1984). Minimal effects seen after subchronic oral exposures to animals include diarrhea, altered renal function, and decreases in erythrocyte counts, hemoglobin, and hematocrit (Domingo et al. 1985, Zaporowska and Wasilewski 1991).

A vanadyl sulfate concentration of 5 μg/mL in drinking water, plus a vanadium level of 3.2 μg/g in the diet (4.1 mg vanadium/kg total) of mice, was reported to cause no adverse effects over a lifetime exposure period (Schroeder and Balassa 1967). In similar lifetime studies, rats and mice exhibited no adverse effects when exposed to 5 ppm vanadium (as vanadyl sulfate) in drinking water (Schroeder et al. 1970, Schroeder and Mitchener 1975b). The estimated dose levels were 0.7 mg vanadium/kg per day for rats and 0.9 mg vanadium/kg per day for mice. Vanadium pentoxide in the diet of rats at levels of 10 and 100 ppm for their entire lifetime resulted in no significant toxicological effects except for a reduction in hair cystine content (Stokinger 1981).

White and Dieter (1978) observed no mortality among mallard ducks fed diets containing 1, 10, or 100 ppm vanadyl sulfate for 12 weeks. Altered lipid metabolism was observed among birds fed 100 ppm vanadium; no other effects were observed. Among chickens, 200 to 400 ppm $Ca_2(VO_4)_2$ in the diet produced 100% mortality; weight gain decreased among chicks fed 20 to 40 ppm $Ca_2(VO_4)_2$ (Romoser et al. 1961).

### 6.3.2 Media Toxicity Data

Toxicity tests were performed on ash and water from Chestnut Ridge OU 2. These data are presented and discussed in detail in Sect. 3.2.6.1. and are briefly summarized here.

#### 6.3.2.1 Aquatic toxicity tests

To determine if surface waters from Chestnut Ridge OU 2 are toxic to aquatic organisms, 7-day static renewal fathead minnow and *Ceriodaphnia dubia* tests were performed in the laboratory. These tests were used to evaluate the effects of water-borne contaminants on minnow survival and growth and *Ceriodaphnia* survival and reproduction. Standard EPA methods (method 1000.0, fathead minnow; method 1002.0, *Ceriodaphnia*; Weber et al. 1989) were employed for both tests. In addition, 4-day feeding rate tests were conducted using the snail, *Elimia clavaeformis.* While not EPA-approved, these snail tests have been used to evaluate water quality at the inlet and outfall of Lake Reality in upper East Fork Poplar Creek (Hinzman 1993) and to evaluate storm water quality entering White Oak Creek (Hinzman et al. 1993).

The results of these tests are presented in Sect. 3.2.6.1. Neither fathead minnow nor *Ceriodaphnia* tests provided strong indications of toxicity of surface waters from Chestnut Ridge OU 2 to these species. For 3 of 11 fathead minnow test periods, survival was reduced (<80%); however, the growth rate of minnow larvae in these test periods was high. These samples also displayed high among-replicate variation, suggesting effects from pathogenic bacteria or fungi, not contaminants. In 21 of 24 *Ceriodaphnia* site-test combinations, survivorship was ≥90%. In 18 of 24 site-test combinations, *Ceriodaphnia* reproduction was ≥20 offspring/female, well above the EPA's criterion for acceptability of controls (15

offspring/female). For *Elimia* snails, feeding rates in Chestnut Ridge OU 2 surface water were very similar to those observed for snails in diluted mineral water, providing little evidence of toxicity.

### 6.3.2.2 Soil toxicity tests

To evaluate the toxicity of ash to soil invertebrates, laboratory and in situ earthworm toxicity tests were performed. Methods for these tests are described in detail in Sect. 3.2.6.1. The laboratory earthworm toxicity tests in general followed the EPA-approved procedure described by Green et al. (1989). Tests were performed on ash from the Willow and Sycamore areas of the FCAP and on ash from the Sluice Channel Area. For comparison, tests were performed on soil from the reference area and on artificial soil. An additional treatment was added in which soils from all sources (e.g., from Chestnut Ridge OU 2, the reference site, and artificial soil) were autoclaved to eliminate soil pathogens. Survivorship and growth was evaluated after 14 days. The in situ earthworm tests were conducted by placing earthworms in buckets sunk into the ash at six locations in the FCAP and observing survivorship after 14 days. Four buckets were used at each location: two filled with ash and two filled with peat moss. One pair of buckets (ash and peat filled) at each location received fermented alfalfa as food for earthworms.

The results of these tests are presented in Sect. 3.2.6.1. The two earthworm tests showed that FCAP soils might be problematic for soil organisms. In the in situ test, earthworm survival in FCAP soil at six sites was substantially lower than in peat moss. The addition of food increased survival in both soil types, but survival even in food-augmented FCAP soil was still very low compared even to non-amended peat moss. The low survival of the worms in the FCAP soils did not appear to be explained by pH, water, or organic content of the soil. In the laboratory test, soil from the FCAP Sycamore site adversely affected earthworm survival, and, on average, worms lost more body weight in all FCAP soil treatments than they did in any of the control- or reference-soil treatments. This difference was nearly significant ($p = 0.066$). On average, autoclaving the FCAP soils used in this test increased mean survival of the earthworms, but this effect was not statistically significant ($p = 0.152$). Additionally, mean survival even in autoclaved Sycamore site soil was only 52.5%. FCAP soil texture, or soil-associated contaminants, are factors that could account for earthworm mortality in the in situ and laboratory tests and/or weight-loss patterns noted in the laboratory test.

An additional study was performed in which the effects of fly ash on the biomass of a bacteria involved in symbiotic relationships with leguminous plants was examined (see Sect. 3.2.6.6). *Rhizobium leguminosarum* was cultured in fly ash and sand mixtures and phospholipids were extracted and quantified using thin layer, chromatography-flame, ionization detection. The concentration of all phospholipids decreased with increasing fly ash. On the basis of phospholipid concentration, fly ash alone was able to support less than 10% of the biomass of control cultures (0% fly ash). These results suggest that fly ash is toxic to these microorganisms or is physically unsuitable as a growth substrate. This may have implications for the ability of leguminous plants to colonize nitrogen-poor ash deposits that do not support populations of *Rhizobium* inoculum.

### 6.3.3 Biological Survey Data

To determine population levels and contaminant loadings of biota resident at Chestnut Ridge OU 2, biological surveys were conducted. Results of these surveys are presented and described in detail in Sects. 3.2.6.2 through 3.2.6.6 and are briefly summarized here.

#### 6.3.3.1 Benthic macroinvertebrate surveys

Benthic macroinvertebrates were sampled from two locations in UMB (MCK 2.03 and MCK 2.53) above Rogers Quarry and from a reference site in the upper reaches of White Oak Creek (WCK 6.8). Surveys were conducted on July 15, 1992, and again on March 30, 1993. Sampling methods employed, results, and conclusions are discussed in detail in Sect. 3.2.6.2.

Data from the surveys exhibited no evidence of strong impact at either UMB site. Except for minor differences in taxonomic composition and the relative abundance of existing taxa, the communities existing at both UMB sites and White Oak Creek in March 1993 appeared to be similarly balanced (i.e., a variety of taxa) and to contain several taxa within groups, such as Ephemeroptera, Plecoptera, and Trichoptera, that are relatively intolerant of pollutants (Lenat 1988). The only exception to this may have been at MCK 2.03 where chironomids comprised a much greater proportion of the community than at either MCK 2.53 or WCK 6.8. As conditions become more stressed, the proportion of chironomids generally increases (Wiederholm 1984, Smith 1993). However, since this was a general survey with only one sample being collected in March, this difference could be due to sampling error and thus an artifact.

Taxa richness values were comparable among the three sites in March, which would suggest that if any impacts were occurring, they were minor. However, differences between WCK 6.8 and the two UMB sites in July 1992 were much greater. The benthic community in UMB during summer low flow was dominated by midge taxa with reduced Total and EPT richness. Under higher flow conditions during March, the community was more balanced with values for total and EPT richness comparable to those for White Oak Creek. Thus, these results suggest that water quality in UMB may be impacted by past operations or existing conditions in the watershed. Since arsenic and selenium have previously been implicated as two possible contaminants contributing to moderate stress in the invertebrates in UMB (Tolbert and Smith 1992), it is possible that these data may also be indicative of stress from these metals or some other unidentified factor(s). During summer, when flows are lowest and temperatures are highest, the aquatic communities may be exposed to worst case conditions; thus, any stress associated with contaminants could be amplified.

As an alternative hypothesis, it is also possible that the differences between WCK 6.8 and the UMB sites are reflective of natural regional variability. The differences in taxa composition may also show that White Oak Creek has more available habitat and more stable flow conditions than UMB. Considerable variability exists not only among relatively undisturbed streams flowing south off Chestnut Ridge (Smith 1993), but also among several streams in and near the ORR (Smith 1992). Since UMB is unique among streams in the area in having had fly ash discharged into its headwaters, the ability to distinguish natural variability from changes due to impact will be difficult without more rigorously designed field and laboratory studies.

### 6.3.3.2 Soil invertebrate surveys

A survey for soil invertebrates was conducted in July 1993 (see Sect. 3.2.6.3). Five 1-ft$^3$ samples of ash were collected from the FCAP and hand sorted. A total of two beetle larvae and five millipedes were collected in all five samples. Two additional samples were collected from the Sluice Channel Area. These samples yielded similar organisms in similar abundance to those from the FCAP. Based upon these samples, earthworms do not appear to be present in the ash of the FCAP or the Sluice Channel Area. The specimens collected provided insufficient biomass for laboratory analysis. Therefore, the degree of contaminant bioaccumulation in soil invertebrates is unknown. The apparent absence of earthworms in the ash suggests that this may not be a significant pathway for contaminant exposure to vermivores (e.g., shrews, robins, woodcock, etc.)

### 6.3.3.3 Heavy metal bioaccumulation in vegetation

To determine if elements in coal ash bioaccumulate in plants and would therefore present a risk to herbivorous wildlife, vegetation at Chestnut Ridge OU 2 was sampled and metal content was determined. These data are reported in Sect. 3.2.6.4. Foliage samples from mixed deciduous trees (species listed in Sect. 3.2.6.4.), eastern red cedar (*Juniperus virginiana*), and groundcover species (*Eulalia viminea* and *Lonicera* sp.) were collected and concentrations of As, Cd, Cr, Pb, Hg, Se, and Tl in foliage were determined. Vegetation samples were collected from two areas on the FCAP (the Sycamore or FCAP-1 area and the Willow or FCAP-2 area), the Sluice Channel Area (SLUICE-1), with samples from the Walker Branch Watershed used as controls. Below is a brief summary of the results of these analyses; a more detailed discussion may be found in Sect. 3.2.6.4.

Thallium was not above the detection limit for any of the 70 vegetation samples analyzed. Lead was not detected in any samples from Walker Branch or the Sluice Channel Area and was observed in only 2 of 39 samples from the FCAP. Arsenic was not above the detection limit in any sample from the Sluice Channel Area or Walker Branch. Arsenic concentrations in deciduous foliage from FCAP-1 and FCAP-2 were significantly greater than those from Walker Branch. Cadmium was not detected in any eastern red cedar or groundcover samples; concentrations in FCAP-2 deciduous foliage were significantly greater than those from FCAP-1, the Sluice Channel Area, or Walker Branch. Chromium was not detected above detection limits in any samples except for groundcover foliage from the FCAP. Mercury concentrations in all foliage from the FCAP and Sluice Channel Area were below or slightly above the detection limit (0.05 $\mu$g/g); levels in deciduous foliage from Walker Branch were significantly greater than those from the other areas. Selenium concentrations in deciduous foliage from both FCAP sites were significantly greater than those at either the Sluice Channel Area or Walker Branch. Selenium in groundcover was also elevated at FCAP-1 and the Sluice Channel Area in comparison to Walker Branch.

These data show that some of the metals in the ash, arsenic and selenium in particular, are taken up by plants and retained in foliage. They therefore may be consumed by herbivores that forage within Chestnut Ridge OU 2.

### 6.3.3.4 Heavy metal bioaccumulation in small mammals

To determine if elements in coal ash bioaccumulate in small mammals and would therefore present a risk to predators, small mammals at Chestnut Ridge OU 2 were sampled

and metal content was determined. These data are reported in Sect. 3.2.6.5. Small mammals were collected from two areas on the FCAP (the Sycamore, or FCAP-1 area, and the Willow, or FCAP-2 area), the Sluice Channel Area (SLUICE-1), and the Walker Branch Watershed (the control area). Whole body concentrations of As, Cd, Cr, Pb, Hg, Se, and Tl in small mammals were determined. Below is a brief summary of the results of these analyses; a more detailed discussion may be found in Sect. 3.2.6.4.

A total of 24 specimens representing 5 species were collected from all 4 locations (18 from Chestnut Ridge OU 2 and 6 from Walker Branch). Concentrations of As, Se, and Pb in small mammals collected at Chestnut Ridge OU 2 were greater than those found in small mammals from the reference site. Cd, Hg, and Cr bioaccumulated in small mammals from Chestnut Ridge OU 2. However, concentrations were relatively low and comparable to those from the reference site and published values. With the exception of two animals from the FCAP, thallium was not observed in small mammals.

These data show that small mammals from Chestnut Ridge OU 2 bioaccumulate metals from ash. Small mammals therefore may provide a contaminant transfer pathway for predators that consume small mammals from Chestnut Ridge OU 2.

#### 6.3.3.5 Banded sculpin reintroduction study

To determine if UMB can support a fish population and, if so, how that population might differ from an existing population in a noncontaminated reference stream, a fish introduction study was initiated. In this study, banded sculpins (*Cottus carolinae*) were introduced into UMB and survival, distribution, density, biomass, condition factor, feeding patterns, reproduction, and recruitment were observed. This study is not yet complete and only preliminary data are available. A more detailed description of this study is found in Sect. 3.2.6.6. To date, sculpins have been recovered from UMB, indicating that the water is not acutely toxic to sculpins. However, the number of sculpins recovered has been variable. Several factors that may account for this variability include age-dependent mortality, starvation, predation, cannibalism, introduction-related stress mortality, immigration, emigration, reproduction, pollutant-related stress mortality, changes in sampling techniques, and ineffective sampling of certain habitats.

## 6.4 RISK CHARACTERIZATION

### Introduction

Risk characterization is the phase of risk assessment in which the information concerning exposure (Sect. 6.2) and the information concerning the potential effects of exposure (Sect. 6.3) are integrated to estimate risks (the likelihood of effects given the exposure). Risk characterization in ERA is performed by a weight-of-evidence analysis. The principle lines of evidence concerning effects are biological survey data that indicate the actual state of the receiving environment, media toxicity data that indicate whether the contaminated media are toxic under controlled conditions, and single chemical toxicity data that indicate the toxic effects of the concentrations measured in site media. Procedurally, the risk characterization is performed for each assessment endpoint by (1) screening all measured contaminants against toxicological benchmarks and background concentrations, (2) estimating the effects of the contaminants retained by the screening analysis, (3) estimating the toxicity of the ambient

media based on the media toxicity test results, (4) estimating the effects of exposure on the 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.

### Risk Characterization for Contaminant Screening

Contaminant concentrations potentially toxic to the endpoint biota were identified by comparing the site media concentrations to background levels and toxicological benchmarks. Many of the contaminants commonly found during an RI also occur naturally in the environment. The concentrations of these chemicals found at local background (unimpacted) sites were assumed to be nonhazardous. To ensure that risk management decisions are based on the risks posed by site contaminants, the background constituents must be differentiated from the site-associated contaminants. For each area and medium, the detected contaminants and the detection limits were compared to background. The naturally occurring inorganic chemicals and the ubiquitous radionuclides were compared to local background concentrations. Synthetic organic compounds should not be present at background sites. Any organic compound that was detected and validated was considered above background. The maximum detected contaminant concentration, or the UCL95, was used for detected analytes. For nondetected analytes the maximum nondetected value was used. Nondetected organics without benchmarks were not screened. The resulting list would be relatively long and meaningless, given the lack of criteria for their evaluation. To be conservative, the detected and nondetected values were compared to the lowest of either the UCL95, or the maximum detected value, of the reference data. Contaminants above background levels were then compared to the benchmarks.

Because there is no guidance for screening benchmarks, a set of alternative benchmarks were derived for contaminants found at the ORR (Hull and Suter 1993; Opresko et al. 1993; and Suter et al. 1992b, 1993a). Benchmark development is an ongoing process. Although the methods for derivation remain the same, the most current benchmarks are used. The ecological risks posed by contaminants are discussed as they relate to each endpoint community. These communities may have multiple sources of exposure (e.g., ash, soil, surface water). There also may be more than one set of applicable benchmarks per endpoint community. With the exception of the wildlife benchmarks, each source medium in each area was screened separately against the appropriate benchmarks. The screening results were then integrated for each community. For wildlife, the potential sources of exposure were integrated in the benchmark screening procedure.

### Risk Characterization from Contaminant Concentrations

The screening assessment indicates which contaminants are potentially toxic to the endpoint biota but does not indicate what effects would be expected to occur. Therefore, the next step in the assessment is a comparison of the distributions in space or time of concentrations in the medium being considered to the distributions of effects with increasing concentration derived from the literature (the toxicity profiles, Sect. 6.3.1). For each contaminant, the exposure is expressed variously as a range, a sample mean, and a UCL95 on the population mean. For each contaminant, the effects are expressed as increasing intensity of effects and increasing numbers of species or processes affected as exposure increases. Integration of these two distributions results in a distribution of possible types and levels of effects that will be expressed as a most likely effect and a reasonable maximum effect

on the endpoint. This characterization must also consider the form and speciation of the contaminants in the toxicity studies and in the media on the OU to ensure that the most appropriate toxicity data are used.

### Risk Characterization from Media Toxicity Tests

Risk characterization for the media toxicity tests involves interpretation of the test results in terms of the exposures experienced by the endpoint biota in the field. That is, the durations of the tests relative to the duration of field exposures, the level of contamination at the times and locations sampled relative to the ranges of contamination found in the media analyses, and the influence of sample preparation on the exposure are used to help interpret the effects seen in the tests as indicators of toxic effects in the field. In some cases, there are validation studies for the media toxicity tests that can be used to help interpret the test results in terms of field effects.

### Risk Characterization from Biological Surveys

Risk characterization for the biological survey data involves interpretation of the survey results on terms of the exposures experienced by the surveyed biota. For example, if the exposure levels and the surveyed community characteristics change over time, do the survey results include the period of maximum effects, and, if not, what might those maximum effects be? It also involves interpretation of the survey data in terms of the endpoint definitions. That is, are the survey data consistent with types and levels of effects prescribed in the definition of the assessment endpoints?

### Risk Characterization from Weight-of-Evidence

The weighing of evidence is performed by applying professional judgment to the evidence in the following manner. First, risk characterizations based on biological survey results provide the best evidence of the true effects. For example, no matter how many contaminants occur at concentrations that are lethal to plants, we know that the ash is not lethal to some species of plants because they live on the site. However, the ash could be lethal to some other plant species and it could decrease growth or yield of all plant species. Therefore, the bounds on the possible biological effects indicated by the survey data are characterized. Second, risk characterizations based on toxicity tests provide the best evidence concerning whether toxicity may be responsible for apparent effects or whether toxic effects may be occurring that are not apparent in the survey data. However, their reliability is limited by the limited durations, species, and responses included in the tests, so negative results are not definitive evidence that there is no toxicity. Third, risk characterizations based on contaminant analyses provide evidence concerning which contaminants are responsible for any apparent effects or observed toxicity and concerning whether toxic effects may be occurring that were not detected by the surveys or the media toxicity tests. However, the application of laboratory toxicity data for individual contaminants to measured concentrations of individual contaminants does not address the combined effects of multiple contaminants, the effects of site-specific conditions on contaminant availability and toxicity, or the range of species, life stages, and responses that occur on the site.

Not all of these lines of evidence are available for all endpoints, and the reliability of each line of evidence varies among endpoints. However, there is always a gradient from greatest realism in the biological survey data to best characterization of causation in the

single contaminant toxicity data. The weighing of evidence balances these relative strengths and weaknesses to realistically estimate effects that are caused by the total exposures.

In addition, if prior studies of coal ash have addressed the assessment endpoint, the results of those studies are discussed. Although they are not directly applicable to the Chestnut Ridge OU 2 ash and disposal conditions, they serve to indicate whether the results from assessment of this site are consistent with other ashes at other sites. Such consistency lends credibility to the results.

**Uncertainties**

Following the characterization of risks to each of the ecological assessment endpoints, uncertainties that may affect the conclusions of the assessment are listed and, as far as possible, quantified.

### 6.4.1 Risk Integration for Current Conditions

#### 6.4.1.1 Fish

The evidence concerning risks to fish consists of a survey of fish in UMB, toxicity tests of water from UMB, and analyses of chemicals in water. These lines of evidence are summarized in the following text and in Table 6.39.

**Contaminant Screening of Surface Water**

Contaminants of potential ecological concern in surface water were identified by comparing the concentrations at FCAP and UMB to the reference and benchmark levels. As discussed in Sect. 6.2, fish and water-exposed invertebrates are exposed to contaminant concentrations that vary spatially and temporally. Therefore, for screening the detected concentrations the UCL95 was used for the FCAP and UMB surface water data. To be conservative, these values were compared to the lowest of either the UCL95 or the maximum detected value of the reference data. Inorganics and radionuclides that exceeded background and the detected organic compounds were then compared to the benchmarks (Table 6.27). The same procedure was conducted using the maximum nondetected value. The UCL95 and the maximum nondetected value were also used for the comparison to benchmarks. The results of these screenings are presented in Tables 6.40 through 6.43.

As noted in Sect. 6.1.3, this is a conservative screening because only total recoverable concentrations were measured. Uranium concentrations were converted from pCi/L to mg/L and compared to the available benchmarks. Although no benchmarks are available for the radionuclides, the isotopes of uranium were also carried through the screening process.

The hardness of surface water significantly affects the toxicities of several metals, including copper and zinc. Toxicity of hardness dependent metals increases as hardness decreases. Hardness was measured in water samples from UMB used for the media toxicity tests (Sect. 3.2.6.1 and Table 3.32 in Chap. 3). Hardness was not measured at the FCAP. The benchmark values used for this assessment are based on a hardness of 100 mg/L. Because this is less than both the measured values at UMB and the average values for the ORR, the benchmarks were not adjusted.

The NAWQC for PCP is pH dependent (TDEC 1991). The formulas used are: acute criteria = exp(1.005(pH) - 4.830); chronic criteria = exp(1.005(pH) - 5.290). A pH of 7.8 is used for the standard acute (20 $\mu$g/L) and chronic (13 $\mu$g/L) criteria. Two pH values were reported for FCAP pond water, 6.10 and 6.44 (Chap. 3). Over a 3 year period, water for 11 toxicity tests was collected from MCK 2.03 and tested for physicochemical water quality factors, including pH. The pH ranged from 7.49 to 8.26 (Sect. 3.2.6). The PCP criteria were not changed for UMB. To be conservative, a pH of 6.10 was used to adjust the acute and chronic criteria at the FCAP to 3.7 and 2.3 $\mu$g/L, respectively.

The toxicities of arsenic, chromium, and mercury are dependent upon their chemical species. To support the HHRA, total chromium and chromium VI concentrations were determined in three of the six surface water samples from Chestnut Ridge OU 2. Chromium VI was not detected in any of these samples. Therefore, the total chromium concentrations were assumed to represent the trivalent form of chromium. Arsenic and mercury, which were not speciated for this RI, were conservatively assumed to be in their most toxic forms (Sect. 6.2.3).

Eight chemicals detected in UMB water and seven detected in the FCAP water exceeded both background concentrations and one or more aquatic benchmarks (Tables 6.40 and 6.41). These COPCs are discussed in the next section. Of the COPCs, three in UMB and four on the FCAP exceed NAWQC, which are also Tennessee water quality standards and ARARs. In addition, six chemicals in the FCAP water and seven in UMB water were not detected, but the detection limits exceeded benchmarks (Tables 6.42 and 6.43).

### Aqueous Concentrations of COPCs

**Aluminum.** The sample mean as well as the UCL concentrations of aluminum in water from both the FCAP and UMB exceeded both acute and chronic NAWQC as well as lowest chronic values (LCVs) and chronic $EC_{20}$s for both fish and daphnids. These are whole water concentrations, so they may significantly overestimate the bioavailable concentrations.

**Barium.** There are no known toxicity data for fish exposed to barium. However, the concentrations reported from the FCAP and UMB are well below the concentrations reported to be toxic to aquatic plants and to a marine crustacean and are below the Illinois water quality standard for indigenous aquatic life.

**Chromium.** The UCL concentration for chromium in UMB exceeded the sensitive species $EC_{20}$, which is the most conservative benchmark, corresponding to a 20% reduction in the life-cycle production of a species that lies at the fifth percentile of the species sensitivity distribution specified by the EPA. Chromium was not detected in the FCAP water but the detection limit there exceeded fish and daphnid $EC_{20}$s and LCVs.

**Copper.** The sample mean as well as the UCL concentrations of copper from both the FCAP and UMB exceeded LCVs and chronic $EC_{20}$s for both fish and daphnids. Copper concentrations below those measured on the OU cause significant decreases in survivorship of early life stages in a subchronic test with brook trout (EPA 1985c). The water analyses are whole water concentrations so they may significantly overestimate the bioavailable concentrations.

**Iron.** The sample mean as well as the UCL concentrations of iron in water from both the FCAP and UMB exceeded chronic NAWQC as well as LCVs and chronic $EC_{20}$s for daphnids. These are whole water concentrations so they may significantly overestimate the bioavailable concentrations.

**Lead.** UCL concentrations of lead from the FCAP exceeded the chronic NAWQC, and the sample mean was slightly below the criterion.

**Manganese.** There are no criteria or standards for manganese. However, manganese concentrations on the FCAP exceeded $EC_{20}$s and LCVs for fish and daphnids and those in UMB exceeded the $EC_{20}$ for fish populations.

**Zinc.** The sample mean as well as the UCL concentrations of zinc in water from the FCAP exceeded both acute and chronic NAWQC as well as LCVs for both fish and daphnids and chronic $EC_{20}$s for fish and their populations. In UMB, zinc concentrations exceeded only the sensitive species $EC_{20}$.

**Bis(2-ethylhexyl)phthalate.** There are no criteria or standards for bis(2-ethylhexyl)phthalate. However, concentrations from UMB exceeded $EC_{20}$s and LCVs for fish and daphnids. This phthalate ester was not detected in water from the FCAP but the limits of detection at that site were above the benchmarks. The source of phthalate esters in this OU is unknown. They are unlikely to be a constituent of ash.

**Media Toxicity Data**

The standard EPA 7-day tests of fathead minnow larvae and *Ceriodaphnia* indicate that the water is nontoxic in most sampling periods. In 8 of 11 fathead minnow tests and 21 of 24 *Ceriodaphnia* tests responses were normal. In the other cases, survivorship was reduced, but the individuals who performed the tests believe that effects were due to pathogens or other extraneous factors. The snail feeding studies showed no sign of toxicity.

The chief limitations on these tests are the small number species involved and the short durations of the tests. Exposures in the field cover full life cycles rather than the "sensitive life stages" in the tests and include many more species than the test set. However, the validation studies carried out by the EPA indicate that the 7-day test set protects fish (and benthic invertebrate communities) from severe effects, at least when recolonization from upstream reaches is possible (Mount et al. 1984, 1985, 1986a, 1986b; Mount and Norberg-King 1985, 1986; Norberg-King and Mount 1986).

The minimal processing of the water samples and minimal holding times should ensure that the toxicity of the water is not significantly modified. Therefore, the exposures in the tests realistically represent the field exposures at the time of sample collection. Because the number of tests was relatively high (11 minnow and 24 *Ceriodaphnia* tests), and because the sampling dates included a period when part of UMB was dry, the tests can be interpreted as representative sample of water from low to normal flows. However, they do not include samples from storm events that are likely to suspend ash and flush dissolved metals out of the deposited ash. Therefore, they may miss acutely toxic events.

**Biological Survey Data**

The two ponds on the FCAP are small and isolated and therefore would not be expected to support fish. However, the absence of fish from UMB is remarkable in that similar streams in the area support diverse fish communities (Ryon et al. 1992). Because there is no route for recolonization of UMB by fish, their absence may reflect exposures that occurred when ash was being actively sluiced into the OU rather than current conditions of exposure to ash leachate and ash in sediments.

The reintroduction of banded sculpins has produced ambiguous results except to indicate that the water in UMB is not acutely lethal to all individuals of that species. The low recoveries of introduced fish and lack of clear signs of successful reproduction could be attributed to toxic effects but could also be due to other habitat factors or to difficulties in sampling.

Fish are present in Rogers Quarry and in Lower McCoy Branch. Although these areas are outside OU 2, they receive water and ash from UMB and contain no known sources of toxic contaminants other than the coal ash. Therefore, they provide supplemental information concerning the effects of the coal ash and leachate in this watershed. Largemouth bass in both Rogers Quarry and sunfish in Lower McCoy Branch display high frequencies of gross deformities of the cranium, fins, and scales (Ryon et al. 1992). The fish community of Lower McCoy Branch had an abnormal composition for a stream of its size (Ryon et al. 1992). Largemouth bass in Rogers Quarry contained elevated concentrations of arsenic, selenium, and possibly thallium (Ryon et al. 1992). Because there are no known sources of toxic contaminants in the watershed other than coal ash, it is reasonable to assume that if a fish community were present in UMB, it would experience effects that are at least as severe.

**Weight of Evidence**

Clearly, ash disposal in this OU has had catastrophic effects on the fish community of UMB. However, it is likely that the extinction of the fish community occurred during the period of ash sluicing and that the effects resulted from mechanical effects of the ash as well as the toxicity of chemicals in the leachate. Extreme pH has also contributed to the aqueous effects of coal ash (Sect. 6.1.4) but the ash slurry from Y-12 had a near neutral mean pH of 7.76 (Turner et al. 1986). Therefore, we know that the ash has been hazardous to fish in OU 2, but the hazards may be more relevant to incidents of ash erosion into the branch than to routine leaching of chemicals. However, the ongoing effects on fish in downstream habitats suggests that the ash and water are still toxic. The reintroduction of sculpins in UMB has not yet resulted in establishment of a viable population, which is consistent with chronic toxic effects but does not strongly support such effects. Clearly, that experiment must be given more time and attention. The results of the benthic invertebrate surveys (Sect. 3.2.6) suggest that food is available for fish in UMB. There are not now, nor has there ever been, fish in the FCAP ponds.

The aqueous toxicity tests showed no clear toxicity. This indicates that there is no aqueous toxicity, toxicity is acute and episodic (i.e., associated with storm flows), or toxicity occurs in fish species or life stages (e.g., breeding adults) that are not included in the fish toxicity tests. The *Ceriodaphnia* test results more reliably represent invertebrate responses because they include reproductive endpoints. They suggest that aqueous toxicity is not limiting the availability of prey for fish. The absence of aqueous toxicity does not preclude toxicity due

to dietary exposure. The importance of dietary exposure in fish is only now becoming apparent. Data and methods to address this route of exposure are not available.

Multiple metals occur in both the FCAP ponds and UMB at concentrations that are acutely or chronically toxic to fish and other aquatic organisms in laboratory toxicity tests. The primary limitation on this line of evidence is that the bioavailability of the metals in ambient water is likely to be less than in the toxicity tests. On the other hand, the sampling may have missed episodes of high metal levels due to the flushing effect of storm events.

In sum, the biological survey data indicate that the ash has had drastic effects on fish and is probably still having effects, but they are less severe than when ash was being sluiced. The occurrence of contaminants at concentrations that are higher than background and higher than concentrations that cause toxic effects in the laboratory supports that conclusion. However, the 7-day toxicity tests performed with UMB water did not detect aqueous toxicity. The weight of evidence suggests that chronic toxic effects on fish are occurring or would occur if a fish community were present.

### 6.4.1.2 Benthic macroinvertebrates

Risk characterization involves the integration of multiple lines of evidence. The two areas of potential exposure for benthic macroinvertebrates, the FCAP and UMB, differed in their benchmark screening results and the available lines of evidence. The procedure for benchmark screening was the same for both areas and is presented in the next section. The COPCs, ambient media toxicity tests, and the biomonitoring data were integrated separately for the FCAP and UMB and are summarized in the following text and in Tables 6.44 and 6.45.

#### Contaminant Screening of Sediments

Benthic macroinvertebrate communities are potentially exposed to contaminants in the surface water and sediment. The surface water screening procedure, and a discussion of the resulting COPCs, was presented in Sect. 6.4.1.1 and Tables 6.40 through 6.43. The NAWQC, sensitive species $EC_{20}$, the daphnia $EC_{20}$, and the daphnia LCV are the aqueous benchmarks most relevant to benthic macroinvertebrates.

Contaminants of potential ecological concern in sediment were identified by comparing the concentrations at the FCAP and UMB to the reference concentrations and the benchmark values. As discussed in Sect. 6.2, the maximally exposed, sediment-associated organism could receive 100% of its exposure from the most contaminated sediment. Therefore, for screening the detected concentrations the maximum detected value was used for the FCAP and UMB sediment data. To be conservative, these values were compared to the lowest of either the UCL95, or the maximum detected value, of the reference data. Inorganics and radionuclides that exceeded background, and the detected organic compounds, were then compared to the benchmarks (Tables 6.28 through 6.31). The same procedure was conducted using the maximum nondetected value. The maximum detected and nondetected values were also used for the comparison to benchmarks.

The equilibrium partitioning approach was used to develop benchmarks for nonionic organics. The $K_{ow}$s presented in Hull and Suter (1993) were used, in conjunction with the

TOC results for each area. To be conservative, the lowest reported %TOC values (Chap. 3) were used. That is, the %TOC used for the FCAP and UMB was 2.65 and 2.84, respectively.

Two metals (arsenic and iron) detected in the FCAP sediment exceeded both the background concentrations and the benchmark levels (Table 6.46). These COPCs are discussed in the next section. Five metals (Sb, Cd, Hg, Ag, and Tl) and eleven organics not detected in the sediment had detection limits that exceeded both background and benchmarks (Table 6.47).

Seven metals (Al, Cr, Cu, Fe, Pb, Mn, and Zn) detected in the FCAP surface water exceeded both the background concentrations and at least one benchmark relevant to benthic macroinvertebrates (Table 6.40). These COPCs were discussed in Sect. 6.4.1.1. One metal, chromium VI, and five organics not detected in the surface water had detection limits that exceeded relevant benchmarks (Table 6.42).

Five metals (As, Ba, Fe, Mn, and Ni) detected in the UMB sediment exceeded both background and sediment benchmark levels (Table 6.48). These COPCs are discussed in the next section. Four metals (Sb, Cd, Hg, and Ag) not detected in the sediment had detection limits that exceeded both background concentrations and sediment benchmarks (Table 6.49).

Seven metals (Al, As, Cu, Fe, Pb, Hg, and Zn) detected in the UMB surface water exceeded both the background concentrations and at least one relevant aqueous benchmark (Table 6.41). One organic compound, bis(2-ethylhexyl)phthalate, was detected and exceeded relevant aqueous benchmarks. These COPCs were discussed in Sect. 6.4.1.1. Three metals ($Cr^{+6}$, Co, and Ni) and four organics not detected in the surface water had detection limits that exceeded relevant benchmarks (Table 6.43).

**Contaminants of Potential Concern**

**Arsenic.** The sample mean and the maximum detected concentrations of arsenic in the FCAP and UMB sediment samples exceeded both the lower (MOE) and upper (ER-L) benchmarks. The mean arsenic concentrations at the FCAP and UMB were 1.3 and 37 times higher than the upper benchmark, respectively.

Mean arsenic concentrations at both areas were at least an order of magnitude higher than lowest sediment concentrations, 3.7 mg/L and 7.4 mg/L, respectively, associated with reduced taxa richness in two Illinois rivers (Long and Morgan 1991). Concentrations less than those at Chestnut Ridge OU 2 reduced survival of a freshwater prawn (Long and Morgan 1991).

**Barium.** The sample mean and the maximum detected concentrations in the UMB sediment samples exceeded both the lower (EPA-V) and upper (EPA-V) benchmarks. The mean barium concentration was 3.7 and 7.5 times higher than the mean reference concentration and the upper benchmark, respectively.

These benchmarks should be used with caution because very little sediment information is available for barium. MacDonald et al. (1988) found that aqueous barium concentrations as low as 100 mg/L reduced the survival of yellow crabs *(Cancer anthonyi)*.

**Iron.** The sample mean and the maximum detected concentrations of iron in the UMB sediment samples exceeded both the lower (MOE) and upper (MOE) benchmarks. The mean iron concentration at UMB was 1.5 times higher than the lower benchmark, whereas the maximum detected value at the FCAP was only marginally greater than the lower benchmark.

Although little information is available on iron in sediments, freshwater invertebrates are susceptible to aqueous iron concentrations as low as 0.3 mg/L (Train 1979 cited in Dave 1984).

**Manganese.** Both the sample mean and the maximum detected manganese concentrations in UMB sediment samples exceeded the lower (MOE) and upper (MOE) benchmarks. The mean manganese value was 19 and 8 times higher than the lower and upper benchmarks, respectively.

Very little information is available on manganese toxicity in sediments or surface water. Acute and chronic aqueous toxicity values for *Daphnia magna* were 17.48 mg/L and <1.1 mg/L, respectively (Kimball n.d.).

**Nickel.** The sample mean and the maximum detected concentrations of nickel in the UMB sediment samples exceeded the lower (ER-L) benchmark. The mean concentration was 1.3 times higher than the lower benchmark. Nickel concentrations in the FCAP were below both benchmarks.

UMB sediment concentrations were comparable to concentrations in sediments that caused >90% mortality of two freshwater invertebrates, *H. azteca* and *D. magna* (Maleug et al. 1984).

**Risk Characterization from Media Toxicity Tests**

Media toxicity tests of sediment or surface water from the FCAP were not performed. Media toxicity tests of sediment and surface water from UMB were conducted during both phases of the RI. There was no strong evidence for acute or chronic toxicity of surface water or sediments from the UMB area. The results of these tests are presented in Sect. 3.2.6 of this RI and are summarized below.

Phase one 7-day fathead minnow tests had generally high survival and growth rates. Survival and reproduction of *Ceriodaphnia* during the 7-day laboratory tests of water from UMB were also acceptable. There were some exceptions to this characterization, but even the more sensitive *Ceriodaphnia* life-cycle tests resulted in similar longevity, and only a small decrease in net fecundity, as compared to the control. The results of the 7-day snail feeding-rate test and the *in situ* snail test were not significantly different from the reference site.

Phase two 7-day fathead minnow tests provided no strong evidence of toxicity for water from MCK 2.03 or MCK 2.53. Two of the eleven tests resulted in a greater than 20% reduction in survival in full-strength site water. However, the unusually high variation among replicates was cited as evidence for a pathogenic cause, rather than a chemical stressor. The survival and reproduction rates of *Ceriodaphnia* in the 7-day tests were not generally indicative of acute or chronic toxicity. The laboratory snail feeding-rate tests, several of which

were spiked with UMB sediment, provided no evidence of acute toxicity from either the water or sediments of MCK 2.03.

**Biological Survey Data**

Benthic macroinvertebrate surveys of the FCAP were not conducted during this RI. Benthic macroinvertebrate surveys of UMB were conducted in both phases of this RI. The Phase 1 and Phase 2 surveys provided evidence of moderate stress and no strong impacts, respectively. Both measurement endpoints (i.e., Total Species Richness and EPT Richness) exhibited a steady increase at UMB over the course of these investigations. The results of these surveys, which are presented in Ryon et al. (1992) and Sect. 3.2.6 of this RI, are summarized below.

Phase 1 benthic macroinvertebrate surveys of UMB were performed in April, July, and October 1989 and January 1990. Seven metrics were calculated including the measurement endpoints (Total Species Richness and EPT Richness) for this assessment. The mean annual taxonomic richness and EPT richness were significantly reduced ($\alpha$=0.05) relative to the reference. In April 1989 the total richness and EPT richness were 52% and 35% of the reference, respectively. All measures of community structure increased substantially through the Phase 1 sampling period. Total richness exhibited a fourfold increase at MCK 2.03 in January 1990 relative to the April 1989 sampling event.

Phase 2 surveys, conducted in July 1992 and March 1993, consisted of only one sample at each of two sites, MCK 2.03 and MCK 2.53. These surveys also exhibited a general increase in the measurement endpoints. The total richness was 66% of reference at both sites in July and increased to levels comparable to the reference in March. EPT richness showed a similar, though less substantial, increase from 58% of reference at both sites to 78% and 96% at MCK 2.03 and MCK 2.53, respectively. The differences between July and March could be attributed to the diluting effects of spring flow or to natural variation.

Overall, the results of the Phase 2 surveys, in conjunction with the trend observed in the Phase 1 surveys, give some evidence of improving conditions at UMB. However, the fact remains that only one sample at each site had acceptable (>80% of reference) taxonomic and EPT richness.

**Weight of Evidence**

The weight of evidence for UMB can be summarized as follows:

1. Contaminants associated with coal ash (Sect. 6.1.4) were present in the sediment and surface water at concentrations sufficient to reduce survival, growth, and reproduction of benthic macroinvertebrates.

2. Toxicity tests of water and sediment from UMB provided no clear evidence of toxicity. Media for these tests were collected on a periodic basis, resulting in a minimal integration of the temporal variability that would be likely at the site. Although pulses of contaminants or ash would be missed by this sampling procedure, their impact would be registered by the benthic community at the site. Also, the only test conducted using sediment were the nonstandardized snail tests.

3. At the beginning of the sampling period, the macroinvertebrate community was clearly impacted. The evidence of stress on the community decreased through time coincident with process improvements at the Y-12 steam plant and the cessation of ash discharge to the FCAP.

Combining the three lines of evidence suggests that ash disposal activities at Chestnut Ridge OU 2 have resulted in a moderate to severe impact on the benthic macroinvertebrate community in the UMB area (Table 6.44). Taxa richness has clearly improved since the start of this investigation, but it is unknown whether this trend will be sustained. That the toxicity tests of sediment and water never produced clear evidence of toxicity may be the result of one or more factors: that the ash is a physical stressor, the perturbations are periodic in nature, or that the test organisms were not as sensitive to the stressors as the benthic community.

The weight of evidence is less substantial at the FCAP (Table 6.45). The only available line of evidence is the presence of coal ash-associated contaminants in sediment and surface water at potentially toxic concentrations. Only tenuous conclusions can be made without the corroboration of media toxicity tests or biological surveys. For the COPCs the sediment concentrations at UMB were higher than at the FCAP and the aqueous concentrations at UMB were lower than at the FCAP. Since the effects at UMB were not predicted by the water toxicity tests, and sufficient sediment toxicity tests were not performed, it is reasonable to assume that affects observed in UMB could occur upstream in the FCAP.

### 6.4.1.3 Plants

#### Contaminant Screening of Soil and Ash

Contaminants of potential ecological concern in soil and ash were identified by comparing the concentrations from each area to the reference and benchmark levels. As discussed in Sect. 6.2, the maximally exposed soil- or ash-associated organism could receive 100% of its exposure from the most contaminated sediment. Therefore, for screening the detected concentrations the maximum detected value was used for the soil and ash data. Analytes detected in ash and soil at maximum concentrations greater than in reference soils were screened against toxicological benchmarks for effects on terrestrial plants. The results are given in Table 6.50. Hazard quotients for individual analytes were calculated by dividing the maximum measured concentration in the medium by the toxicity benchmark concentration. Results were rounded to the nearest whole number. Hazard quotients were not calculated for analytes not found in concentrations greater than in the reference soils except in the case of antimony, silver, and thallium in the Sluice Channel Area and the FCAP ashes (Table 6.50). The maximum nondetected concentration of these contaminants exceeded the toxicity benchmarks for plants growing in soil.

Ash and soil data were compared to both the reference soil data and the results of the *Background Soil Characterization Project for the Oak Ridge Reservation* (Energy Systems 1993b). To be conservative, the site data were compared to the lowest of either the UCL95, or the maximum detected value, of the reference data. The FCAP and UMB are located in the Chepultepec Formation and the Sluice Channel Area is located in the Copper Ridge Formation (see Chap. 3). Therefore, the ash results for all three areas, and the soil results for the FCAP and UMB, were compared to the upper 95% confidence bound on the median for the A horizon of the Chepultepec Formation. The soil results for the Sluice

Channel Area were compared to the upper 95% confidence bound on the median for the A horizon of the Copper ridge Formation. If the soil or ash maximum detected concentration or maximum detection limit exceeded either the reference or the levels in Energy Systems 1993b, it was retained for comparison to benchmarks. The maximum detected and nondetected values were also used for the comparison to benchmarks. The results of these screenings are presented in Tables 6.51 through 6.57.

Toxicity of mercury is dependent upon its chemical species. Mercury, which was not speciated for this RI, was compared to the benchmarks for both the inorganic and methylated forms.

### Contaminant Screening of Shallow Groundwater

Contaminants of potential ecological concern in the shallow groundwater were identified by comparing the concentrations at the site to the benchmark levels. As discussed in Sect. 6.2, the maximally exposed, sediment-associated organism could receive 100% of its exposure from the most contaminated sediment. Therefore, for screening the detected concentrations the maximum detected value was used. An appropriate reference for shallow groundwater was not available. The site data were compared directly to the benchmarks (Table 6.32). The same procedure was conducted using the maximum nondetected value. The toxicity of mercury is dependent upon its chemical species. Mercury, which was not speciated for this RI, was compared to the benchmarks for both the inorganic and methylated forms. The results of this screening are presented in Table 6.58.

Several analytes were detected in shallow groundwater at maximum concentrations exceeding the toxicological benchmarks for effects on terrestrial plants grown in solution (Table 6.59). hazard quotients were calculated as outlined above.

Screening results are integrated with media toxicity test and biosurvey results to assess risk to plants at Chestnut Ridge OU 2 posed by contaminants in ash, soil, and ash/soil pore water.

**Aluminum.** Ash in all areas of Chestnut Ridge OU 2 have maximum aluminum contents several hundred times the benchmark for plants growing in soil. Maximum concentrations of aluminum in soil were approximately the same as in the overlying ash in each subarea of the site. The benchmark concentration is associated with a reduction in clover seedling viability of ~ 30% (Mackay et al. 1990). Maximum concentrations of aluminum in the shallow groundwater were 40 times the benchmark. Soil solution levels near the benchmark are associated experimentally with a 25% reduction in length of the longest root of rye grass (Wong and Bradshaw 1982).

These results indicate possible phytotoxic effects of aluminum at levels found in soil, ash, and groundwater of the OU. Although there are no other lines of evidence to support these conclusions, it is possible that plant growth at the site is being inhibited to some degree. The shallow groundwater aluminum concentration indicates that a significant part of the aluminum in the ash and soil may be in pore water in a form that is available to plants at the near neutral pH conditions of the soil and ash at the site.

**Antimony.** The maximum nondetected concentration of antimony in the Sluice Channel Area and the FCAP ashes were ~ 3 and 5 times the phytotoxicity benchmark, respectively.

This benchmark was derived from a secondary source in which phytotoxic effects were not defined (Kloke 1979). As there is no other information available on phytotoxic effects, it is not possible to predict effects that plants may experience from antimony exposure in Chestnut Ridge OU 2.

**Arsenic.** Maximum concentrations of arsenic in ash exceeded the phytotoxicity benchmark in all areas. In soil, maximum arsenic concentrations were ~ 5 times the benchmark. The benchmark concentration is associated with a reduction in yield of cotton (22%) and soybeans (45%) (Deuel and Swoboda 1972). The measured arsenic concentrations at the site and the benchmark concentration are based on total arsenic. Based on the oxidation-reduction behavior of arsenic, it is possible that it occurs in the oxidized, less toxic arsenic (V) form in the Sluice Channel Area ash and soil. Moisture conditions in the FCAP and UMB ashes and soils vary greatly with seasonal rainfall. It is possible that arsenic exists in the more toxic arsenic (III) state during the winter and spring in these areas. Turner (1981) sampled pore water from the saturated zone in the FCAP (4-6 m depth) between February and July. He found from 7 to 100% of the arsenic to be in the +3 oxidation state. This suggests that arsenic in the ash and soil of OU may be inhibiting plant growth, particularly in seasons when soil and ash are wet.

**Barium.** Maximum concentrations of barium were approximately equivalent to the phytotoxicity benchmark in the FCAP and UMB ash. The benchmark concentration is associated with a reduction in leaf weight (38%) in barley plants (Chaudhry et al. 1977). Plants growing in the FCAP particularly may be negatively impacted by barium in the ash. Vegetation in UMB may able to exploit the less contaminated soil beneath the ash.

**Cadmium.** Cadmium levels at or above benchmark were found in the ash and soil of the UMB area. The maximum nondetected level in the FCAP ash was equivalent to the benchmark. Soil levels of cadmium near the benchmark concentration are associated with a reduction in plant weight of spinach (33%), soybean (40%), lettuce (40%), radish roots (35%), and corn (30%) (Sadana and Singh 1987, Strickland et al. 1979, Haghiri 1973, Miller et al. 1977). Soil levels of cadmium in the UMB are of particular concern in relation to inhibiting plant growth in that area. The wide range of physiological effects occurring at low cadmium concentrations makes it advisable to consider the vegetation of the FCAP and Sluice Channel Area to be at risk from cadmium in the ash also.

**Chromium.** Maximum concentration of chromium in the ash were approximately 10 (Sluice Channel Area and UMB) to 20 (FCAP) times the benchmark for toxic effects on plants. Chromium was found in the UMB soil at concentrations almost 3 times as high as in the overlying ash. Soil levels of chromium near the benchmark concentration are associated with a reduction in lettuce shoot weight (50%) (Adema and Henzen 1989). The possibility of toxic effects on plant growth at the OU is supported by chromium concentrations in ground cover vegetation on the FCAP approximately 15 times lettuce leaf tissue concentrations associated with the benchmark soil level. Chromium likely exists in the available and toxic chromium (VI) form in the Sluice Channel Area with its toxic, neutral pH conditions. The seasonally variable moisture conditions of the soil and ash of the FCAP and UMB will control speciation of chromium in those areas.

**Cobalt.** Maximum concentrations of cobalt in the Sluice Channel Area and the FCAP ash and in the UMB soil were approximately equivalent to the phytotoxicity benchmark. This benchmark was derived from a secondary source in which the nature and magnitude of

phytotoxic effects were not defined (Linzon 1978), so it is difficult to assess their relevance to risk to plant growth posed by cobalt at Chestnut Ridge OU 2.

**Copper.** Maximum concentrations of copper in ash and soil were approximately equivalent to the phytotoxicity benchmark. This benchmark was derived from a secondary source in which effects were not defined (Kovalskiy 1974). Copper may be sorbed by iron and manganese oxides at neutral to alkaline pH conditions of coal ash and not be very available to plants (Adriano et al. 1980).

Maximum concentrations of copper in shallow groundwater were approximately twice the phytotoxicity benchmark. Soil solution levels near the benchmark are associated with a 70% reduction in length of the longest root of rye grass (Wong and Bradshaw 1982). This indicates that copper is available to plants under conditions prevailing at the OU and at concentrations in ash, soil, and soil solution that may inhibit plant growth.

**Iron.** Iron concentrations in ash and soil at the site did not exceed benchmark vales for phytotoxicity. It is not considered readily available to plants under neutral and alkaline pH conditions generally found in coal ash (Adriano et al. 1980). Maximum concentrations of iron in shallow groundwater, however, exceeded the soil solution phytotoxicity benchmark by a factor of ~ 3. This benchmark was derived from a secondary source in which the nature and magnitude of phytotoxic effects were not defined (Chapman 1966), so it is difficult to assess their relevance to risk to plant growth posed by iron at Chestnut Ridge OU 2.

**Manganese.** Maximum concentrations of manganese in the Sluice Channel Area soil and UMB soil were ~ 5 times the phytotoxicity benchmark. The benchmark concentration is associated with a 25% reduction in stem weight of bush beans (Wallace et al. 1977b).

Maximum concentrations of manganese in shallow groundwater were ~ 3 times the soil solution phytotoxicity benchmark. Soil solution levels near the benchmark are associated with ~ a 30% reduction in length of the longest root of rye grass (Wong and Bradshaw 1982). These results taken together indicate the likelihood that plant growth in the Sluice Channel Area and UMB is inhibited to some degree by the manganese present in soil and soil solution, and that growth on the FCAP may be inhibited by manganese in soil solution.

**Mercury.** Maximum concentrations of mercury in ash were over the phytotoxicity benchmark by factors of 2 (Sluice Channel Area and UMB) and 4 (FCAP). This benchmark was derived from a secondary source in which the magnitude and nature of phytotoxic effects were not defined (Kloke 1979) so it is difficult to assess the effect mercury exposure may have on plants at Chestnut Ridge OU 2.

**Nickel.** Maximum concentrations of nickel in the Sluice Channel Area, FCAP and UMB ash, and the UMB soil were twice the phytotoxicity benchmark. The benchmark concentration is associated with a 75% reduction in shoot weight of barley (Wallace et al. 1977a). This suggests that excess nickel may be limiting plant growth in all areas of the OU, especially in the FCAP where vegetation roots are exposed entirely to ash.

**Selenium.** Maximum concentrations of selenium in ash exceeded the phytotoxicity benchmark by factors of 5 (Sluice Channel Area) to 31 (FCAP). Maximum concentrations in soil were approximately equivalent to the benchmark in the Sluice Channel Area and twice the benchmark in UMB. The benchmark concentration is associated with an ~ 60%

reduction in shoot weight of sorgrass (Carlson et al. 1991). This suggests that excess selenium may be limiting plant growth in all areas of the OU.

**Silver.** The maximum nondetected concentration of silver in the Sluice Channel Area and FCAP ashes were approximately twice the phytotoxicity benchmark. This benchmark was derived from a secondary source in which the nature and extent of phytotoxic effects were not defined (Linzon 1978), so it is difficult to assess the effect the silver exposure may have on plants at Chestnut Ridge OU 2.

**Thallium.** Maximum concentrations of thallium in the FCAP ash exceeded the phytotoxicity benchmark by a factor of 3. Maximum nondetected concentrations of thallium in the Sluice Channel Area ash was approximately equivalent to the benchmark. This benchmark was derived from a secondary source in which the nature and magnitude of phytotoxic effects were not defined (Kloke 1979), so it is difficult to assess the effect thallium exposure may have on plants at Chestnut Ridge OU 2.

**Vanadium.** Maximum concentrations of vanadium in ash exceeded the phytotoxicity benchmark by factors of 18 (Sluice Channel Area) to 33 (UMB). Maximum concentrations of vanadium in the Sluice Channel Area and UMB soil exceeded the benchmark by a factors of 13 and 38. This benchmark was derived from a secondary source in which the nature and magnitude of phytotoxic effects were not defined (EPA 1975). However, the ubiquitous presence and high hazard quotients associated with vanadium at the OU require further consideration of vanadium as posing a risk to plant growth.

**Zinc.** Maximum concentrations of zinc in ash and soil were 2 to 4 times the phytotoxicity benchmark. The benchmark concentration is associated with an ~ 30% reduction in number of seeds produced per soybean plant (Aery and Sakar 1991).

Maximum concentrations of zinc in shallow groundwater were ~ 5 times the soil solution phytotoxicity benchmark. Soil solution levels near the benchmark are associated with reductions in plant weights of clover (60%), barrel medic (80%), and lucerne (60%) (Carroll and Loneragan 1968). These results indicate the possibility of severe inhibition of plant growth and reproduction for plants growing in ash and soil at the OU.

### Media Toxicity and Biosurvey Data

There were no tests performed to assess the toxicity of the soil and ash of the OU to plants. The survey of vegetation recorded a wide variety of plant types but did not attempt to evaluate growth potential or plant variety with respect to a reference site. The analysis of vegetation for contaminants was useful in the evaluation of the risk posed by chromium to plant growth in the FCAP.

### Conclusion

The summaries of hazard quotients for contaminants in ash and soil at Chestnut OU 2 (Tables 6.60 and 6.61) allow one to make a few general conclusions concerning the potential phytotoxic effects of these media over the site. These are summarized on a site-wide basis in Table 6.62.

Contaminants were present in coal ash, soil, and shallow groundwater at concentrations sufficient to reduce growth of terrestrial plants. This is the only available line of evidence; however, it suggests that the ash disposal activities at Chestnut Ridge OU 2 have had at least a moderate impact on the plant growth at the site.

Comparing the Sluice Channel Area and UMB soils, it is evident that the latter contains a considerably larger number of contaminants at higher levels than does the Sluice Channel Area soil. The ash at each of the three areas in OU 2 contains basically the same suite of phytotoxic contaminants. The FCAP ash tends to have higher concentrations than the Sluice Channel Area and UMB ashes, between which no consistent difference exists.

#### 6.4.1.4 Soil invertebrates

**Earthworms**

Analytes detected in ash and soil at maximum concentrations greater than in reference soils were screened against toxicological benchmarks for effects on earthworms. The results are given in Table 6.63. Hazard quotients for individual analytes were calculated by dividing the maximum measured concentration in the medium by the toxicity benchmark concentration. Results were rounded to the nearest whole number. Hazard quotients were not calculated for analytes not found in concentrations greater than in the reference soils.

Screening results are integrated with media toxicity test and biosurvey results to assess risk to earthworms at Chestnut Ridge OU 2 posed by contaminants in ash and soil.

**Cadmium.** The maximum detected concentration of cadmium in UMB soil was approximately equivalent to the earthworm toxicity benchmark. The benchmark is associated with a 23% decrease in cocoons produced per worm and a 22% decrease in juveniles produced per worm of *Eisenia andrei* during a 21-day exposure in artificial soil (van Gestel et al. 1992). This indicates the potential for negative effects on earthworm reproduction caused by the presence of cadmium.

**Chromium.** The maximum detected concentration of chromium in ash was 9 times (Sluice Channel Area and UMB) and 21 times (FCAP) the earthworm toxicity benchmark. The maximum detected concentration of chromium in UMB soil was 25 times the benchmark. The benchmark is associated with a 25% mortality of *Octochaetus pattoni* caused by exposure to chromium (VI) for 60 days in soil mixed with animal dung (Abassi and Soni 1983). For the purposes of the risk assessment, the assumption is made that all of the measured chromium at the OU is in the chromium (VI) form. The high hazard quotients associated with the measured chromium concentrations indicate that this contaminant should be considered as potentially inhibitory to survival of earthworms in ash and soil of the OU, even if this conservative assumption concerning its speciation is not correct.

**Copper.** The maximum detected concentration of copper in the FCAP ash was equivalent to the earthworm toxicity benchmark. The benchmark is associated with a 70% decrease in cocoons produced per worm and a 64% decrease in hatchlings per cocoon of *Dendrbaena rubida* during a 120-day exposure in soil mixed with animal dung (Bengtsson et al. 1986). This is a strong indication that earthworms in the FCAP ash would be under reproductive stress.

**Mercury.** The maximum detected concentration of mercury in the Sluice Channel Area and UMB ash was approximately equivalent to the earthworm toxicity benchmark. The maximum detected concentration in the FCAP ash was 3 times the benchmark. The benchmark is associated with a 65% reduction in survival and a 40% reduction in cocoon production in *Octochaetus pattoni* caused by exposure to mercury (II) for 60 days in soil mixed with animal dung (Abassi and Soni 1983). These results would suggest that earthworms in ash at all portions of the OU would be under survival and reproductive stress due to the presence of mercury.

### Media Toxicity and Biosurvey Data

The results of the laboratory and in situ earthworm toxicity tests indicate that the FCAP ash may be detrimental to earthworm survival (see Sect. 3.2.6.1). Survival in nonautoclaved ash from the Willow area of the FCAP was reduced by 66% compared to controls, and in ash from the Sycamore area it was reduced by 21%. In the in situ test, survival was reduced by at least 90% in ash as compared to that in peat moss. The total lack of earthworms found in the Sluice Channel Area and FCAP ash during the preliminary invertebrate survey supports the media toxicity data in suggesting that the ash at Chestnut Ridge OU 2 is inhibitory to earthworm survival.

### Conclusion

The summaries of hazard quotients for contaminants in ash and soil at Chestnut OU 2 (Tables 6.60 and 6.61) allow one to make a few general conclusions concerning the potential toxicity to earthworms of soil and ash over the site. These are summarized on a site-wide basis in Table 6.64.

Contaminants were present in coal ash and soil at concentrations sufficient to reduce survival and reproduction of earthworms. No earthworms were found in ash at the Sluice Channel Area and FCAP portions of the site. Combining these two available lines of evidence suggests that the ash disposal activities at Chestnut Ridge OU 2 have had a moderate to severe impact on the earthworm population at the site.

Comparing the Sluice Channel Area and UMB soils one can see that chromium particularly occurs in higher concentration in the UMB soil. A comparison of the ash from the three areas of the site shows no consistent differences between the three areas based on the limited number of benchmarks available from the literature. The media toxicity test data, however, indicated that the FCAP ash is at least more toxic than the Sluice Channel Area ash. This points out the importance of having multiple lines of evidence for the correct assessment of the risk posed by the coal ash at this site. There are inhibitory chemical or physical characteristics of the ash that are not evaluated in the conventional earthworm toxicity literature.

### Soil/Litter-dwelling Invertebrates other than Earthworms

Analytes detected in vegetation samples from the OU were screened against toxicological benchmarks for effects on soil invertebrates other than earthworms. No analytes measured in the FCAP vegetation for which there are benchmarks (cadmium, lead, mercury) exceeded those benchmarks. There are no benchmarks for arsenic, chromium, selenium, or thallium.

### Media Toxicity and Biosurvey Data

There were no media toxicity tests performed to evaluate the effect of soil and ash on soil invertebrates other than earthworms. During the preliminary invertebrate survey, very few invertebrates were found in Sluice Channel Area and FCAP ash (Sect. 3.2.6.3). Those discovered consisted of millipedes and beetle larvae.

### Conclusions

The summary of risk characterization given in Table 6.65 indicates the inconclusive nature of the evidence on the effects of contaminants in vegetation on the soil/litter-dwelling invertebrate populations at the site. Contaminants were not present in the vegetation at concentrations sufficient to reduce survival and reproduction of soil/litter-dwelling invertebrates. The only other line of evidence is the biosurvey data, which indicated that very few soil-dwelling invertebrates were living in the Sluice Channel Area and FCAP ash. A more conclusive assessment of the risk of these populations requires more complete field sampling and vegetation analysis programs.

## 6.4.1.5 Soil microbes and microbial processes

Analytes detected in ash and soil at maximum concentrations greater than in a reference soil were screened against toxicological benchmarks for effects on microbes and microbial processes. The results are given in Table 6.66. Hazard quotients for individual analytes were calculated by dividing the maximum measured concentration in the medium by the toxicity benchmark concentration. Results were rounded to the nearest whole number. Hazard quotients were not calculated for analytes not found in concentrations greater than in the reference soils.

Screening results are integrated with media toxicity test and biosurvey results to assess risk to plants at Chestnut Ridge OU 2 posed by contaminants in ash and soil.

**Aluminum.** Aluminum concentrations exceeded background soil and benchmark levels by a large margin in all ash and soil samples taken at the site. Sluice Channel Area soil had the lowest and the FCAP ash had the highest concentrations. The benchmark is associated with a 20% decrease in nitrogen mineralization by native soil microflora in a loam soil (Liang and Tabatabai 1977).

**Cadmium.** Concentrations of cadmium exceeding reference soil and benchmark levels were found only in the UMB ash and soil. The benchmark is associated with a 50% decrease in phosphatase activity by native soil microflora in a sandy soil (Doelman and Haanstra 1989).

**Chromium.** Chromium was measured at concentrations exceeding background soil and benchmark levels in ash samples taken from all three areas of the site and from UMB soil. The benchmark is associated with a 50% decrease in arylsulfatase activity by native soil microflora in a sandy soil (Haanstra and Doelman 1991).

**Copper.** Copper concentrations at the site exceeded background soil and benchmark levels in ash samples from all areas from the site and in UMB soil. It was approximately equivalent to the benchmark in Sluice Channel Area soil. The benchmark is associated with

a 50% decrease in arylsulfatase activity by native soil microflora in a sandy soil (Haanstra and Doelman 1991).

**Iron.** The maximum detected concentrations of iron in Chestnut Ridge OU 2 ash and soil were very high and exceeded benchmark levels by factors of several hundred to several thousand times. Sluice Channel Area soil was the only medium with levels of iron not exceeding background soil levels. The benchmark is associated with a 20% decrease in nitrogen mineralization by native soil microflora in two loam soils (Liang and Tabatabai 1977).

**Manganese.** Manganese was only found in elevated concentrations in the Sluice Channel Area and UMB soils. The benchmark is associated with a 67% decrease in nitrification by native soil microflora in a sandy loam soil (Premi and Cornfield 1969).

**Mercury.** The maximum detected concentrations of mercury in Chestnut Ridge OU 2 ash ranged from 6 times (Sluice Channel Area) to 13 times (FCAP) the benchmark for toxicity to soil microbes and microbial processes. It was not found in elevated concentrations in the Sluice Channel Area and UMB soils. The benchmark is associated with a 87% decrease in carbon mineralization by native soil microflora in a clay loam soil (Landa and Fang 1978).

**Vanadium.** Vanadium was found in slightly elevated concentrations in all media. The benchmark is associated with a 21% decrease in respiration by native soil microflora in a soil/litter microcosm (Lighthart et al. 1977).

**Zinc.** Zinc was found in concentrations exceeding background soils and benchmark levels in all ash and in UMB soil. The benchmark is associated with a 50% decrease in arylsulfatase activity by native soil microflora in a sandy soil (Haanstra and Doelman 1991).

**Media Toxicity and Biosurvey Data**

Results of experiments investigating the effects of fly ash on the biomass of *Rhizobium leguminosarum* are given in Sect. 3.2.6.6. The ~ 90% decrease in biomass of *Rhizobium* cultured in 100% coal ash suggests that fly ash is toxic to these microorganisms or is physically unsuitable as a growth substrate. This may have implications for the ability leguminous plants to colonize nitrogen-poor ash deposits that do not support populations of *Rhizobium* inoculum.

There was no biological survey conducted to evaluate the toxicity of the soil and ash of the OU to soil microbes or microbial processes.

**Conclusion**

The summaries of hazard quotients for contaminants in ash and soil at Chestnut Ridge OU 2 (Tables 6.60 and 6.61) allow one to make a few general conclusions concerning the potential toxicity to microbes and microbial processes of soil and ash over the site. These are summarized on a site-wide basis in Table 6.67.

Contaminants were present in coal ash and soil at concentrations sufficient to reduce survival and functioning of soil microorganisms. The media toxicity test used to determine the effects of ash on a species of *Rhizobium* provides clear evidence that survival of this organism was severely impaired.

Combining the two available lines of evidence suggests that the ash disposal activities at Chestnut Ridge OU 2 have had a moderate to severe impact on the microbial population and its functioning. Comparing the Sluice Channel Area and UMB soils, it is evident that the latter contains a larger number of contaminants toxic to microbes at higher levels than does the Sluice Channel Area soil. A comparison of the ash from the three areas of the site shows the FCAP ash tends to have higher toxic contaminant concentrations than the Sluice Channel Area and UMB ashes, between which no consistent difference exists.

**Integration of Risks to Plants, Soil Invertebrates, and Microbes: Conclusions**

**Sluice Channel Area.** The Sluice Channel Area supports a mature, secondary, mixed pine and hardwood forest with a hardwood understory and ground cover. It is not possible to compare the vigor of this plant community to a reference community on the basis of data gathered for this RI report. The extreme variability in the depth of the ash makes it possible that the larger trees and understory are deriving water and nutrients from below the soil layers chemically and physically impacted by the ash. Growth of ground cover plants will more likely be negatively influenced by phytotoxic levels of contaminants in ash and soil. Analyses of soils and ash sampled in the depth interval of 1 to 12 in. (2.5 to 30 cm) indicate the existence of a number of metals in potentially phytotoxic concentrations that may be limiting growth of the plant community of the Sluice Channel Area.

The extreme variability of the depth of the ash of the Sluice Channel Area makes it difficult to assess the impact of ash-derived contaminants on soil heterotrophs. Soil and ash samples were composited across transects of variable ash depth, so it is not possible to evaluate what proportion of the area may have surface soil relatively free of contaminants and contain unstressed soil invertebrate and microbial populations. Other confounding factors include past ash coverings of presently ash-free portions of the slope and lateral movement of ash and of contaminants in surface and vadose zone flow. Analyses of soil and ash samples indicate the existence of metals in potentially toxic concentrations that may be limiting growth and functioning of the soil heterotrophic community of the Sluice Channel Area. The lack of soil invertebrates in the Sluice Channel Area ash and the reduced survival of earthworms in the media toxicity tests support the likelihood of an inhibitory effect of ash on reproduction and survival. Further site investigation and comparisons to reference sites is necessary to evaluate the magnitude of the effects on populations and processes (e.g., litter decomposition, nutrient cycling) in the Sluice Channel Area.

**FCAP.** The growth of vegetation and soil organisms is dependent on, and limited by, the physical and chemical characteristics of the ash of the FCAP. The highest concentrations of most of the phytotoxic analytes found at Chestnut Ridge OU 2 occurred in the FCAP ash. Without reference site growth comparisons, it is difficult to evaluate the condition of the plant communities and effects of the contaminants.

There are reports of vigorous plant growth, both volunteer and introduced, on coal ash deposits (Horton and McMinn 1977, Furr et al. 1975). Scanlon and Duggan (1979), however, found a mean survival rate of various trees (alder, birch, sycamore, olive, and dogwood) on fly ash to be 53%. These results occurred with concentrations of some contaminants below those found in FCAP ash, while others were higher than in FCAP ash (Scanlon and Duggan 1979). Differences in analytical techniques make such comparisons difficult to interpret; however, they indicate the possibility of limitations to growth of the hardwood forest of the lower FCAP and the willow and grass community of the upper FCAP.

There is very little information on the effects of coal ash on the growth and functioning of soil heterotrophic populations. Additions of fly ash to soil have been found to reduce microbial respiration and nitrification (Wong and Wong 1986, Cervelli et al. 1986); however, the negative effects may have been due to alkalinity rather than metal or organic contaminants. Other research indicates metals in fly ash added to soil act to reduce microbial respiration (Arthur et al. 1984). Several metals were found in the FCAP ash in concentrations in some cases significantly exceeding toxicity benchmarks for earthworms and soil microbes. The lack of earthworms in the ash and the reduced survival of earthworms in the media toxicity tests support the likelihood of an inhibitory effect of ash on survival of earthworms. Although Cd, Pb, and mercury in sampled FCAP vegetation were not above benchmark levels for toxicity to soil invertebrates, the dearth of these organisms in the FCAP ash samples investigated indicates the potential for negative effects on these communities due to other chemical or physical factors. Further analyses and comparisons to reference sites is necessary to evaluate the magnitude of the effects on populations and processes (e.g., litter decomposition, nutrient cycling) in the FCAP ash.

**UMB.** As with the Sluice Channel Area, depth of ash deposits in the UMB is variable and the same problems exist with regard to determining the extent of the area impacted presently and in the past. It would appear that UMB is a dynamic system with frequent movement, burial and uncovering of ash deposits. The predominant vegetation is grasses that may be able to take advantage of soil below the ash deposits. The existence in UMB of ash and soil containing metals above benchmark concentrations for toxicity to plants, earthworms, and microorganisms and microbial processes suggests the possibility that these communities are negatively impacted. Comparisons of populations and activities to reference sites is required to assess the nature, extent, and magnitude of the impacts.

#### 6.4.1.6 Small mammals

As stated in the introduction to Sect. 6.4, risk characterization consists of the integration of multiple lines of evidence. These lines of evidence consist of data from toxicity tests of ambient media, biomonitoring data, and published toxicity data for COPCs.

Toxicity tests using ambient media, such as those that are regularly used for soil and aquatic organisms (e.g., earthworm, fathead minnow, *Ceriodaphnia* tests) are not generally conducted for small mammals and were not performed for Chestnut Ridge OU 2. Therefore, this line of evidence is unavailable for small mammals at Chestnut Ridge OU 2.

The biomonitoring data for small mammals at Chestnut Ridge OU 2 consists of species presence/absence data and body burdens of As, Cd, Cr, Pb, Hg, Se, and Tl. These data are only available for the FCAP, Sluice Channel Area, and the reference site. Three conclusions may be made from these data:

1. Arsenic and selenium are being bioaccumulated by small mammals at Chestnut Ridge OU 2. Levels in small mammals from Chestnut Ridge OU 2 are significantly greater than those from the reference site, with levels of both arsenic and selenium being highest in small mammals from the FCAP as compared to those from the Sluice Channel Area.

   An extensive literature search (Sect. 6.3.1.6) provided no data to indicate if whole-body arsenic concentrations comparable to those observed among small mammals at Chestnut Ridge OU 2 cause adverse effects. For selenium, no effects were observed among

*Microtus* sp. or *Reithrodontomys* sp. (small mammal species within the same genera as those observed at Chestnut Ridge OU 2) with selenium body burdens as high as 33 ppm (Clark 1987). For comparison, the maximum selenium body burden for rodents at Chestnut Ridge OU 2 was 5.64 ppm. Body burdens ranging from 10 to 100 ppm produced no visible adverse effects among ornate shrews (*Sorex ornatus;* Clark 1987). The selenium body burden for the single shrew collected from Chestnut Ridge OU 2 was 2.33 ppm. It is unknown, however, if the small mammal populations at Chestnut Ridge OU 2 are at equilibrium with the bioaccumulative contaminants. For example, Charters (1991) observed that *Microtus* collected from Love Canal in New York were younger than those collected from nearby reference areas. This age difference was attributed to the lack of overwintering individuals at the waste site. Arsenic and selenium levels observed in small mammals at Chestnut Ridge OU 2 may therefore represent values for juvenile or dispersing individuals and may not be indicative of actual body burdens in a mature, resident population.

2. Contamination at Chestnut Ridge OU 2 is not so great as to render the site uninhabitable to rodents. Due to the relatively small number of specimens collected, these data do not allow estimates of the abundance or production of the rodent populations at Chestnut Ridge OU 2 to be evaluated. However, they do suggest that Chestnut Ridge OU 2 supports a richer rodent fauna than does the reference area (Walker Branch); four rodent species were observed at Chestnut Ridge OU 2 as opposed to only one at the reference location. The higher species richness at Chestnut Ridge OU 2, however, may simply reflect the higher diversity of habitat types at Chestnut Ridge OU 2 compared to the reference site.

   Another observation relating to rodents is that of 11 *Peromyscus* collected at Chestnut Ridge OU 2, all 11 were males. If it is assumed that the proportion of males in the *Peromyscus* population is 50%, the binomial probability of collecting 11 individuals and having all be males is $(0.5)^{11} = 0.00048$ (Dowdy and Weardon 1983). This low probability suggests that some factor is affecting the sex ratio of mice collected at Chestnut Ridge OU 2. Factors that could produce a skewed sex ratio include sex-related trap preference or avoidance, greater dispersal or movement by males such that they are more likely to encounter traps, or sex-related reaction to contaminants.

3. No short-tailed shrews were collected from the FCAP and only one individual was obtained from the Sluice Channel Area. As with rodents, the small number of shrew specimens collected does not allow estimates of the abundance or production of the shrew population at Chestnut Ridge OU 2 to be evaluated. However, the data suggest that Chestnut Ridge OU 2 is not suitable for shrews. This may be due to either the lack of appropriate habitat or toxicity of contaminants at the site.

The last line of evidence is the use of published toxicity data. To determine if the contaminant exposure that small mammals (represented by white-footed mice and short-tailed shrews) at Chestnut Ridge OU 2 receive is potentially hazardous, the total contaminant exposure estimates for mice and shrews (Sect. 6.2.2.3, Tables 6.19 and 6.20), were compared to estimated NOAELs for mice and shrews (Sect. 6.3.1.6, Table 6.36). To quantify the magnitude of hazard, a hazard quotient (HQ) was calculated where: HQ = exposure/NOAEL. Hazard quotients greater than 1 indicate that individuals may be experiencing exposures that are in excess of NOAELs and suggest that adverse effects may be occurring. Hazard quotients for mice and shrews are presented in Tables 6.68 and 6.69.

For white-footed mice at Chestnut Ridge OU 2, four contaminants had HQs>1 (Table 6.68): aluminum and arsenic at the FCAP, Sluice Channel Area, and UMB; and selenium and thallium at the FCAP and Sluice Channel Area only. For short-tailed shrews at Chestnut Ridge OU 2, seven contaminants had HQs>1 (Table 6.69): Al, As, Ba, Se, and V at the FCAP, Sluice Channel Area, and UMB; cadmium at UMB only and thallium at the FCAP only.

Small mammals at the reference site also received contaminant exposures that exceeded NOAELs. Exposure to Al, Se, and Tl exceeded NOAELs for white-footed mice (Table 6.68); and exposure to Al, As, Ba, Mn, Se, and V exceeded NOAELs for short-tailed shrews (Table 6.69). These data may indicate that the reference area is contaminated or naturally occurring, background metal concentrations are high such that they exceed toxicological benchmarks.

Combining both available lines of evidence suggests that small mammals are being adversely affected by contaminants at Chestnut Ridge OU 2 (Table 6.70). The levels of contaminants to which small mammals at Chestnut Ridge OU 2 are exposed are sufficient to exert the following effects: Al: reduced growth and survival of offspring; As: reduced lifespan and number of offspring; Ba: reduced lifespan and increased blood pressure (shrews only); Cd: reduced reproductive success (shrews only); Se: increased juvenile mortality and reduced reproductive success; Tl: testicular toxicity; and V: reduced lifespan and survival (shrews only). Of the three areas that comprise Chestnut Ridge OU 2, the greatest effects are occurring on the FCAP, with effects in the Sluice Channel Area and UMB being less but approximately equal to each other. Of the two small mammal species considered, short-tailed shrews are experiencing the greatest effects, with effects experienced by white-footed mice being substantially less.

#### 6.4.1.7 Reservation-wide endpoints

The approach to the ecological assessment of Chestnut Ridge OU 2 is similar to that for other source OUs in that it is limited to those biota that can be definitively associated with the site. This approach does not address contaminant exposure and effects that may be experienced by wide-ranging species (deer, avian and mammalian predator, etc.) that may periodically use the site. While impacts to the **populations** of these wide-ranging species must be addressed at the reservation level, potential effects to **individuals** of wide-ranging species that exploit the site can be assessed.

Characterization of the risks that Chestnut Ridge OU 2 poses to individuals of wide-ranging species was performed using the weight-of-evidence approach previously employed for other endpoint species. Reservation-wide endpoint species for which risks were characterized include white-tailed deer, red fox, and red-tailed hawk. Toxicity tests using ambient media from Chestnut Ridge OU 2 are not available for these species. No biomonitoring surveys were specifically conducted for these species at Chestnut Ridge OU 2; however, incidental observations indicate that white-tailed deer use the site (Sect. 2.7.4). Both red fox and red-tailed hawk are common on the reservation (Parr and Evans 1992) and individuals of both species would be expected to use the site. However, use of Chestnut Ridge OU 2 by these species has not been documented.

Because data from media toxicity tests and biological surveys are not available, published toxicity data provide the only line of evidence with which to characterize risk to wide-ranging

species. HQs were calculated as described in Sect. 6.4.1.6 (small mammals) for white-tailed deer (Table 6.71), red fox (Table 6.72), and red-tailed hawk (Table 6.73). HQs were calculated for both exposure scenarios (absolute maximum exposure and maximum proportional exposure; for a detailed description of their derivation, see Sect. 6.2.2.3) for all portions of Chestnut Ridge OU 2. Separate HQs were calculated for contaminant exposure experienced by deer consuming ash to meet sodium needs.

For white-tailed deer consuming ash from the FCAP to meet sodium requirements, six contaminants had HQs>1 (Table 6.71): Al, As, Ba, Se, Tl, and V. Six contaminants also had HQs>1 under the absolute maximum exposure scenario for white-tailed deer: Al, As, and Ba at the FCAP, Sluice Channel Area, and UMB; Se and Tl at the FCAP and Sluice Channel Area only; and Cd at the FCAP only. White-tailed deer at the reference site also received exposures to Al, As, Se, and Tl that exceeded NOAELs (Table 6.71). Under the maximum proportional exposure scenario for white-tailed deer, four contaminants had HQs>1: Al, As, Se, and Tl at the FCAP and Sluice Channel Area; at UMB, only Al had HQ>1.

For red fox under the absolute maximum exposure scenario, six contaminants had HQs>1: Al and As at the FCAP, Sluice Channel Area, and UMB; Ba at UMB only; and Cr, Hg, Se, and Tl at the FCAP and Sluice Channel Area only (Table 6.72). Red fox at the reference site also received exposures to Al, Cr, Hg, Se, and Tl that exceeded NOAELs (Table 6.72). Under the maximum proportional exposure scenario for red fox, two contaminants had HQs>1: Al at the FCAP and UMB, and Se at the FCAP.

Under the absolute maximum exposure scenario for red-tailed hawks, four contaminants had HQs>1: As, Cr, Hg, and Se at the FCAP, Sluice Channel Area, and background location; no HQs exceeded 1 at UMB (Table 6.73). No HQs exceeded 1 for any contaminant at any of the locations under the maximum proportional exposure scenario (Table 6.73).

Using either exposure scenario (absolute maximum or maximum proportional), the available data suggests that survival and reproduction of individual white-tailed deer or red fox that use Chestnut Ridge OU 2 may be adversely affected by contaminants at the site (Table 6.74). The levels of contaminants to which white-tailed deer and red fox at Chestnut Ridge OU 2 are exposed are sufficient to exert the following effects: Al: reduced growth and survival of offspring; As: reduced lifespan and number of offspring; Ba: reduced lifespan and increased blood pressure; Cd: reduced reproductive success (deer only); Cr: reduced growth (fox only); Hg: increased mortality (fox only); Se: increased juvenile mortality and reduced reproductive success; Tl: testicular toxicity; and V: reduced lifespan and survival (deer only).

While survival and reproduction of individual red-tailed hawks using Chestnut Ridge OU 2 is likely to be depressed under the absolute maximum exposure scenario (with arsenic reducing growth and altering behavior, and Cr, Hg, and Se reducing reproductive success), no adverse effects are expected to these hawks under the proportional exposure scenario.

Of the three portions of Chestnut Ridge OU 2, the FCAP exerts the greatest adverse effect on the wide-ranging species, followed by the Sluice Channel Area and UMB. As suggested in Sect. 6.4.1.6, contaminant exposures for wide-ranging species at the reference site that exceed NOAELs may indicate that the reference area is contaminated or naturally occurring, background metal concentrations are high such that they exceed toxicological benchmarks.

While no federally threatened or endangered bird or mammal species are known to use any portion of Chestnut Ridge OU 2, four bird species listed as either threatened (Cooper's hawk) or in need of management (common barn owl, red-shouldered hawk, and black vulture) by the state of Tennessee have been observed on or are expected to use Chestnut Ridge OU 2 (see Sect. 2.7.3). Similar to red-tailed hawks, individuals of these species that use Chestnut Ridge OU 2 may experience significant exposures. However, because all four species have home ranges considerably larger than Chestnut Ridge OU 2, evaluation of effects to their populations is most appropriate at the reservation-wide scale. In addition, because foraging behaviors and food habits of these species differ dramatically from those of red-tailed hawks, exposures experienced by these species are not likely to be significantly greater. For example, black vultures feed on carrion (little of which is expected to be found on Chestnut Ridge OU 2) and have home ranges of ~ 18000 ha (Rabenold 1987); Cooper's hawks are primarily forest birds (DeGraaf et al. 1981) whose diets contain 50% to 85% birds (Reynolds and Meslow 1984); red-shouldered hawks prefer wooded wetlands such as those found on Chestnut Ridge OU 2, however, their diet consists primarily of reptiles and amphibians (DeGraaf et al. 1981), few of which were observed on Chestnut Ridge OU 2; and while small mammals make up 60% to 90% of the diet of barn owls, they forage in open, grassland habitats (Johnsgard 1988) of which little is present at Chestnut Ridge OU 2.

### 6.4.2 Risk Integration for Future Conditions

Future risks to ecological receptors at Chestnut Ridge OU 2 are not expected to increase and may very well decrease relative to current levels of contaminants. Based on data from toxicity tests of ambient media, biomonitoring data, and published toxicity data for COPCs, it was determined that ecological effects are currently occurring (see Sect. 6.4.1). Because the disposal of coal ash at Chestnut Ridge OU 2 has been discontinued, future exposure to coal ash contamination is not expected to increase, except in the event of dam failure. If the dam fails, further exposure of fish and benthic macroinvertebrate populations to coal ash contamination can be expected. Heavy ash intrusion will increase turbidity, reduce aquatic invertebrate density, limit macrobenthic organisms, inhibit growth of benthic algae, and alter nutrient cycling, etc.

The area is expected to remain in natural vegetation. Ecological recovery will continue and will be indicated by natural succession of terrestrial plants and increasing diversity among plant species. The weathering of soils and sediments in the terrestrial environment will result in the loss of substantial amounts of some soluble species, such as aluminum and magnesium, over time (El-Mogazi et al. 1988). However, leachability of some trace elements from the ash will decrease as the age of the material increases (Roy et al. 1981). Increased pH may decrease the mobility of heavy metals and toxic elements in the soil, thus decreasing their availability for uptake by plants. As a result, heavy metal concentrations in vegetation should decrease along with the detrimental effects of contaminated vegetation consumption by herbivorous wildlife and soil heterotrophs.

As the vegetation of the FCAP and surrounding areas develops, a gradual decrease in the water table at the FCAP site may be expected, increasing exposure in the aquatic environment. Drying the surface ash could lead to increased concentrations of most heavy metals and toxic elements, potentially increasing exposures of aquatic plants and animals (Cherry et al. 1979).

Conservative estimates used in toxicity tests and the derivation of benchmarks have indicated that no increase is expected in system sensitivity or value. Therefore, it is not expected that any new effects will manifest at the site.

### 6.4.3 Uncertainties

#### 6.4.3.1 Conceptual model

The EPA's framework for ERA specifies that pathways and receptors that are excluded from the conceptual model and therefore are not assessed should be presented in the uncertainties section (RAF 1992).

- Aquatic primary producers are not included as endpoint receptors. This is justified because in first order forest streams like UMB, allochthonous material is the primary energy source. Also, it is conventionally assumed that the benthic invertebrate and fish communities are more sensitive than the algal community.

- Amphibians are not included as endpoint receptors. This is justified because there is relatively little toxicity data on effects of metals on amphibians, and they are assumed to be represented by other endpoint biota for which there are more data. Amphibian larvae could be seasonally important components of the FCAP ponds if the water quality was suitable. However, they have not been observed there. The screening against water quality criteria for aquatic life and other aquatic benchmarks must serve to address that possibility.

- Reptiles are not included as endpoint receptors. This is justified because there is almost no information on the toxicity of metals to reptiles, so they must be assumed to be represented by mammalian and avian endpoint biota. Reptiles are not known to be ecologically important by the criteria of EPA Region IV.

- Piscivorous wildlife are not included as endpoint receptors. This is justified because, even if fish were present, the local piscivorous species are unlikely to make significant use of a small stream like UMB.

- Herbivorous insects are not included as endpoint receptors although they are important to energy and nutrient dynamics. This is justified because there is almost no information on the response of herbivorous insects to the metals associated with coal ash. It must be assumed that the assessment of risks to herbivorous mammals will serve to indicate potential effects on forest insects.

- Inhalation is not included as a pathway of exposure. This is a minor source of uncertainty because significantly volatile compounds are not involved, and the site is too wet and well vegetated for suspended dust to be a significant route of exposure.

#### 6.4.3.2 Fish

- The limits of detection for seven chemicals are higher than the threshold concentrations for ecological effects.

- There are no analyses of water samples that were filtered before extraction. Only the dissolved phase is bioavailable to any significant extent, so this results in a conservative estimate of exposure.

- Water toxicity data are much more standardized than soil or sediment toxicity data, and water composition is less variable than soil or sediments. However, alternative methods for calculating thresholds for aquatic toxic effects from laboratory tests produce benchmarks that vary over a range greater than two orders of magnitude (Suter et al. 1992b).

- The aqueous toxicity tests are only 7 days long and incorporate only one life stage of fish. Although this is the standard EPA test for ambient waters and effluents, it does not constitute a true test of chronic toxicity and may in fact significantly underestimate chronic toxicity of water to fish (Suter 1990).

- The effects of ash on fish may include physical effects (abrasion of gills by suspended ash or burying of eggs) or effects of dietary exposures due to accumulation of contaminants by prey or incidental ingestion of ash with prey. Such effects are not included in the aqueous toxicity test or in the assessments of contaminant concentrations.

- Because the ash apparently killed all fish in UMB, there was no fish community to be monitored for current effects. Results of the sculpin reintroduction are too preliminary to be conclusive.

#### 6.4.3.3 Benthic macroinvertebrates

The following uncertainties are associated with assessment of the risks posed by Chestnut Ridge OU 2 to benthic macroinvertebrates in the FCAP and UMB:

1. **Bioavailability of contaminants:** Total contaminant concentrations are required for the HHRA and were the only values determined in surface water at Chestnut Ridge OU 2. ERAs based on these data are conservative because the bio-available concentration is generally less than 100% of the total recoverable concentration. Metals, for example, generally include 30-95% particle-bound material (HECD 1992). In sediments the toxicity of several metals, including nickel, has been related to the acid volatile sulfide content of sediments (Di Toro et al. 1992). Acid volatile sulfide is a reactive pool of solid-phase sulfide that is available to bind metals and render that portion unavailable and therefore nontoxic to biota.

2. **Species of metal present in media:** The aqueous toxicities of arsenic and mercury depend on the species present. Arsenic can occur in either the trivalent and pentavalent states of which arsenic (V) is often the most toxic form. Methylmercury is generally more toxic to aquatic biota than inorganic mercury and was not determined for this assessment. These contaminants were conservatively assumed to be present in their most toxic form, which varied depending on the test organism used.

3. **Missing Benchmarks:** Benchmarks and toxicity data were not available for all the detected contaminants at Chestnut Ridge OU 2. Several metals and radionuclides that had no benchmarks were detected in the sediment and surface water of Chestnut Ridge

OU 2 at concentrations above background levels. One organic compound without a benchmark was also detected in the surface water.

4. **Insufficient Media Tests:** Although the sediment toxicity tests using snails provide useful information, standardized sediment toxicity data for the site would be more readily interpreted. The surface water tests used for this assessment provided no conclusive evidence of toxicity even though the benthic community was clearly stressed, as indicated by the biological surveys. This disparity could be the result of sediment-associated contaminants not being adequately assessed.

5 **Biosurvey Sampling Locations:** Standard benthic macroinvertebrate samples are collected from riffles instead of pools. Sediment is eroded from riffles and deposited in pools. The riffle community, therefore, is likely to experience less sediment-associated stress and underestimate the true toxicity.

6. **Sediment Benchmark Methods:** Data for calculating ER-Ls are often field studies in which the sediments were contaminated with several inorganic and organics chemicals. However, for each individual ER-L, the concentration of that particular contaminant was presumed to be 100% of the cause of the toxicity observed. This assumption results in conservative assessments of the toxic concentration of any one particular contaminant. Also, most of the field studies used to derive the benchmarks were conducted at estuarian sites. Most of the laboratory studies that were performed by spiking the sediments used metal salts. These tests result in conservative toxicity values because metal salts are highly bioavailable.

### 6.4.3.4 Plants

COPCs for Chestnut Ridge OU 2 include elements that act as micronutrients in plants up to a certain level but are phytotoxic beyond that level. Other elements are not necessary for growth but are not particularly toxic, or are toxic to plants even in small quantities under certain conditions. The exact degree of toxicity of any one contaminant as measured in the soil and ash at the site is difficult to estimate. The following factors create uncertainty in assessing the risk posed by the COPCs:

**"Bioavailability" of Elements.** The extraction methods used (EPA 1986c) may remove from the soil quantities of elements and compounds greater than are available to plants. The double-acid extraction removes the exchangeable fraction of metals, thereby giving a concentration that reflects the total potential pool of contaminants, not that to which the plant is exposed at any one time. Under field conditions these contaminants will be in the soil/ash solution in concentrations reflecting a dynamic equilibrium between the solid and liquid phase. Therefore, it is difficult to assess the types of interactions between contaminants and between contaminants and plant roots that may occur under field conditions at the site from the reported media metal concentrations. This is confounded by the concentration- and species-dependent synergistic and antagonistic interactions between metals during uptake by roots and inside plants. The analytical techniques also fail to differentiate between species of metals present in the media (particularly arsenic and chromium). In both cases, one prevalent species is considerably more toxic to plants and other life forms than another. Without specific analyses for these forms, or medium chemical and physical data sufficient to evaluate the probable occurrence of the species, it is not possible to accurately assess the risk posed by these contaminants at the site.

**Variable Response to Toxicants.** There is a lack of information on toxicity of contaminants to specific plant species and growth stages found at the Chestnut Ridge OU 2. There is a considerable amount of variability between plant species and plant growth stages in tolerance to specific, and combinations of, contaminants. The literature from which benchmarks were derived is not based on experiments using plants found in ecosystems representative of Chestnut Ridge OU 2. It is difficult to extrapolate from largely agricultural crops in early growth stage used in most of the published literature to hardwood and pine trees and the varied ground vegetation found over the site. Because of our lack of understanding of the complex interactions between contaminants, benchmark levels are necessarily derived from experiments in which plants are exposed to single contaminants. The reality of exposure at a site is usually one of multiple contaminant exposure, which may not be adequately assessed on the basis of literature-derived benchmarks.

**Lack of Benchmarks for Some Metals and Most Organic Compounds and Radionuclides.** Little research has been conducted on the phytotoxic effects of the radionuclides and organic compounds. Thus, it is not possible to assess the risk to plant growth posed by some of the analytes found in concentrations exceeding those found in reference soils.

Analytes found in ash and soil in concentrations above reference for which there are no benchmarks include: Ca, Mg, K, Na, $^{238}$Th, $^{232}$Th, $^{234}$U, $^{235}$U, $^{238}$U, 2-methylnaphthalene, alpha BHC, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, and chloroform.

**Sufficiency of media toxicity test data and biosurvey data.** A complete and accurate risk assessment requires input from as many lines of evidence as possible. An assessment for which there is no data from direct experimentation with the contaminated media and organisms representing assessment endpoints cannot be validated with site-specific data. Survey information is also required to understand how the populations in the area of concern have or have not adapted to exposure in a longer time frame than is seen in chemical analyses and short-term media toxicity tests. This can only be accomplished through survey of the contaminated site and comparison with reference sites.

### 6.4.3.5 Soil invertebrates

#### Earthworms

COPCs for Chestnut Ridge OU 2 include elements that are necessary for growth, non-toxic, and toxic to earthworms even in small quantities under certain conditions. The following factors create uncertainty in assessing the risk posed by the COPCs in the soil and ash.

**"Bioavailability" of Elements.** This factor is discussed in Sect. 6.4.3.4, focusing on the uncertainty associated with evaluating the risk to plants. The extraction methods used may remove from the soil quantities of elements and compounds greater than are effective on earthworms externally, or greater than will be extracted from soil as it passes through the gut.

**Variable Response to Contaminants.** There is variability between earthworm genera and growth stages in tolerance to specific, and combinations of, contaminants. Benchmark levels are derived from experiments in which earthworms are exposed to single contaminants. The reality of exposure at a site is usually one of multiple contaminant exposure.

**Lack of Benchmarks for Some Metals and Most Organic Compounds and Radionuclides.** Little research has been conducted on the toxic effects of the radionuclides and organic compounds, other than pesticides. Thus, it is not possible to assess the risk to earthworm populations posed by many of the analytes found in concentrations exceeding those found in reference soils.

**Soil/Litter-dwelling invertebrates other than earthworms**

COPCs for Chestnut Ridge OU 2 include elements that are necessary for growth, non-toxic, and toxic to soil/litter-dwelling invertebrates even in small quantities under certain conditions. The following factors create uncertainty in assessing the risk posed by the COPCs in vegetation.

**Variable Response to Contaminants.** There is variability between types of invertebrates and growth stages in tolerance to specific, and combinations of, contaminants. Benchmark levels are derived from experiments in which organisms are exposed to single contaminants. The reality of exposure at a site is usually one of multiple contaminant exposure.

**Lack of Benchmarks for Some Metals and Most Organic Compounds and Radionuclides.** No research has been conducted on the toxic effects of the radionuclides and organic compounds, other than pesticides. Thus, it is not possible to assess the risk to soil/litter-dwelling invertebrate populations posed by some of the analytes found in elevated concentrations in vegetation at the site.

**Sufficiency of Media Toxicity Test Data and Incomplete Biosurvey Data.** A complete and accurate risk assessment requires input from as many lines of evidence as possible. This point is discussed in the section on uncertainties of assessing the risk posed to plants by contaminants at the site.

**6.4.3.6 Soil microbes and microbial processes**

COPCs for Chestnut Ridge OU 2 include elements that are necessary for growth, non-toxic, and toxic to microbes and microbial processes even in small quantities under certain conditions. The following factors create uncertainty in assessing the risk posed by the COPCs in soil and ash.

**"Bioavailability" of Elements.** This factor is discussed in Sect. 6.4.3.4, focusing on the uncertainty associated with evaluating the risk to plants. The extraction methods used may remove from the soil quantities of elements and compounds greater than are available to microbes or effective on soil processes.

**Variable Response to Contaminants.** There is a considerable amount of variability between microbes at all taxonomic levels and between enzyme systems with regard to tolerance to specific, and combinations of, contaminants. Benchmark levels are necessarily derived from experiments in which microbial populations are exposed to single contaminants. The reality of exposure at a site is usually one of multiple contaminant exposure.

**Lack of Benchmarks for Some Metals and Most Organic Compounds and Radionuclides.** No research has been conducted on the toxic effects of the radionuclides and organic compounds on soil microbes. Thus, it is not possible to assess the risk to microbial activity

posed by some of the analytes found in concentrations exceeding those found in reference soils.

**Sufficiency of Media Toxicity Test Data and Biosurvey Data.** This point is discussed in the section on uncertainties of assessing the risk posed to plants by contaminants at the site.

### 6.4.3.7. Small mammals and reservation-wide endpoints

The following uncertainties are associated with the assessment of the risk that Chestnut Ridge OU 2 poses to small mammals and reservation-wide endpoints.

1. **Bioavailability of contaminants.** It was assumed that 100% of the contaminant concentration reported in the various media (water, ash, vegetation, etc.) was bioavailable. Much of the contaminant burden in biotic media (plants and small mammals) is likely to be bioavailable; however, it is very unlikely that 100% of the contaminants in abiotic media (ash or water) are bioavailable. While contaminants in biotic media may be bioavailable, the uptake efficiencies for wildlife species that consume plants or small mammals 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.

2. **Species/form of metal present in the environment.** Toxicity of metals varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. For example, $As^{+3}$, $Cr^{+6}$, and methyl mercury are more toxic than $As^{+5}$, $Cr^{+3}$, and inorganic mercury, respectively. The available data on the contaminant concentrations in media do not report which species or form of contaminant was observed. While species may be predicted for some metals (e.g., due to the high oxidizing potential of $Cr^{+6}$, most chromium in soil is likely to be $Cr^{+3}$; Eisler 1986), this was not possible for all metals. Because benchmarks used for comparison represented the more toxic species/forms of the metals (particularly for As, Cr, and Hg), if the less toxic species/form of the metal was actually present at Chestnut Ridge OU 2, potential toxicity at the site may be overestimated.

3. **Detection limits.** While thallium was observed in few media samples, it is listed as a contaminant of concern because the detection limit for this metal is higher than NOAELs. To evaluate risk, it was assumed that thallium concentrations were equal to the detection limit. This assumption is likely to overestimate the exposure to and risks from thallium.

4. **Interactions between contaminants.** Animal studies have shown that some of the metals observed at Chestnut Ridge OU 2 interact antagonistically such that the exposure to one reduces effects of the other. For example, Se protects mammals and birds from toxic effects of As, Cd, Hg, and Tl (Levander and Argrett 1969, Eisler 1985b). In contrast, while inorganic As reduces the toxicity of inorganic Se compounds, inorganic As enhances the toxicity of organic Se compounds (Eisler 1988a). The interactive effects of the metals to which wildlife at Chestnut Ridge OU 2 are exposed cannot be predicted and are wholly unknown. Therefore, risks may be either over- or underestimated.

5. **Single contaminant tests vs exposure to multiple contaminants in the field.** While wildlife residents at Chestnut Ridge OU 2 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 (see above), single contaminant studies may overestimate their toxic potential at Chestnut Ridge OU 2. Similarly, for those contaminants that interact synergistically, single contaminant studies may underestimate their toxic potential at Chestnut Ridge OU 2.

6. **Extrapolation from published toxicity data.** No published NOAELs were found for wildlife species present at Chestnut Ridge OU 2. To estimate the toxicity of contaminants at the site, it was necessary to extrapolate from NOAELs observed for test species (frequently mice, rats, or mallard ducks) to NOAELs for the wildlife species of interest at Chestnut Ridge OU 2. While it was assumed that toxicity could be estimated as a function of body size (see Sect. 6.3.1.6), the accuracy of the estimates is not known. For example, a wildlife species may be either more or less sensitive to a contaminant than would be predicted.

   Additional extrapolation uncertainty exists for those contaminants for which data either consisted of LOAELs or were subchronic in duration. For either case, a 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.

7. **Incomplete data to estimate exposure.** Contaminant residue data were not available for all media from all components of Chestnut Ridge OU 2. For example, surface water data are lacking for the Sluice Channel Area, vegetation and small mammal bioaccumulation data were unavailable for UMB, and no earthworms were available from any of the three areas. Because all exposure pathways are not represented, exposure estimates for the Sluice Channel Area and UMB and for shrews represent underestimates; actual exposures experienced by wildlife residents in the Sluice Channel Area and UMB and by shrews are likely to be greater than the estimated exposures.

8. **Variable food and water consumption.** While food consumption by wildlife at Chestnut Ridge OU 2 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 at Chestnut Ridge OU 2 could 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.

   For all estimates of contaminant exposure through food and water consumption, it was assumed that 100% of food or water that was consumed was obtained from a contaminated source and that the level of contamination was constant. Because wildlife are mobile, they may travel off the site to drink or forage, thus reducing their exposure. Additionally, contaminant concentrations in media at the site may vary both spatially and temporally. Therefore, the assumption that 100% of food and water consumed is contaminated is likely to overestimate actual contaminant exposure.

9. **Actual amount of use that Chestnut Ridge OU 2 receives.** The home ranges of mice and shrews are sufficiently small such that they are likely to be contained entirely within the

confines of Chestnut Ridge OU 2. Therefore, mice and shrews are likely to drink and forage exclusively within and receive 100% exposure from the site. In contrast, because the home ranges of deer, fox, and hawks are substantially larger than Chestnut Ridge OU 2, it is unlikely that individuals of these species will forage exclusively within the site. The absolute maximum exposure scenario (see Sect. 6.2.2.3) therefore is likely to overestimate the exposure that these wide-ranging species experience.

While the maximum proportional exposure scenario (see Sect. 6.2.2.3) may provide a more reasonable estimate, it assumes that use of Chestnut Ridge OU 2 will be proportional to published home range values and does not consider factors, such as habitat quality, that may affect use. Sources of uncertainty with this scenario include:

1. Variation in home range. If home ranges on the ORR are larger or smaller than those in the literature, exposure will be either over- or underestimated.

2. Attractancy or repellency of the site. Chestnut Ridge OU 2 may have favorable or undesirable attributes that may either increase or decrease use. For example, the high mineral content may be attracting deer from great distances to consume ash, greatly increasing the exposure for both individuals and the population as a whole.

### 6.4.4 Relation to Human Health Risk Assessment

ERAs and HHRAs have the same regulatory significance and the efforts should be coordinated. However, there are significant differences between the two types that result in distinct data needs, procedures, and interpretations for ERAs. The following discussion is drawn from Suter et al. (1993b).

1. Ecological effects that are used as ecological assessment endpoints can often be observed in the field. This allows the use of epidemiological approaches; for example, fish communities of a contaminated stream can be sampled and their characteristics compared to those of fish from an uncontaminated stream. In contrast, human health effects are very seldom measurable and are often hypothetical, so risks must be modeled.

2. EPA provides standard exposure parameters for humans in guidance documents and standard toxicological benchmarks in their data bases. It is up to ecological risk assessors to generate their own exposure parameters and benchmarks.

3. The selection of a limited but meaningful number of assessment endpoints for an ecological risk assessment is a problem not confronted by human health risk assessors who deal only with human health as an endpoint. The dilemma arises because there are numerous nonhuman species at a site, and because there are ecosystem properties as well as species which must be protected. At the Chestnut Ridge site there are contaminants that pose potential risks to terrestrial plant communities, aquatic and benthic populations, wildlife and organisms involved in nutrient cycling through organic matter decomposition. It is sometimes difficult to achieve a consensus on which properties of species and ecosystems are important enough to be used as assessment endpoints.

4. Many more routes of exposure must be considered in ecological risk assessments. In addition to the routes considered for humans at Chestnut Ridge OU 2, aquatic organisms

respire water, benthic invertebrates consume sediments, earthworms ingest soil, plant roots take up soil water, wildlife drink from surface water and ingest ash and soil, etc.

5. The forms and amounts of chemicals available to nonhuman organisms are often different. For example, dissolved metals are the form available to aquatic biota and terrestrial plants, but total metals are assumed to be available to human consumers. Similarly, humans consume only part of a fish while piscivorous wildlife consume whole fish. The present ecological risk assessment has shown that several contaminants not considered threatening to human health are present in concentrations for which severe problems for other species are documented. Examples include Al, which occurs above levels toxic to plants, soil microbes, and small mammals; Cu, which occurs above levels toxic to plants, earthworms, soil microbes; and Se and V, which occur above levels toxic to small mammals.

6. Spatial scale is an important consideration in ecological risk assessment. A small site like Chestnut Ridge OU 2 that may support populations of herbaceous plants, earthworms, and shrews may be only a part of the home range of local deer or hawk populations. It is reasonable to sample the plants, earthworms and shrews to assess the risks of chronic toxicity at the site. It would not be reasonable, however, to sample the deer and hawk populations and use them as assessment endpoints unless acute lethality is a possibility.

7. Temporal issues are more important for ecological risk assessment because of the seasonality of ecological processes and of some of the processes that mobilize, transform, and transport contaminants. For example, an aquatic system may be of importance as a spawning area for fish during one season, and the flux of contaminants during that season may be higher or lower than the mean. On the FCAP portion of Chestnut Ridge OU 2, the water content of the ash varies with season. This may alter the speciation (and toxicity) and availability of contaminants in the ash. Consequently, ecological sampling and analysis should be conducted seasonally for at least a year.

## 6.5 SUMMARY AND CONCLUSIONS

An ERA was conducted to evaluate potential adverse effects associated with coal ash disposal-related contamination at Chestnut Ridge OU 2 sites: the FCAP, Sluice Channel Area, and UMB areas. Potential impacts to both terrestrial and aquatic organisms were considered. Exposure to contaminants in surface water, sediments, shallow groundwater, surface soil, ash, and prey species were evaluated for selected endpoints, including fish, benthic macroinvertebrates, plants, soil invertebrates, soil microbes, small mammals, and several wide-ranging wildlife species. These organisms are exposed through ingestion, uptake, or direct contact with surface and groundwater, soils, and sediments, and ingestion of vegetation, invertebrates, or small animals by predators.

Exposures were quantified in Sect. 6.2.2. The results of media toxicity tests, biological surveys, and comparison of exposures to toxicological benchmarks were used in a weight-of-evidence approach to determine the likelihood of adverse ecological effects occurring due to the disposal of coal ash at Chestnut Ridge OU 2 sites. Evaluation of all lines of evidence available for fish, benthic macroinvertebrates, plants, soil invertebrates, small mammals, and wide-ranging mammals indicates the potential for adverse effects from current contaminant levels at Chestnut Ridge OU 2 sites. The risk characterization for each endpoint

is briefly summarized here (see Sect. 6.4 for details), followed by a listing of the contaminants affecting endpoints at each site. All future risks are expected to be similar to or less than current risks except in the event of a catastrophic failure of the dam holding back the FCAP. Dam failure would result in higher risks to aquatic organisms in UMB.

**Fish**

The two ponds at the FCAP do not support fish populations, but because they are small and isolated, they were not expected to support fish. Al, Cr, Cu, Fe, Pb, Mn, and Zn concentrations in water from FCAP ponds are at levels that could limit fish survival or reproduction (Sect. 6.4.1.1).

Ash disposal at Chestnut Ridge OU 2 has had catastrophic effects on the fish community of UMB. Consideration of available evidence indicates contamination has had and is continuing to influence the capability of UMB to support fish populations. No fish were observed during biological surveys. Al; Cu, Fe, and Zn are all present in concentrations high enough to reduce survival or reproduction; and Cr, Pb, and Mn concentrations are sufficient to reduce the population abundance or production of sensitive species (Table 6.39). However, ambient water tests produced no clear indications of toxicity.

**Benthic Macroinvertebrates**

Metal concentrations in sediment and surface water at the FCAP were high enough to reduce survival (sediment: As) or survival and reproduction (surface water: Al, Cr, Cu, Fe, Pb, Mn, and Zn) of benthic macroinvertebrates, but the lack of other lines of evidence prevents a definitive characterization of risks for these organisms (Sect. 6.4.1.2).

Metal concentrations in sediments and surface water of UMB exceeded benchmarks and were sufficient to reduce survival (sediments: Ba, Mn, and Ni) or survival and reproduction (surface water: Al, As, Cu, Fe, Pb, Hg, and Zn) of benthic macroinvertebrates (Table 6.44). Water and sediment toxicity tests did not provide evidence of acute or chronic toxicity, and there was inconclusive evidence from biological surveys to determine impacts.

**Plants**

Growth and possibly reproduction of plants at the FCAP, Sluice Channel Area, and UMB sites may be depressed by $\geq$20% based on results of media analyses. Al, Sb, As, Ba, Cd, Cr, Co, Cu, Fe, Mn, Hg, Ni, Se, Ag, V, and Zn were all found in ash, soil, and shallow groundwater at concentrations high enough to cause reduced growth (Table 6.62). FCAP ash tended to have higher concentrations of all contaminants than the Sluice Channel Area or UMB ashes. Al, As, Se, and V represented the highest risks to plants.

**Soil Invertebrates**

Growth and reproduction of earthworms is likely to have been depressed by $\geq$20% at the FCAP, Sluice Channel Area, and UMB sites. No earthworms were found during surveys of the FCAP and Sluice Channel Area (UMB was not surveyed), and toxicity tests suggest possible toxicity of ash to earthworms. Cd, Cr, Cu, Hg, Ag, and Tl were all found in ash and soil at concentrations high enough to cause reduced growth or reproduction in earthworms (Table 6.64).

Evidence of risks to soil or litter-dwelling invertebrates other than earthworms was inconclusive. Few invertebrates were observed in surveys of ash from the Sluice Channel Area and FCAP, indicating negative effects. However, contaminant levels measured in vegetation from the Sluice Channel Area and FCAP were lower than levels considered toxic to litter-decomposing invertebrates. Because the number of contaminants with available benchmarks is limited, and vegetation was analyzed for only a small number of contaminants relative to those known to occur at the sites, the failure to demonstrate toxicity may not accurately characterize risks to these invertebrates.

### Soil Microbes and Microbial Processes

Growth and functioning of microbial populations in ash and soil at Chestnut Ridge OU 2 sites are likely to have been depressed by $\geq$20%. Al, Cd, Cr, Cu, Fe, Mn, Hg, V, and Zn were all found in ash and soil in concentrations that could cause reduced microbial growth and reduction in enzyme activities involved in organic matter breakdown and nutrient cycling (Table 6.67). UMB soils contain more contaminants at higher levels than Sluice Channel Area soils. Ash at the FCAP tended to have higher toxic contaminant concentrations than Sluice Channel Area or UMB ash.

### Small Mammals

The abundance of shrews and possibly other small mammals is likely to have been depressed by $\geq$20% at the FCAP, Sluice Channel Area, or UMB. Biological surveys suggest minimal effects on rodents; data for shrews may suggest toxicity but are inconclusive. Measured body burdens in small mammals from Chestnut Ridge OU 2 were less than those found to have no effects in other studies. Toxicological data that indicate concentrations of Al, As, Ba, Cd, Se, Tl, and V in ash, food, and water are sufficient to cause reduced survival or reproduction in small mammals (Table 6.70).

### Reservation-wide Endpoints

Survival or reproduction of individual white-tailed deer and red fox occurring at Chestnut Ridge OU 2 sites is likely to be depressed under both maximum and proportional exposure scenarios. Red-tailed hawk survival or reproduction is likely to be depressed under the maximum exposure scenario, but not under the proportional scenario. Assuming maximum exposures, metal concentrations in ash, food, and water are sufficient to cause reduced survival or reproduction in the white-tailed deer (Al, As, Ba, Cd, Se, and Tl), red fox (Al, As, Ba, Cr, Hg, Se, and Tl), and red-tailed hawk (As, Cr, Hg, and Se) (Table 6.74).

### Filled Coal Ash Pond

Contaminants in the soil, ash, and water of the FCAP present a hazard to fish, benthic macroinvertebrates, plants, soil invertebrates, soil microbes, small mammals, and wide-ranging wildlife species. Contaminants of concern include Al, As, Cr, Cu, Fe, Pb, Mn, Hg, Ni, Se, Tl, V, and Zn.

Although no fish occupy the FCAP, concentrations of Al, Cr, Cu, Fe, Pb, Mn, and Zn would likely result in adverse effects to fish if they were present. Sediment concentrations of Al, As, Cu, Cr, Fe, Pb, Mn, and Zn represent a hazard to benthic macroinvertebrate populations.

Al, As, Cr, Cu, Hg, Ni, Se, Tl, V, and Zn all exceeded soil benchmarks for phytotoxicity in ash at the FCAP; Al, As, Cr, Se, and V had hazard quotients $\geq$19. Al, Cu, Fe, Mn, and Zn in shallow groundwater exceeded benchmarks for phytotoxicity in soil solution. Cr and Hg in FCAP ash exceeded benchmarks for toxicity to earthworms. Hazard quotients for Al, Cr, Cu, Fe, Hg, V, and Zn in ash were all $\geq$6, suggesting significant risk to soil microorganisms and microbial processes.

Terrestrial wildlife are exposed to contaminants in soil, ash, water, and food. Contaminants representing a hazard to small mammals at the FCAP include Al, As, Ba, Se, Tl, and V. Shrews received greater exposures than mice. Wide-ranging species also experienced exposures likely to result in adverse effects: white-tailed deer (Al, As, Ba, Cd, and V), red fox (Al, As, Cr, Hg, Se, and Tl), and red-tailed hawk (As, Cr, Hg, and Se).

**Sluice Channel Area**

Contaminants in the soil, ash, and shallow groundwater of the Sluice Channel Area present a hazard to plants, soil invertebrates, soil microbes, small mammals, and wide-ranging wildlife species. Contaminants of concern include Al, As, Ba, Cd, Cr, Cu, Fe, Mn, Hg, Ni, Se, Tl, V, and Zn.

Contaminant concentrations in surface soil (Al, As, Mn, and V) and ash (Al, As, Cr, Hg, Ni, Se, V, and Zn) at the Sluice Channel Area exceeded benchmarks and could result in reduced plant growth. The detection limits for Sb and Ag in Sluice Channel Area ash exceeded benchmarks, thus they may also be of concern. Al, Cu, Fe, Mn, and Zn in shallow groundwater exceeded benchmarks for phytotoxicity in soil solution. Cr and Hg in Sluice Channel Area ash exceeded benchmarks for toxicity to earthworms. Al, Mn, and V in soil and Al, Cr, Cu, Fe, Hg, V, and Zn in ash exceeded benchmarks for toxicity to soil microorganisms and microbial processes.

Contaminants representing a hazard to small mammals at the Sluice Channel Area include Al, As, Ba, Se, Tl, and V. Shrews received greater exposures than mice. Wide-ranging species also experienced exposures likely to result in adverse effects: white-tailed deer (Al, As, Ba, Cd, and V), red fox (Al, As, Cr, Hg, Se, and Tl), and red-tailed hawk (As, Cr, Hg, and Se).

**Upper McCoy Branch**

Contaminants in the soil, ash, surface water, and shallow groundwater of UMB present a hazard to fish, benthic macroinvertebrates, plants, soil invertebrates, soil microbes, small mammals, and wide-ranging wildlife species. Contaminants of concern include Al, As, Ba, Cd, Cr, Cu, Fe, Pb, Mn, Hg, Ni, Se, V, and Zn.

Concentrations of Al, Cu, Cr, Fe, Pb, Mn, and Zn in UMB surface water are high enough to impact survival and reproduction of fish. Bis(2-ethylhexyl)phthalate may represent a hazard to fish at UMB. Bis(2-ethylhexyl)phthalate, a common laboratory contaminant, is not believed to be associated with Chestnut Ridge OU 2 operations and therefore is not believed to be a COPC; however, no statistical tests/analyses allowed it to be eliminated from the COPC list for this ERA. Sediment concentrations of Al, As, Ba, Cu, Fe, Pb, Mn, Hg, Ni, and Zn represent a hazard to benthic macroinvertebrate populations.

Al, As, Cr, Ni, Se, V, and Zn concentrations in UMB soil and ash; Mn in soil; and Cd in ash exceeded soil benchmarks for phytotoxicity. Al, Cu, Fe, Mn, and Zn in shallow groundwater exceeded benchmarks for phytotoxicity in soil solution. Cr in soil and ash exceeded benchmarks for toxicity to earthworms. In UMB soil, Al Cd, Cr, Cu, Fe, Mn, V, and Zn exceeded soil benchmarks for soil microorganisms and microbial processes; the same compounds were a problem in UMB ash except Mn was replaced by Hg.

Contaminants representing a hazard to small mammals at UMB include Al, As, Ba, Cd, Se, and V. Shrews received greater exposures than mice. Wide-ranging species also experienced exposures likely to result in adverse effects: white-tailed deer (Al, As, Ba, Cd, and V), red fox (Al, As, and Ba), and red-tailed hawk (As, Cr, Hg, and Se).

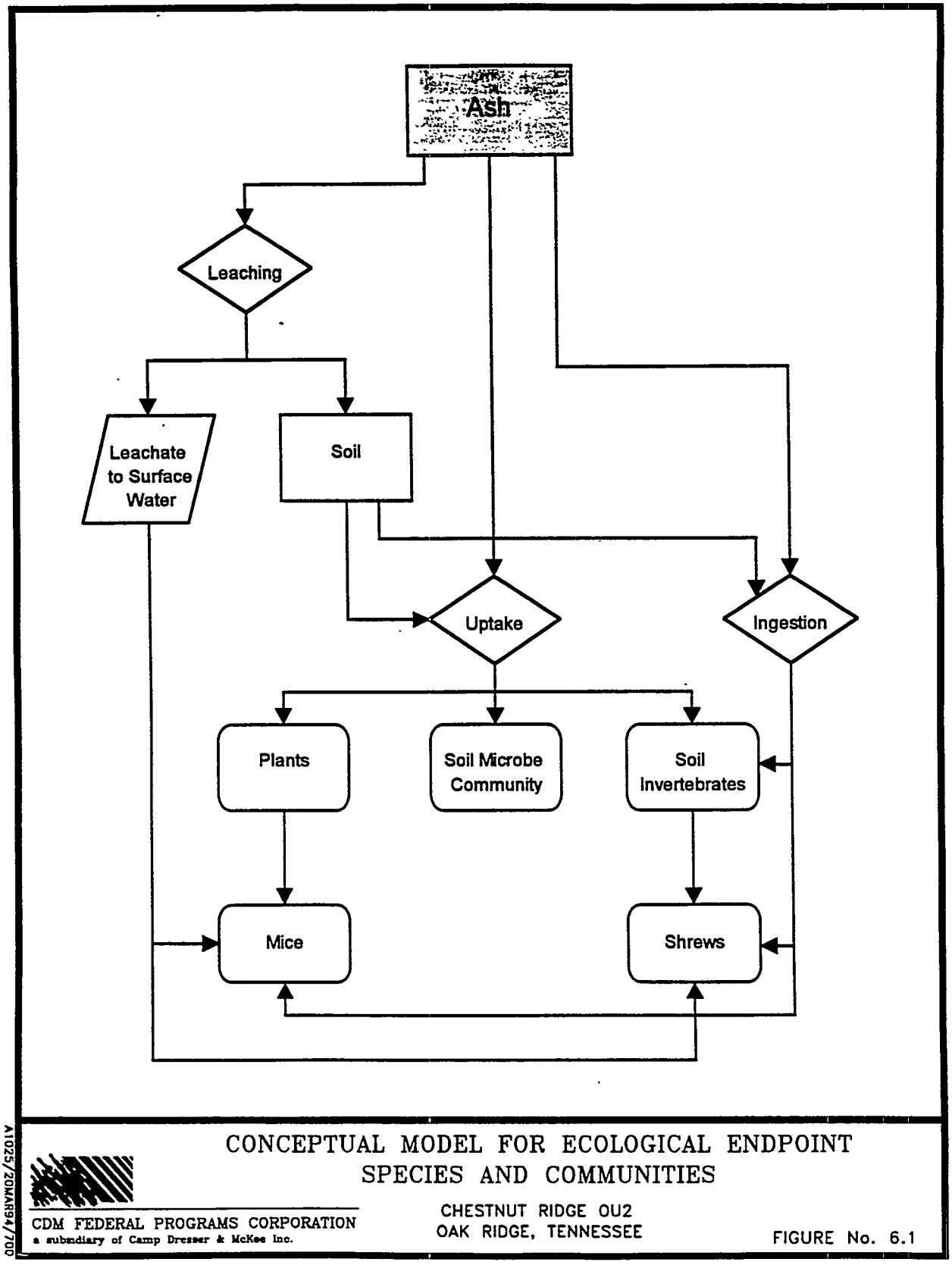



Fig. 6.1. Conceptual model for ecological endpoint species and communities.

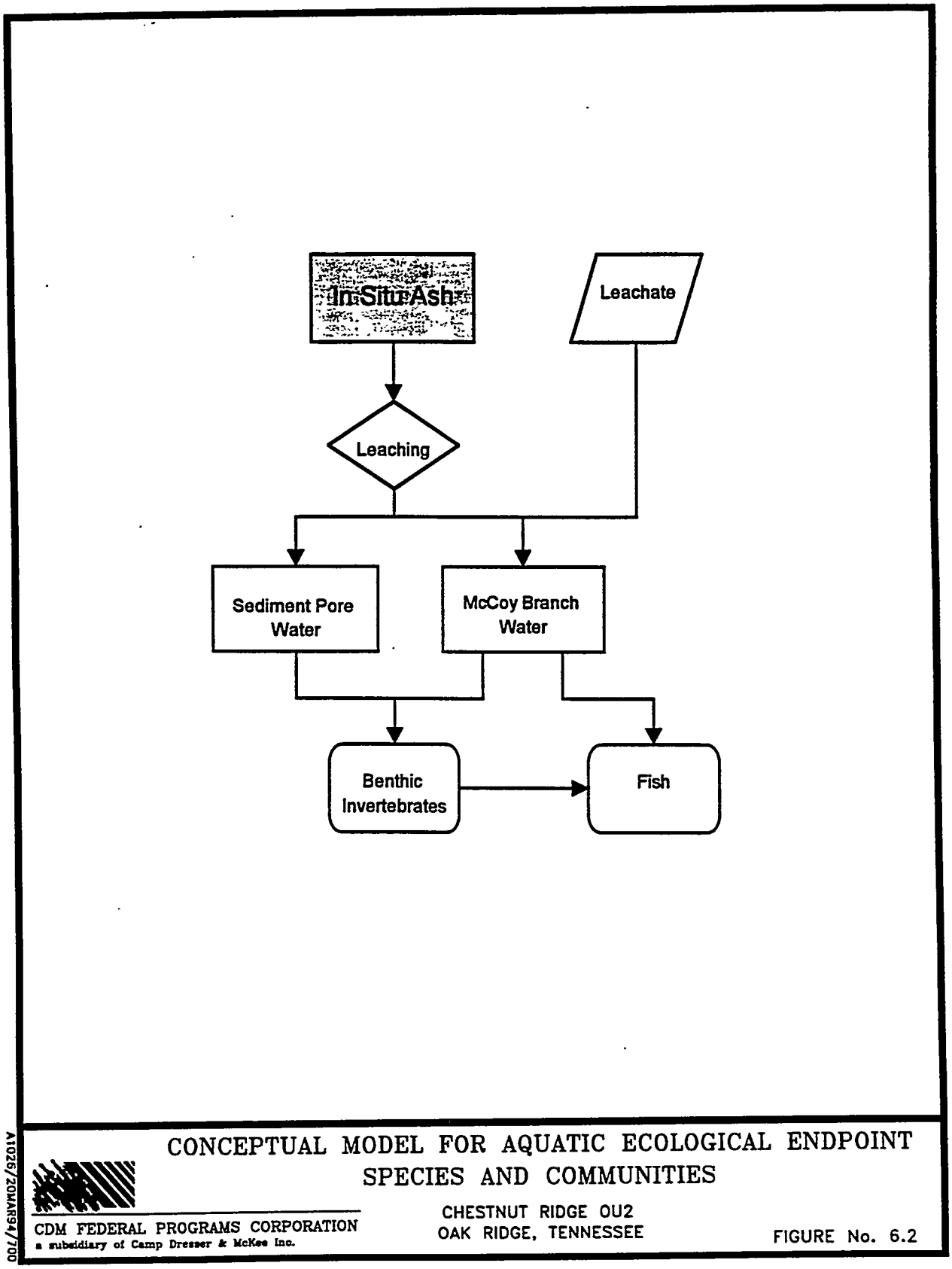



Fig. 6.2. Conceptual model for aquatic ecological endpoint species and communities.

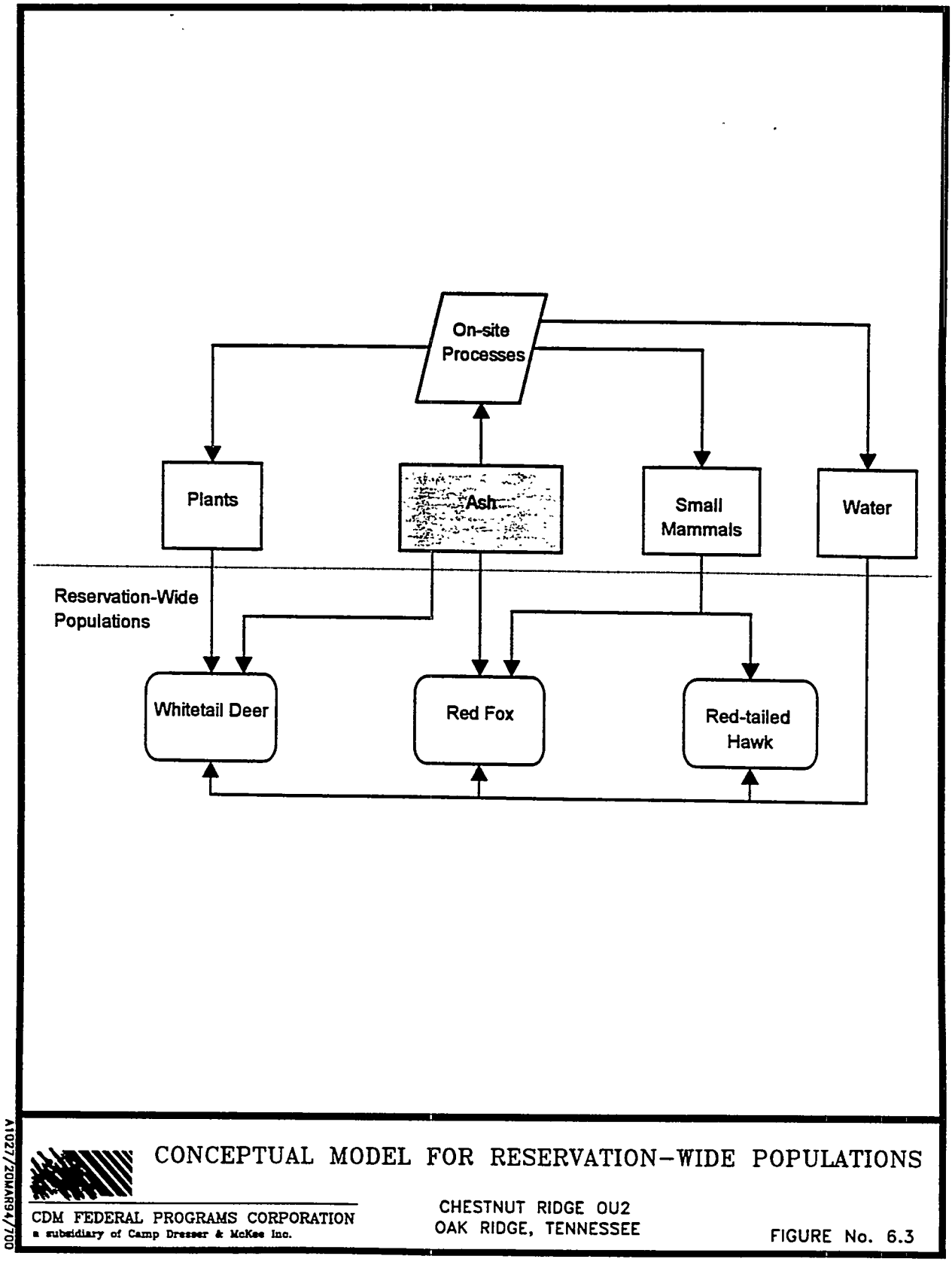



Fig. 6.3. Conceptual model for reservation—wide populations.

**Table 6.1 Assessment and measurement endpoints for Chestnut Ridge OU 2**

| Endpoint community | Assessment endpoint | Measurement endpoint |
|---|---|---|
| Fish | Species richness | Species richness |
| | Fish population viability | Age-class distribution for banded sculpins |
| Benthic macroinvertebrate | Total species richness | Total species richness |
| | EPT[1] richness | EPT[1] richness |
| Plant | Abundance and productivity | Community type representative of habitat |
| | | Laboratory phytotoxicity tests[2] for productivity endpoints |
| Soil heterotroph | Earthworm abundance | Earthworm abundance |
| | Soil heterotrophic function | Toxicity tests for earthworms and bacterial nitrification using ambient media |
| | | Laboratory toxicity tests[2] for reproductive and nutrient cycling endpoints |
| Small mammal | Abundance | Laboratory toxicity tests[2] for mortality and reproductive endpoints |

[1]EPT = Ephemeroptera, Plecoptera, and Trichoptera.
[2]From published literature.

**Table 6.2a. Reference sediment data for Chestnut Ridge OU 2**

| Analyte | Number of samples | | Mean | UCL95 on mean |
|---|---|---|---|---|
| | Detected | Analyzed | | |
| *Inorganics (mg/kg)* | | | | |
| Aluminum | 4 | 4 | 4.40e+03 | 6.03e+03 |
| Antimony | 0 | 4 | 1.12e+01 | 1.56e+01 |
| Arsenic | 4 | 4 | 7.23e+00 | 1.25e+01 |
| Barium | 4 | 4 | 1.23e+02 | 2.01e+02 |
| Beryllium | 3 | 4 | 8.15e-01 | 1.16e+00 |
| Cadmium | 0 | 4 | 1.48e+00 | 2.20e+00 |
| Calcium | 4 | 4 | 7.49e+03 | 1.71e+04 |
| Chromium | 4 | 4 | 1.92e+01 | 2.73e+01 |
| Cobalt | 4 | 4 | 1.31e+01 | 1.76e+01 |
| Copper | 3 | 4 | 6.40e+00 | 1.33e+01 |
| Iron | 4 | 4 | 1.81e+04 | 2.09e+04 |
| Lead | 4 | 4 | 8.43e+00 | 1.18e+01 |
| Magnesium | 4 | 4 | 2.42e+03 | 5.56e+03 |
| Manganese | 4 | 4 | 2.05e+03 | 3.15e+03 |
| Mercury | 0 | 4 | 1.48e-01 | 2.20e-01 |
| Nickel | 3 | 4 | 8.55e+00 | 1.14e+01 |
| Potassium | 1 | 4 | 5.02e+02 | 7.56e+02 |
| Selenium | 0 | 3 | 9.33e-01 | 1.30e+00 |
| Silver | 0 | 4 | 2.23e+00 | 3.00e+00 |
| Sodium | 1 | 4 | 1.15e+02 | 3.24e+01 |
| Thallium | 0 | 4 | 1.16e+00 | 1.70e+00 |
| Vanadium | 4 | 4 | 2.91e+01 | 3.50e+01 |
| Zinc | 4 | 4 | 1.39e+02 | 2.89e+02 |
| *Radionuclides (pCi/g)* | | | | |
| Thorium-228 | 4 | 4 | 6.35e-01 | 8.40e-01 |
| Thorium-230 | 0 | 4 | 1.39e+00 | 1.80e+00 |
| Thorium-232 | 4 | 4 | 5.53e-01 | 7.10e-01 |
| Uranium-234 | 4 | 4 | 8.68e-01 | 1.00e+00 |
| Uranium-235 | 3 | 4 | 1.39e-01 | 2.20e-01 |
| Uranium-238 | 4 | 4 | 9.28e-01 | 1.30e+00 |

**Table 6.2b. Reference soil data for Chestnut Ridge OU 2**

| Analyte | Number of samples | | Mean | UCL95 on mean |
|---|---|---|---|---|
| | Detected | Analyzed | | |
| *Inorganics (mg/kg)* | | | | |
| Aluminum | 5 | 5 | 8.19e+03 | 1.03e+04 |
| Antimony | 0 | 3 | 1.12e+01 | 1.25e+01 |
| Arsenic | 5 | 5 | 3.80e+00 | 5.76e+00 |
| Barium | 5 | 5 | 8.58e+01 | 1.08e+02 |
| Beryllium | 4 | 5 | 5.92e-01 | 7.65e-01 |
| Cadmium | 2 | 5 | 1.40e+00 | 1.82e+00 |
| Calcium | 5 | 5 | 1.68e+03 | 2.23e+03 |
| Chromium | 5 | 5 | 1.15e+01 | 2.03e+01 |
| Cobalt | 5 | 5 | 1.16e+01 | 2.07e+01 |
| Copper | 3 | 5 | 6.34e+00 | 6.80e+00 |
| Iron | 5 | 5 | 1.03e+04 | 1.71e+04 |
| Lead | 5 | 5 | 2.37e+01 | 3.34e+01 |
| Magnesium | 5 | 5 | 5.15e+02 | 6.14e+02 |
| Manganese | 5 | 5 | 1.59e+03 | 2.37e+03 |
| Mercury | 5 | 5 | 2.10e-01 | 3.07e-01 |
| Nickel | 4 | 5 | 7.54e+00 | 8.50e+00 |
| Phosphorus | 2 | 2 | 2.69e+02 | 3.26e+02 |
| Potassium | 3 | 5 | 5.81e+02 | 7.39e+02 |
| Selenium | 1 | 5 | 6.44e-01 | 1.03e+00 |
| Silver | 0 | 3 | 2.37e+00 | 2.70e+00 |
| Sodium | 4 | 5 | 1.23e+02 | 1.88e+02 |
| Thallium | 0 | 3 | 1.03e+00 | 1.10e+00 |
| Vanadium | 5 | 5 | 2.20e+01 | 3.51e+01 |
| Zinc | 5 | 5 | 4.04e+01 | 4.95e+01 |
| *Radionuclides (pCi/g)* | | | | |
| Cesium-137 | 2 | 2 | 2.69e-01 | 3.30e-01 |
| Thorium-228 | 3 | 3 | 3.50e-01 | 5.10e-01 |
| Thorium-230 | 1 | 3 | 3.60e+00 | 8.60e+00 |
| Thorium-232 | 3 | 3 | 4.37e-01 | 5.20e-01 |
| Uranium-234 | 3 | 3 | 3.20e+00 | 3.50e+00 |
| Uranium-235 | 3 | 3 | 2.23e-01 | 3.40e-01 |
| Uranium-238 | 5 | 5 | 1.69e+00 | 2.22e+00 |

Table 6.2c. Reference surface water data for Chestnut Ridge OU 2

| Analyte | Number of samples | | Mean | UCL95 on mean |
|---|---|---|---|---|
| | Detected | Analyzed | | |
| *Inorganics (mg/L)* | | | | |
| Aluminum | 1 | 5 | 1.72e-01 | 3.78e-01 |
| Antimony | 0 | 5 | 3.60e-02 | 3.60e-02 |
| Arsenic | 0 | 5 | 3.00e-03 | 3.00e-03 |
| Barium | 5 | 5 | 6.85e-02 | 1.14e-01 |
| Beryllium | 0 | 5 | 1.00e-03 | 1.00e-03 |
| Cadmium | 0 | 5 | 5.00e-03 | 5.00e-03 |
| Calcium | 5 | 5 | 2.39e+01 | 3.56e+01 |
| Chromium | 0 | 5 | 8.00e-03 | 8.00e-03 |
| Chromium VI | 0 | 1 | 1.00e-02 | 1.00e-02 |
| Cobalt | 0 | 5 | 8.00e-03 | 8.00e-03 |
| Copper | 0 | 5 | 4.00e-03 | 4.00e-03 |
| Iron | 4 | 5 | 5.04e-01 | 1.48e+00 |
| Lead | 0 | 5 | 2.00e-03 | 2.00e-03 |
| Magnesium | 5 | 5 | 1.22e+01 | 1.74e+01 |
| Manganese | 5 | 5 | 1.01e-01 | 2.79e-01 |
| Mercury | 0 | 5 | 2.00e-04 | 2.00e-04 |
| Nickel | 0 | 5 | 1.70e-02 | 1.70e-02 |
| Potassium | 1 | 5 | 1.51e+00 | 1.86e+00 |
| Selenium | 0 | 5 | 3.00e-03 | 3.00e-03 |
| Silver | 0 | 5 | 7.00e-03 | 7.00e-03 |
| Sodium | 5 | 5 | 6.58e-01 | 8.39e-01 |
| Thallium | 0 | 5 | 4.00e-03 | 4.00e-03 |
| Vanadium | 0 | 5 | 6.00e-03 | 6.00e-03 |
| Zinc | 5 | 5 | 6.72e-03 | 9.52e-03 |
| *Radionuclides (pCi/L)* | | | | |
| Thorium-228 | 0 | 5 | 1.64e-02 | 4.50e-02 |
| Thorium-230 | 0 | 5 | 2.98e-01 | 5.90e-01 |
| Thorium-232 | 1 | 5 | 1.36e-01 | 4.00e-01 |
| Uranium-234 | 5 | 5 | 2.20e-01 | 3.82e-01 |
| Uranium-235 | 0 | 5 | 1.08e-02 | 5.00e-02 |
| Uranium-238 | 1 | 5 | 6.16e-02 | 1.31e-01 |

**Table 6.3. Background soil data from the BSCP[a]**

| Analyte | 95% UCB on median[b] | |
|---|---|---|
| | Chepultepec | Copper Ridge |
| *Inorganics (mg/kg)* | | |
| Aluminum | 9.51e+03 | 1.18e+04 |
| Arsenic | 1.44e+01 | 3.07e+01 |
| Barium | 6.95e+01 | 9.32e+01 |
| Beryllium | 4.60e-01 | 6.34e-01 |
| Calcium | 6.11e+02 | 6.96e+02 |
| Chromium | 1.74e+01 | 1.83e+01 |
| Cobalt | 1.53e+01 | 1.03e+01 |
| Copper | 5.26e+00 | 8.19e+00 |
| Iron | 1.65e+04 | 1.39e+04 |
| Lead | 2.46e+01 | 5.22e+01 |
| Lithium | 4.99e+00 | 3.48e+00 |
| Magnesium | 4.43e+02 | 5.57e+02 |
| Manganese | 1.26e+03 | 1.46e+03 |
| Mercury | 1.53e-01 | 1.84e-01 |
| Molybdenum | N/A | 1.75e+00 |
| Nickel | N/A | 9.71e+00 |
| Potassium | N/A | 4.51e+02 |
| Selenium | 6.25e-01 | 8.03e-01 |
| Silicon | 5.95e+02 | 7.64e+02 |
| Sodium | 3.44e+02 | 3.81e+02 |
| Strontium | 3.33e+00 | 4.81e+00 |
| Sulfate | 1.03e+02 | 8.82e+01 |
| Vanadium | 3.43e+01 | 3.03e+01 |
| Zinc | 4.86e+01 | 4.32e+01 |
| *Radionuclides (pCi/g)* | | |
| Cesium-137 | 2.36e+00 | 1.99e+00 |
| Thorium-228 | 8.63e-01 | 4.84e-01 |
| Thorium-230 | 9.27e-01 | 1.33e+00 |

**Table 6.3 (continued)**

| Analyte | 95% UCB on median[b] | |
|---|---|---|
| | Chepultepec | Copper Ridge |
| Thorium-232 | 7.22e-01 | 7.89e-01 |
| Uranium-233/234 | 1.34e+00 | 1.75e+00 |
| Uranium-235 | 1.04e-01 | 1.77e-01 |
| Uranium-238 | 1.26e+00 | 1.54e+00 |

[a]*Background Soil Characterization Project for the Oak Ridge Reservation* (Energy Systems 1993b).

[b]95% Upper Confidence Bound on the median, for each formation, from the *Background Soil Characterization Project for the Oak Ridge Reservation*, DOE/OR/01-1174/V1, and Appendixes, DOE/OR/01-1175/V2 (Energy Systems 1993b).

**Table 6.4. Shallow groundwater data for Chestnut Ridge OU 2[a]**

| Analyte | Samples | | Maximum nondetect | Maximum detect | Mean |
|---|---|---|---|---|---|
| | Detected | Analyzed | | | |
| *Inorganics (mg/L)* | | | | | |
| Aluminum | 3 | 6 | 8.80e-02 | 2.18e+01 | 7.62e+00 |
| Antimony | 0 | 3 | 3.60e-02 | | 3.60e-02 |
| Arsenic | 2 | 6 | 3.00e-03 | 3.20e-03 | 2.87e-03 |
| Barium | 6 | 6 | | 6.74e-01 | 2.34e-01 |
| Beryllium | 3 | 6 | 1.00e-03 | 4.00e-03 | 1.50e-03 |
| Cadmium | 1 | 6 | 5.00e-03 | 8.00e-03 | 4.83e-03 |
| Calcium | 6 | 6 | | 1.59e+02 | 9.75e+01 |
| Chloride | 3 | 3 | | 5.00e+00 | 3.33e+00 |
| Chromium | 3 | 6 | 8.00e-03 | 3.14e-02 | 1.71e-02 |
| Cobalt | 3 | 6 | 8.00e-03 | 2.76e-02 | 1.43e-02 |
| Copper | 3 | 6 | 4.00e-03 | 7.35e-02 | 2.18e-02 |
| Fluoride | 3 | 3 | | 2.00e-01 | 1.67e-01 |
| Iron | 5 | 6 | 4.10e-02 | 3.35e+01 | 1.14e+01 |
| Lead | 3 | 6 | 2.00e-03 | 5.29e-02 | 1.23e-02 |
| Magnesium | 6 | 6 | | 3.74e+01 | 2.50e+01 |
| Manganese | 6 | 6 | | 2.75e+00 | 1.53e+00 |
| Mercury | 2 | 6 | 2.00e-04 | 3.00e-04 | 1.50e-04 |
| Nickel | 3 | 6 | 1.70e-02 | 4.11e-02 | 2.18e-02 |
| Niobium | 1 | 3 | 7.00e-03 | 1.02e-02 | 8.07e-03 |
| Nitrate | 2 | 3 | 1.00e+00 | 7.00e+00 | 3.67e+00 |
| Phosphorus | 1 | 3 | 2.00e-01 | 5.30e-01 | 3.10e-01 |
| Potassium | 5 | 6 | 1.35e+00 | 7.71e+00 | 3.92e+00 |
| Selenium | 1 | 6 | 8.00e-03 | 5.80e-03 | 4.63e-03 |
| Silver | 0 | 3 | 7.00e-03 | | 7.00e-03 |
| Sodium | 6 | 6 | | 1.05e+01 | 4.58e+00 |
| Sulfate | 3 | 3 | | 3.70e+01 | 2.47e+01 |
| Thallium | 0 | 3 | 4.00e-03 | | 4.00e-03 |
| Vanadium | 3 | 6 | 6.00e-03 | 6.27e-02 | 2.22e-02 |
| Zinc | 6 | 6 | | 2.07e+00 | 4.15e-01 |
| *Radionuclides (pCi/L)* | | | | | |
| Radium-226 | 2 | 2 | | 1.16e+00 | 9.99e-01 |
| Thorium-228 | 0 | 2 | 1.80e-02 | | 2.00e-03 |

**Table 6.4 (continued)**

| Analyte | Samples | | Maximum nondetect | Maximum detect | Mean |
|---|---|---|---|---|---|
| | Detected | Analyzed | | | |
| Thorium-230 | 0 | 2 | 3.90e-01 | | 2.75e-01 |
| Thorium-232 | 0 | 2 | 2.70e-02 | | 1.10e-02 |
| Uranium-234 | 2 | 2 | | 3.50e-01 | 2.85e-01 |
| Uranium-235 | 0 | 2 | 5.90e-02 | | 3.85e-02 |
| Uranium-238 | 4 | 4 | | 6.73e-01 | 4.11e-01 |
| *Organics (mg/L)* | | | | | |
| 1,1,1-Trichloroethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,1,2,2-Tetrachloroethane | 0 | 1 | 1.00e-02 | | 1.00e-02 |
| 1,1,2-Trichloroethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,1-Dichloroethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,1-Dichloroethene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,2,4-Trichlorobenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,2-Dichlorobenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,2-Dichloroethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,2-Dichloroethene (total) | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,2-Dichloropropane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,3-Dichlorobenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 1,4-Dichlorobenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2,2'-Oxybis(1-Chloropropane) | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2,4,5-Trichlorophenol | 0 | 2 | 2.50e-02 | | 2.50e-02 |
| 2,4,6-Trichlorophenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2,4-Dichlorophenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2,4-Dimethylphenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2,4-Dinitrophenol | 0 | 2 | 2.50e-02 | | 2.50e-02 |
| 2,4-Dinitrotoluene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2,6-Dinitrotoluene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2-Butanone | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2-Chloronaphthalene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2-Chlorophenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2-Hexanone | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2-Methylnaphthalene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2-Methylphenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 2-Nitroaniline | 0 | 2 | 2.50e-02 | | 2.50e-02 |

Table 6.4 (continued)

| Analyte | Samples | | Maximum nondetect | Maximum detect | Mean |
|---|---|---|---|---|---|
| | Detected | Analyzed | | | |
| 2-Nitrophenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 3,3'-Dichlorobenzidine | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 3-Nitroaniline | 0 | 2 | 2.50e-02 | | 2.50e-02 |
| 4,6-Dinitro-2-Methylphenol | 0 | 2 | 2.50e-02 | | 2.50e-02 |
| 4-Bromophenyl-phenylether | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 4-Chloro-3-Methylphenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 4-Chloroaniline | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 4-Chlorophenyl-phenylether | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 4-Methyl-2-Pentanone | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 4-Methylphenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| 4-Nitroaniline | 0 | 2 | 2.50e-02 | | 2.50e-02 |
| 4-Nitrophenol | 0 | 2 | 2.50e-02 | | 2.50e-02 |
| Acenaphthene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Acenaphthylene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Acetone | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Anthracene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Benzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Benzo(a)Anthracene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Benzo(a)Pyrene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Benzo(b)Fluoranthene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Benzo(g,h,i)Perylene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Benzo(k)Fluoranthene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| bis(2-Chloroethoxy)Methane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| bis(2-Chloroethyl)Ether | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| bis(2-Ethylhexyl)Phthalate | 1 | 2 | 1.00e-02 | 1.00e-03 | 5.50e-03 |
| Bromodichloromethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Bromoform | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Bromomethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Butylbenzylphthalate | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Carbazole | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Carbon Disulfide | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Carbon Tetrachloride | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Chlorobenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |

**Table 6.4 (continued)**

| Analyte | Samples | | Maximum nondetect | Maximum detect | Mean |
|---|---|---|---|---|---|
| | Detected | Analyzed | | | |
| Chloroethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Chloroform | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Chloromethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Chrysene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| cis-1,3-Dichloropropene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Di-n-Butylphthalate | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Di-n-Octylphthalate | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Dibenz(a,h)Anthracene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Dibenzofuran | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Dibromochloromethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Diethylphthalate | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Dimethyl Phthalate | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Ethylbenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Fluoranthene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Fluorene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Hexachlorobenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Hexachlorobutadiene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Hexachlorocyclopentadiene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Hexachloroethane | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Indeno(1,2,3-cd)Pyrene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Isophorone | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Methylene Chloride | 0 | 2 | 1.40e-02 | | 1.20e-02 |
| N-Nitroso-Di-n-Propylamine | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| N-Nitrosodiphenylamine | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Naphthalene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Nitrobenzene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Pentachlorophenol | 0 | 2 | 2.50e-02 | | 2.50e-02 |
| Phenanthrene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Phenol | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Pyrene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Styrene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Tetrachloroethene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Toluene | 0 | 2 | 1.00e-02 | | 1.00e-02 |

**Table 6.4 (continued)**

| Analyte | Samples | | Maximum nondetect | Maximum detect | Mean |
|---|---|---|---|---|---|
| | Detected | Analyzed | | | |
| trans-1,3-Dichloropropene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Trichloroethene | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Vinyl Chloride | 0 | 2 | 1.00e-02 | | 1.00e-02 |
| Xylene (total) | 0 | 2 | 1.00e-02 | | 1.00e-02 |

[a]Data are from well stations 672, 674, and 676 only, as decribed in Sect. 6.2.2.3.

**Table 6.5. Total inorganic uranium concentrations for Chestnut Ridge OU 2 (mg/L)[a]**

| Area | Detected | Analyzed | Maximum detect | Mean | UCL95 on mean |
|---|---|---|---|---|---|
| FCAP | 2 | 3 | 4.83e-04 | 3.11e-04 | 6.18e-04 |
| UMB | 7 | 11 | 1.24e-03 | 1.17e-03 | 1.25e-03 |
| Reference | 5 | 5 | 8.07e-04 | 2.87e-04 | 5.70e-04 |

[a]Values calculated using formula and specific activities of each isotope from the *Radiological Health Handbook*, (D.H.E.W 1970).

**Table 6.6. Analytes in ash from Chestnut Ridge OU 2 and in reference soil**

| Analyte | Maximum detected value or upper 95% confidence interval for analyte concentration (mg/kg) | | | |
|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| Aluminum | 21900 | 14100[m] | 19800 | 10300 |
| Arsenic | 131 | 159[m] | 59.2[m] | 5.76 |
| Barium | 450 | 421[m] | 560 | 108 |
| Beryllium | 4.39 | 1.6[m] | 2.9 | 0.765 |
| Cadmium | ND[1] | ND | 3.67 | 1.82 |
| Calcium | 5600 | 4920 [m] | 5520 | 2230[m] |
| Chromium | 25.1 | 17.1[m] | 16.9 | 20.3 |
| Cobalt | 19.3 | 28.6[m] | 19.3 | 20.7 |
| Copper | 69.1 | 70.1[m] | 58.5 | 6.8[m] |
| Iron | 37700 | 183000[m] | 51900 | 17100 |
| Lead | 18.8 | 17.8[m] | 15.9 | 33.4 |
| Magnesium | 1750 | 1010[m] | 1620 | 614 |
| Manganese | 152 | 344[m] | 876 | 2370 |
| Mercury | 0.705 | 0.61[m] | 0.581 | 0.307 |
| Nickel | 36 | 39.9[m] | 35 | 8.5[m] |
| Phosphorus | ND | ND | 1510 | 326[m] |
| Potassium | 4500 | 1920[m] | 3720 | 739[m] |
| Selenium | 14.8 | 4.7[m] | 6.13 | 1.03 |
| Sodium | 741 | 342[m] | 361 | 188 |
| Thallium | 2.21 | ND | ND | ND |
| Vanadium | 84.9 | 47.5[m] | 72.5 | 35.1 |
| Zinc | 53.9 | 40[m] | 70.5 | 49.5 |
| Cesium-137[2] | ND | ND | 0.129 | 0.33[m] |
| Thorium-228[2] | 2.17 | 1.1[m] | 9.2[m] | 0.51[m] |
| Thorium-230[2] | 4.02 | 2.9[m] | ND | 8.6[m] |
| Thorium-232[2] | 2.12 | 1.3[m] | ND | 0.52[m] |
| Thorium-234[2] | ND | ND | 6.3[m] | ND |
| Uranium-234[2] | 3.87 | 4.4[m] | ND | 3.5[m] |
| Uranium-235[2] | 0.235 | 0.17[m] | ND | 0.34[mm] |
| Uranium-238[2] | 3.87 | 3.3[m] | 3.43 | 2.22 |
| 2-Methylnaphthalene | 0.093[m] | ND | ND | ND |

**Table 6.6 (continued)**

| Analyte | Maximum detected value or upper 95% confidence interval for analyte concentration (mg/kg) | | | |
|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| 4-Nitrophenol | ND | ND | ND | 0.14[m] |
| Butylbenzylphthalate | 0.11[m] | ND | ND | 0.054[m] |
| bis(2-Ethylhexyl)Phthalate | 0.27[m] | ND | ND | ND |
| Chloroform | 0.002[m] | ND | ND | ND |
| alpha-BHC | 0.0003[m] | ND | ND | ND |

[m]Maximum detected value; all other vales are Upper 95% confidence interval.
[1]ND = not detected; concentration in ash or soil was below detection limit.
[2]Units for radionuclides were pCi/g.

**Table 6.7. Exposure estimates for white-footed mice and short-tailed shrews consuming ash or reference soil**

| Analyte | Estimated exposure for white-footed mice (mg/kg per day) | | | | Estimated exposure for short-tailed shrew (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| Aluminum | 36.792 | 23.688 | 33.264 | 17.304 | 1708.2 | 1099.8 | 1544.4 | 803.4 |
| Arsenic | 0.22008 | 0.26712 | 0.09946 | 0.00968 | 10.218 | 12.402 | 4.6176 | 0.44928 |
| Barium | 0.756 | 0.70728 | 0.9408 | 0.18144 | 35.1 | 32.838 | 43.68 | 8.424 |
| Beryllium | 0.00738 | 0.00269 | 0.00487 | 0.00129 | 0.34242 | 0.1248 | 0.2262 | 0.05967 |
| Cadmium | 0 | 0 | 0.00617 | 0.00306 | 0 | 0 | 0.28626 | 0.14196 |
| Calcium | 9.408 | 8.2656 | 9.2736 | 3.7464 | 436.8 | 383.76 | 430.56 | 173.94 |
| Chromium | 0.04217 | 0.02873 | 0.02839 | 0.0341 | 1.9578 | 1.3338 | 1.3182 | 1.5834 |
| Cobalt | 0.03242 | 0.04805 | 0.03242 | 0.03478 | 1.5054 | 2.2308 | 1.5054 | 1.6146 |
| Copper | 0.11609 | 0.11777 | 0.09828 | 0.01142 | 5.3898 | 5.4678 | 4.563 | 0.5304 |
| Iron | 63.336 | 307.44 | 87.192 | 28.728 | 2940.6 | 14274 | 4048.2 | 1333.8 |
| Lead | 0.03158 | 0.0299 | 0.02671 | 0.05611 | 1.4664 | 1.3884 | 1.2402 | 2.6052 |
| Magnesium | 2.94 | 1.6968 | 2.7216 | 1.03152 | 136.5 | 78.78 | 126.36 | 47.892 |
| Manganese | 0.25536 | 0.57792 | 1.47168 | 3.9816 | 11.856 | 26.832 | 68.328 | 184.86 |
| Mercury | 0.00118 | 0.00102 | 0.00098 | 0.00052 | 0.05499 | 0.04758 | 0.04532 | 0.02395 |
| Nickel | 0.06048 | 0.06703 | 0.0588 | 0.01428 | 2.808 | 3.1122 | 2.73 | 0.663 |
| Phosphorus | 0 | 0 | 2.5368 | 0.54768 | 0 | 0 | 117.78 | 25.428 |
| Potassium | 7.56 | 3.2256 | 6.2496 | 1.24152 | 351 | 149.76 | 290.16 | 57.642 |
| Selenium | 0.02486 | 0.0079 | 0.0103 | 0.00173 | 1.1544 | 0.3666 | 0.478 | 0.08034 |

Table 6.7 (continued)

| Analyte | Estimated exposure for white-footed mice (mg/kg per day) | | | | Estimated exposure for short-tailed shrew (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| Sodium | 1.24488 | 0.57456 | 0.60648 | 0.31584 | 57.798 | 26.676 | 28.158 | 14.664 |
| Thallium | 0.00371 | 0 | 0 | 0 | 0.17238 | 0 | 0 | 0 |
| Vanadium | 0.14263 | 0.0798 | 0.1218 | 0.05897 | 6.6222 | 3.705 | 5.655 | 2.7378 |
| Zinc | 0.09055 | 0.0672 | 0.11844 | 0.08316 | 4.2042 | 3.12 | 5.499 | 3.861 |
| Cesium-137[2] | 0 | 0 | 0.00022 | 0.00055 | 0 | 0 | 0.01006 | 0.02574 |
| Thorium-228[2] | 0.00365 | 0.00185 | 0.01546 | 0.00086 | 0.16926 | 0.0858 | 0.7176 | 0.03978 |
| Thorium-230[2] | 0.00675 | 0.00487 | 0 | 0.01445 | 0.31356 | 0.2262 | 0 | 0.6708 |
| Thorium-232[2] | 0.00356 | 0.00218 | 0 | 0.00087 | 0.16536 | 0.1014 | 0 | 0.04056 |
| Thorium-234[2] | 0 | 0 | 0.01058 | 0 | 0 | 0 | 0.4914 | 0 |
| Uranium-234[2] | 0.0065 | 0.00739 | 0 | 0.00588 | 0.30186 | 0.3432 | 0 | 0.273 |
| Uranium-235[2] | 0.00039 | 0.00029 | 0 | 0.00057 | 0.01833 | 0.01326 | 0 | 0.02652 |
| Uranium-238[2] | 0.0065 | 0.00554 | 0.00576 | 0.00373 | 0.30186 | 0.2574 | 0.26754 | 0.17316 |
| 2-Methylnaphthalene | 0.00016 | 0 | 0 | 0 | 0.00725 | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 | 0.00024 | 0 | 0 | 0 | 0.01092 |
| Butylbenzylphthalate | 0.00018 | 0 | 0 | 0.00009 | 0.00858 | 0 | 0 | 0.00421 |
| bis(2-Ethylhexyl)Phthalate | 0.00045 | 0 | 0 | 0 | 0.02106 | 0 | 0 | 0 |

Table 6.7 (continued)

| Analyte | Estimated exposure for white-footed mice (mg/kg per day) | | | | Estimated exposure for short-tailed shrew (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| Chloroform | 0 | 0 | 0 | 0 | 0.00016 | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 | 0 | 0.00002 | 0 | 0 | 0 |

[1]ND = not detected; concentration in ash or soil was below detection limit.
[2]Units for radionuclides were pCi/g per day.

**Table 6.8. Absolute maximum exposure estimates for white-tailed deer and red fox consuming ash or reference soil**

| Analyte | Estimated exposure for white-tailed deer (mg/kg per day) | | | | Estimated exposure for red fox (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| Aluminum | 12.775 | 8.225 | 11.55 | 6.00833 | 8.03 | 5.17 | 7.26 | 3.77667 |
| Arsenic | 0.07642 | 0.09275 | 0.03453 | 0.00336 | 0.04803 | 0.0583 | 0.02171 | 0.00211 |
| Barium | 0.2625 | 0.24558 | 0.32667 | 0.063 | 0.165 | 0.15437 | 0.20533 | 0.0396 |
| Beryllium | 0.00256 | 0.00093 | 0.00169 | 0.00045 | 0.00161 | 0.00059 | 0.00106 | 0.00028 |
| Cadmium | 0 | 0 | 0.00214 | 0.00106 | 0 | 0 | 0.00135 | 0.00067 |
| Calcium | 3.26667 | 2.87 | 3.22 | 1.30083 | 2.05333 | 1.804 | 2.024 | 0.81767 |
| Chromium | 0.01464 | 0.00998 | 0.00986 | 0.01184 | 0.0092 | 0.00627 | 0.0062 | 0.00744 |
| Cobalt | 0.01126 | 0.01668 | 0.01126 | 0.01208 | 0.00708 | 0.01049 | 0.00708 | 0.00759 |
| Copper | 0.04031 | 0.04089 | 0.03413 | 0.00397 | 0.02534 | 0.0257 | 0.02145 | 0.00249 |
| Iron | 21.99167 | 106.75 | 30.275 | 9.975 | 13.82333 | 67.1 | 19.03 | 6.27 |
| Lead | 0.01097 | 0.01038 | 0.00928 | 0.01948 | 0.00689 | 0.00653 | 0.00583 | 0.01225 |
| Magnesium | 1.02083 | 0.58917 | 0.945 | 0.35817 | 0.64167 | 0.37033 | 0.594 | 0.22513 |
| Manganese | 0.08867 | 0.20067 | 0.511 | 1.3825 | 0.05573 | 0.12613 | 0.3212 | 0.869 |
| Mercury | 0.00041 | 0.00036 | 0.00034 | 0.00018 | 0.00026 | 0.00022 | 0.00021 | 0.00011 |
| Nickel | 0.021 | 0.02328 | 0.02042 | 0.00496 | 0.0132 | 0.01463 | 0.01283 | 0.00312 |
| Phosphorus | 0 | 0 | 0.88083 | 0.19017 | 0 | 0 | 0.55367 | 0.11953 |

Table 6.8 (continued)

| Analyte | Estimated exposure for white-tailed deer (mg/kg per day) | | | | Estimated exposure for red fox (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| Potassium | 2.625 | 1.12 | 2.17 | 0.43108 | 1.65 | 0.704 | 1.364 | 0.27097 |
| Selenium | 0.00863 | 0.00274 | 0.00358 | 0.0006 | 0.00543 | 0.00172 | 0.00225 | 0.00038 |
| Sodium | 0.43225 | 0.1995 | 0.21058 | 0.10967 | 0.2717 | 0.1254 | 0.13237 | 0.06893 |
| Thallium | 0.00129 | 0 | 0 | 0 | 0.00081 | 0 | 0 | 0 |
| Vanadium | 0.04953 | 0.02771 | 0.04229 | 0.02048 | 0.03113 | 0.01742 | 0.02658 | 0.01287 |
| Zinc | 0.03144 | 0.02333 | 0.04113 | 0.02888 | 0.01976 | 0.01467 | 0.02585 | 0.01815 |
| Cesium-137[2] | 0 | 0 | 0.00008 | 0.00019 | 0 | 0 | 0.00005 | 0.00012 |
| Thorium-228[2] | 0.00127 | 0.00064 | 0.00537 | 0.0003 | 0.0008 | 0.0004 | 0.00337 | 0.00019 |
| Thorium-230[2] | 0.00235 | 0.00169 | 0 | 0.00502 | 0.00147 | 0.00106 | 0 | 0.00315 |
| Thorium-232[2] | 0.00124 | 0.00076 | 0 | 0.0003 | 0.00078 | 0.00048 | 0 | 0.00019 |
| Thorium-234[2] | 0 | 0 | 0.00368 | 0 | 0 | 0 | 0.00231 | 0 |
| Uranium-234[2] | 0.00226 | 0.00257 | 0 | 0.00204 | 0.00142 | 0.00161 | 0 | 0.00128 |
| Uranium-235[2] | 0.00014 | 0.0001 | 0 | 0.0002 | 0.00009 | 0.00006 | 0 | 0.00012 |
| Uranium-238[2] | 0.00226 | 0.00193 | 0.002 | 0.0013 | 0.00142 | 0.00121 | 0.00126 | 0.00081 |
| 2-Methylnaphthalene | 0.00005 | 0 | 0 | 0 | 0.00003 | 0 | 0 | 0 |

Table 6.8 (continued)

| Analyte | Estimated exposure for white-tailed deer (mg/kg per day) | | | | Estimated exposure for red fox (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil | FCAP ash | Sluice Channel Area ash | UMB ash | Walker Branch soil |
| 4-Nitrophenol | 0 | 0 | 0 | 0.00008 | 0 | 0 | 0 | 0.00005 |
| Butylbenzylphthalate | 0.00006 | 0 | 0 | 0.00003 | 0.00004 | 0 | 0 | 0.00002 |
| bis(2-Ethylhexyl)Phthalate | 0.00016 | 0 | 0 | 0 | 0.0001 | 0 | 0 | 0 |
| Chloroform | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

[1]ND = not detected; concentration in ash or soil was below detection limit.
[2]Units for radionuclides were pCi/g per day.

**Table 6.9. Maximum proportional exposure estimates for white-tailed deer and red fox consuming ash**

| Analyte | Estimated exposure for white-tailed deer (mg/kg per day) | | | Estimated exposure for red fox (mg/kg per day) | | |
|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | FCAP ash | Sluice Channel Area ash | UMB ash |
| Aluminum | 1.30305 | 0.83895 | 1.1781 | 0.4745 | 0.3055 | 0.429 |
| Arsenic | 0.00779 | 0.00946 | 0.00352 | 0.00284 | 0.00345 | 0.00128 |
| Barium | 0.02678 | 0.02505 | 0.03332 | 0.00975 | 0.00912 | 0.01213 |
| Beryllium | 0.00026 | 0.0001 | 0.00017 | 0.0001 | 0.00003 | 0.00006 |
| Cadmium | 0 | 0 | 0.00022 | 0 | 0 | 0.00008 |
| Calcium | 0.3332 | 0.29274 | 0.32844 | 0.12133 | 0.1066 | 0.1196 |
| Chromium | 0.00149 | 0.00102 | 0.00101 | 0.00054 | 0.00037 | 0.00037 |
| Cobalt | 0.00115 | 0.0017 | 0.00115 | 0.00042 | 0.00062 | 0.00042 |
| Copper | 0.00411 | 0.00417 | 0.00348 | 0.0015 | 0.00152 | 0.00127 |
| Iron | 2.24315 | 10.8885 | 3.08805 | 0.81683 | 3.965 | 1.1245 |
| Lead | 0.00112 | 0.00106 | 0.00095 | 0.00041 | 0.00039 | 0.00034 |
| Magnesium | 0.10413 | 0.0601 | 0.09639 | 0.03792 | 0.02188 | 0.0351 |
| Manganese | 0.00904 | 0.02047 | 0.05212 | 0.00329 | 0.00745 | 0.01898 |
| Mercury | 0.00004 | 0.00004 | 0.00003 | 0.00002 | 0.00001 | 0.00001 |
| Nickel | 0.00214 | 0.00237 | 0.00208 | 0.00078 | 0.00086 | 0.00076 |
| Phosphorus | 0 | 0 | 0.08985 | 0 | 0 | 0.03272 |
| Potassium | 0.26775 | 0.11424 | 0.22134 | 0.0975 | 0.0416 | 0.0806 |
| Selenium | 0.00088 | 0.00028 | 0.00036 | 0.00032 | 0.0001 | 0.00013 |
| Sodium | 0.04409 | 0.02035 | 0.02148 | 0.01606 | 0.00741 | 0.00782 |
| Thallium | 0.00013 | 0 | 0 | 0.00005 | 0 | 0 |
| Vanadium | 0.00505 | 0.00283 | 0.00431 | 0.00184 | 0.00103 | 0.00157 |

Table 6.9 (continued)

| Analyte | Estimated exposure for white-tailed deer (mg/kg per day) | | | Estimated exposure for red fox (mg/kg per day) | | |
|---|---|---|---|---|---|---|
| | FCAP ash | Sluice Channel Area ash | UMB ash | FCAP ash | Sluice Channel Area ash | UMB ash |
| Zinc | 0.00321 | 0.00238 | 0.00419 | 0.00117 | 0.00087 | 0.00153 |
| Cesium-137[2] | 0 | 0 | 0.00001 | 0 | 0 | 0 |
| Thorium-228[2] | 0.00013 | 0.00007 | 0.00055 | 0.00005 | 0.00002 | 0.0002 |
| Thorium-230[2] | 0.00024 | 0.00017 | 0 | 0.00009 | 0.00006 | 0 |
| Thorium-232[2] | 0.00013 | 0.00008 | 0 | 0.00005 | 0.00003 | 0 |
| Thorium-234[2] | 0 | 0 | 0.00037 | 0 | 0 | 0.00014 |
| Uranium-234[2] | 0.00023 | 0.00026 | 0 | 0.00008 | 0.0001 | 0 |
| Uranium-235[2] | 0.00001 | 0.00001 | 0 | 0.00001 | 0 | 0 |
| Uranium-238[2] | 0.00023 | 0.0002 | 0.0002 | 0.00008 | 0.00007 | 0.00007 |
| 2-Methylnaphthalene | 0.00001 | 0 | 0 | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 | 0 | 0 | 0 |
| Butylbenzylphthalate | 0.00001 | 0 | 0 | 0 | 0 | 0 |
| bis(2-Ethylhexyl)Phthalate | 0.00002 | 0 | 0 | 0.00001 | 0 | 0 |
| Chloroform | 0 | 0 | 0 | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 | 0 | 0 | 0 |

[1]ND = not detected; concentration in ash or soil was below detection limit.
[2]Units for radionuclides were pCi/g per day.

**Table 6.10. Exposure estimates for white-tailed deer consuming ash from the FCAP to meet sodium requirements**

| Analyte | Estimated exposure for white-tailed deer (mg/kg per day) | Analyte | Estimated exposure for white-tailed deer (mg/kg per day) |
|---|---|---|---|
| Aluminum | 93.8 | Sodium | 3.17395 |
| Arsenic | 0.5611 | Thallium | 0.00947 |
| Barium | 1.9275 | Vanadium | 0.36366 |
| Beryllium | 0.0188 | Zinc | 0.23087 |
| Cadmium | 0 | Cesium-137[2] | 0 |
| Calcium | 23.9866 | Thorium-228[2] | 0.00929 |
| Chromium | 0.1075 | Thorium-230[2] | 0.1722 |
| Cobalt | 0.08267 | Thorium-232[2] | 0.00908 |
| Copper | 0.2959 | Thorium-234[2] | 0 |
| Iron | 161.4817 | Uranium-234[2] | 0.01658 |
| Lead | 0.08053 | Uranium-235[2] | 0.00101 |
| Magnesium | 7.49583 | Uranium-238[2] | 0.01658 |
| Manganese | 0.65107 | 2-Methylnaphthalene | 0.0004 |
| Mercury | 0.00302 | 4-Nitrophenol | 0 |
| Nickel | 0.1542 | Butylbenzylphthalate | 0.00047 |
| Phosphorus | 0 | bis(2-Ethylhexyl)Phthalate | 0.00116 |
| Potassium | 19.275 | Chloroform | 0.0001 |
| Selenium | 0.06339 | alpha-BHC | 0 |

[1]ND = not detected; concentration in ash or soil was below detection limit.
[2]Units for radionuclides were pCi/g per day.

**Table 6.11. Analytes in surface water from Chestnut Ridge OU 2 and background locations**

| Analyte | Maximum detected value or upper 95% confidence interval for analyte concentration (mg/L) | | |
|---|---|---|---|
| | FCAP surface water | UMB surface water | Background surface water |
| Aluminum | 7.92[m] | 0.429 | 0.378 |
| Arsenic | 0.01[m] | 0.0755 | 0.003 |
| Barium | 0.0689[m] | 0.0908 | 0.114 |
| Chloride | ND[1] | 2.72 | ND |
| Calcium | 27[m] | 48.6 | 35.6[m] |
| Chromium | 0.0088[m] | ND | 0.008 |
| Copper | 0.0073[m] | 0.00526 | 0.004 |
| Fluoride | ND | 0.197 | ND |
| Iron | 6.11[m] | 1.08 | 1.48 |
| Lead | 0.0035[m] | 0.00163 | 0.002 |
| Magnesium | 15.3[m] | 15.6 | 17.4[m] |
| Manganese | 1.7[m] | 0.514 | 0.279 |
| Mercury | ND | 0.00024[m] | 0.0002 |
| Niobium | ND | 0.00802 | ND |
| Nitrate | ND | 1[m] | ND |
| Potassium | 6.67[m] | 3.94 | 1.86 |
| Sodium | 1.19[m] | 2.51 | 0.839[m] |
| Sulphate | ND | 24.1 | ND |
| Vanadium | 0.0156[m] | ND | 0.006 |
| Zinc | 0.106[m] | 0.0173 | 0.00952 |
| Radium-226[2] | ND | 0.567[m] | ND |
| Thorium-228[2] | 0.22[m] | 0.041[m] | 0.045 |
| Thorium-230[2] | 2[m] | 0.62[m] | 0.59 |
| Thorium-232[2] | ND | 0.077[m] | 0.4 |
| Uranium-234[2] | 0.33[m] | 0.5[m] | 0.382 |
| Uranium-238[2] | 0.11[m] | 0.336[m] | 0.131 |
| Di-N-octylphthalate | ND | 0.0171 | ND |
| Di-N-butylphthalate | ND | ND | 0.001[m] |
| Phenol | ND | ND | 0.002[m] |
| bis(2-Ethylhexyl)Phthalate | ND | 0.048[m] | 0.006[m] |

[m]Maximum detected value; all other vales are upper 95% confidence interval.
[1]ND = not detected; concentration in surface water was below detection limit.
[2]Units for radionuclides were pCi/L.

**Table 6.12. Exposure estimates for white-footed mice and short-tailed shrew consuming surface water**

| Analyte | Estimated exposure for white-footed mouse (mg/kg per day) | | | Estimated exposure for short-tailed shrew (mg/kg per day) | | |
|---|---|---|---|---|---|---|
| | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water |
| Aluminum | 1.16028 | 0.06285 | 0.05538 | 1.19328 | 0.06464 | 0.05695 |
| Arsenic | 0.00147 | 0.01106 | 0.00044 | 0.00151 | 0.01138 | 0.00045 |
| Barium | 0.01009 | 0.0133 | 0.0167 | 0.01038 | 0.01368 | 0.01718 |
| Chloride | 0 | 0.39848 | 0 | 0 | 0.40981 | 0 |
| Calcium | 3.9555 | 7.1199 | 5.2154 | 4.068 | 7.3224 | 5.36373 |
| Chromium | 0.00129 | 0 | 0.00117 | 0.00133 | 0 | 0.00121 |
| Copper | 0.00107 | 0.00077 | 0.00059 | 0.0011 | 0.00079 | 0.0006 |
| Fluoride | 0 | 0.02886 | 0 | 0 | 0.02968 | 0 |
| Iron | 0.89512 | 0.15822 | 0.21682 | 0.92057 | 0.16272 | 0.22299 |
| Lead | 0.00051 | 0.00024 | 0.00029 | 0.00053 | 0.00025 | 0.0003 |
| Magnesium | 2.24145 | 2.2854 | 2.5491 | 2.3052 | 2.3504 | 2.6216 |
| Manganese | 0.24905 | 0.0753 | 0.04087 | 0.25613 | 0.07744 | 0.04204 |
| Mercury | 0 | 0.00004 | 0.00003 | 0 | 0.00004 | 0.00003 |
| Niobium | 0 | 0.00117 | 0 | 0 | 0.00121 | 0 |
| Nitrate | 0 | 0.1465 | 0 | 0 | 0.15067 | 0 |
| Potassium | 0.97716 | 0.57721 | 0.27249 | 1.00495 | 0.59363 | 0.28024 |
| Sodium | 0.17434 | 0.36772 | 0.12291 | 0.17929 | 0.37817 | 0.12641 |
| Sulphate | 0 | 3.53065 | 0 | 0 | 3.63107 | 0 |
| Vanadium | 0.00229 | 0 | 0.00088 | 0.00235 | 0 | 0.0009 |
| Zinc | 0.01553 | 0.00253 | 0.00139 | 0.01597 | 0.00261 | 0.00143 |
| Radium-226[2] | 0 | 0.08307 | 0 | 0 | 0.08543 | 0 |
| Thorium-228[2] | 0.03223 | 0.00601 | 0.00659 | 0.03315 | 0.00618 | 0.00678 |
| Thorium-230[2] | 0.293 | 0.09083 | 0.08644 | 0.30133 | 0.09341 | 0.08889 |
| Thorium-232[2] | 0 | 0.01128 | 0.0586 | 0 | 0.0116 | 0.06027 |
| Uranium-234[2] | 0.04835 | 0.07325 | 0.05596 | 0.04972 | 0.07533 | 0.05755 |
| Uranium-238[2] | 0.01612 | 0.04922 | 0.01919 | 0.01657 | 0.05062 | 0.01974 |
| Di-N-octylphthalate | 0 | 0.00251 | 0 | 0 | 0.00258 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0.00015 | 0 | 0 | 0.00015 |
| Phenol | 0 | 0 | 0.00029 | 0 | 0 | 0.0003 |
| bis(2-Ethylhexyl)Phthalate | 0 | 0.00703 | 0.00088 | 0 | 0.00723 | 0.0009 |

[1]ND = not detected; concentration in surface water was below detection limit.
[2]Units for radionuclides were pCi/L per day.

**Table 6.13. Absolute maximum exposure estimates for red fox, white-tailed deer, and red-tailed hawk consuming surface water**

| Analyte | Estimated exposure for red fox (mg/kg per day) | | | Estimated exposure for white-tailed deer (mg/kg per day) | | | Estimated exposure for red-tailed hawk (mg/kg per day) | | |
|---|---|---|---|---|---|---|---|---|---|
| | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water |
| Aluminum | 0.65604 | 0.03554 | 0.03131 | 0.52061 | 0.0282 | 0.02485 | 0.45016 | 0.02438 | 0.02148 |
| Arsenic | 0.00083 | 0.00625 | 0.00025 | 0.00066 | 0.00496 | 0.0002 | 0.00057 | 0.00429 | 0.00017 |
| Barium | 0.00571 | 0.00752 | 0.00944 | 0.00453 | 0.00597 | 0.00749 | 0.00392 | 0.00516 | 0.00648 |
| Chloride | 0 | 0.22531 | 0 | 0 | 0.17879 | 0 | 0 | 0.1546 | 0 |
| Calcium | 2.2365 | 4.0257 | 2.94887 | 1.7748 | 3.19464 | 2.34011 | 1.53464 | 2.76234 | 2.02345 |
| Chromium | 0.00073 | 0 | 0.00066 | 0.00058 | 0 | 0.00053 | 0.0005 | 0 | 0.00045 |
| Copper | 0.0006 | 0.00044 | 0.00033 | 0.00048 | 0.00035 | 0.00026 | 0.00041 | 0.0003 | 0.00023 |
| Fluoride | 0 | 0.01632 | 0 | 0 | 0.01295 | 0 | 0 | 0.0112 | 0 |
| Iron | 0.50611 | 0.08946 | 0.12259 | 0.40163 | 0.07099 | 0.09729 | 0.34728 | 0.06139 | 0.08412 |
| Lead | 0.00029 | 0.00014 | 0.00017 | 0.00023 | 0.00011 | 0.00013 | 0.0002 | 0.00009 | 0.00011 |
| Magnesium | 1.26735 | 1.2922 | 1.4413 | 1.00572 | 1.02544 | 1.14376 | 0.86963 | 0.88668 | 0.98899 |
| Manganese | 0.14082 | 0.04258 | 0.02311 | 0.11175 | 0.03379 | 0.01834 | 0.09663 | 0.02921 | 0.01586 |
| Mercury | 0 | 0.00002 | 0.00002 | 0 | 0.00002 | 0.00001 | 0 | 0.00001 | 0.00001 |
| Niobium | 0 | 0.00066 | 0 | 0 | 0.00053 | 0 | 0 | 0.00046 | 0 |
| Nitrate | 0 | 0.08283 | 0 | 0 | 0.06573 | 0 | 0 | 0.05684 | 0 |
| Potassium | 0.5525 | 0.32636 | 0.15407 | 0.43844 | 0.25899 | 0.12226 | 0.37911 | 0.22394 | 0.10572 |
| Sodium | 0.09857 | 0.20791 | 0.0695 | 0.07822 | 0.16499 | 0.05515 | 0.06764 | 0.14266 | 0.04769 |

**Table 6.13 (continued)**

| Analyte | Estimated exposure for red fox (mg/kg per day) | | | Estimated exposure for white-tailed deer (mg/kg per day) | | | Estimated exposure for red-tailed hawk (mg/kg per day) | | |
|---|---|---|---|---|---|---|---|---|---|
| | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water |
| Sulphate | 0 | 1.99628 | 0 | 0 | 1.58417 | 0 | 0 | 1.3698 | 0 |
| Vanadium | 0.00129 | 0 | 0.0005 | 0.00103 | 0 | 0.00039 | 0.00089 | 0 | 0.00034 |
| Zinc | 0.00878 | 0.00143 | 0.00079 | 0.00697 | 0.00114 | 0.00063 | 0.00602 | 0.00098 | 0.00054 |
| Radium-226[2] | 0 | 0.04697 | 0 | 0 | 0.03727 | 0 | 0 | 0.03223 | 0 |
| Thorium-228[2] | 0.01822 | 0.0034 | 0.00373 | 0.01446 | 0.0027 | 0.00296 | 0.0125 | 0.00233 | 0.00256 |
| Thorium-230[2] | 0.16567 | 0.05136 | 0.04887 | 0.13147 | 0.04075 | 0.03878 | 0.11368 | 0.03524 | 0.03353 |
| Thorium-232[2] | 0 | 0.00638 | 0.03313 | 0 | 0.00506 | 0.02629 | 0 | 0.00438 | 0.02274 |
| Uranium-234[2] | 0.02734 | 0.04142 | 0.03164 | 0.02169 | 0.03287 | 0.02511 | 0.01876 | 0.02842 | 0.02171 |
| Uranium-238[2] | 0.00911 | 0.02783 | 0.01085 | 0.00723 | 0.02209 | 0.00861 | 0.00625 | 0.0191 | 0.00745 |
| Di-N-octylphthalate | 0 | 0.00142 | 0 | 0 | 0.00112 | 0 | 0 | 0.00097 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0.00008 | 0 | 0 | 0.00007 | 0 | 0 | 0.00006 |
| Phenol | 0 | 0 | 0.00017 | 0 | 0 | 0.00013 | 0 | 0 | 0.00011 |
| bis(2-Ethylhexyl)Phthalate | 0 | 0.00398 | 0.0005 | 0 | 0.00316 | 0.00039 | 0 | 0.00273 | 0.00034 |

[1]ND = not detected; concentration in ash or soil was below detection limit.
[2]Units for radionuclides were pCi/L per day.

**Table 6.14. Maximum proportional exposure estimates for red fox, white-tailed deer, and red-tailed hawk consuming surface water**

| Analyte | Estimated exposure for red fox (mg/kg per day) | | | Estimated exposure for white-tailed deer (mg/kg per day) | | | Estimated exposure for red-tailed hawk (mg/kg per day) | | |
|---|---|---|---|---|---|---|---|---|---|
| | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water |
| Aluminum | 0.03828 | 0.00207 | 0.00183 | 0.05306 | 0.00287 | 0.00253 | 0.01055 | 0.00057 | 0.0005 |
| Arsenic | 0.00005 | 0.00036 | 0.00001 | 0.00007 | 0.00051 | 0.00002 | 0.00001 | 0.0001 | 0 |
| Barium | 0.00033 | 0.00044 | 0.00055 | 0.00046 | 0.00061 | 0.00076 | 0.00009 | 0.00012 | 0.00015 |
| Chloride | 0 | 0.01315 | 0 | 0 | 0.01822 | 0 | 0 | 0.00362 | 0 |
| Calcium | 0.1305 | 0.2349 | 0.17207 | 0.1809 | 0.32562 | 0.23852 | 0.03597 | 0.06474 | 0.04742 |
| Chromium | 0.00004 | 0 | 0.00004 | 0.00006 | 0 | 0.00005 | 0.00001 | 0 | 0.00001 |
| Copper | 0.00004 | 0.00003 | 0.00002 | 0.00005 | 0.00004 | 0.00003 | 0.00001 | 0.00001 | 0.00001 |
| Fluoride | 0 | 0.00095 | 0 | 0 | 0.00132 | 0 | 0 | 0.00026 | 0 |
| Iron | 0.02953 | 0.00522 | 0.00715 | 0.04094 | 0.00724 | 0.00992 | 0.00814 | 0.00144 | 0.00197 |
| Lead | 0.00002 | 0.00001 | 0.00001 | 0.00002 | 0.00001 | 0.00001 | 0 | 0 | 0 |
| Magnesium | 0.07395 | 0.0754 | 0.0841 | 0.10251 | 0.10452 | 0.11658 | 0.02038 | 0.02078 | 0.02318 |
| Manganese | 0.00822 | 0.00248 | 0.00135 | 0.01139 | 0.00344 | 0.00187 | 0.00226 | 0.00068 | 0.00037 |
| Mercury | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Niobium | 0 | 0.00004 | 0 | 0 | 0.00005 | 0 | 0 | 0.00001 | 0 |
| Nitrate | 0 | 0.00483 | 0 | 0 | 0.0067 | 0 | 0 | 0.00133 | 0 |
| Potassium | 0.03224 | 0.01904 | 0.00899 | 0.04469 | 0.0264 | 0.01246 | 0.00889 | 0.00525 | 0.00248 |
| Sodium | 0.00575 | 0.01213 | 0.00406 | 0.00797 | 0.01682 | 0.00562 | 0.00159 | 0.00334 | 0.00112 |
| Sulphate | 0 | 0.11648 | 0 | 0 | 0.16147 | 0 | 0 | 0.0321 | 0 |
| Vanadium | 0.00008 | 0 | 0.00003 | 0.0001 | 0 | 0.00004 | 0.00002 | 0 | 0.00001 |

**Table 6.14 (continued)**

| Analyte | Estimated exposure for red fox (mg/kg per day) | | | Estimated exposure for white-tailed deer (mg/kg per day) | | | Estimated exposure for red-tailed hawk (mg/kg per day) | | |
|---|---|---|---|---|---|---|---|---|---|
| | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water | FCAP surface water | UMB surface water | Background surface water |
| Zinc | 0.00051 | 0.00008 | 0.00005 | 0.00071 | 0.00012 | 0.00006 | 0.00014 | 0.00002 | 0.00001 |
| Radium-226[2] | 0 | 0.00274 | 0 | 0 | 0.0038 | 0 | 0 | 0.00076 | 0 |
| Thorium-228[2] | 0.00106 | 0.0002 | 0.00022 | 0.00147 | 0.00027 | 0.0003 | 0.00029 | 0.00005 | 0.00006 |
| Thorium-230[2] | 0.00967 | 0.003 | 0.00285 | 0.0134 | 0.00415 | 0.00395 | 0.00266 | 0.00083 | 0.00079 |
| Thorium-232[2] | 0 | 0.00037 | 0.00193 | 0 | 0.00052 | 0.00268 | 0 | 0.0001 | 0.00053 |
| Uranium-234[2] | 0.0016 | 0.00242 | 0.00185 | 0.00221 | 0.00335 | 0.00256 | 0.00044 | 0.00067 | 0.00051 |
| Uranium-238[2] | 0.00053 | 0.00162 | 0.00063 | 0.00074 | 0.00225 | 0.00088 | 0.00015 | 0.00045 | 0.00017 |
| Di-N-octylphthalate | 0 | 0.00008 | 0 | 0 | 0.00011 | 0 | 0 | 0.00002 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0 | 0 | 0.00001 | 0 | 0 | 0 |
| Phenol | 0 | 0 | 0.00001 | 0 | 0 | 0.00001 | 0 | 0 | 0 |
| bis(2-Ethylhexyl)Phthalate | 0 | 0.00023 | 0.00003 | 0 | 0.00032 | 0.00004 | 0 | 0.00006 | 0.00001 |

[1]ND = not detected; concentration in ash or soil was below detection limit.
[2]Units for radionuclides were pCi/L per day.

**Table 6.15. Exposure estimates for white-tailed deer consuming foliage**

| Analyte | Upper 95% confidence interval analyte in foliage (mg/kg) | | Absolute maximum exposure estimate (mg/kg per day) | | Maximum proportional exposure estimate (mg/kg per day) | | Walker Branch vegetation | |
|---|---|---|---|---|---|---|---|---|
| | FCAP vegetation | Sluice Channel Area vegetation | FCAP vegetation | Sluice Channel Area vegetation | FCAP vegetation | Sluice Channel Area vegetation | Upper 95% confidence interval analyte in foliage (mg/kg) | Estimated exposure (mg/kg per day) |
| Arsenic | 1.77 | 1[1] | 0.05133 | 0.029 | 0.00522 | 0.00295 | 1.06 | 0.03074 |
| Cadmium | 1.23 | 0.37 | 0.03567 | 0.01073 | 0.00363 | 0.00109 | 0.26 | 0.00754 |
| Chromium | 5.3 | 3[1] | 0.1537 | 0.087 | 0.01564 | 0.00885 | 3[1] | 0.087 |
| Lead | 1.07 | 1[1] | 0.03103 | 0.029 | 0.00316 | 0.00295 | 1[1] | 0.029 |
| Mercury | 0.06 | 0.07 | 0.00174 | 0.00203 | 0.00018 | 0.00021 | 0.12 | 0.00348 |
| Selenium | 23.61 | 1.95 | 0.68469 | 0.05655 | 0.06965 | 0.00575 | 1[1] | 0.029 |
| Thallium[1] | 1[1] | 1[1] | 0.029 | 0.029 | 0.00295 | 0.00295 | 1[1] | 0.029 |

[1]Below detection limit, therefore value for detection limit used in calculation.

Table 6.16. Exposure estimates for white-footed mice consuming plant material

| Analyte | Upper 95% confidence interval analyte in foliage (mg/kg) | | Exposure estimate (mg/kg per day) | | Walker Branch vegetation | |
|---|---|---|---|---|---|---|
| | FCAP vegetation | Sluice Channel Area vegetation | FCAP vegetation | Sluice Channel Area vegetation | Upper 95% confidence interval analyte in foliage (mg/kg) | Estimated exposure (mg/kg per day) |
| Arsenic | 1.77 | 1[1] | 0.14868 | 0.084 | 1.06 | 0.08904 |
| Cadmium | 1.23 | 0.37 | 0.10332 | 0.03108 | 0.26 | 0.02184 |
| Chromium | 5.3 | 3[1] | 0.4452 | 0.252 | 3[1] | 0.252 |
| Lead | 1.07 | 1[1] | 0.08988 | 0.084 | 1[1] | 0.084 |
| Mercury | 0.06 | 0.07 | 0.00504 | 0.00588 | 0.12 | 0.01008 |
| Selenium | 23.61 | 1.95 | 1.98324 | 0.1638 | 1[1] | 0.084 |
| Thallium[1] | 1[1] | 1[1] | 0.084 | 0.084 | 1[1] | 0.084 |

[1]Below detection limit, therefore value for detection limit used in calculation.

**Table 6.17. Exposure estimates for red fox consuming small mammals**

| Analyte | Upper 95% confidence interval analyte in small mammals (mg/kg) | | Absolute maximum exposure estimate (mg/kg per day) | | Maximum proportional exposure estimate (mg/kg per day) | | Walker Branch small mammals | |
|---|---|---|---|---|---|---|---|---|
| | FCAP | Sluice Channel Area | FCAP | Sluice Channel Area | FCAP | Sluice Channel Area | Upper 95% confidence interval analyte (mg/kg) | Estimated exposure (mg/kg per day) |
| Arsenic | 0.219 | 0.12 | 0.02 | 0.01096 | 0.00117 | 0.00064 | 0.062 | 0.00566 |
| Cadmium | 0.033 | 0.047 | 0.00301 | 0.00429 | 0.00018 | 0.00025 | 0.037 | 0.00338 |
| Chromium | 1.528 | 1.267 | 0.13956 | 0.11572 | 0.00815 | 0.00676 | 3.822 | 0.34908 |
| Lead | 2.861 | 0.718 | 0.2613 | 0.06558 | 0.01526 | 0.00383 | 1.261 | 0.11517 |
| Mercury | 0.054 | 0.254 | 0.00493 | 0.0232 | 0.00029 | 0.00135 | 0.269 | 0.02457 |
| Selenium | 3.191 | 1.423 | 0.29144 | 0.12997 | 0.01702 | 0.00759 | 0.977 | 0.08923 |
| Thallium[1] | 0.211 | 0.2 | 0.01927 | 0.01827 | 0.00113 | 0.00107 | 0.2 | 0.01827 |

[1]Below detection limit, therefore value for detection limit used in calculation.

**Table 6.18. Exposure estimates for red-tailed hawks consuming small mammals**

| Analyte | Upper 95% confidence interval analyte in small mammals (mg/kg) | | Absolute maximum exposure estimate (mg/kg per day) | | Maximum proportional exposure estimate (mg/kg per day) | | Walker Branch small mammals | |
|---|---|---|---|---|---|---|---|---|
| | FCAP | Sluice Channel Area | FCAP | Sluice Channel Area | FCAP | Sluice Channel Area | Upper 95% confidence interval analyte (mg/kg) | Estimated exposure (mg/kg per day) |
| Arsenic | 0.219 | 0.12 | 0.13809 | 0.07567 | 0.00311 | 0.00171 | 0.062 | 0.03909 |
| Cadmium | 0.033 | 0.047 | 0.02081 | 0.02964 | 0.00047 | 0.00067 | 0.037 | 0.02333 |
| Chromium | 1.528 | 1.267 | 0.96348 | 0.79891 | 0.02171 | 0.018 | 3.822 | 2.40996 |
| Lead | 2.861 | 0.718 | 1.80401 | 0.45274 | 0.04065 | 0.0102 | 1.261 | 0.79512 |
| Mercury | 0.054 | 0.254 | 0.03405 | 0.16016 | 0.00077 | 0.00361 | 0.269 | 0.16962 |
| Selenium | 3.191 | 1.423 | 2.01209 | 0.89727 | 0.04534 | 0.02022 | 0.977 | 0.61605 |
| Thallium[1] | 0.211 | 0.2 | 0.13305 | 0.12611 | 0.003 | 0.00284 | 0.2 | 0.12611 |

[1]Below detection limit, therefore value for detection limit used in calculation.

**Table 6.19. Total exposure experienced by white-footed mice from all sources**

| Analyte | Exposure to white-footed mice (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Aluminum | 37.95228 | 23.688 | 33.32685 | 17.35938 |
| Arsenic | 0.37023 | 0.35112 | 0.11052 | 0.09916 |
| Barium | 0.76609 | 0.70728 | 0.9541 | 0.19814 |
| Beryllium | 0.00738 | 0.00269 | 0.00487 | 0.00129 |
| Cadmium | 0.10332 | 0.03108 | 0.00617 | 0.0249 |
| Calcium | 13.3635 | 8.2656 | 16.3935 | 8.9618 |
| Chloride | 0 | 0 | 0.39848 | 0 |
| Chromium | 0.48866 | 0.28073 | 0.02839 | 0.28727 |
| Cobalt | 0.03242 | 0.04805 | 0.03242 | 0.03478 |
| Copper | 0.11716 | 0.11777 | 0.09905 | 0.01201 |
| Fluoride | 0 | 0 | 0.02886 | 0 |
| Iron | 64.23112 | 307.44 | 87.35022 | 28.94482 |
| Lead | 0.12197 | 0.1139 | 0.02695 | 0.1404 |
| Magnesium | 5.18145 | 1.6968 | 5.007 | 3.58062 |
| Manganese | 0.50441 | 0.57792 | 1.54698 | 4.02247 |
| Mercury | 0.00622 | 0.0069 | 0.00102 | 0.01063 |
| Nickel | 0.06048 | 0.06703 | 0.0588 | 0.01428 |
| Niobium | 0 | 0 | 0.00117 | 0 |
| Nitrate | 0 | 0 | 0.1465 | 0 |
| Phosphorus | 0 | 0 | 2.5368 | 0.54768 |
| Potassium | 8.53716 | 3.2256 | 6.82681 | 1.51401 |
| Selenium | 2.0081 | 0.1717 | 0.0103 | 0.08573 |
| Sodium | 1.41922 | 0.57456 | 0.9742 | 0.43875 |
| Sulphate | 0 | 0 | 3.53065 | 0 |
| Thallium | 0.08771 | 0.084 | 0 | 0.084 |
| Vanadium | 0.14492 | 0.0798 | 0.1218 | 0.05985 |
| Zinc | 0.10608 | 0.0672 | 0.12097 | 0.08455 |
| Cesium-137[1] | 0 | 0 | 0.00022 | 0.00055 |
| Radium-226[1] | 0 | 0 | 0.08307 | 0 |
| Thorium-228[1] | 0.03588 | 0.00185 | 0.02147 | 0.00745 |
| Thorium-230[1] | 0.29975 | 0.00487 | 0.09083 | 0.10089 |
| Thorium-232[1] | 0.00356 | 0.00218 | 0.01128 | 0.05947 |

**Taable 6.19 (continued)**

| Analyte | Exposure to white-footed mice (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Thorium-234[1] | 0 | 0 | 0.01058 | 0 |
| Uranium-234[1] | 0.05485 | 0.00739 | 0.07325 | 0.06184 |
| Uranium-235[1] | 0.00039 | 0.00029 | 0 | 0.00057 |
| Uranium-238[1] | 0.02262 | 0.00554 | 0.05498 | 0.02292 |
| 2-Methylnaphthalene | 0.00016 | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 | 0.00024 |
| Butylbenzylphthalate | 0.00018 | 0 | 0 | 0.00009 |
| bis(2-Ethylhexyl)Phthalate | 0.00045 | 0 | 0.00703 | 0.00088 |
| Chloroform | 0 | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0.00251 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0.00015 |
| Phenol | 0 | 0 | 0 | 0.00029 |

[1]Units for radionuclides were pCi/g per day.

**Table 6.20. Total exposure experienced by short-tailed shrew from all sources**

| Analyte | Exposure to short-tailed shrew (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Aluminum | 1709.39328 | 1099.8 | 1544.46464 | 803.45695 |
| Arsenic | 10.21951 | 12.402 | 4.62898 | 0.44973 |
| Barium | 35.11038 | 32.838 | 43.69368 | 8.44118 |
| Beryllium | 0.34242 | 0.1248 | 0.2262 | 0.05967 |
| Cadmium | 0 | 0 | 0.28626 | 0.14196 |
| Calcium | 440.868 | 383.76 | 437.8824 | 179.30373 |
| Chloride | 0 | 0 | 0.40981 | 0 |
| Chromium | 1.95913 | 1.3338 | 1.3182 | 1.58461 |
| Cobalt | 1.5054 | 2.2308 | 1.5054 | 1.6146 |
| Copper | 5.3909 | 5.4678 | 4.56379 | 0.531 |
| Fluoride | 0 | 0 | 0.02968 | 0 |
| Iron | 2941.52057 | 14274 | 4048.36272 | 1334.02299 |
| Lead | 1.46693 | 1.3884 | 1.24045 | 2.6055 |
| Magnesium | 138.8052 | 78.78 | 128.7104 | 50.5136 |
| Manganese | 12.11213 | 26.832 | 68.40544 | 184.90204 |
| Mercury | 0.05499 | 0.04758 | 0.04536 | 0.02398 |
| Nickel | 2.808 | 3.1122 | 2.73 | 0.663 |
| Niobium | 0 | 0 | 0.00121 | 0 |
| Nitrate | 0 | 0 | 0.15067 | 0 |
| Phosphorus | 0 | 0 | 117.78 | 25.428 |
| Potassium | 352.00495 | 149.76 | 290.75363 | 57.92224 |
| Selenium | 1.1544 | 0.3666 | 0.47814 | 0.08034 |
| Sodium | 57.97729 | 26.676 | 28.53617 | 14.79041 |
| Sulphate | 0 | 0 | 3.63107 | 0 |
| Thallium | 0.17238 | 0 | 0 | 0 |
| Vanadium | 6.62455 | 3.705 | 5.655 | 2.7387 |
| Zinc | 4.22017 | 3.12 | 5.50161 | 3.86243 |
| Cesium-137[1] | 0 | 0 | 0.01006 | 0.02574 |
| Radium-226[1] | 0 | 0 | 0.08543 | 0 |
| Thorium-228[1] | 0.20241 | 0.0858 | 0.72378 | 0.04656 |
| Thorium-230[1] | 0.61489 | 0.2262 | 0.09341 | 0.75969 |
| Thorium-232[1] | 0.16536 | 0.1014 | 0.0116 | 0.10083 |

Table 6.20 (continued)

| Analyte | Exposure to short-tailed shrew (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Thorium-234[1] | 0 | 0 | 0.4914 | 0 |
| Uranium-234[1] | 0.35158 | 0.3432 | 0.07533 | 0.33055 |
| Uranium-235[1] | 0.01833 | 0.01326 | 0 | 0.02652 |
| Uranium-238[1] | 0.31843 | 0.2574 | 0.31816 | 0.1929 |
| 2-Methylnaphthalene | 0.00725 | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 | 0.01092 |
| Butylbenzylphthalate | 0.00858 | 0 | 0 | 0.00421 |
| bis(2-Ethylhexyl)Phthalate | 0.02106 | 0 | 0 | 0.0009 |
| Chloroform | 0.00016 | 0 | 0 | 0 |
| alpha-BHC | 0.00002 | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0.00015 |
| Phenol | 0 | 0 | 0 | 0.0003 |

[1]Units for radionuclides were pCi/g per day.

**Table 6.21. Total exposure experienced by red fox from all sources under the absolute maximum exposure scenario**

| Analyte | Absolute maximum exposure to red fox (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Aluminum | 8.68604 | 5.17 | 7.29554 | 3.80798 |
| Arsenic | 0.06886 | 0.07009 | 0.02796 | 0.00796 |
| Barium | 0.17071 | 0.15437 | 0.21285 | 0.04904 |
| Beryllium | 0.00161 | 0.00059 | 0.00106 | 0.00028 |
| Cadmium | 0.00301 | 0.00429 | 0.00135 | 0.00405 |
| Calcium | 4.28983 | 1.804 | 6.0497 | 3.76654 |
| Chloride | 0 | 0 | 0.22531 | 0 |
| Chromium | 0.140553 | 0.12272 | 0.0062 | 0.35718 |
| Cobalt | 0.00708 | 0.01049 | 0.00708 | 0.00759 |
| Copper | 0.02594 | 0.0257 | 0.02189 | 0.00282 |
| Fluoride | 0 | 0 | 0.01632 | 0 |
| Iron | 14.32944 | 67.1 | 19.11946 | 6.39259 |
| Lead | 0.26848 | 0.0724 | 0.00597 | 0.12759 |
| Magnesium | 1.90902 | 0.37033 | 1.8862 | 1.66643 |
| Manganese | 0.19655 | 0.12613 | 0.36378 | 0.89211 |
| Mercury | 0.00519 | 0.02342 | 0.00023 | 0.0247 |
| Nickel | 0.0132 | 0.01463 | 0.01283 | 0.00312 |
| Niobium | 0 | 0 | 0.00066 | 0 |
| Nitrate | 0 | 0 | 0.08283 | 0 |
| Phosphorus | 0 | 0 | 0.55367 | 0.11953 |
| Potassium | 2.2025 | 0.704 | 1.69036 | 0.42504 |
| Selenium | 0.29687 | 0.13169 | 0.00225 | 0.08961 |
| Sodium | 0.37027 | 0.1254 | 0.34028 | 0.13843 |
| Sulphate | 0 | 0 | 1.99628 | 0 |
| Thallium | 0.02008 | 0.01827 | 0 | 0.01827 |
| Vanadium | 0.03242 | 0.01742 | 0.02658 | 0.01337 |
| Zinc | 0.02854 | 0.01467 | 0.02728 | 0.01894 |
| Cesium-137[1] | 0 | 0 | 0.00005 | 0.00012 |
| Radium-226[1] | 0 | 0 | 0.04697 | 0 |
| Thorium-228[1] | 0.01902 | 0.0004 | 0.00677 | 0.00392 |
| Thorium-230[1] | 0.16714 | 0.00106 | 0.05136 | 0.05202 |

Table 6.21 (continued)

| Analyte | Absolute maximum exposure to red fox (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Thorium-232[1] | 0.00078 | 0.00048 | 0.00638 | 0.03332 |
| Thorium-234[1] | 0 | 0 | 0.00231 | 0 |
| Uranium-234[1] | 0.02876 | 0.00161 | 0.04142 | 0.03292 |
| Uranium-235[1] | 0.00009 | 0.00006 | 0 | 0.00012 |
| Uranium-238[1] | 0.01053 | 0.00121 | 0.02909 | 0.01166 |
| 2-Methylnaphthalene | 0.00003 | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 | 0.00005 |
| Butylbenzylphthalate | 0.00004 | 0 | 0 | 0.00002 |
| bis(2-Ethylhexyl)Phthalate | 0.0001 | 0 | 0.00398 | 0.0005 |
| Chloroform | 0 | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0.00142 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0.00008 |
| Phenol | 0 | 0 | 0 | 0.00017 |

[1]Units for radionuclides were pCi/g per day.

**Table 6.22. Total exposure experienced by red fox from all sources under the maximum proportional exposure scenario**

| Analyte | Maximum proportional exposure to red fox (mg/kg per day) | | |
|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB |
| Aluminum | 0.51278 | 0.3055 | 0.43107 |
| Arsenic | 0.00406 | 0.00409 | 0.00164 |
| Barium | 0.01008 | 0.00912 | 0.01257 |
| Beryllium | 0.0001 | 0.00003 | 0.00006 |
| Cadmium | 0.00018 | 0.00025 | 0.00008 |
| Calcium | 0.25183 | 0.1066 | 0.3545 |
| Chloride | 0 | 0 | 0.01315 |
| Chromium | 0.00873 | 0.00713 | 0.00037 |
| Cobalt | 0.00042 | 0.00062 | 0.00042 |
| Copper | 0.00154 | 0.00152 | 0.0013 |
| Fluoride | 0 | 0 | 0.00095 |
| Iron | 0.84636 | 3.965 | 1.12972 |
| Lead | 0.01569 | 0.00422 | 0.00035 |
| Magnesium | 0.11187 | 0.02188 | 0.1105 |
| Manganese | 0.01151 | 0.00745 | 0.02146 |
| Mercury | 0.00031 | 0.00136 | 0.00001 |
| Nickel | 0.00078 | 0.00086 | 0.00076 |
| Niobium | 0 | 0 | 0.00004 |
| Nitrate | 0 | 0 | 0.00483 |
| Phosphorus | 0 | 0 | 0.03272 |
| Potassium | 0.12974 | 0.0416 | 0.09964 |
| Selenium | 0.01734 | 0.00769 | 0.00013 |
| Sodium | 0.02181 | 0.00741 | 0.01995 |
| Sulphate | 0 | 0 | 0.11648 |
| Thallium | 0.00118 | 0.00107 | 0 |
| Vanadium | 0.00192 | 0.00103 | 0.00157 |
| Zinc | 0.00168 | 0.00087 | 0.00161 |
| Cesium-137[1] | 0 | 0 | 0 |
| Radium-226[1] | 0 | 0 | 0.00274 |
| Thorium-228[1] | 0.00111 | 0.00002 | 0.0004 |
| Thorium-230[1] | 0.00976 | 0.00006 | 0.003 |
| Thorium-232[1] | 0.00005 | 0.00003 | 0.00037 |

Table 6.22 (continued)

| Analyte | Maximum proportional exposure to red fox (mg/kg per day) | | |
|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB |
| Thorium-232[1] | 0.00005 | 0.00003 | 0.00037 |
| Thorium-234[1] | 0 | 0 | 0.00014 |
| Uranium-234[1] | 0.00168 | 0.0001 | 0.00242 |
| Uranium-235[1] | 0.00001 | 0 | 0 |
| Uranium-238[1] | 0.00061 | 0.00007 | 0.00169 |
| 2-Methylnaphthalene | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 |
| Butylbenzylphthalate | 0 | 0 | 0 |
| bis(2-Ethylhexyl)Phthalate | 0.00001 | 0 | 0.00023 |
| Chloroform | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0.00008 |
| Di-N-butylphthalate | 0 | 0 | 0 |
| Phenol | 0 | 0 | 0 |

[1]Units for radionuclides were pCi/g per day.

**Table 6.23. Total exposure experienced by white-tailed deer from all sources under the absolute maximum exposure scenario**

| Analyte | Absolute maximum exposure to white-tailed deer (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Aluminum | 13.29561 | 8.225 | 11.5782 | 6.03318 |
| Arsenic | 0.12841 | 0.12175 | 0.03949 | 0.0343 |
| Barium | 0.26703 | 0.24558 | 0.33264 | 0.07049 |
| Beryllium | 0.00256 | 0.00093 | 0.00169 | 0.00045 |
| Cadmium | 0.03567 | 0.01073 | 0.00214 | 0.0086 |
| Calcium | 5.04147 | 2.87 | 6.41464 | 3.64094 |
| Chloride | 0 | 0 | 0.17879 | 0 |
| Chromium | 0.16892 | 0.09698 | 0.00986 | 0.09937 |
| Cobalt | 0.01126 | 0.01668 | 0.01126 | 0.01208 |
| Copper | 0.04079 | 0.04089 | 0.03448 | 0.00423 |
| Fluoride | 0 | 0 | 0.01295 | 0 |
| Iron | 22.3933 | 106.75 | 30.34599 | 10.07229 |
| Lead | 0.04223 | 0.03938 | 0.00939 | 0.04861 |
| Magnesium | 2.02655 | 0.58917 | 1.97044 | 1.50193 |
| Manganese | 0.20042 | 0.20067 | 0.54479 | 1.40084 |
| Mercury | 0.00215 | 0.00239 | 0.00036 | 0.00367 |
| Nickel | 0.021 | 0.02328 | 0.02042 | 0.00496 |
| Niobium | 0 | 0 | 0.00053 | 0 |
| Nitrate | 0 | 0 | 0.06573 | 0 |
| Phosphorus | 0 | 0 | 0.88083 | 0.19017 |
| Potassium | 3.06344 | 1.12 | 2.42899 | 0.55334 |
| Selenium | 0.69332 | 0.05929 | 0.00358 | 0.0296 |
| Sodium | 0.51047 | 0.1995 | 0.37557 | 0.16482 |
| Sulphate | 0 | 0 | 1.58417 | 0 |
| Thallium | 0.03029 | 0.029 | 0 | 0.029 |
| Vanadium | 0.05056 | 0.02771 | 0.04229 | 0.02087 |
| Zinc | 0.03841 | 0.02333 | 0.04227 | 0.02951 |
| Cesium-137[1] | 0 | 0 | 0.00008 | 0.00019 |
| Radium-226[1] | 0 | 0 | 0.03727 | 0 |
| Thorium-228[1] | 0.01573 | 0.00064 | 0.00807 | 0.00326 |
| Thorium-230[1] | 0.13382 | 0.00169 | 0.04075 | 0.0438 |

Table 6.23 (continued)

| Analyte | Absolute maximum exposure to white-tailed deer (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Thorium-232[1] | 0.00124 | 0.00076 | 0.00506 | 0.02659 |
| Thorium-234[1] | 0 | 0 | 0.00368 | 0 |
| Uranium-234[1] | 0.02395 | 0.00257 | 0.03287 | 0.02715 |
| Uranium-235[1] | 0.00014 | 0.0001 | 0 | 0.0002 |
| Uranium-238[1] | 0.00949 | 0.00193 | 0.02409 | 0.00991 |
| 2-Methylnaphthalene | 0.00005 | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 | 0.00008 |
| Butylbenzylphthalate | 0.00006 | 0 | 0 | 0.00003 |
| bis(2-Ethylhexyl)Phthalate | 0.00016 | 0 | 0.00316 | 0.00039 |
| Chloroform | 0 | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0.00112 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0.00007 |
| Phenol | 0 | 0 | 0 | 0.00013 |

[1]Units for radionuclides were pCi/g per day.

**Table 6.24. Total exposure experienced by white-tailed deer from all sources under the maximum proportional exposure scenario**

| Analyte | Maximum proportional exposure to white-tailed deer (mg/kg per day) | | |
|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB |
| Aluminum | 1.35611 | 0.83895 | 1.18097 |
| Arsenic | 0.01308 | 0.01241 | 0.00403 |
| Barium | 0.02724 | 0.02505 | 0.03393 |
| Beryllium | 0.00026 | 0.0001 | 0.00017 |
| Cadmium | 0.00363 | 0.00109 | 0.00022 |
| Calcium | 0.5141 | 0.29274 | 0.65406 |
| Chloride | 0 | 0 | 0.01822 |
| Chromium | 0.01719 | 0.00987 | 0.00101 |
| Cobalt | 0.00115 | 0.0017 | 0.00115 |
| Copper | 0.00416 | 0.00417 | 0.00352 |
| Fluoride | 0 | 0 | 0.00132 |
| Iron | 2.28409 | 10.8885 | 3.09529 |
| Lead | 0.0043 | 0.00401 | 0.00096 |
| Magnesium | 0.20664 | 0.0601 | 0.20091 |
| Manganese | 0.02061 | 0.02047 | 0.05556 |
| Mercury | 0.00352 | 0.00025 | 0.00003 |
| Nickel | 0.00214 | 0.00237 | 0.00208 |
| Niobium | 0 | 0 | 0.00005 |
| Nitrate | 0 | 0 | 0.0067 |
| Phosphorus | 0 | 0 | 0.08985 |
| Potassium | 0.31244 | 0.11424 | 0.24774 |
| Selenium | 0.07053 | 0.00603 | 0.00036 |
| Sodium | 0.05206 | 0.02035 | 0.0383 |
| Sulphate | 0 | 0 | 0.16147 |
| Thallium | 0.00308 | 0.00295 | 0 |
| Vanadium | 0.00515 | 0.00283 | 0.00431 |
| Zinc | 0.00392 | 0.00238 | 0.00431 |
| Cesium-137[1] | 0 | 0 | 0.00001 |

Table 6.24 (continued)

| Analyte | Maximum proportional exposure to white-tailed deer (mg/kg per day) | | |
|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB |
| Radium-226[1] | 0 | 0 | 0.0038 |
| Thorium-228[1] | 0.0016 | 0.00007 | 0.00082 |
| Thorium-230[1] | 0.01364 | 0.00017 | 0.00415 |
| Thorium-232[1] | 0.00013 | 0.00008 | 0.00052 |
| Thorium-234[1] | 0 | 0 | 0.00037 |
| Uranium-234[1] | 0.00244 | 0.00026 | 0.00335 |
| Uranium-235[1] | 0.00001 | 0.00001 | 0 |
| Uranium-238[1] | 0.00097 | 0.0002 | 0.00245 |
| 2-Methylnaphthalene | 0.00001 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 |
| Butylbenzylphthalate | 0.00001 | 0 | 0 |
| bis(2-Ethylhexyl)Phthalate | 0.00002 | 0 | 0.00032 |
| Chloroform | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0.00011 |
| Di-N-butylphthalate | 0 | 0 | 0 |
| Phenol | 0 | 0 | 0 |

[1]Units for radionuclides were pCi/g per day.

**Table 6.25. Total exposure experienced by red-tailed hawk from all sources under the absolute maximum exposure scenario**

| Analyte | Absolute maximum exposure to red-tailed hawk (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Aluminum | 0.45016 | 0 | 0.02438 | 0.02148 |
| Arsenic | 0.13866 | 0.07567 | 0.00429 | 0.03926 |
| Barium | 0.00392 | 0 | 0.00516 | 0.00648 |
| Beryllium | 0 | 0 | 0 | 0 |
| Cadmium | 0.02081 | 0.02964 | 0 | 0.02333 |
| Calcium | 1.53464 | 0 | 2.76234 | 2.02345 |
| Chloride | 0 | 0 | 0.1546 | 0 |
| Chromium | 0.96398 | 0.79891 | 0 | 2.41041 |
| Cobalt | 0 | 0 | 0 | 0 |
| Copper | 0.00041 | 0 | 0.0003 | 0.00023 |
| Fluoride | 0 | 0 | 0.0112 | 0 |
| Iron | 0.34728 | 0 | 0.06139 | 0.08412 |
| Lead | 1.80421 | 0.45274 | 0.00009 | 0.79523 |
| Magnesium | 0.86963 | 0 | 0.88668 | 0.98899 |
| Manganese | 0.09663 | 0 | 0.02921 | 0.01586 |
| Mercury | 0.03405 | 0.16016 | 0.00001 | 0.16963 |
| Nickel | 0 | 0 | 0 | 0 |
| Niobium | 0 | 0 | 0.00046 | 0 |
| Nitrate | 0 | 0 | 0.05684 | 0 |
| Phosphorus | 0 | 0 | 0 | 0 |
| Potassium | 0.37911 | 0 | 0.22394 | 0.10572 |
| Selenium | 2.01209 | 0.89727 | 0 | 0.61605 |
| Sodium | 0.06764 | 0 | 0.14266 | 0.04769 |
| Sulphate | 0 | 0 | 1.3698 | 0 |
| Thallium | 0.13305 | 0.12611 | 0 | 0.12611 |
| Vanadium | 0.00089 | 0 | 0 | 0.00034 |
| Zinc | 0.00602 | 0 | 0.00098 | 0.00054 |
| Cesium-137[1] | 0 | 0 | 0 | 0 |
| Radium-226[1] | 0 | 0 | 0.03223 | 0 |

Table 6.25 (continued)

| Analyte | Absolute maximum exposure to red-tailed hawk (mg/kg per day) | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB | Background |
| Thorium-228[1] | 0.0125 | 0 | 0.00233 | 0.00256 |
| Thorium-230[1] | 0.11368 | 0 | 0.03524 | 0.03353 |
| Thorium-232[1] | 0 | 0 | 0.00438 | 0.02274 |
| Thorium-234[1] | 0 | 0 | 0 | 0 |
| Uranium-234[1] | 0.01876 | 0 | 0.02842 | 0.02171 |
| Uranium-235[1] | 0 | 0 | 0 | 0 |
| Uranium-238[1] | 0.00625 | 0 | 0.0191 | 0.00745 |
| 2-Methylnaphthalene | 0 | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 | 0 |
| Butylbenzylphthalate | 0 | 0 | 0 | 0 |
| bis(2-Ethylhexyl)Phthalate | 0 | 0 | 0 | 0.00034 |
| Chloroform | 0 | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0.00006 |
| Phenol | 0 | 0 | 0 | 0.00011 |

[1]Units for radionuclides were pCi/g per day.

**Table 6.26. Total exposure experienced by red-tailed hawk from all sources under the maximum proportional exposure scenario**

| Analyte | Maximum proportional exposure to red-tailed hawk (mg/kg per day) | | |
|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB |
| Aluminum | 0.01055 | 0 | 0.02438 |
| Arsenic | 0.00312 | 0.00171 | 0.00429 |
| Barium | 0.00009 | 0 | 0.00516 |
| Beryllium | 0 | 0 | 0 |
| Cadmium | 0.00047 | 0.00067 | 0 |
| Calcium | 0.03597 | 0 | 2.76234 |
| Chloride | 0 | 0 | 0.1546 |
| Chromium | 0.02172 | 0.018 | 0 |
| Cobalt | 0 | 0 | 0 |
| Copper | 0.00001 | 0 | 0.0003 |
| Fluoride | 0 | 0 | 0.0112 |
| Iron | 0.00814 | 0 | 0.06139 |
| Lead | 0.04065 | 0.0102 | 0.00009 |
| Magnesium | 0.02038 | 0 | 0.88668 |
| Manganese | 0.00226 | 0 | 0.02921 |
| Mercury | 0.00077 | 0.00361 | 0.00001 |
| Nickel | 0 | 0 | 0 |
| Niobium | 0 | 0 | 0.00046 |
| Nitrate | 0 | 0 | 0.05684 |
| Phosphorus | 0 | 0 | 0 |
| Potassium | 0.00889 | 0 | 0.22394 |
| Selenium | 0.04529 | 0.0202 | 0 |
| Sodium | 0.00159 | 0 | 0.14266 |
| Sulphate | 0 | 0 | 1.3698 |
| Thallium | 0.003 | 0.00284 | 0 |
| Vanadium | 0.00002 | 0 | 0 |
| Zinc | 0.00014 | 0 | 0.00098 |
| Cesium-137[1] | 0 | 0 | 0 |
| Radium-226[1] | 0 | 0 | 0.03223 |
| Thorium-228[1] | 0.00029 | 0 | 0.00233 |
| Thorium-230[1] | 0.00266 | 0 | 0.03524 |

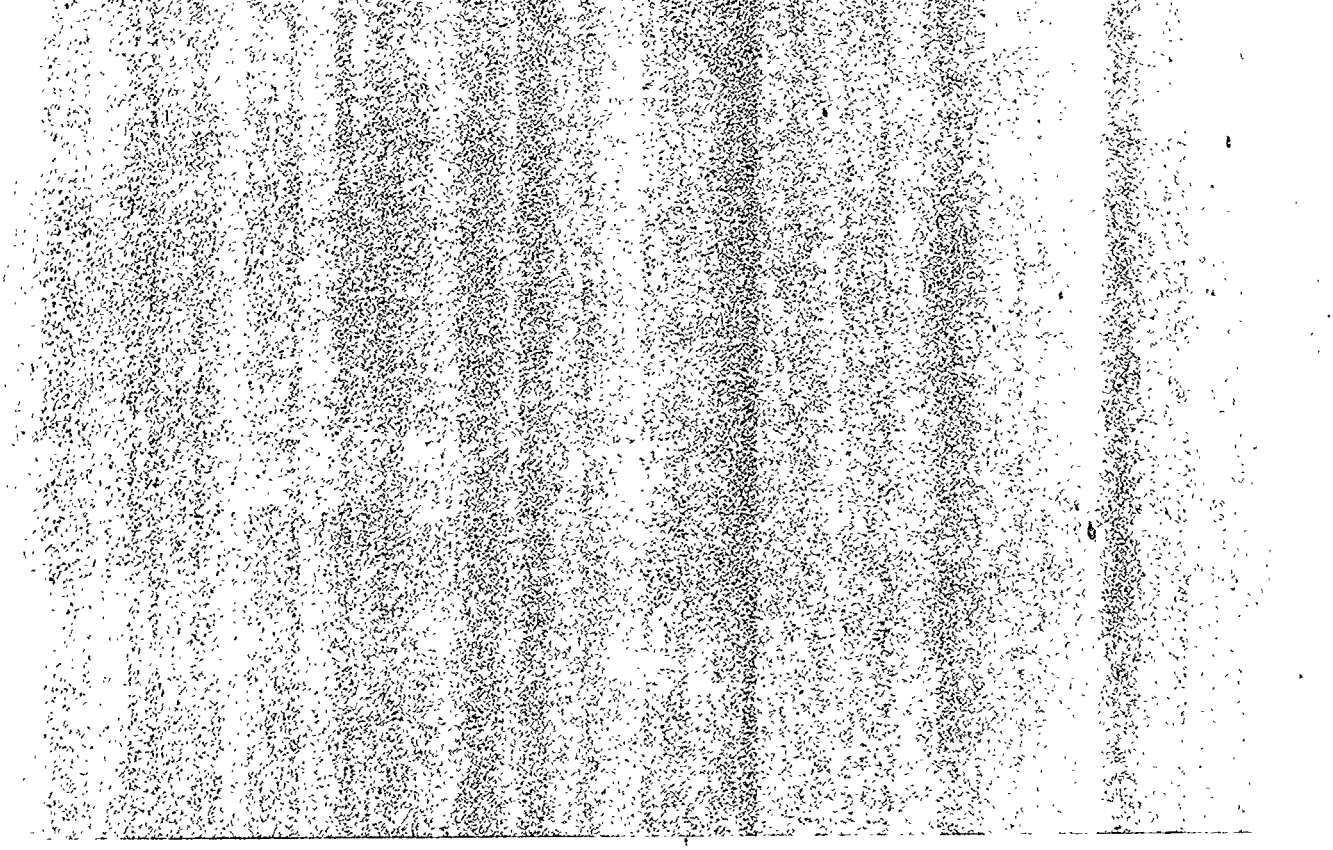




**Table 6.26 (continued)**

| Analyte | Maximum proportional exposure to red-tailed hawk (mg/kg per day) | | |
|---|---|---|---|
| | FCAP | Sluice Channel Area | UMB |
| Thorium-232[1] | 0 | 0 | 0.00438 |
| Thorium-234[1] | 0 | 0 | 0 |
| Uranium-234[1] | 0.00044 | 0 | 0.02842 |
| Uranium-235[1] | 0 | 0 | 0 |
| Uranium-238[1] | 0.00015 | 0 | 0.0191 |
| 2-Methylnaphthalene | 0 | 0 | 0 |
| 4-Nitrophenol | 0 | 0 | 0 |
| Butylbenzylphthalate | 0 | 0 | 0 |
| bis(2-Ethylhexyl)Phthalate | 0 | 0 | 0 |
| Chloroform | 0 | 0 | 0 |
| alpha-BHC | 0 | 0 | 0 |
| Di-N-octylphthalate | 0 | 0 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 |
| Phenol | 0 | 0 | 0 |

[1]Units for radionuclides were pCi/g per day.

Table 6.27. Nonadvisory surface water benchmarks

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| *INORGANICS* | | |
| ALUMINUM | 1.90e+00 | LCV_DAPHNIDS |
| | 5.80e+00 | LCV_FISH |
| | 5.40e-01 | LTV_DAPHNIDS |
| | 4.70e+00 | LTV_FISH |
| | 7.50e-01 | NAWQ_ACUTE |
| | 8.70e-02 | NAWQ_CHRONIC |
| | 7.50e-02 | SS_EC20 |
| ANTIMONY | 7.90e-02 | EC20_POPULATION |
| | 5.40e+00 | LCV_DAPHNIDS |
| | 1.60e+00 | LCV_FISH |
| | 1.90e+00 | LTV_DAPHNIDS |
| | 2.31e+00 | LTV_FISH |
| ARSENIC III | 2.00e+00 | EC20_POPULATION |
| | 9.14e-01 | LCV_DAPHNIDS |
| | 2.96e+00 | LCV_FISH |
| | 6.33e-01 | LTV_DAPHNIDS |
| | 2.13e+00 | LTV_FISH |
| | 3.60e-01 | NAWQ_ACUTE |
| | 1.90e-01 | NAWQ_CHRONIC |
| | 5.50e-02 | SS_EC20 |
| ARSENIC V | 1.85e-01 | EC20_POPULATION |
| | 4.50e-01 | LCV_DAPHNIDS |
| | 8.92e-01 | LCV_FISH |
| | 9.32e-01 | LTV_DAPHNIDS |
| | 1.50e+00 | LTV_FISH |
| BARIUM | 2.03e+01 | LCV_DAPHNIDS |
| BERYLLIUM | 2.10e-02 | EC20_POPULATION |
| | 5.30e-03 | LCV_DAPHNIDS |
| | 5.70e-02 | LCV_FISH |
| | 3.80e-03 | LTV_DAPHNIDS |
| | 1.48e-01 | LTV_FISH |
| BORON | 9.30e-03 | LCV_DAPHNIDS |
| | 7.00e-03 | LTV_DAPHNIDS |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| CADMIUM | 4.30e-03 | EC20_POPULATION |
| | 1.50e-04 | LCV_DAPHNIDS |
| | 1.70e-03 | LCV_FISH |
| | 7.50e-04 | LTV_DAPHNIDS |
| | 1.80e-03 | LTV_FISH |
| | 3.90e-03 | NAWQ_ACUTE |
| | 1.10e-03 | NAWQ_CHRONIC |
| | 1.30e-05 | SS_EC20 |
| CHLORIDE | 2.30e+02 | NAWQ_ACUTE |
| | 8.60e+02 | NAWQ_CHRONIC |
| CHROMIUM III | 1.26e-01 | EC20_POPULATION |
| | 4.40e-02 | LCV_DAPHNIDS |
| | 6.88e-02 | LCV_FISH |
| | 8.90e-02 | LTV_FISH |
| | 1.70e+00 | NAWQ_ACUTE |
| | 2.10e-01 | NAWQ_CHRONIC |
| | 8.44e-03 | SS_EC20 |
| CHROMIUM VI | 3.16e-01 | EC20_POPULATION |
| | 6.13e-03 | LCV_DAPHNIDS |
| | 7.32e-02 | LCV_FISH |
| | 5.00e-04 | LTV_DAPHNIDS |
| | 5.10e-02 | LTV_FISH |
| | 1.60e-02 | NAWQ_ACUTE |
| | 1.10e-02 | NAWQ_CHRONIC |
| | 2.66e-04 | SS_EC20 |
| COBALT | 3.98e-03 | EC20_POPULATION |
| | 5.10e-03 | LCV_DAPHNIDS |
| | 2.90e-01 | LCV_FISH |
| | 4.40e-03 | LTV_DAPHNIDS |
| | 8.10e-01 | LTV_FISH |
| COPPER | 8.60e-03 | EC20_POPULATION |
| | 2.30e-04 | LCV_DAPHNIDS |
| | 3.87e-03 | LCV_FISH |
| | 2.05e-04 | LTV_DAPHNIDS |
| | 5.00e-03 | LTV_FISH |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| | 1.80e-02 | NAWQ_ACUTE |
| | 1.20e-02 | NAWQ_CHRONIC |
| | 2.60e-04 | SS_EC20 |
| CYANIDE | 1.10e-02 | EC20_POPULATION |
| | 7.80e-03 | LCV_FISH |
| | 5.30e-03 | LTV_FISH |
| | 2.20e-02 | NAWQ_ACUTE |
| | 5.20e-03 | NAWQ_CHRONIC |
| | 1.17e-03 | SS_EC20 |
| FLUORIDE | 1.70e+01 | LITERATURE |
| IRON | 1.58e-01 | LCV_DAPHNIDS |
| | 1.60e-02 | LTV_DAPHNIDS |
| | 1.00e+00 | NAWQ_CHRONIC |
| LEAD | 7.10e-02 | EC20_POPULATION |
| | 1.23e-02 | LCV_DAPHNIDS |
| | 1.89e-02 | LCV_FISH |
| | 2.20e-02 | LTV_FISH |
| | 8.20e-02 | NAWQ_ACUTE |
| | 3.20e-03 | NAWQ_CHRONIC |
| | 3.50e-04 | SS_EC20 |
| MANGANESE | 1.12e-01 | EC20_POPULATION |
| | 1.10e+00 | LCV_DAPHNIDS |
| | 1.77e+00 | LCV_FISH |
| | 1.10e+00 | LTV_DAPHNIDS |
| | 1.27e+00 | LTV_FISH |
| MERCURY, INORGANIC | 3.20e-04 | EC20_POPULATION |
| | 9.60e-04 | LCV_DAPHNIDS |
| | 2.30e-04 | LCV_FISH |
| | 8.70e-04 | LTV_DAPHNIDS |
| | 8.70e-04 | LTV_FISH |
| | 2.40e-03 | NAWQ_ACUTE |
| | 1.30e-03 | NAWQ_FINAL CHRONIC VALUE |
| | 1.80e-04 | SS_EC20 |
| MERCURY, METHYL | 2.80e-04 | EC20_POPULATION |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| | 4.00e-05 | LCV_DAPHNIDS |
| | 5.20e-04 | LCV_FISH |
| | 8.70e-04 | LTV_DAPHNIDS |
| | 3.00e-05 | LTV_FISH |
| MOLYBDENUM | 8.80e-01 | LCV_DAPHNIDS |
| | 3.60e-01 | LTV_DAPHNIDS |
| NICKEL | 2.15e-01 | EC20_POPULATION |
| | 5.00e-03 | LCV_DAPHNIDS |
| | 3.50e-02 | LCV_FISH |
| | 4.50e-02 | LTV_DAPHNIDS |
| | 6.20e-02 | LTV_FISH |
| | 1.40e+00 | NAWQ_ACUTE |
| | 1.60e-01 | NAWQ_CHRONIC |
| | 1.10e-02 | SS_EC20 |
| NITRATE | 1.00e+02 | LITERATURE |
| SELENIUM | 9.17e-02 | LCV_DAPHNIDS |
| | 8.83e-02 | LCV_FISH |
| | 2.50e-02 | LTV_DAPHNIDS |
| | 4.00e-02 | LTV_FISH |
| | 2.00e-02 | NAWQ_ACUTE |
| | 3.50e-02 | NAWQ_CHRONIC |
| | 2.60e-03 | SS_EC20 |
| SILVER | 3.20e-04 | EC20_POPULATION |
| | 2.60e-03 | LCV_DAPHNIDS |
| | 1.20e-04 | LCV_FISH |
| | 5.60e-04 | LTV_DAPHNIDS |
| | 2.00e-04 | LTV_FISH |
| | 4.10e-03 | NAWQ_ACUTE |
| | 1.40e-04 | SS_EC20 |
| SODIUM | 8.50e+01 | LCV_DAPHNIDS |
| THALLIUM | 6.70e-02 | EC20_POPULATION |
| | 1.35e-01 | LCV_DAPHNIDS |
| | 4.00e-02 | LCV_FISH |
| | 6.40e-02 | LTV_DAPHNIDS |
| | 8.10e-02 | LTV_FISH |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| URANIUM | 2.70e-02 | EC20_POPULATION |
| | 1.42e-01 | LCV_FISH |
| | 4.55e-01 | LTV_FISH |
| VANADIUM | 3.20e-02 | EC20_POPULATION |
| | 9.40e-01 | LCV_DAPHNIDS |
| | 8.00e-02 | LCV_FISH |
| | 4.30e-01 | LTV_DAPHNIDS |
| | 4.10e-02 | LTV_FISH |
| ZINC | 8.00e-02 | EC20_POPULATION |
| | 4.67e-02 | LCV_DAPHNIDS |
| | 3.64e-02 | LCV_FISH |
| | 4.70e-02 | LTV_FISH |
| | 1.20e-01 | NAWQ_ACUTE |
| | 1.10e-01 | NAWQ_CHRONIC |
| | 2.10e-02 | SS_EC20 |
| ZIRCONIUM | 2.51e-01 | EC20_POPULATION |
| | 5.48e-01 | LCV_FISH |
| | 2.40e+00 | LTV_FISH |
| *ORGANICS* | | |
| 1,1,1-TRICHLOROETHANE | 2.51e-01 | EC20_POPULATION |
| | 1.77e+00 | LCV_DAPHNIDS |
| | 3.49e+00 | LCV_FISH |
| | 1.30e+00 | LTV_DAPHNIDS |
| | 2.46e+00 | LTV_FISH |
| 1,1,2,2-TETRACHLOROETHANE | 1.59e+00 | EC20_POPULATION |
| | 9.90e+00 | LCV_DAPHNIDS |
| | 2.40e+00 | LCV_FISH |
| | 4.20e-01 | LTV_DAPHNIDS |
| | 1.40e+00 | LTV_FISH |
| 1,1,2-TRICHLOROETHANE | 1.58e+01 | EC20_POPULATION |
| | 1.84e+01 | LCV_DAPHNIDS |
| | 9.40e+00 | LCV_FISH |
| | 1.30e+01 | LTV_DAPHNIDS |
| | 1.48e+01 | LTV_FISH |
| 1,1-DICHLOROETHANE | 1.59e+00 | EC20_POPULATION |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| | 1.47e+01 | LCV_FISH |
| | 8.22e+00 | LTV_FISH |
| | 4.47e-01 | EC20_POPULATION |
| | 4.72e+00 | LCV_DAPHNIDS |
| | 2.80e+00 | LCV_FISH |
| 1,2-DICHLOROETHANE | 1.26e+00 | EC20_POPULATION |
| | 1.52e+01 | LCV_DAPHNIDS |
| | 4.14e+01 | LCV_FISH |
| | 1.10e+01 | LTV_DAPHNIDS |
| | 2.90e+01 | LTV_FISH |
| 1,2-DICHLOROETHENE | 9.54e+00 | LCV_FISH |
| | 5.72e+00 | LTV_FISH |
| 1,3-DICHLOROPROPENE TOTAL | 4.00e-02 | EC20_POPULATION |
| | 8.05e-01 | LCV_DAPHNIDS |
| | 2.44e-01 | LCV_FISH |
| | 3.50e-01 | LTV_FISH |
| 1-METHYL NAPHTHALENE | 3.16e-02 | EC20_POPULATION |
| | 5.26e-01 | LCV_FISH |
| | 5.00e-01 | LTV_FISH |
| 1-PENTANOL | 3.55e+00 | EC20_POPULATION |
| | 3.05e+01 | LCV_FISH |
| | 1.52e+01 | LTV_FISH |
| 2,4-DINITROPHENOL | 7.00e-01 | LCV_FISH |
| 2-BUTANONE | 1.78e+01 | EC20_POPULATION |
| | 1.39e+03 | LCV_DAPHNIDS |
| | 2.82e+02 | LCV_FISH |
| | 9.88e+01 | LTV_FISH |
| 2-HEXANONE | 1.26e+00 | EC20_POPULATION |
| | 3.28e+01 | LCV_FISH |
| | 1.62e+01 | LTV_FISH |
| 2-METHYLPHENOL | 7.40e-02 | EC20_POPULATION |
| | 1.32e+00 | LCV_DAPHNIDS |
| | 4.89e-01 | LCV_FISH |
| | 4.70e-01 | LTV_FISH |
| 2-NITROPHENOL | 4.30e+00 | LCV_OTHER |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| 2-PROPANOL | 3.16e+00 | EC20_POPULATION |
| | 8.33e+01 | LCV_FISH |
| | 3.54e+01 | LTV_FISH |
| 3-OCTANONE | 1.10e+02 | LCV_DAPHNIDS |
| | 5.45e+00 | LCV_FISH |
| | 3.57e+00 | LTV_FISH |
| 4,4'-DDD | 6.10e-04 | EC20_POPULATION |
| | 1.69e-03 | LCV_FISH |
| | 3.99e-03 | LTV_FISH |
| 4,4'-DDT | 1.60e-05 | LCV_DAPHNIDS |
| | 7.30e-04 | LCV_FISH |
| | 3.50e-04 | LTV_FISH |
| | 1.10e-03 | NAWQ_ACUTE |
| | 1.00e-06 | NAWQ_CHRONIC |
| | 8.00e-06 | SS_EC20 |
| 4-METHYL-2-PENTANONE | 1.59e+00 | EC20_POPULATION |
| | 7.74e+01 | LCV_FISH |
| ACENAPHTHENE | 6.65e+00 | LCV_DAPHNIDS |
| | 4.13e-01 | LCV_FISH |
| | 1.97e-01 | LTV_FISH |
| ACETONE | 2.37e+01 | EC20_POPULATION |
| | 3.11e+03 | LCV_DAPHNIDS |
| | 5.08e+02 | LCV_FISH |
| | 1.62e+02 | LTV_FISH |
| AMMONIA | 6.30e-01 | LCV_DAPHNIDS |
| | 1.70e-03 | LCV_FISH |
| ANTHRACENE | 2.10e-03 | LCV_DAPHNIDS |
| | 9.00e-05 | LCV_FISH |
| | 8.20e-03 | LTV_DAPHNIDS |
| | 3.50e-04 | LTV_FISH |
| AROCLOR-1221 | 1.00e-02 | EC20_POPULATION |
| | 6.00e-02 | LCV_FISH |
| | 8.00e-02 | LTV_FISH |
| AROCLOR-1232 | 1.60e-02 | EC20_POPULATION |
| | 1.24e-01 | LCV_FISH |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| | 1.48e-01 | LTV_FISH |
| AROCLOR-1242 | 1.58e-03 | EC20_POPULATION |
| | 9.00e-03 | LCV_FISH |
| | 2.90e-03 | LTV_FISH |
| AROCLOR-1248 | 1.26e-03 | EC20_POPULATION |
| | 4.30e-03 | LCV_DAPHNIDS |
| | 2.00e-04 | LCV_FISH |
| | 2.50e-03 | LTV_DAPHNIDS |
| | 4.00e-04 | LTV_FISH |
| AROCLOR-1254 | 6.30e-04 | EC20_POPULATION |
| | 2.10e-03 | LCV_DAPHNIDS |
| | 1.00e-03 | LCV_FISH |
| | 1.20e-03 | LTV_DAPHNIDS |
| | 5.20e-04 | LTV_FISH |
| AROCLOR-1260 | 3.16e-01 | EC20_POPULATION |
| | 2.30e-03 | LCV_FISH |
| | 2.10e-03 | LTV_FISH |
| BENZENE | 2.29e-01 | EC20_POPULATION |
| | 9.80e+01 | LCV_DAPHNIDS |
| | 2.99e-01 | LCV_FISH |
| | 2.10e-02 | LTV_FISH |
| BENZIDENE | 6.80e-02 | EC20_POPULATION |
| | 1.34e-01 | LCV_FISH |
| | 1.58e-01 | LTV_FISH |
| BENZO(A)ANTHRACENE | 6.50e-04 | LCV_DAPHNIDS |
| BENZO(A)PYRENE | 3.00e-04 | LCV_DAPHNIDS |
| | 2.99e-03 | LTV_FISH |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| BENZOIC ACID | 1.26e+00 | EC20_POPULATION |
| | 1.30e+01 | LCV_FISH |
| | 7.41e+00 | LTV_FISH |
| BENZYL ALCOHOL | 3.75e-01 | EC20_POPULATION |
| | 5.89e-01 | LCV_FISH |
| | 5.50e-01 | LTV_FISH |
| BHC (OTHER) | 9.50e-02 | LCV_DAPHNIDS |
| BIS(2-ETHYLHEXYL)PHTHALATE | 5.00e-02 | EC20_POPULATION |
| | 3.00e-03 | LCV_DAPHNIDS |
| | 8.40e-03 | LCV_FISH |
| | 3.00e-03 | LTV_DAPHNIDS |
| | 5.40e-02 | LTV_FISH |
| CARBON DISULFIDE | 1.00e+00 | EC20_POPULATION |
| | 2.44e-01 | LCV_DAPHNIDS |
| | 9.54e+00 | LCV_FISH |
| | 5.72e+00 | LTV_FISH |
| CARBON TETRACHLORIDE | 2.24e-01 | EC20_POPULATION |
| | 5.58e+00 | LCV_DAPHNIDS |
| | 3.40e+00 | LCV_FISH |
| | 6.50e-02 | LTV_FISH |
| CHLORDANE | 7.10e-04 | EC20_POPULATION |
| | 1.60e-02 | LCV_DAPHNIDS |
| | 1.60e-03 | LCV_FISH |
| | 1.21e-02 | LTV_DAPHNIDS |
| | 2.50e-04 | LTV_FISH |
| | 2.40e-03 | NAWQ_ACUTE |
| | 4.30e-06 | NAWQ_CHRONIC |
| | 1.70e-04 | NAWQ_FINAL CHRONIC VALUE |
| | 5.00e-04 | SS_EC20 |
| CHLOROBENZENE | 1.65e-01 | EC20_POPULATION |
| | 1.50e+01 | LCV_DAPHNIDS |
| | 1.20e+00 | LCV_FISH |
| | 1.00e+00 | LTV_FISH |
| CHLOROFORM | 5.62e-01 | EC20_POPULATION |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| | 4.48e+00 | LCV_DAPHNIDS |
| | 1.24e+00 | LCV_FISH |
| | 8.40e+00 | LTV_FISH |
| DECANE | 7.87e+00 | LCV_DAPHNIDS |
| DI-N-BUTYLPHTHALATE | 2.51e-01 | EC20_POPULATION |
| | 7.17e-01 | LCV_DAPHNIDS |
| | 7.17e-01 | LCV_FISH |
| | 5.00e-01 | LTV_DAPHNIDS |
| | 2.70e-01 | LTV_FISH |
| DI-N-OCTYLPHTHALATE | 2.00e+00 | EC20_POPULATION |
| | 5.70e-01 | LCV_DAPHNIDS |
| | 5.70e+00 | LCV_FISH |
| | 3.10e-01 | LTV_DAPHNIDS |
| | 1.00e-01 | LTV_FISH |
| DIBENZOFURAN | 1.00e+00 | LCV_DAPHNIDS |
| ETHYL BENZENE | 3.98e-01 | EC20_POPULATION |
| | 1.29e+01 | LCV_DAPHNIDS |
| | 4.40e-01 | LCV_FISH |
| FLUORANTHENE | 3.20e-02 | EC20_POPULATION |
| | 6.58e+01 | LCV_DAPHNIDS |
| | 2.20e-01 | LCV_FISH |
| FLUORINE | 1.08e+00 | EC20_POPULATION |
| | 4.40e+00 | LCV_DAPHNIDS |
| | 8.78e+00 | LCV_FISH |
| | 3.71e+00 | LTV_DAPHNIDS |
| | 5.34e+00 | LTV_FISH |
| GAMMA-BHC (LINDANE) | 1.45e-02 | LCV_DAPHNIDS |
| | 1.46e-02 | LCV_FISH |
| | 1.10e-02 | LTV_DAPHNIDS |
| | 1.10e-03 | LTV_FISH |
| | 2.00e-03 | NAWQ_ACUTE |
| | 8.00e-05 | NAWQ_CHRONIC |
| | 1.10e-04 | SS_EC20 |
| HEPTACHLOR | 1.00e-04 | EC20_POPULATION |
| | 3.18e-03 | LCV_DAPHNIDS |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| | 1.26e-03 | LCV_FISH |
| | 8.60e-04 | LTV_FISH |
| | 5.20e-04 | NAWQ_ACUTE |
| | 3.80e-06 | NAWQ_CHRONIC |
| | 4.00e-06 | SS_EC20 |
| HEXANE | 6.57e+01 | LCV_FISH |
| | 2.90e+01 | LTV_FISH |
| METHYLENE CHLORIDE | 1.26e+00 | EC20_POPULATION |
| | 4.27e+01 | LCV_DAPHNIDS |
| | 1.08e+02 | LCV_FISH |
| | 4.10e-01 | LTV_FISH |
| N-NITROSODIPHENYLAMINE | 4.00e-02 | EC20_POPULATION |
| | 1.04e+00 | LCV_DAPHNIDS |
| | 3.32e-01 | LCV_FISH |
| | 3.39e-01 | LTV_FISH |
| NAPHTHALENE | 1.00e+00 | EC20_POPULATION |
| | 1.16e+00 | LCV_DAPHNIDS |
| | 6.20e-01 | LCV_FISH |
| | 6.00e-01 | LTV_DAPHNIDS |
| | 4.50e-01 | LTV_FISH |
| 4-NITROPHENOLS | 6.00e-02 | EC20_POPULATION |
| | 7.10e+00 | LCV_DAPHNIDS |
| | 4.81e-01 | LCV_FISH |
| | 5.00e+00 | LTV_DAPHNIDS |
| | 4.64e-01 | LTV_FISH |
| NITROPHENOLS TOTAL | 1.50e-01 | LCV_OTHER |
| PHENANTHRENE | 2.00e-01 | LCV_DAPHNIDS |
| | 1.10e-01 | LTV_DAPHNIDS |
| PHENOL | 4.47e+00 | EC20_POPULATION |
| | 2.01e+00 | LCV_DAPHNIDS |
| | 1.37e+00 | LCV_FISH |
| | 2.30e-01 | LTV_FISH |
| TETRACHLOROETHENE | 5.00e-02 | EC20_POPULATION |
| | 7.50e-01 | LCV_DAPHNIDS |
| | 8.40e-01 | LCV_FISH |

Table 6.27 (continued)

| Analysis | Benchmark value (mg/L) | Benchmark name |
|---|---|---|
| | 5.10e-01 | LTV_DAPHNIDS |
| | 5.00e-01 | LTV_FISH |
| TOLUENE | 2.00e-01 | EC20_POPULATION |
| | 2.52e+01 | LCV_DAPHNIDS |
| | 1.27e+00 | LCV_FISH |
| | 2.60e-02 | LTV_FISH |
| TRICHLOROETHENE | 2.32e-01 | EC20_POPULATION |
| | 7.26e+00 | LCV_DAPHNIDS |
| | 1.11e+01 | LCV_FISH |
| | 5.76e+00 | LTV_FISH |
| VINYL ACETATE | 1.08e-01 | EC20_POPULATION |
| | 8.10e-01 | LCV_FISH |
| | 7.18e-01 | LTV_FISH |
| VINYL CHLORIDE | 2.89e+01 | LCV_FISH |
| | 1.45e+01 | LTV_FISH |
| XYLENE | 6.23e+01 | LCV_FISH |
| | 2.68e+00 | LTV_FISH |

[a]Benchmarks, which are described in Sect. 6.3, are from Suter et al., 1992b: LCV, for daphnids or fish, is the lowest chronic values reported in the literature; LTV, for daphnids or fish, is the lowest Test $EC_{20}$ value; SS_$EC_{20}$ is the sensative species $EC_{20}$.

**Table 6.28 Benchmarks for toxicity of polar organic compounds to sediment-associated biota**

| Analyte | Benchmark | |
|---|---|---|
| | Value (mg/kg) | Name[a] |
| Benzoic acid | 6.50e-04 | WA. SQS |
| Benzyl alcohol | 5.70e-05 | WA. SQS |
| Pentachlorophenol | 3.60e-04 | WA. SQS |
| Phenol | 4.20e-04 | WA. SQS |
| 2-Methylphenol | 6.30e-05 | WA. SQS |
| 4-Methylphenol | 6.70e-04 | WA. SQS |
| 2,4-Dimethylphenol | 2.90e-05 | WA. SQS |

[a]The Washington State Sediment Quality Standards are listed in Hull and Suter (1993) and discussed in Sect. 6.3.

**Table 6.29. Benchmarks for toxicity of metals to sediment-associated biota**

| Analyte | Lower | | Upper | |
|---|---|---|---|---|
| | Value (mg/kg) | Name[a] | Value (mg/kg) | Name[a] |
| Antimony | 2.00e+00 | ER-L | 2.50e+01 | AET |
| Arsenic | 6.00e+00 | MOE | 3.30e+01 | ER-L |
| Barium | 2.00e+01 | EPA-V | 6.00e+01 | EPA-V |
| Cadmium | 6.00e-01 | MOE | 5.00e+00 | ER-L |
| Chromium | 8.00e+01 | ER-L | 1.45e+02 | ER-M |
| Copper | 7.00e+01 | ER-L | 3.00e+01 | AET |
| Iron | 3.00e+04 | MOE | 4.00e+04 | MOE |
| Lead | 3.50e+01 | ER-L | 3.00e+02 | AET |
| Manganese | 4.60e+02 | MOE-L | 1.11e+03 | MOE |
| Mercury | 1.50e-01 | ER-L | 1.00e+00 | AET |
| Nickel | 3.00e+01 | ER-L | 5.00e+01 | ER-M |
| Silver | 1.00e+00 | ER-L | 1.70e+00 | AET |
| Zinc | 1.20e+02 | ER-L | 2.60e+02 | AET |

[a]Benchmarks are from Hull and Suter (1993) and are described in Sect. 6.3.

**Table 6.30. Benchmarks for toxicity of organic compounds to sediment-associated biota[a]**

| Analyte | Benchmark value (mg/kg)[a] | | Log $K_{ow}$[a] | Benchmark name[b] |
|---|---|---|---|---|
| | FCAP | UMB | | |
| 1,1,1-TRICHLOROETHANE | 2.01e+00 | 2.15e+00 | 2.48 | EC20_POPULATION |
| | 1.42e+01 | 1.52e+01 | 2.48 | LCV_DAPHNIDS |
| | 2.80e+01 | 3.00e+01 | 2.48 | LCV_FISH |
| | 1.04e+01 | 1.11e+01 | 2.48 | LTV_DAPHNIDS |
| | 1.97e+01 | 2.11e+01 | 2.48 | LTV_FISH |
| 1,1,2,2-TETRACHLOROETHANE | 1.03e+01 | 1.10e+01 | 2.39 | EC20_POPULATION |
| | 6.44e+01 | 6.90e+01 | 2.39 | LCV_DAPHNIDS |
| | 1.56e+01 | 1.67e+01 | 2.39 | LCV_FISH |
| | 2.73e+00 | 2.93e+00 | 2.39 | LTV_DAPHNIDS |
| | 9.11e+00 | 9.76e+00 | 2.39 | LTV_FISH |
| 1,1,2-TRICHLOROETHANE | 1.27e+02 | 1.36e+02 | 2.48 | EC20_POPULATION |
| | 1.47e+02 | 1.58e+02 | 2.48 | LCV_DAPHNIDS |
| | 7.52e+01 | 8.06e+01 | 2.48 | LCV_FISH |
| | 1.04e+02 | 1.11e+02 | 2.48 | LTV_DAPHNIDS |
| | 1.18e+02 | 1.27e+02 | 2.48 | LTV_FISH |
| 1,1-DICHLOROETHANE | 2.59e+00 | 2.78e+00 | 1.79 | EC20_POPULATION |
| | 2.40e+01 | 2.57e+01 | 1.79 | LCV_FISH |
| | 1.34e+01 | 1.44e+01 | 1.79 | LTV_FISH |
| | 3.58e-01 | 3.83e-01 | 1.48 | EC20_POPULATION |
| | 3.78e+00 | 4.05e+00 | 1.48 | LCV_DAPHNIDS |
| | 2.24e+00 | 2.40e+00 | 1.48 | LCV_FISH |
| 1,2-DICHLOROETHANE | 9.85e-01 | 1.06e+00 | 1.47 | EC20_POPULATION |
| | 1.19e+01 | 1.27e+01 | 1.47 | LCV_DAPHNIDS |
| | 3.23e+01 | 3.47e+01 | 1.47 | LCV_FISH |
| | 8.60e+00 | 9.22e+00 | 1.47 | LTV_DAPHNIDS |
| | 2.27e+01 | 2.43e+01 | 1.47 | LTV_FISH |
| 4,4'-DDD | 1.69e+01 | 1.81e+01 | 6.02 | EC20_POPULATION |
| | 4.69e+01 | 5.03e+01 | 6.02 | LCV_FISH |
| | 1.11e+02 | 1.19e+02 | 6.02 | LTV_FISH |
| 4,4'-DDT | 2.08e-01 | 2.23e-01 | 5.69 | LCV_DAPHNIDS |
| | 9.47e+00 | 1.02e+01 | 5.69 | LCV_FISH |
| | 4.54e+00 | 4.87e+00 | 5.69 | LTV_FISH |
| | 1.43e+01 | 1.53e+01 | 5.69 | NAWQ_ACUTE |
| | 1.30e-02 | 1.39e-02 | 5.69 | NAWQ_CHRONIC |
| | 1.04e-01 | 1.11e-01 | 5.69 | SS_EC20 |

**Table 6.30 (continued)**

| Analyte | Benchmark value (mg/kg)[a] | | Log $K_{ow}$[a] | Benchmark name[b] |
|---|---|---|---|---|
| | FCAP | UMB | | |
| ACENAPHTHENE | 1.46e+03 | 1.57e+03 | 3.92 | LCV_DAPHNIDS |
| | 9.10e+01 | 9.76e+01 | 3.92 | LCV_FISH |
| | 4.34e+01 | 4.65e+01 | 3.92 | LTV_FISH |
| ANTHRACENE | 1.57e+00 | 1.68e+00 | 4.45 | LCV_DAPHNIDS |
| | 6.72e-02 | 7.20e-02 | 4.45 | LCV_FISH |
| | 6.12e+00 | 6.56e+00 | 4.45 | LTV_DAPHNIDS |
| | 2.61e-01 | 2.80e-01 | 4.45 | LTV_FISH |
| AROCLOR-1221 | 3.66e+03 | 3.92e+03 | 7.14 | EC20_POPULATION |
| | 2.19e+04 | 2.35e+04 | 7.14 | LCV_FISH |
| | 2.93e+04 | 3.14e+04 | 7.14 | LTV_FISH |
| AROCLOR-1232 | 5.85e+03 | 6.27e+03 | 7.14 | EC20_POPULATION |
| | 4.54e+04 | 4.86e+04 | 7.14 | LCV_FISH |
| | 5.41e+04 | 5.80e+04 | 7.14 | LTV_FISH |
| AROCLOR-1242 | 5.78e+02 | 6.19e+02 | 7.14 | EC20_POPULATION |
| | 3.29e+03 | 3.53e+03 | 7.14 | LCV_FISH |
| | 1.06e+03 | 1.14e+03 | 7.14 | LTV_FISH |
| AROCLOR-1248 | 4.61e+02 | 4.94e+02 | 7.14 | EC20_POPULATION |
| | 1.57e+03 | 1.69e+03 | 7.14 | LCV_DAPHNIDS |
| | 7.32e+01 | 7.84e+01 | 7.14 | LCV_FISH |
| | 9.15e+02 | 9.80e+02 | 7.14 | LTV_DAPHNIDS |
| | 1.46e+02 | 1.57e+02 | 7.14 | LTV_FISH |
| AROCLOR-1254 | 1.79e+01 | 1.92e+01 | 6.03 | EC20_POPULATION |
| | 5.96e+01 | 6.39e+01 | 6.03 | LCV_DAPHNIDS |
| | 2.84e+01 | 3.04e+01 | 6.03 | LCV_FISH |
| | 3.41e+01 | 3.65e+01 | 6.03 | LTV_DAPHNIDS |
| | 1.48e+01 | 1.58e+01 | 6.03 | LTV_FISH |
| AROCLOR-1260 | 1.16e+05 | 1.24e+05 | 7.14 | EC20_POPULATION |
| | 8.41e+02 | 9.02e+02 | 7.14 | LCV_FISH |
| | 7.68e+02 | 8.23e+02 | 7.14 | LTV_FISH |
| BENZENE | 8.57e-01 | 9.19e-01 | 2.15 | EC20_POPULATION |
| | 3.67e+02 | 3.93e+02 | 2.15 | LCV_DAPHNIDS |
| | 1.12e+00 | 1.20e+00 | 2.15 | LCV_FISH |
| | 7.86e-02 | 8.42e-02 | 2.15 | LTV_FISH |
| BENZIDENE | 3.94e-02 | 4.23e-02 | 1.34 | EC20_POPULATION |
| | 7.77e-02 | 8.33e-02 | 1.34 | LCV_FISH |

**Table 6.30 (continued)**

| Analyte | Benchmark value (mg/kg)[a] | | Log $K_{ow}$[a] | Benchmark name[b] |
|---|---|---|---|---|
| | FCAP | UMB | | |
| | 9.16e-02 | 9.82e-02 | 1.34 | LTV_FISH |
| BENZO(A)ANTHRACENE | 7.02e+00 | 7.52e+00 | 5.61 | LCV_DAPHNIDS |
| BENZO(A)PYRENE | 8.72e+00 | 9.34e+00 | 6.04 | LCV_DAPHNIDS |
| | 8.69e+01 | 9.31e+01 | 6.04 | LTV_FISH |
| BIS(2-ETHYLHEXYL)PHTHALATE | 1.03e+02 | 1.10e+02 | 4.89 | EC20_POPULATION |
| | 6.17e+00 | 6.61e+00 | 4.89 | LCV_DAPHNIDS |
| | 1.73e+01 | 1.85e+01 | 4.89 | LCV_FISH |
| | 6.17e+00 | 6.61e+00 | 4.89 | LTV_DAPHNIDS |
| | 1.11e+02 | 1.19e+02 | 4.89 | LTV_FISH |
| CARBON TETRACHLORIDE | 4.01e+00 | 4.30e+00 | 2.83 | EC20_POPULATION |
| | 1.00e+02 | 1.07e+02 | 2.83 | LCV_DAPHNIDS |
| | 6.09e+01 | 6.53e+01 | 2.83 | LCV_FISH |
| | 1.16e+00 | 1.25e+00 | 2.83 | LTV_FISH |
| CHLORDANE | 1.13e-02 | 1.22e-02 | 2.78 | EC20_POPULATION |
| | 2.55e-01 | 2.74e-01 | 2.78 | LCV_DAPHNIDS |
| | 2.55e-02 | 2.74e-02 | 2.78 | LCV_FISH |
| | 1.93e-01 | 2.07e-01 | 2.78 | LTV_DAPHNIDS |
| | 3.99e-03 | 4.28e-03 | 2.78 | LTV_FISH |
| | 3.83e-02 | 4.11e-02 | 2.78 | NAWQ_ACUTE |
| | 6.87e-05 | 7.36e-05 | 2.78 | NAWQ_CHRONIC |
| | 2.71e-03 | 2.91e-03 | 2.78 | NAWQ_FINAL CHRONIC VALUE |
| CHLORDANE | 7.98e-03 | 8.56e-03 | 2.78 | SS_EC20 |
| CHLOROBENZENE | 3.03e+00 | 3.24e+00 | 2.84 | EC20_POPULATION |
| | 2.76e+02 | 2.96e+02 | 2.84 | LCV_DAPHNIDS |
| | 2.21e+01 | 2.36e+01 | 2.84 | LCV_FISH |
| | 1.84e+01 | 1.97e+01 | 2.84 | LTV_FISH |
| CHLOROFORM | 1.30e+00 | 1.39e+00 | 1.94 | EC20_POPULATION |
| | 1.03e+01 | 1.11e+01 | 1.94 | LCV_DAPHNIDS |
| | 2.86e+00 | 3.07e+00 | 1.94 | LCV_FISH |
| | 1.94e+01 | 2.08e+01 | 1.94 | LTV_FISH |
| DI-N-BUTYLPHTHALATE | 1.05e+03 | 1.13e+03 | 5.2 | EC20_POPULATION |
| | 3.01e+03 | 3.23e+03 | 5.2 | LCV_DAPHNIDS |
| | 3.01e+03 | 3.23e+03 | 5.2 | LCV_FISH |
| | 2.10e+03 | 2.25e+03 | 5.2 | LTV_DAPHNIDS |

Table 6.30 (continued)

| Analyte | Benchmark value (mg/kg)[a] | | Log $K_{ow}$[a] | Benchmark name[b] |
|---|---|---|---|---|
| | FCAP | UMB | | |
| | 1.13e+03 | 1.22e+03 | 5.2 | LTV_FISH |
| DI-N-OCTYLPHTHALATE | 1.88e+04 | 2.01e+04 | 5.55 | EC20_POPULATION |
| | 5.36e+03 | 5.74e+03 | 5.55 | LCV_DAPHNIDS |
| | 5.36e+04 | 5.74e+04 | 5.55 | LCV_FISH |
| | 2.91e+03 | 3.12e+03 | 5.55 | LTV_DAPHNIDS |
| | 9.40e+02 | 1.01e+03 | 5.55 | LTV_FISH |
| DIBENZOFURAN | 3.50e+02 | 3.76e+02 | 4.12 | LCV_DAPHNIDS |
| ETHYL BENZENE | 1.49e+01 | 1.60e+01 | 3.15 | EC20_POPULATION |
| | 4.84e+02 | 5.18e+02 | 3.15 | LCV_DAPHNIDS |
| | 1.65e+01 | 1.77e+01 | 3.15 | LCV_FISH |
| FLUORANTHENE | 6.74e+01 | 7.22e+01 | 4.9 | EC20_POPULATION |
| | 1.38e+05 | 1.48e+05 | 4.9 | LCV_DAPHNIDS |
| | 4.63e+02 | 4.96e+02 | 4.9 | LCV_FISH |
| GAMMA-BHC (LINDANE) | 2.02e+00 | 2.16e+00 | 3.72 | LCV_DAPHNIDS |
| | 2.03e+00 | 2.18e+00 | 3.72 | LCV_FISH |
| | 1.53e+00 | 1.64e+00 | 3.72 | LTV_DAPHNIDS |
| | 1.53e-01 | 1.64e-01 | 3.72 | LTV_FISH |
| | 2.78e-01 | 2.98e-01 | 3.72 | NAWQ_ACUTE |
| | 1.11e-02 | 1.19e-02 | 3.72 | NAWQ_CHRONIC |
| | 1.53e-02 | 1.64e-02 | 3.72 | SS_EC20 |
| HEPTACHLOR | 7.30e-01 | 7.82e-01 | 5.44 | EC20_POPULATION |
| | 2.32e+01 | 2.49e+01 | 5.44 | LCV_DAPHNIDS |
| | 9.20e+00 | 9.86e+00 | 5.44 | LCV_FISH |
| | 6.28e+00 | 6.73e+00 | 5.44 | LTV_FISH |
| | 3.80e+00 | 4.07e+00 | 5.44 | NAWQ_ACUTE |
| | 2.77e-02 | 2.97e-02 | 5.44 | NAWQ_CHRONIC |
| | 2.92e-02 | 3.13e-02 | 5.44 | SS_EC20 |
| METHYLENE CHLORIDE | 5.93e-01 | 6.36e-01 | 1.25 | EC20_POPULATION |
| | 2.01e+01 | 2.15e+01 | 1.25 | LCV_DAPHNIDS |
| | 5.09e+01 | 5.45e+01 | 1.25 | LCV_FISH |
| | 1.93e-01 | 2.07e-01 | 1.25 | LTV_FISH |
| NAPHTHALENE | 1.03e+02 | 1.10e+02 | 3.59 | EC20_POPULATION |
| | 1.20e+02 | 1.28e+02 | 3.59 | LCV_DAPHNIDS |
| | 6.39e+01 | 6.85e+01 | 3.59 | LCV_FISH |
| | 6.19e+01 | 6.63e+01 | 3.59 | LTV_DAPHNIDS |

**Table 6.30 (continued)**

| Analyte | Benchmark value (mg/kg)[a] | | Log $K_{ow}$[a] | Benchmark name[b] |
|---|---|---|---|---|
| | FCAP | UMB | | |
| | 4.64e+01 | 4.97e+01 | 3.59 | LTV_FISH |
| TETRACHLOROETHENE | 3.33e+00 | 3.57e+00 | 3.4 | EC20_POPULATION |
| | 4.99e+01 | 5.35e+01 | 3.4 | LCV_DAPHNIDS |
| | 5.59e+01 | 5.99e+01 | 3.4 | LCV_FISH |
| | 3.39e+01 | 3.64e+01 | 3.4 | LTV_DAPHNIDS |
| | 3.33e+01 | 3.57e+01 | 3.4 | LTV_FISH |
| TOLUENE | 2.02e+00 | 2.16e+00 | 2.58 | EC20_POPULATION |
| | 2.54e+02 | 2.72e+02 | 2.58 | LCV_DAPHNIDS |
| | 1.28e+01 | 1.37e+01 | 2.58 | LCV_FISH |
| | 2.62e-01 | 2.81e-01 | 2.58 | LTV_FISH |
| TRICHLOROETHENE | 1.20e+00 | 1.28e+00 | 2.29 | EC20_POPULATION |
| | 3.75e+01 | 4.02e+01 | 2.29 | LCV_DAPHNIDS |
| | 5.71e+01 | 6.12e+01 | 2.29 | LCV_FISH |
| | 2.98e+01 | 3.19e+01 | 2.29 | LTV_FISH |
| VINYL CHLORIDE | 2.31e+01 | 2.48e+01 | 1.48 | LCV_FISH |
| | 1.16e+01 | 1.25e+01 | 1.48 | LTV_FISH |

[a]Benchmarks calculated using the equilibrium partioning formula, and the $K_{ow}$s, from Hull and Suter (1993), as described in Sect. 6.3. The percent of TOCs used were 2.65 and 2.84 for FCAP and UMB, respectively.

[b]Benchmarks are from Suter et al. (1992b) and are described in Sect. 6.3.

**Table 6.31. Benchmarks for toxicity of nonpolar organic compounds to sediment-associated biota**

| Analyte | Benchmark | |
|---|---|---|
| | Value (mg/kg) | Name[a] |
| 2-Methylnaphthalene | 6.50e-02 | ER-L |
| Acenaphthene | 1.50e-01 | ER-L |
| Anthracene | 8.50e-02 | ER-L |
| Benzo(à)Anthracene | 2.30e-01 | ER-L |
| Chrysene | 4.00e-01 | ER-L |
| Dibenz(a,h)Anthracene | 6.00e-02 | ER-L |
| Fluoranthene | 6.00e-01 | ER-L |
| Fluorene | 3.50e-02 | ER-L |
| Naphthalene | 3.40e-01 | ER-L |
| Phenanthrene | 2.25e-01 | ER-L |
| Pyrene | 3.50e-01 | ER-L |

[a]Values represent ER-Ls,( Effects Range—Low) and are from Long and Morgan (1991).

**Table 6.32. Screening benchmark concentrations for the phytotoxicity of chemicals in soil and soil solution**

| Chemical | Soil (mg/kg)[a] | Solution (mg/L)[a] |
|---|---|---|
| Aluminum | 50 | 0.5 |
| Antimony | 5[b] | |
| Arsenic | 10 | 0.01 |
| Barium | 500 | 500d |
| Beryllium | 10[b] | 0.5 |
| Bismuth | | 27[c] |
| Boron | 0.5 | 1[e] |
| Bromine | 10[b] | 15[d] |
| Cadmium | 3 | 0.05 |
| Chromium | 2 | 0.05 |
| Cobalt | 25[f] | 0.06 |
| Copper | 60[g] | 0.03 |
| Fluorine | 200[b] | 5[c] |
| Iodine | 4 | 0.5 |
| Iron | | 10[d] |
| Lead | 50 | 0.1 |
| Lithium | 2 | 3 |
| Manganese | 500 | 1 |
| Methyl mercury | | 0.002 |
| Mercury | 0.3[b] | 0.01 |
| Molybdenum | 2[f] | 0.5[d] |
| Nickel | 25 | 0.1 |
| Selenium | 1 | 1 |
| Silver | 2[f] | 0.16 |
| Tellurium | | 2 |
| Thallium | 1[b] | 1[h] |
| Tin | 50[b] | 40[c] |
| Titanium | | 0.07 |
| Vanadium | 2.5[i] | 0.51 |
| Zinc | 25 | 0.4 |
| 2,4 Dinitrophenol | 20 | |
| Di-n-butyl phthalate | 200 | |

Table 6.32 (continued)

| Chemical | Soil (mg/kg)[a] | Solution (mg/L)[a] |
|---|---|---|
| PCBs | 40 | |
| Toluene | 200 | |

[a]Letters after concentrations denote values said in secondary sources to represent phytotoxicity thresholds. See Sect. 6.3.1.3 for method of benchmark derivation;
[b]Kloke 1979;
[c]Scharrer 1955;
[d]Chapman 1966;
[e]Bowen 1979;
[f]Linzon 1978;
[g]Kovalskiy 1974;
[h]Stiles 1958;
[i]EPA 1975.

**Table 6.33. Screening benchmark concentrations for the toxicity of chemicals in soil and soil/litter microcosms to earthworms[a]**

| Chemical | Soil (mg/kg) |
|---|---|
| Cadmium | 20 |
| Chromium | 2 |
| Copper | 100 |
| Lead | 500 |
| Methyl mercury | 10 |
| Mercury | 0.5 |
| Nickel | 200 |
| Zinc | 660[b] |
| Carbaryl | 20[c] |
| Chloroacetamide | 10 |
| 3,4-dichloroaniline | 100 |
| 1,2-dichloropropane | 3880 |
| Dieldrin | 25 |
| Dimethylphthalate | 1060 |
| Fluorene | 170 |
| Ninitrosodiphenylamine | 110 |
| 4-nitrophenol | 40 |
| Pentachlorophenol | 15 |
| Phenol | 190 |
| 2,4,6-trichlorophenol | 60 |
| 1,2,4-trichlorobenzene | 225 |

[a]See Sect. 6.3.1.4 for method of benchmark derivation;
[b]Benchmark based on $LC_{50}$ value;
[c]All organic compound benchmarks are based on $LC_{50}$ values.

**Table 6.34. Screening benchmark concentrations for the toxicity of chemicals in vegetation[a] to invertebrates other than earthworms**

| Chemical | Vegetation (mg/kg) |
|---|---|
| Cadmium | 3.5 |
| Copper | 100 |
| Fluoride | 320 |
| Iron | 7530 |
| Lead | 1000 |
| Mercury | 1000 |
| Zinc | 10 |

[a]See Sect. 6.3.1.4 for method of benchmark derivation.

**Table 6.35. Screening benchmark concentrations for the toxicity of chemicals in soil[a] to soil microbes**

| Chemical | Soil (mg/kg) |
|---|---|
| Aluminum | 135 |
| Cadmium | 3[b] |
| Chromium | 4[b] |
| Cobalt | 1360 |
| Copper | 6[b] |
| Fluorine | 30 |
| Iron | 280 |
| Lanthanum | 55 |
| Lead | 35[b] |
| Lithium | 15 |
| Manganese | 100 |
| Mercury | 0.1 |
| Molybdenum | 480 |
| Nickel | 250 |
| Selenium | 485 |
| Silver | 540 |
| Tin | 595 |
| Vanadium | 20 |
| Zinc | 15[b] |
| BHC | 10 |

[a]See Sect. 6.3.1.5 for method of benchmark derivation.
[b]Benchmark based on $LC_{50}$ value.

Table 6.36. Estimated NOAELs for mammalian wildlife at Chestnut Ridge OU 2

| Contaminant | Experimental information | | | | | Estimated NOAELs for mammals (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Form | Test species | NOAEL (mg/kg per day) and duration | Endpoint | Citation | White-footed mouse | Short-tailed shrew | Red fox | White-tailed deer |
| Aluminum | | mouse | 1.93 3 gen. | reproduction/ growth | Ondreicka et al. 1966 | 2.2 | 2.42 | 0.33 | 0.16 |
| Arsenic | $As^{+3}$ | mouse | 0.095[2] 3 gen. | reproduction | Schroeder and Mitchner 1971 | 0.1086 | 0.119 | 0.0165 | 0.0077 |
| Barium | | rat | 0.51 16 months | cardiovasc. | Perry et al. 1983 | 1.311 | 1.44 | 0.199 | 0.093 |
| Beryllium | | rat | 0.54 1126 days | bone/ wt. loss | Schroeder and Mitchner 1975a | 1.39 | 1.52 | 0.211 | 0.098 |
| Cadmium | | mouse | 0.19[2] 2 gen. | reproduction | Schroeder and Mitchner 1971 | 0.217 | 0.24 | 0.033 | 0.015 |
| Chromium | $Cr^{+6}$ | rat | 2.4 1 year | growth, food consumption | Mackenzie et al. 1958 | 6.17 | 6.79 | 0.094 | 0.44 |
| Copper | sulfate | mink | 10 1 year | reproduction | Aulerich et al. 1982 | 41.6 | 45.7 | 6.32 | 2.96 |
| Fluoride | | mouse | 1.9 lifetime | longevity survival | Schroeder et al. 1968 | 2.17 | 2.38 | 0.33 | 0.155 |
| Lead | acetate | rat | 5 3 gen. | reproduction | Azar et al. 1973 | 2 | 2.2 | 0.31 | 0.14 |
| Manganese | sulfate | pig | 4.06 120 days | reproduction | Grummer et al. 1955 | 67.56 | 74.29 | 10.28 | 4.81 |
| Mercury | methyl | rat mink | 0.024-3 gen 0.007[1]-93 days | reproduction mortality | Verschuuren et al. 1976c Wobeser et al. 1976 | 0.06172 | 0.06786 | 0.0044 | 0.0044 |

**Table 6.36 (continued)**

| Contaminant | Experimental information | | | | | Estimated NOAELs for mammals (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Form | Test species | NOAEL (mg/kg per day) and duration | Endpoint | Citation | White-footed mouse | Short-tailed shrew | Red fox | White-tailed deer |
| Nickel | sulfate | rat | 24.15<br>3 gen. | reproduction | Ambrose et al. 1976 | 62.1 | 68.28 | 9.46 | 4.42 |
| Niobium | | mouse | 1.16<br>lifetime | longevity survival | Schroeder et al. 1968 | 1.32 | 1.46 | 0.2 | 0.094 |
| Nitrate | | human | 1.6<br>8 months | methemo-globinemia | Bosch et al. 1950 | 23.64 | 25.99 | 3.6 | 1.68 |
| Selenium | selenate | mouse | 0.057[2]<br>3 gen | reproduction | Schroeder and Mitchner 1971 | 0.065 | 0.07 | 0.0099 | 0.0046 |
| Thallium | sulfate | rat | 0.011[1]<br>60 days | reproduction | Formigli et al. 1986 | 0.028 | 0.031 | 0.0043 | 0.002 |
| Vanadium | | mouse | 0.95<br>lifetime | longevity survival | Schroeder and Balassa 1967 | 1.08 | 1.19 | 0.165 | 0.07 |
| Zinc | carbonate | rat | 9.7[1]<br>37 days | reproduction | Kinnamon 1963 | 24.9 | 27.42 | 3.8 | 1.77 |
| bis(2-Ethylhexyl)Phthalate | | mouse | 1.41[1]<br>105 days | reproduction | Lamb et al. 1987 | 1.61 | 1.77 | 0.25 | 0.11 |
| Chloroform | | rat<br>dog | 9[2] - 78 weeks<br>1.29[2] -7.5 years | kidney<br>liver | Reuber 1979<br>Heywood et al. 1979 | 23.14 | 25.44 | 1.65 | 0.77 |

**Table 6.36 (continued)**

| Contaminant | Experimental information | | | | | Estimated NOAELs for mammals (mg/kg per day) | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Form | Test species | NOAEL (mg/kg per day) and duration | Endpoint | Citation | White-footed mouse | Short-tailed shrew | Red fox | White-tailed deer |
| Di-N-butylphthalate | | mouse | 42.3[1] 105 days | reproduction | Lamb et al. 1987 | 48.36 | 53.17 | 7.36 | 3.44 |

[1]Estimated NOAEL: subchronic to chronic factor applied;
[2]Estimated NOAEL: LOAEL to NOAEL factor applied.

**Table 6.37. Estimated NOAELs for avian wildlife at Chestnut Ridge OU 2**

| Contaminant | Experimental information | | | | | Estimated NOAELs for birds (mg/kg per day) |
|---|---|---|---|---|---|---|
| | Form | Test species | NOAEL (mg/kg per day) and duration | Endpoint | Citation | Red-tailed hawk |
| Aluminum | sulfate | ring dove | 111.4<br>4 months | reproduction | Carriere et al. 1986 | 57.90 |
| Arsenic | arsenate | mallard duck | 0.036[1,2]<br>10 weeks | growth/ behavior | Camardese et al. 1990<br>Whitworth et al. 1991 | 0.033 |
| Cadmium | | mallard duck | 1.9<br>90 days | reproduction | White and Finley 1978 | 1.91 |
| Chromium | $Cr^{+3}$ | black duck | 0.54<br>10 months | reproduction | Haseltine et al. | 0.557 |
| Copper | oxide | chicken | 2.28[1]<br>10 weeks | wt. gain/ mortality | Mehring et al. 1960 | 2.24 |
| Lead | metal | american kestrel | 5.9<br>7 months | reproduction | Pattee 1984 | 2.89 |
| Mercury | methyl | mallard duck | 0.0064[2]<br>3 gen. | reproduction | Heinz 1979 | 0.0062 |
| Nickel | sulfate | chicken | 2.14[1]<br>4 weeks | wt. gain | Weber and Reid 1968 | 1.7 |
| Selenium | selenite | mallard duck | 0.1[1]<br>10 weeks | reproduction | Heinz et al. 1987 | 0.096 |
| Vanadium | | mallard duck | 10.34<br>12 weeks | survival/ blood chem. | White and Dieter 1978 | 10.47 |
| Zinc | carbonate | mallard duck | 1.7[1,2]<br>60 days | body wt./ mortality | Gasaway and Buss 1972 | 1.68 |

Table 6.37 (continued)

| Contaminant | Experimental information | | | | | Estimated NOAELs for birds (mg/kg per day) |
|---|---|---|---|---|---|---|
| | Form | Test species | NOAEL (mg/kg per day) and duration | Endpoint | Citation | Red-tailed hawk |
| bis(2-Ethylhexyl)Phthalate | | ring dove | 0.11[1] 4 weeks | reproduction | Peakall 1974 | 0.057 |
| Di-N-butylphthalate | | ring dove | 0.011[1,2] 4 weeks | reproduction | Peakall 1974 | 0.0057 |

[1]Estimated NOAEL—subchronic to chronic factor applied;
[2]Estimated NOAEL—LOAEL to NOAEL factor applied.

Table 6.38. Literature on metal body burdens in small mammals

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| **Arsenic** | | | | | |
| | **Rodents:** | | | | |
| | *Microtus pennsylvanicus* | NY orchard | 0.051-0.630 | fresh | Elfving et al. 1979 |
| | *Microtus pinetorum* | NY orchard | 0.16-0.956 | fresh | Elfving et al. 1979 |
| | *Peromyscus leucopus* | NY orchard | 0.079-0.589 | fresh | Elfving et al. 1979 |
| | *Microtus pennsylvanicus* | reference orchard | 0.0-0.005 | fresh | Elfving et al. 1979 |
| | *Microtus pinetorum* | reference orchard | 0.0-0.028 | fresh | Elfving et al. 1979 |
| | *Peromyscus leucopus* | reference orchard | 0.006-0.021 | fresh | Elfving et al. 1979 |
| | *Peromyscus maniculatus* | reference | <0.1 | fresh | Smith and Rongstad 1982 |
| | **Shrews:** | | | | |
| | *Blarina brevicauda* | Zn-Cu mine | 0.62 | fresh | Smith and Rongstad 1982 |
| | *Blarina brevicauda* | reference | <0.1 | fresh | Smith and Rongstad 1982 |
| **Cadmium** | | | | | |
| | **Rodents:** | | | | |
| | *Apodemus sylvaticus* | Cu-Cd refinery | <1.0 | dry | Hunter and Johnson 1982 |
| | *Apodemus sylvaticus* | Pb-Zn mine M | 2.6 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |
| | *Apodemus sylvaticus* | Pb-Zn mine Y | 1 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |
| | *Apodemus sylvaticus* | smelter | 0.8 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Clethrionomys glareolus* | Pb-Zn mine Y | 0.9 | dry | Johnson et al.1978 |
| | *Microtus agrestis* | mine-waste site | 1.84 | dry | Andrews et al. 1984 |
| | *Microtus agrestis* | sewage farm | 14.4[1] | dry | Beardsley et al. 1978 |
| | *Microtus agrestis* | Cu-Cd refinery | 1.7 | dry | Hunter and Johnson 1982 |
| | *Microtus agrestis* | Pb-Zn mine Y | 0.6 | dry | Johnson et al. 1978 |
| | *Microtus pennsylvanicus* | Zn-Cu mine | 1.4 | dry | Anthony and Kozlowski 1982 |
| | *Microtus pennsylvanicus* | roadside w/ 92.4k veh/day | 0.37 | dry | Scanlon 1987 |
| | *Microtus pennsylvanicus* | roadside w/ 24.1k veh/day | 0.19 | dry | Scanlon 1987 |
| | *Microtus pennsylvanicus* | roadside w/ 7.5 veh/day | 0.22 | dry | Scanlon 1987 |
| | *Peromyscus leucopus* | Zn smelter | 2.6[2] | dry | Beyer et al. 1985b |
| | *Peromyscus leucopus* | roadside w/ 92.4k veh/day | 0.81 | dry | Scanlon 1987 |
| | *Peromyscus leucopus* | roadside w/ 24.1k veh/day | 0.3 | dry | Scanlon 1987 |
| | *Peromyscus leucopus* | roadside w/ 7.5 veh/day | 0.2 | dry | Scanlon 1987 |
| | *Microtus pennsylvanicus* | Zn-Cu mine | 0.45 | fresh | Smith and Rongstad 1982 |
| | *Microtus spp.* | Smelter complex | 0.50-0.84 | fresh | Blus et al. 1987 |
| | *Peromyscus maniculatus* | Smelter complex | 0.42-0.81 | fresh | Blus et al. 1987 |
| | *Peromyscus maniculatus* | Zn-Cu mine | 0.49 | fresh | Smith and Rongstad 1982 |
| | *Apodemus sylvaticus* | reference M | 0.3 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |

**Table 6.38 (continued)**

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Apodemus sylvaticus* | reference | 0.3 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |
| | *Apodemus sylvaticus* | reference | 0.3 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |
| | *Clethrionomys glareolus* | reference | 0.2 | dry | Johnson et al. 1978 |
| | *Microtus agrestis* | reference | 0.88 | dry | Andrews et al. 1984 |
| | *Microtus agrestis* | ref. farm | 7.5[1] | dry | Beardsley et al. 1978 |
| | *Microtus agrestis* | lab. stock | 5[1] | dry | Beardsley et al. 1978 |
| | *Microtus agrestis* | reference | 0.1 | dry | Johnson et al. 1978 |
| | *Microtus pennsylvanicus* | reference | <0.4 | dry | Anthony and Kozlowski 1982 |
| | *Microtus pennsylvanicus* | Reference >400 m from hwy | 0.11-0.18 | dry | Scanlon 1987 |
| | *Peromyscus leucopus* | reference | 1.2[2] | dry | Beyer et al. 1985b |
| | *Peromyscus leucopus* | Reference >400 m from hwy | 0.18-0.40 | dry | Scanlon 1987 |
| | *Peromyscus maniculatus* | reference | 1.4 | dry | Schlesinger and Potter 1974 |
| | *Microtus pennsylvanicus* | reference | <0.1 | fresh | Smith and Rongstad 1982 |
| | *Peromyscus maniculatus* | reference | <0.1 | fresh | Smith and Rongstad 1982 |
| | **Shrews:** | | | | |
| | *Blarina brevicauda* | Zn smelter | 7.3[2] | dry | Beyer et al. 1985b |
| | *Blarina brevicauda* | roadside w/ 92.4k veh/day | 1.71 | dry | Scanlon 1987 |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Blarina brevicauda* | roadside w/ 24.1k veh/day | 0.46 | dry | Scanlon 1987 |
| | *Blarina brevicauda* | roadside w/ 7.5 veh/day | 0.47 | dry | Scanlon 1987 |
| | *Sorex araneus* | mine waste | 52.7 | dry | Andrews et al. 1984 |
| | *Sorex araneus* | Cu refinery | 27.5 | dry | Hunter and Johnson 1982 |
| | *Sorex araneus* | Pb-Zn mine M | 40 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |
| | *Sorex araneus* | 10m from mine M | 17 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |
| | *Blarina brevicauda* | Zn-Cu mine | 2 | fresh | Smith and Rongstad 1982 |
| | *Blarina brevicauda* | reference | 4.8[2] | dry | Beyer et al. 1985b |
| | *Blarina brevicauda* | Reference >400 m from hwy | 0.47-0.66 | dry | Scanlon 1987 |
| | *Blarina brevicauda* | reference | 1.2 | dry | Schlesinger and Potter 1974 |
| | *Sorex araneus* | reference | 1.19 | dry | Andrews et al. 1984 |
| | *Sorex araneus* | reference M | 4 | dry | Johnson et al. 1978; Roberts and Johnson 1978 |
| | *Blarina brevicauda* | reference | 0.28 | fresh | Smith and Rongstad 1982 |
| | *Sorex cinereus* | reference | 0.24 | fresh | Smith and Rongstad 1982 |
| | | | | | |
| **Chromium** | | | | | |
| | **Rodents:** | | | | |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Microtus agrestis* | sewage farm | 6.9[1] | dry | Beardsley et al. 1978 |
| | *Microtus agrestis* | reference | 7.2[1] | dry | Beardsley et al. 1978 |
| | *Microtus agrestis* | lab. colony | 3.8[1] | dry | Beardsley et al. 1978 |
| | *Sigmodon hispidus* | 100-130 m from cooling tower drift | 0.4 | dry | Taylor and Parr 1978; |
| | *Sigmodon hispidus* | controls | 0.06 | dry | Taylor and Parr 1978; |
| | *Sigmodon hispidus* | 100-130 m from cooling tower drift | 0.12 | fresh | Taylor and Parr 1978; |
| | *Sigmodon hispidus* | controls | 0.19 | fresh | Taylor and Parr 1978; |
| | | | | | |
| Mercury | | | | | |
| | Rodents: | | | | |
| | *Apodemus sylvaticus* | fields w/ Hg soaked wheat | 1.3-2.7 | dry | Jefferies and French 1976, Jefferies et al. 1973 |
| | *Apodemus sylvaticus* | presowed fields | 0.1 | dry | Jefferies and French 1976, Jefferies et al. 1973 |
| | *Apodemus sylvaticus* | fields w/ Hg soaked wheat | 0.1-2.6 | dry | Westlake et al. 1980 |
| | Shrews: | | | | |
| | *Blarina brevicauda* | Zn-Cu mine | <0.1 | fresh | Smith and Rongstad 1982 |
| | *Blarina brevicauda* | reference | <0.1 | fresh | Smith and Rongstad 1982 |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | | | | | |
| **Lead** | | | | | |
| | **Rodents:** | | | | |
| | *Apodemus sylvaticus* | major road | 5.3 | dry | Jefferies and French 1972 |
| | *Apodemus sylvaticus* | minor road | 5.1 | dry | Jefferies and French 1972 |
| | *Clethrionomys glareolus* | major roadside | 5.7 | dry | Jefferies and French 1972 |
| | *Clethrionomys glareolus* | minor roadside | 6.2 | dry | Jefferies and French 1972 |
| | *Microtus agrestis* | mine tailings | 60.1 | dry | Andrews et al. 1989 |
| | *Microtus agrestis* | sewage farm | 39[1] | dry | Beardsley et al. 1978 |
| | *Microtus agrestis* | major roadside | 10 | dry | Jefferies and French 1972 |
| | *Microtus agrestis* | minor roadside | 6.8 | dry | Jefferies and French 1972 |
| | *Microtus ochrogaster* | roadside: 19.6k veh/day | 8.1 | dry | Getz et al. 1977 |
| | *Microtus ochrogaster* | roadside: 1360 veh/day | 4.3 | dry | Getz et al. 1977 |
| | *Microtus ochrogaster* | roadside: 340 veh/day | 2.6 | dry | Getz et al. 1977 |
| | *Microtus pennsylvanicus* | roadside: 8120 veh/day | 12.1 | dry | Goldsmith and Scanlon 1977, Scanlon 1979 |
| | *Microtus pennsylvanicus* | roadside: 1085 veh/day | 6.9 | dry | Goldsmith and Scanlon 1977, Scanlon 1979 |
| | *Microtus pennsylvanicus* | roadside: 12470 veh/day | 16.3 | dry | Quarles et al. 1974 |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Microtus pennsylvanicus* | 25-45m from road | 5.8 | dry | Quarles et al. 1974 |
| | *Microtus pennsylvanicus* | roadside w/ 92.4k veh/day | 23.1 | dry | Scanlon 1987 |
| | *Microtus pennsylvanicus* | roadside w/ 24.1k veh/day | 16.6 | dry | Scanlon 1987 |
| | *Microtus pennsylvanicus* | roadside w/ 7.5 veh/day | 7.2 | dry | Scanlon 1987 |
| | *Microtus pinetorum* | Pb-Arsenate trt. orch. | 19.5-40.67[2] | dry | Scanlon et al. 1983 |
| | *Mus musculus* | roadside: 19.6k veh/day | 6.9 | dry | Getz et al. 1977 |
| | *Mus musculus* | roadside:1360 veh/day | 3.4 | dry | Getz et al. 1977 |
| | *Mus musculus* | roadside: 300 veh/day | 4 | dry | Getz et al. 1977 |
| | *Peromyscus leucopus* | near metal smelter | 17[2] | dry | Beyer et al. 1985b |
| | *Peromyscus leucopus* | road:21k veh/day | 15.6 | dry | Goldsmith and Scanlon 1977; Scanlon 1979 |
| | *Peromyscus leucopus* | road: 8120 | 9.7 | dry | Goldsmith and Scanlon 1977; Scanlon 1979 |
| | *Peromyscus leucopus* | road:12470 | 6.8 | dry | Quarles et al. 1974 |
| | *Peromyscus leucopus* | 30-55m fr rd | 3.9 | dry | Quarles et al. 1974 |
| | *Peromyscus leucopus* | roadside w/ 92.4k veh/day | 22 | dry | Scanlon 1987 |
| | *Peromyscus leucopus* | roadside w/ 24.1k veh/day | 13.4 | dry | Scanlon 1987 |
| | *Peromyscus leucopus* | roadside w/ 7.5 veh/day | 5.2 | dry | Scanlon 1987 |
| | *Peromyscus maniculatus* | roadside: 19.6k veh/day | 5.5 | dry | Getz et al. 1977 |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Peromyscus maniculatus* | roadside: 1360 veh/day | 3.7 | dry | Getz et al. 1977 |
| | *Peromyscus maniculatus* | roadside: 340 veh/day | 2.4 | dry | Getz et al. 1977 |
| | *Reithrodontomys megalotis* | roadside: 19.6k veh/day | 10.8 | dry | Getz et al. 1977 |
| | *Reithrodontomys megalotis* | roadside: 1360 veh/day | 3.8 | dry | Getz et al. 1977 |
| | *Reithrodontomys megalotis* | roadside: 340 veh/day | 3.1 | dry | Getz et al. 1977 |
| | *Apodemus sylvaticus* | Pb-Zn mine M | 8.6 | fresh | Roberts et al. 1978 |
| | *Apodemus sylvaticus* | Pb-Zn mine Y | 13.9 | fresh | Roberts et al. 1978 |
| | *Clethrionomys glareolus* | Pb-Zn mine M | 16.3 | fresh | Roberts et al. 1978 |
| | *Clethrionomys glareolus* | Pb-Zn mine Y | 20.7 | fresh | Roberts et al. 1978 |
| | *Microtus agrestis* | Pb-Zn mine M | 45.3 | fresh | Roberts et al. 1978 |
| | *Microtus agrestis* | Pb-Zn mine Y | 42.8 | fresh | Roberts et al. 1978 |
| | *Microtus pennsylvanicus* | roadside: 35k veh/day | 1.45 | fresh | Clark 1979 |
| | *Microtus pennsylvanicus* | Zn-Cu mine | 0.66 | fresh | Smith and Rongstad 1982 |
| | *Microtus sp.* | metal smelter | 23.5-173 | fresh | Blus et al. 1987 |
| | *Peromyscus leucopus* | road:35k veh/day | 4.91 | fresh | Clark 1979 |
| | *Peromyscus maniculatus* | metal smelter | 52.7-58.0 | fresh | Blus et al. 1987 |
| | *Peromyscus maniculatus* | Zn-Cu mine | 4.54 | fresh | Smith and Rongstad 1982 |
| | *Zapus hudsonius* | Zn-Cu mine | 1.34 | fresh | Smith and Rongstad 1982 |
| | | | | | |

**Table 6.38 (continued)**

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Apodemus sylvaticus* | reference fields | 3.5 | dry | Jefferies and French 1972 |
| | *Clethrionomys glareolus* | reference fields | 4.7 | dry | Jefferies and French 1972 |
| | *Clethrionomys glareolus* | reference Y | 2.64 | dry | Roberts et al. 1978 |
| | *Microtus agrestis* | reference | 7.7 | dry | Andrews et al. 1989 |
| | *Microtus agrestis* | reference | 25[1] | dry | Beardsley et al. 1978 |
| | *Microtus agrestis* | lab. stock | 19[1] | dry | Beardsley et al. 1978 |
| | *Microtus ochrogaster* | reference | 3.3 | dry | Getz et al. 1977 |
| | *Microtus pennsylvanicus* | 135-185m from road | 3.8 | dry | Quarles et al. 1974 |
| | *Microtus pennsylvanicus* | reference | 4.9 | dry | Quarles et al. 1974 |
| | *Microtus pennsylvanicus* | Reference >400 m from hwy | 1.1-3.92 | dry | Scanlon 1987 |
| | *Mus musculus* | reference | 4.6 | dry | Getz et al. 1977 |
| | *Peromyscus leucopus* | reference | 7.4[2] | dry | Beyer et al. 1985b |
| | *Peromyscus leucopus* | reference | 6.4 | dry | Goldsmith and Scanlon 1977; Scanlon 1979 |
| | *Peromyscus leucopus* | 115-195m fr rd | 2.6 | dry | Quarles et al. 1974 |
| | *Peromyscus leucopus* | reference | 2.6 | dry | Quarles et al. 1974 |
| | *Peromyscus leucopus* | Reference >400 m from hwy | 2.39-2.60 | dry | Scanlon 1987 |
| | *Peromyscus maniculatus* | reference | 2.8 | dry | Getz et al. 1977 |
| | *Reithrodontomys megalotis* | reference | 3.1 | dry | Getz et al. 1977 |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Apodemus sylvaticus* | reference M | 0.9 | fresh | Roberts et al. 1978 |
| | *Apodemus sylvaticus* | reference Y | 1.2 | fresh | Roberts et al. 1978 |
| | *Clethrionomys glareolus* | reference M | 2.36 | fresh | Roberts et al. 1978 |
| | *Microtus agrestis* | reference Y | 2.76 | fresh | Roberts et al. 1978 |
| | *Microtus pennsylvanicus* | reference | 0.84 | fresh | Clark 1979 |
| | *Microtus pennsylvanicus* | reference | 0.52 | fresh | Smith and Rongstad 1982 |
| | *Peromyscus leucopus* | reference | 1.16 | fresh | Clark 1979 |
| | *Peromyscus maniculatus* | reference | 0.6 | fresh | Smith and Rongstad 1982 |
| | *Zapus hudsonius* | reference | 0.46 | fresh | Smith and Rongstad 1982 |
| | | | | | |
| | **Shrews:** | | | | |
| | *Blarina brevicauda* | Zn smelter | 109[2] | dry | Beyer et al. 1985b |
| | *Blarina brevicauda* | road:19600 | 17.2 | dry | Getz et al. 1977 |
| | *Blarina brevicauda* | road:1360 | 6.4 | dry | Getz et al. 1977 |
| | *Blarina brevicauda* | road:340 | 3.9 | dry | Getz et al. 1977 |
| | *Blarina brevicauda* | road: 21010 veh/day | 34.8 | dry | Goldsmith and Scanlon 1977; Scanlon 1979 |
| | *Blarina brevicauda* | road: 8120 | 15.8 | dry | Goldsmith and Scanlon 1977; Scanlon 1979 |

Table 6.38 (continued)

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Blarina brevicauda* | road: 1085 | 11.6 | dry | Goldsmith and Scanlon 1977; Scanlon 1979 |
| | *Blarina brevicauda* | road:12345 | 22.7 | dry | Quarles et al. 1974 |
| | *Blarina brevicauda* | roadside w/ 24.1k veh/day | 35.1 | dry | Scanlon 1987 |
| | *Blarina brevicauda* | roadside w/ 7.5 veh/day | 12 | dry | Scanlon 1987 |
| | *Blarina brevicauda* | roadside w/ 92.4k veh/day | 72.6 | dry | Scanlon 1987 |
| | *Cryptotis parva* | roadside:8120 veh/day | 10.4 | dry | Goldsmith and Scanlon 1977, Scanlon 1979 |
| | *Cryptotis parva* | roadside:1085 veh/day | 6.5 | dry | Goldsmith and Scanlon 1977, Scanlon 1979 |
| | *Sorex araneus* | mine tailings | 98.6 | dry | Andrews et al. 1989 |
| | *Sorex cinereus* | roadside | 13.7 | dry | Goldsmith and Scanlon 1977, Scanlon 1979 |
| | *Blarina brevicauda* | Near hwy w/ 35k veh/day | 26.2 | fresh | Clark 1979 |
| | *Blarina brevicauda* | Zn-Cu mine | 0.9 | fresh | Smith and Rongstad 1982 |
| | *Sorex araneus* | Pb-Zn mine M | 11.2 | fresh | Roberts et al. 1978 |
| | | | | | |
| | *Blarina brevicauda* | reference | 18[2] | dry | Beyer et al. 1985b |
| | *Blarina brevicauda* | reference | 3.3 | dry | Getz et al. 1977 |

**Table 6.38 (continued)**

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Blarina brevicauda* | reference | 13.9 | dry | Goldsmith and Scanlon 1977; Scanlon 1979 |
| | *Blarina brevicauda* | 25-345m fr rd | 5.2 | dry | Quarles et al. 1974 |
| | *Blarina brevicauda* | reference | 5.4 | dry | Quarles et al. 1974 |
| | *Blarina brevicauda* | Reference >400 m from hwy | 2.7-7.28 | dry | Scanlon 1987 |
| | *Sorex araneus* | reference | 11 | dry | Andrews et al. 1989 |
| | *Sorex cinereus* | reference | 16.6 | dry | Goldsmith and Scanlon 1977, Scanlon 1979 |
| | *Blarina brevicauda* | reference | 1.85 | fresh | Clark 1979 |
| | *Blarina brevicauda* | reference | 5.84 | fresh | Smith and Rongstad 1982 |
| | *Sorex araneus* | reference | 0.88 | fresh | Roberts et al. 1978 |
| | *Sorex cinereus* | reference | 0.88 | fresh | Smith and Rongstad 1982 |
| | | | | | |
| **Selenium** | | | | | |
| | **Rodents:** | | | | |
| | *Microtus californicus* | Irrigated w/ Se-laden drainwater | 0-33 | dry | Clark 1987 |
| | *Reithrodontomys megalotis* | Irrigated w/ Se-laden drainwater | 2.4-27 | dry | Clark 1987 |
| | **Shrews:** | | | | |

**Table 6.38 (continued)**

| Analyte | Species | Location | Whole body concentration (ppm) | Weight | Reference |
|---|---|---|---|---|---|
| | *Sorex ornatus* | Irrigated w/ Se-laden drainwater | 10-100 | dry | Clark 1987 |

[1]Sum of all body tissues;
[2]Carcass concentration.

**Table 6.39. Summary of risk characterization for fish in Upper McCoy Branch**

| Evidence | Result[a] | Explanation |
|---|---|---|
| Biological surveys | + | The absence of fish indicates that some agent induced severe effects. Apparently poor recolonization is consistent with ongoing toxicity. Deformaties, contaminant bioaccumulation, and abnormal species compositions observed in downstream areas. |
| Toxicity tests | — | No clear indications of toxicity in water samples. |
| Media analyses | + | Al, Cu, Fe, and Zn in water sufficient to reduce survival or reproduction. Cr, Pb, and Mn in water sufficient to reduce population abundance or production in sensitive species. |
| Weight-of-evidence | + | The biological survey results in combination with the occurrence of multiple contaminants at potentially toxic concentrations outweigh the ambient water test results. |

[a] + indicates that the evidence is consistent with the occurrence of the endpoint effect; — indicates that the evidence is inconsistent with the occurrence of the endpoint effect; ± indicates that the evidence is too ambiguous to interpret.

**Table 6.40. Comparison of FCAP surface water data to the aquatic benchmarks**

| Total Concentrations[a] | | | | | |
|---|---|---|---|---|---|
| Analyte | Detected | Analyzed | UCL95 | Mean[b] | Benchmarks exceeded[c] |
| *Inorganics (mg/L)* | | | | | |
| Aluminum | 1 | 3 | 1.03e+01 | **2.72e+00** | EC20 for daphnids and fish, LCV for daphnids and fish, NAWQC acute and chronic, SS test EC20 |
| Barium | 3 | 3 | 8.51e-02 | 3.92e-02 | No Benchmarks |
| Chromium | 1 | 3 | 9.05e-03 | 8.27e-03 | SS test EC20 |
| Copper | 1 | 3 | 8.31e-03 | **5.10e-03** | EC20 for daphnids and fish, LCV for daphnids and fish, SS test EC20 |
| Iron | 3 | 3 | 7.87e+00 | **2.32e+00** | EC20 for daphnids, LCV for daphnids, NAWQC chronic, |
| Lead | 1 | 3 | 3.96e-03 | **2.50e-03** | NAWQC chronic, SS test EC20 |
| Manganese | 3 | 3 | 2.17e+00 | **6.79e-01** | EC20 for daphnids, fish, and population, LCV for daphnids and fish |
| Potassium | 1 | 3 | 8.30e+00 | 3.12e+00 | No Benchmarks |
| Zinc | 2 | 3 | 1.36e-01 | **4.12e-02** | EC20 for fish and population, LCV for daphnids and fish, SS test EC20, NAWQC acute and chronic |
| *Radionuclides (pCi/L)* | | | | | |
| Thorium-228 | 1 | 3 | 2.81e-01 | 8.70e-02 | No Benchmarks |
| Thorium-230 | 1 | 3 | 2.57e+00 | 8.33e-01 | No Benchmarks |
| Uranium-234 | 2 | 3 | 4.38e-01 | 1.94e-01 | No Benchmarks |
| Uranium-238 | 2 | 3 | 1.34e-01 | 7.13e-02 | No Benchmarks |

[a]Results are for detected analytes that had a UCL95 concentration greater than background and exceeded at least one non-advisory WQB.
[b]Values in bold exceed at least one nonadvisory benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.41. Comparison of UMB surface water data to the aquatic benchmarks**

| Total Concentrations[a] | | | | | |
|---|---|---|---|---|---|
| Analyte | Detected | Analyzed | UCL95 | Mean[b] | Benchmarks exceeded[c] |
| *Inorganics (mg/L)* | | | | | |
| Aluminum | 10 | 11 | 1.46e+00 | **2.00e-01** | EC20 for daphnids, NAWQC acute and chronic, SS test EC20 |
| Arsenic | 9 | 11 | 2.29e-01 | 3.86e-02 | EC20 for population as As-V; NAWQC chronic and SS test EC20 as As-III |
| Calcium | 11 | 11 | 5.12e+01 | 4.07e+01 | No Benchmarks |
| Copper | 2 | 11 | 8.90e-03 | **4.45e-03** | EC20 for daphnids and fish, LCV for daphnids and fish, SS test EC20 |
| Iron | 7 | 11 | 3.31e+00 | **5.26e-01** | EC20 for daphnids, LCV for daphnids, NAWQC chronic, |
| Lead | 1 | 11 | 3.00e-03 | **1.22e-03** | SS test EC20 |
| Manganese | 8 | 11 | 1.09e+00 | **2.84e-01** | EC20 for population |
| Mercury | 1 | 3 | 2.50e-04 | **2.10e-04** | LCV for fish and SS test EC20 as Hg; LCV for daphnids and EC20 for fish as MeHg |
| Niobium | 2 | 8 | 9.50e-03 | 7.43e-03 | No Benchmarks |
| Potassium | 10 | 11 | 4.94e+00 | 3.16e+00 | No Benchmarks |
| Sulfate | 7 | 8 | 2.60e+01 | 1.84e+01 | No Benchmarks |
| Zinc | 11 | 11 | 3.35e-02 | 1.14e-02 | SS test EC20 |
| *Radionuclides (pCi/L)* | | | | | |
| Radium-226 | 7 | 8 | 6.08e-01 | 4.25e-01 | No Benchmarks |
| Thorium-228 | 1 | 3 | 5.16e-02 | 2.00e-02 | No Benchmarks |
| Thorium-230 | 2 | 3 | 7.75e-01 | 4.57e-01 | No Benchmarks |
| Uranium-234 | 3 | 3 | 5.74e-01 | 4.33e-01 | No Benchmarks |
| Uranium-238 | 7 | 11 | 3.40e-01 | 3.20e-01 | No Benchmarks |
| *Organics (mg/L)* | | | | | |
| bis(2-Ethylhexyl)Phthalate | 2 | 3 | 6.09e-02 | **2.07e-02** | EC20 for daphnids, fish and population, LCV for daphnids and fish |

[a]Results are for detected analytes that had a UCL95 concentration greater than background and exceeded at least one non-advisory WQB.
[b]Values in bold exceed at least one nonadvisory benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.42. Comparison of FCAP surface water nondetected analyte data to the aquatic benchmarks**

| Total Concentrations[a] | | | | | | |
|---|---|---|---|---|---|---|
| Analyte | Detected | Analyzed | Max non-det | Min nondet[b] | Mean[b] | Benchmarks exceeded[c] |
| *Inorganics (mg/L)* | | | | | | |
| Chromium VI | 0 | 2 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LTV_DAPHNIDS, LCV_DAPHNIDS, SS_EC20 |
| *Organics (mg/kg)* | | | | | | |
| Anthracene | 0 | 3 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LTV_DAPHNIDS, LTV_FISH, LCV_DAPHNIDS, LCV_FISH |
| Benzo(a)Anthracene | 0 | 3 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LCV_DAPHNIDS |
| Benzo(a)Pyrene | 0 | 3 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LCV_DAPHNIDS, LTV_FISH |
| bis(2-Ethylhexyl)Phthalate | 0 | 3 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LCV_DAPHNIDS, LCV_FISH, LTV_DAPHNIDS |
| Pentachlorophenol | 0 | 3 | 2.50e-02 | **2.50e-02** | **2.50e-02** | NAWQ_ACUTE, NAWQ_CHRONIC |

[a]Results are for analytes that had a maximum non-det. concentration greater than background and exceeded at least one non-advisory WQB.
[b]Values in bold exceed at least one nonadvisory benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.43. Comparison of UMB surface water nondetected analyte data to the aquatic benchmarks**

| Total Concentrations[a] | | | | | | |
|---|---|---|---|---|---|---|
| Analyte | Detected | Analyzed | Max non-det | Min non-det[b] | Mean[b] | Benchmarks exceeded[c] |
| *Inorganics (mg/kg)* | | | | | | |
| Chromium VI | 0 | 1 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LCV_DAPHNIDS, LTV_DAPHNIDS, SS_EC20 |
| Cobalt | 0 | 3 | 9.00e-03 | **8.00e-03** | **8.67e-03** | EC20_POPULATION, LCV_DAPHNIDS, LTV_DAPHNIDS |
| Nickel | 0 | 3 | 1.80e-02 | **1.70e-02** | **1.77e-02** | LCV_DAPHNIDS, SS_EC20 |
| *Organics (mg/kg)* | | | | | | |
| Anthracene | 0 | 3 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LCV_DAPHNIDS, LCV_FISH, LTV_DAPHNIDS, LTV_FISH |
| Benzo(a)Anthracene | 0 | 3 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LCV_DAPHNIDS |
| Benzo(a)Pyrene | 0 | 3 | 1.00e-02 | **1.00e-02** | **1.00e-02** | LCV_DAPHNIDS, LTV_FISH |
| Pentachlorophenol | 0 | 3 | 2.50e-02 | **2.50e-02** | **2.50e-02** | NAWQ_ACUTE, NAWQ_CHRONIC |

[a]Results are for analytes that had a maximum non-det. concentration greater than background and exceeded at least one non-advisory WQB.
[b]Values in bold exceed at least one nonadvisory benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.44. Summary of risk characterization for the benthic macroinvertebrate community in the UMB**

| Evidence | Result[1] | Explanation |
|---|---|---|
| Contaminant concentrations | + | As, Ba, Fe, Mn and Ni in sediment sufficient to cause reduced survival.<br>Al, As, Cu, Fe, Pb, Hg, and Zn in surface water sufficient to cause reduced survival and reproduction. |
| Media toxicity tests | — | Water and sediment toxicity tests, conducted in both Phases of the RI, provide no strong evidence of acute or chronic toxicity. |
| Biological surveys | ± | Initial survey results clearly indicated a moderate to severe level of impact. Although successive survey results improved steadily only the final survey was acceptable. There is insufficient evidence to determine the stability of this community. |
| Weight-of-evidence | + | Current and future recovery is not assured at this time and the only negative line of evidence was not corroborated by the concurrent survey results. |

[1]NA indicates data are not available; + indicates that the evidence is consistent with the occurrence of endpoint effect; — indicates that the evidence is inconsistent with the occurrence of endpoint effect; ± indicates that the evidence is too ambiguous to interpret.

**Table 6.45. Summary of risk characterization for the benthic macroinvertebrate community in the FCAP**

| Evidence | Result[1] | Explanation |
|---|---|---|
| Contaminant concentrations | + | As and Fe in sediment sufficient to cause reduced survival.<br>Al, Cr, Cu, Fe, Pb, Mn, and Zn in surface water sufficient to cause reduced survival and reproduction. |
| Media toxicity tests | NA | These tests were not performed for media above the FCAP dam. |
| Biological surveys | NA | Surveys were not conducted above the FCAP dam. |
| Weight-of-evidence | ± | Although the one available line of evidence supports a positive assessment there is insufficient evidence available for a definitive characterization. |

[1]NA indicates data are not available; + indicates that the evidence is consistent with the occurrence of endpoint effect; — indicates that the evidence is inconsistent with the occurrence of endpoint effect; ± indicates that the evidence is too ambiguous to interpret.

**Table 6.46. Comparison of FCAP sediment data to benchmarks[a]**

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | |
| Aluminum | 2 | 2 | 1.15e+04 | 1.11e+04 | No Benchmarks |
| Arsenic | 2 | 2 | 7.66e+01 | 4.33e+01 | Lower, Upper |
| Beryllium | 2 | 2 | 2.10e+00 | 1.44e+00 | No Benchmarks |
| Calcium | 2 | 2 | 3.71e+03 | 2.58e+03 | No Benchmarks |
| Iron | 2 | 2 | 3.01e+04 | 1.98e+04 | Lower |
| Magnesium | 2 | 2 | 1.23e+03 | 9.79e+02 | No Benchmarks |
| Potassium | 1 | 2 | 7.48e+02 | 1.22e+03 | No Benchmarks |
| Selenium | 1 | 2 | 1.00e+00 | 1.65e+00 | No Benchmarks |
| Sodium | 2 | 2 | 5.41e+02 | 3.80e+02 | No Benchmarks |
| Vanadium | 2 | 2 | 3.55e+01 | 2.96e+01 | No Benchmarks |
| *Radionuclides (pCi/g)* | | | | | |
| Thorium-228 | 2 | 2 | 7.30e-01 | 6.80e-01 | No Benchmarks |
| Thorium-232 | 2 | 2 | 1.60e+00 | 1.03e+00 | No Benchmarks |
| Uranium-234 | 2 | 2 | 3.90e+00 | 2.95e+00 | No Benchmarks |
| Uranium-235 | 2 | 2 | 2.90e-01 | 2.70e-01 | No Benchmarks |
| Uranium-238 | 2 | 2 | 3.00e+00 | 2.30e+00 | No Benchmarks |
| *Organics (mg/kg)* | | | | | |
| Butylbenzylphthalate | 1 | 2 | 1.70e-01 | 3.15e-01 | No Benchmarks |

[a]Results are for detected analytes that had a Maximum concentration greater than background and the Benchmarks.

**Table 6.47. Comparison of FCAP sediment nondetected analyte data to the benchmarks[a]**

| Analyte | Detected | Analyzed | Max non-det | Min non-det[b] | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | | |
| Antimony | 0 | 2 | 3.48e+01 | **1.28e+01** | **2.38e+01** | Lower, Upper |
| Cadmium | 0 | 2 | 3.80e+00 | **1.40e+00** | **2.60e+00** | Lower |
| Mercury | 0 | 2 | 3.80e-01 | 1.40e-01 | **2.60e-01** | Lower |
| Silver | 0 | 2 | 7.60e+00 | **2.80e+00** | **5.20e+00** | Lower, Upper |
| Thallium | 0 | 2 | 3.00e+00 | 1.10e+00 | 2.05e+00 | No Benchmarks |
| *Radionuclides (pCi/g)* | | | | | | |
| Thorium-230 | 0 | 2 | 3.80e+00 | 1.70e+00 | 2.75e+00 | No Benchmarks |
| *Organics (mg/kg)* | | | | | | |
| 2-Methylnaphthalene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Acenaphthene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Anthracene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L, SQB |
| Benzo(a)Anthracene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Chrysene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Dibenz(a,h)Anthracene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Fluoranthene | 0 | 2 | 1.30e+00 | 4.60e-01 | **8.80e-01** | ER-L |
| Fluorene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Naphthalene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Phenanthrene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |
| Pyrene | 0 | 2 | 1.30e+00 | **4.60e-01** | **8.80e-01** | ER-L |

[a]Results are for analytes that had a maximum non-det. concentration greater than background and the Benchmarks.
[b]Values in bold exceed at least one benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.48. Comparison of UMB sediment data to benchmarks[a]**

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | |
| Aluminum | 2 | 2 | 2.27e+04 | 1.60e+04 | No Benchmarks |
| Arsenic | 2 | 2 | 2.30e+03 | **1.23e+03** | Lower, Upper |
| Barium | 2 | 2 | 5.96e+02 | **4.53e+02** | Lower, Upper |
| Beryllium | 2 | 2 | 4.70e+00 | 3.05e+00 | No Benchmarks |
| Calcium | 2 | 2 | 3.04e+04 | 1.70e+04 | No Benchmarks |
| Cobalt | 2 | 2 | 8.62e+01 | 5.35e+01 | No Benchmarks |
| Iron | 2 | 2 | 6.33e+04 | **4.32e+04** | Lower, Upper |
| Magnesium | 2 | 2 | 1.80e+04 | 9.51e+03 | No Benchmarks |
| Manganese | 2 | 2 | 1.37e+04 | **8.87e+03** | Lower, Upper |
| Nickel | 2 | 2 | 4.51e+01 | **3.79e+01** | Lower |
| Potassium | 2 | 2 | 2.60e+03 | 2.18e+03 | No Benchmarks |
| Selenium | 1 | 2 | 2.40e+00 | 2.40e+00 | No Benchmarks |
| Sodium | 2 | 2 | 5.20e+02 | 4.53e+02 | No Benchmarks |
| Vanadium | 2 | 2 | 4.87e+01 | 3.80e+01 | No Benchmarks |
| *Radionuclides (pCi/g)* | | | | | |
| Thorium-228 | 2 | 2 | 4.30e+00 | 2.65e+00 | No Benchmarks |
| Thorium-232 | 2 | 2 | 1.20e+00 | 1.07e+00 | No Benchmarks |
| Uranium-238 | 2 | 2 | 2.20e+00 | 2.05e+00 | No Benchmarks |

[a]Results are for detected analytes that had a Maximum concentration greater than background and the Benchmarks.
[b]Values in bold exceed at least one benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.49. Comparison of UMB sediment nondetected analyte data to the benchmarks[a]**

| Analyte | Detected | Analyzed | Max non-det | Min non-det[b] | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | | |
| Antimony | 0 | 2 | 3.77e+01 | **2.03e+01** | **2.90e+01** | Lower, Upper |
| Cadmium | 0 | 2 | 4.10e+00 | **2.20e+00** | **3.15e+00** | Lower |
| Mercury | 0 | 2 | 4.10e-01 | **2.20e-01** | **3.15e-01** | Lower |
| Silver | 0 | 2 | 8.20e+00 | **4.40e+00** | **6.30e+00** | Lower, Upper |
| Thallium | 0 | 2 | 3.30e+00 | 1.80e+00 | 2.55e+00 | No Benchmarks |
| *Organics (mg/kg)* | | | | | | |
| 2-Methylnaphthalene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Acenaphthene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Anthracene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L, SQB |
| Benzo(a)Anthracene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Chrysene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Dibenz(a,h)Anthracene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Fluoranthene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Fluorene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Naphthalene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |
| Pyrene | 0 | 2 | 8.90e-01 | **7.30e-01** | **8.10e-01** | ER-L |

[a]Results are for analytes that had a maximum non-det. concentration greater than background and the benchmarks.
[b]Values in bold exceed at least one benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.50. Maximum concentrations of analytes in soil and ash exceeding background soil levels and benchmarks for phytotoxicity in soil**

| Analyte | Benchmark (ppm) | Sluice Channel Area soil (ppm) | HQ[a] | Sluice Channel Area ash (ppm) | HQ | FCAP ash (ppm) | HQ | UMB soil (ppm) | HQ | UMB ash (ppm) | HQ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Aluminum | 50 | 11400 | 228 | 14100 | 282 | 32900 | 658 | 18600 | 372 | 20800 | 416 |
| Antimony | 5 | NEB[b] | | 15.6[c] | 3 | 22.6[c] | 5 | NEB | | NEB | |
| Arsenic | 10 | 51.2 | 5 | 159 | 16 | 193 | 19 | 43.6 | 4 | 59.2 | 6 |
| Barium | 500 | NEB | | NEB | | 630 | 1 | NEB | | 684 | 1 |
| Cadmium | 3 | NEB | | NEB | | 3.1[c] | 1 | 22.8 | 8 | 4.2 | 1 |
| Chromium | 2 | NEB | | 17.1 | 9 | 42.1 | 21 | 50.6 | 25 | 18.2 | 9 |
| Cobalt | 25 | NEB | | 28.6 | 1 | 29.8 | 1 | 29.5 | 1 | NEB | |
| Copper | 60 | 6.4 | <1 | 70.1 | 1 | 101 | 2 | 70.2 | 1 | 74.9 | 1 |
| Manganese | 500 | 1920 | 4 | NEB | | NEB | | 2380 | 5 | NEB | |
| Mercury | 0.3 | NEB | | 0.61 | 2 | 1.3 | 4 | NEB | | 0.73 | 2 |
| Nickel | 25 | NEB | | 39.9 | 2 | 56.0 | 2 | 40.4 | 2 | 40.7 | 2 |
| Selenium | 1 | 1.10 | 1 | 4.7 | 5 | 30.8 | 30 | 2.2 | 2 | 9.1 | 9 |
| Silver | 2 | NEB | | 3.4[c] | 2 | 4.5[c] | 2 | NEB | | NEB | |
| Thallium | 1 | NEB | | 1.4[c] | 1 | 3.2 | 3 | NEB | | NEB | |
| Vanadium | 2.5 | 32.4 | 13 | 47.5 | 19 | 123 | 49 | 94.0 | 38 | 83.7 | 33 |
| Zinc | 25 | NEB | | 40.0 | 2 | 83.1 | 3 | 102 | 4 | 55.7 | 2 |

[a] HQ = Hazard Quotient = concentration of analyte in sample/benchmark concentration.
[b] NEB = concentration of analyte did not exceed background soil concentration.
[c] Maximum non-detected concentration exceeding benchmark.

Table 6.51. Comparison of ash from the Sluice Channel Area to toxicological benchmarks for soil biota[a]

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | |
| Aluminum | 3 | 3 | 1.41e+04 | **1.08e+04** | Processes, Vegetation |
| Arsenic | 3 | 3 | 1.59e+02 | **1.34e+02** | Vegetation |
| Calcium | 3 | 3 | 4.92e+03 | 4.57e+03 | No Benchmarks[b] |
| Chromium | 3 | 3 | 1.71e+01 | **1.18e+01** | Annelids, Processes, Vegetation |
| Cobalt | 3 | 3 | 2.86e+01 | 1.43e+01 | Vegetation |
| Copper | 3 | 3 | 7.01e+01 | **4.02e+01** | Processes, Vegetation |
| Iron | 3 | 3 | 1.83e+05 | **7.08e+04** | Processes |
| Magnesium | 3 | 3 | 1.01e+03 | 7.12e+02 | No Benchmarks |
| Mercury | 3 | 3 | 6.10e-01 | **4.60e-01** | Annelids, Processes |
| Nickel | 3 | 3 | 3.99e+01 | 2.34e+01 | Vegetation |
| Potassium | 2 | 3 | 1.92e+03 | 1.18e+03 | No Benchmarks |
| Selenium | 3 | 3 | 4.70e+00 | **3.67e+00** | Vegetation |
| Sodium | 3 | 3 | 3.42e+02 | 1.91e+02 | No Benchmarks |
| Vanadium | 3 | 3 | 4.75e+01 | **3.85e+01** | Processes, Vegetation |
| Zinc | 3 | 3 | 4.00e+01 | **3.20e+01** | Processes, Vegetation |
| *Radionuclides (pCi/g)* | | | | | |
| Thorium-228 | 3 | 3 | 1.10e+00 | 1.06e+00 | No Benchmarks |
| Thorium-232 | 3 | 3 | 1.30e+00 | 1.12e+00 | No Benchmarks |
| Uranium-234 | 3 | 3 | 4.40e+00 | 3.73e+00 | No Benchmarks |
| Uranium-238 | 3 | 3 | 3.30e+00 | 2.83e+00 | No Benchmarks |

[a]Results are for detected analytes that had a maximum concentration greater than background soil and the soil benchmarks.
[b]Values in bold exceed at least one benchmark.
[c]Benchmarks presented in Sect. 6.3. "No Benchmarks" indicates analytes for which benchmarks have note been derived.

Table 6.52. Comparison of ash from the FCAP to toxicological benchmarks for soil biota[a]

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | |
| Aluminum | 15 | 15 | 3.29e+04 | **1.93e+04** | Processes, Vegetation |
| Arsenic | 15 | 15 | 1.93e+02 | **1.10e+02** | Vegetation |
| Barium | 15 | 15 | 6.30e+02 | 4.02e+02 | Vegetation |
| Calcium | 15 | 15 | 6.94e+03 | 4.97e+03 | No benchmarks |
| Chromium | 15 | 15 | 4.21e+01 | **2.14e+01** | Annelids, Processes, Vegetation |
| Cobalt | 15 | 15 | 2.98e+01 | 1.61e+01 | Vegetation |
| Copper | 15 | 15 | 1.01e+02 | **5.91e+01** | Processes, Vegetation |
| Iron | 15 | 15 | 8.82e+04 | **2.95e+04** | Processes |
| Magnesium | 15 | 15 | 2.40e+03 | 1.58e+03 | No benchmarks |
| Mercury | 14 | 15 | 1.30e+00 | **5.47e-01** | Annelids, Processes, Vegetation |
| Nickel | 14 | 15 | 5.60e+01 | **3.00e+01** | Vegetation |
| Potassium | 15 | 15 | 6.21e+03 | 4.04e+03 | No benchmarks |
| Selenium | 15 | 15 | 3.08e+01 | **1.12e+01** | Vegetation |
| Sodium | 15 | 15 | 1.08e+03 | 6.74e+02 | No benchmarks |
| Thallium | 4 | 15 | 3.20e+00 | **1.98e+00** | Vegetation |
| Vanadium | 15 | 15 | 1.23e+02 | **7.34e+01** | Processes, Vegetation |
| Zinc | 15 | 15 | 8.31e+01 | **4.64e+01** | Processes, Vegetation |
| *Radionuclides (pCi/g)* | | | | | |
| Thorium-228 | 15 | 15 | 2.70e+00 | 1.95e+00 | No benchmarks |
| Thorium-232 | 15 | 15 | 2.70e+00 | 1.92e+00 | No benchmarks |
| Uranium-234 | 15 | 15 | 4.80e+00 | 3.49e+00 | No benchmarks |

Table 6.52 (continued)

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| Uranium-235 | 15 | 15 | 3.50e-01 | 1.98e-01 | No benchmarks |
| Uranium-238 | 15 | 15 | 5.10e+00 | 3.47e+00 | No benchmarks |
| *Organics (mg/kg)* | | | | | |
| 2-Methylnaphthalene | 1 | 14 | 9.30e-02 | 6.55e-01 | No benchmarks |
| alpha-BHC | 1 | 3 | 3.00e-04 | 2.37e-03 | No benchmarks |
| bis(2-Ethylhexyl)Phthalate | 2 | 15 | 2.70e-01 | 6.40e-01 | No benchmarks |
| Butylbenzylphthalate | 4 | 14 | 1.10e-01 | 5.48e-01 | No benchmarks |
| Chloroform | 1 | 3 | 2.00e-03 | 1.40e-02 | No benchmarks |

[a] Results are for detected analytes that had a maximum concentration greater than background soil and the soil benchmarks.
[b] Values in bold exceed at least one benchmark.
[c] Benchmarks presented in Sect. 6.3. "No Benchmarks" indicates analytes for which benchmarks have note been derived.

Table 6.53. Comparison of ash from the UMB to toxicological benchmarks for soil biota[a]

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | |
| Aluminum | 6 | 6 | 2.08e+04 | **1.70e+04** | Processes, Vegetation |
| Arsenic | 6 | 6 | 5.92e+01 | **4.72e+01** | Vegetation |
| Barium | 6 | 6 | 6.84e+02 | 4.19e+02 | Vegetation |
| Cadmium | 5 | 6 | 4.20e+00 | 2.50e+00 | Processes, Vegetation |
| Calcium | 6 | 6 | 6.95e+03 | 4.31e+03 | No Benchmarks |
| Chromium | 6 | 6 | 1.82e+01 | **1.42e+01** | Annelids, Processes, Vegetation |
| Copper | 6 | 6 | 7.49e+01 | **4.34e+01** | Processes, Vegetation |
| Iron | 6 | 6 | 6.43e+04 | **3.77e+04** | Processes |
| Magnesium | 6 | 6 | 1.69e+03 | 1.39e+03 | No Benchmarks |
| Mercury | 6 | 6 | 7.30e-01 | **3.67e-01** | Annelids, Processes, Vegetation |
| Nickel | 6 | 6 | 4.07e+01 | **2.96e+01** | Vegetation |
| Phosphorus | 6 | 6 | 2.18e+03 | 9.41e+02 | No Benchmarks |
| Potassium | 6 | 6 | 3.90e+03 | 3.16e+03 | No Benchmarks |
| Selenium | 6 | 6 | 9.10e+00 | **3.62e+00** | Vegetation |
| Sodium | 6 | 6 | 3.84e+02 | 3.24e+02 | No Benchmarks |
| Vanadium | 6 | 6 | 8.37e+01 | **5.74e+01** | Processes, Vegetation |
| Zinc | 6 | 6 | 7.57e+01 | **5.57e+01** | Processes, Vegetation |
| *Radionuclides (pCi/g)* | | | | | |
| Thorium-228 | 1 | 1 | 9.20e+00 | 9.20e+00 | No Benchmarks |
| Thorium-234 | 1 | 1 | 6.30e+00 | 6.30e+00 | No Benchmarks |
| Uranium-238 | 6 | 6 | 4.37e+00 | 2.75e+00 | No Benchmarks |

[a] Results are for detected analytes that had a maximum concentration greater than background soil and the soil benchmarks.
[b] Values in bold exceed at least one benchmark.
[c] Benchmarks presented in Sect. 6.3. "No Benchmarks" indicates analytes for which benchmarks have note been derived.

Table 6.54. Comparison of Chestnut Ridge OU 2 soil-beneath-ash data to benchmarks[a]

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | |
| Aluminum | 3 | 3 | 1.14e+04 | **7.71e+03** | Processes, Vegetation |
| Arsenic | 3 | 3 | 5.12e+01 | **3.69e+01** | Vegetation |
| Calcium | 3 | 3 | 8.94e+02 | 6.80e+02 | No Benchmarks |
| Copper | 2 | 3 | 6.40e+00 | **6.17e+00** | Processes |
| Iron | 3 | 3 | 1.64e+04 | **1.45e+04** | Processes |
| Manganese | 3 | 3 | 1.92e+03 | **1.36e+03** | Processes, Vegetation |
| Selenium | 2 | 2 | 1.10e+00 | **1.04e+00** | Vegetation |
| Vanadium | 3 | 3 | 3.24e+01 | **2.77e+01** | Processes, Vegetation |
| *Radionuclides (pCi/g)* | | | | | |
| Thorium-228 | 3 | 3 | 8.40e-01 | 6.33e-01 | No Benchmarks |
| Thorium-230 | 1 | 3 | 1.40e+00 | 1.40e+00 | No Benchmarks |
| Thorium-232 | 3 | 3 | 9.20e-01 | 7.20e-01 | No Benchmarks |
| *Organics (mg/kg)* | | | | | |
| Butylbenzylphthalate | 2 | 3 | 5.30e-02 | 1.67e-01 | No Benchmarks |

[a] Results are for detected analytes that had a Maximum concentration greater than background soil and the soil benchmark.
[b] Values in bold exceed at least one benchmark.
[c] Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.55. Comparison of UMB soil-beneath-ash data to benchmarks[a]**

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | |
| Aluminum | 11 | 11 | 1.86e+04 | **9.81e+03** | Processes, Vegetation |
| Arsenic | 11 | 11 | 4.36e+01 | **1.94e+01** | Vegetation |
| Cadmium | 11 | 11 | 2.28e+01 | **3.95e+00** | Annelids, Processes, Vegetation |
| Calcium | 11 | 11 | 2.50e+05 | 4.61e+04 | No Benchmarks |
| Chromium | 10 | 11 | 5.06e+01 | **1.48e+01** | All |
| Cobalt | 10 | 11 | 2.95e+01 | 1.70e+01 | Annelids, Processes, Vegetation |
| Copper | 10 | 11 | 7.02e+01 | **1.85e+01** | Processes, Vegetation |
| Iron | 11 | 11 | 3.72e+05 | **4.99e+04** | Processes |
| Magnesium | 11 | 11 | 1.12e+04 | 2.84e+03 | No Benchmarks |
| Manganese | 11 | 11 | 2.38e+03 | **9.81e+02** | Processes, Vegetation |
| Nickel | 10 | 11 | 4.04e+01 | 2.19e+01 | Vegetation |
| Potassium | 10 | 11 | 2.19e+03 | 1.61e+03 | No Benchmarks |
| Selenium | 5 | 11 | 2.20e+00 | 6.18e-01 | Vegetation |
| Sodium | 11 | 11 | 6.26e+02 | 1.70e+02 | No Benchmarks |
| Vanadium | 11 | 11 | 9.40e+01 | **3.30e+01** | Processes, Vegetation |
| Zinc | 11 | 11 | 1.02e+02 | **7.07e+01** | Processes, Vegetation |
| *Radionuclides (pCi/g)* | | | | | |
| Thorium-234 | 2 | 2 | 5.15e+00 | 5.13e+00 | No Benchmarks |
| Uranium-238 | 11 | 11 | 2.35e+00 | 1.16e+00 | No Benchmarks |

[a] Results are for detected analytes that had a Maximum concentration greater than background soil and the soil benchmark.
[b] Values in bold exceed at least one benchmark.
[c] Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.56. Comparison of Sluice Channel Area ash nondetected analyte data to soil benchmarks[a]**

| Analyte | Detected | Analyzed | Max non-det | Min non-det[b] | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | | |
| Antimony | 0 | 3 | 1.56e+01 | **1.20e+01** | **1.35e+01** | Vegetation |
| Silver | 0 | 3 | 3.40e+00 | **2.60e+00** | **2.93e+00** | Vegetation |
| Thallium | 0 | 3 | 1.40e+00 | **1.00e+00** | **1.17e+00** | Vegetation |

[a] Results are for detected analytes that had a maximum non-det. concentration greater than background soil and the benchmarks.
[b] Values in bold exceed at least one benchmark.
[c] Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.57. Comparison of FCAP ash nondetected analyte data to soil benchmarks[a]**

| Analyte | Detected | Analyzed | Max non-det | Min non-det[b] | Mean[b] | Benchmarks exceeded[c] |
|---|---|---|---|---|---|---|
| *Inorganics (mg/kg)* | | | | | | |
| Antimony | 0 | 15 | 2.26e+01 | **1.26e+01** | **1.73e+01** | Vegetation |
| Cadmium | 0 | 15 | 3.10e+00 | 1.70e+00 | 2.19e+00 | Processes, Vegetation |
| Silver | 0 | 15 | 4.50e+00 | **2.40e+00** | **3.50e+00** | Vegetation |

[a] Results are for detected analytes that had a maximum non-det. concentration greater than background soil and the benchmarks.
[b] Values in bold exceed at least one benchmark.
[c] Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.58. Maximum concentrations of analytes in shallow groundwater exceeding background soil levels and benchmarks for phytotoxicity in solution**

| Analyte | Benchmark (ppm) | Groundwater (ppm) | HQ[a] |
|---|---|---|---|
| Aluminum | 0.5 | 21.8 | 44 |
| Copper | 0.03 | 0.07 | 2 |
| Iron | 10 | 33.5 | 4 |
| Manganese | 1 | 2.75 | 3 |
| Zinc | 0.4 | 2.07 | 5 |

[a] HQ = Hazard Quotient = concentration of analyte in sample/benchmark concentration.

**Table 6.59. Comparison of Sluice Channel Area shallow groundwater detected analyte data to vegetation benchmarks[a]**

| Analyte | Detected | Analyzed | Max det | Mean[b] | Benchmarks exceede |
|---|---|---|---|---|---|
| *Inorganics (mg/L)* | | | | | |
| Aluminum | 3 | 6 | 2.18e+01 | **7.62e+00** | Vegetation |
| Copper | 3 | 6 | 7.35e–02 | 2.18e–02 | Vegetation |
| Iron | 5 | 6 | 3.35e+01 | **1.14e+01** | Vegetation |
| Manganese | 6 | 6 | 2.75e+00 | **1.53e+00** | Vegetation |
| Zinc | 6 | 6 | 2.07e+00 | **4.15e–01** | Vegetation |
| *Radionuclides (pCi/L)* | | | | | |
| Radium-226 | 2 | 2 | 1.16e+00 | 9.99e–01 | No benchmark |
| Uranium-234 | 2 | 2 | 3.50e–01 | 2.85e–01 | No benchmark |
| Uranium-238 | 4 | 4 | 6.73e–01 | 4.11e–01 | No benchmark |
| *Organics (mg/L)* | | | | | |
| bis(2-Ethylhexyl)Phthalate | 1 | 2 | 1.00e–03 | 5.50e–03 | No benchmark |

[a] Results are for detected analytes that had a maximum detected concentration greater than the benchmark.
[b] Values in bold exceed at least one benchmark.
[c] Benchmarks presented in Sect. 6.3. "No Benchmarks" listed for analytes without a benchmark.

**Table 6.60. Hazard quotients for contaminants in ash of Chestnut Ridge OU 2 by site section**

| Contaminant | Sluice Channel Area | | | FCAP | | | UMB | | |
|---|---|---|---|---|---|---|---|---|---|
| | Plant | Earthworm | Microbe | Plant | Earthworm | Microbe | Plant | Earthworm | Microbe |
| Aluminum | 282 | n.d.[a] | 104 | 658 | n.d. | 244 | 416 | n.d. | 154 |
| Antimony | 3 | n.d. | n.d. | 5 | n.d. | n.d. | n.d. | n.d. | n.d. |
| Arsenic | 16 | n.d. | n.d. | 19 | n.d. | n.d. | 6 | n.d. | n.d. |
| Barium | n.d. | n.d. | n.d. | 1 | n.d. | n.d. | 1 | n.d. | n.d. |
| Cadmium | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 1 | <1 | 1 |
| Chromium | 9 | 9 | 4 | 21 | 21 | 11 | 9 | 9 | 5 |
| Cobalt | 1 | n.d. | <1 | 1 | n.d. | <1 | n.d. | n.d. | <1 |
| Copper | 1 | <1 | 12 | 2 | 1 | 17 | 1 | <1 | 12 |
| Iron | n.d. | n.d. | 654 | n.d. | n.d. | 315 | n.d. | n.d. | 230 |
| Manganese | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Mercury | 2 | 1 | 6 | 4 | 3 | 13 | 2 | 1 | 7 |
| Nickel | 2 | <1 | <1 | 2 | <1 | <1 | 2 | <1 | <1 |
| Selenium | 5 | n.d. | <1 | 30 | n.d. | <1 | 9 | n.d. | <1 |
| Silver | n.d. | 2 | <1 | 2 | n.d. | <1 | n.d. | n.d. | <1 |
| Thallium | n.d. | 1 | n.d. | 3 | n.d. | n.d. | n.d. | n.d. | n.d. |
| Vanadium | 19 | n.d. | 2 | 49 | n.d. | 6 | 33 | n.d. | 4 |
| Zinc | 2 | <1 | 3 | 3 | <1 | 6 | 2 | <1 | 4 |

[a]n.d. = Not determined: No benchmark or maximum concentration of analyte in ash did not exceed background soil level.

**Table 6.61. Hazard quotients for contaminants in soil of Chestnut Ridge OU2 by site section**

| Contaminant | Sluice Channel Area | | | UMB | | |
|---|---|---|---|---|---|---|
| | Plant | Earthworm | Microbe | Plant | Earthworm | Microbe |
| Aluminum | 228 | n.d.[a] | 84 | 372 | n.d. | 138 |
| Antimony | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Arsenic | 5 | n.d. | n.d. | 4 | n.d. | n.d. |
| Barium | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Cadmium | n.d. | <1 | n.d. | 8 | 1 | 8 |
| Chromium | n.d. | <1 | n.d. | 25 | 25 | 13 |
| Cobalt | n.d. | n.d. | <1 | 1 | n.d. | <1 |
| Copper | <1 | <1 | 1 | 1 | <1 | 12 |
| Iron | n.d. | n.d. | n.d. | n.d. | n.d. | 1329 |
| Manganese | 4 | n.d. | 19 | 5 | n.d. | 24 |
| Mercury | n.d. | <1 | n.d. | n.d. | n.d. | n.d. |
| Nickel | n.d. | <1 | <1 | 2 | <1 | <1 |
| Selenium | 1 | n.d. | <1 | 2 | n.d. | <1 |
| Silver | n.d. | n.d. | <1 | n.d. | n.d. | <1 |
| Thallium | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Vanadium | 13 | n.d. | 2 | 38 | n.d. | 5 |
| Zinc | n.d. | <1 | n.d. | 4 | <1 | 7 |

[a] n.d. = Not determined: No benchmark or maximum concentration of analyte did not exceed background soil level.

**Table 6.62. Summary of risk characterization for terrestrial plants**

| Evidence | Result[a] | Explanation |
|---|---|---|
| Biological survey | NA | No quantitative evaluation of growth, species composition, or diversity at site |
| Body burdens | NA | No organized body of data defining relationship between levels of chemicals in plants and toxic effects for comparison to limited data on vegetation concentrations |
| Media analyses | + | Al, Sb, As, Ba, Cd, Cr, Co, Cu, Fe, Mn, Hg, Ni, Se, Ag, V, and Zn are found in ash, soil, and shallow groundwater in concentrations high enough to cause reduced growth |
| Weight-of-evidence | + | Growth, and possible reproduction, of vegetation are likely to have been depressed by at least 20% in all three areas |

[a] + indicates that the evidence is consistent with the occurrence of the endpoint effect.
- indicates that the evidence is inconsistent with the occurrence of the endpoint effect.
$\underline{+}$ indicates that the evidence is too ambiguous to interpret.



**Table 6.63. Maximum concentrations of analytes in soil and ash exceeding background soil levels and benchmarks for toxicity to earthworms**

| Analyte | Benchmark (ppm) | Sluice Channel Area ash (ppm) | HQ[a] | FCAP ash (ppm) | HQ | UMB soil (ppm) | HQ | UMB ash (ppm) | HQ |
|---|---|---|---|---|---|---|---|---|---|
| Cadmium | 20 | NEB[b] | | NEB | | 22.8 | 1 | 4.2 | <1 |
| Chromium | 2 | 17.1 | 9 | 42.1 | 21 | 50.6 | 25 | 18.2 | 9 |
| Copper | 100 | 70.1 | <1 | 101 | 1 | 70.2 | <1 | 74.9 | <1 |
| Mercury | 0.5 | 0.61 | 1 | 1.30 | 3 | NEB | | 0.73 | 1 |

[a] HQ = Hazard Quotient: concentration of analyte in sample/benchmark concentration.
[b] NEB = concentration of analyte did not exceed background soil concentration.

**Table 6.64. Summary of risk characterization for earthworms**

| Evidence | Result[a] | Explanation |
|---|---|---|
| Biological survey | + | Earthworms are absent from the Sluice Channel Area and FCAP; UMB not sampled |
| Body burdens | NA | No earthworms found |
| Media analyses | + | Cd, Cr, Cu, Hg, Ag, and Tl are found in ash and soil in concentrations high enough to cause reduced growth or reproduction |
| Weight-of-evidence | + | Growth and reproduction of earthworms are likely to have been depressed by at least 20% in all three areas |

[a] + indicates that the evidence is consistent with the occurrence of the endpoint effect.
- indicates that the evidence is inconsistent with the occurrence of the endpoint effect.
± indicates that the evidence is too ambiguous to interpret.

**Table 6.65. Summary of risk characterization for soil/litter-dwelling invertebrates**

| Evidence | Result[a] | Explanation |
|---|---|---|
| Biological survey | + | Very few invertebrates of limited type were found in the ash of the Sluice Channel and FCAP |
| Body burdens | NA | Invertebrates not collected for analysis |
| Media analyses | - | Contaminants measured in vegetation of the Sluice and FCAP areas were not at levels considered to be toxic to litter-decomposing invertebrates |
| Weight-of-evidence | ± | Results from the two lines of evidence available are conflicting; however, the number of contaminants analyzed for in vegetation was small compared to the full list of those known to be at the site. Furthermore, the number of contaminants for which there are benchmarks is limited. The biological survey results suggest that the ash may be toxic to these organisms |

[a] + indicates that the evidence is consistent with the occurrence of the endpoint effect.
- indicates that the evidence is inconsistent with the occurrence of the endpoint effect.
± indicates that the evidence is too ambiguous to interpret.

**Table 6.66. Maximum concentrations of analytes in soil and ash exceeding background soil levels and benchmarks for toxicity to soil microorganisms and microbial processes**

| Analyte | Benchmark (ppm) | Sluice Channel Area soil (ppm) | HQ[a] | Sluice Channel Area ash (ppm) | HQ | FCAP ash (ppm) | HQ | UMB soil (ppm) | HQ | UMB ash (ppm) | HQ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Aluminum | 135 | 11400 | 84 | 14100 | 104 | 32900 | 244 | 18600 | 138 | 20800 | 154 |
| Cadmium | 3 | NEB[b] | | NEB | | NEB | | 22.8 | 8 | 4.2 | 1 |
| Chromium | 4 | NEB | | 17.1 | 4 | 42.1 | 11 | 50.6 | 13 | 18.2 | 5 |
| Copper | 6 | 6.4 | 1 | 70.1 | 12 | 101 | 17 | 70.2 | 12 | 74.9 | 12 |
| Iron | 280 | NEB | | 183000 | 654 | 88200 | 315 | 372000 | 1329 | 64300 | 230 |
| Manganese | 100 | 1920 | 19 | NEB | | NEB | | 2380 | 24 | NEB | |
| Mercury | 0.1 | NEB | | 0.61 | 6 | 1.3 | 13 | NEB | | 0.73 | 7 |
| Vanadium | 20 | 32.4 | 2 | 47.5 | 2 | 123 | 6 | 94.0 | 5 | 83.7 | 4 |
| Zinc | 15 | NEB | | 40.0 | 3 | 83.1 | 6 | 102 | 7 | 55.7 | 4 |

[a] HQ = Hazard Quotient: concentration of analyte in sample/benchmark concentration.
[b] NEB = concentration of analyte did not exceed background soil concentration.



**Table 6.67. Summary of risk characterization for soil microbes and microbial processes**

| Evidence | Result[a] | Explanation |
|---|---|---|
| Biological survey | NA | No evaluation of microbial populations or processes occurring at the site. |
| Media toxicity yests | + | Rhizobium leguminosarum biomass decreased by approximately 90% when cultured in ash from FCAP. |
| Media analyses | + | Al, Cd, Cr, Cu, Fe, Mn, Hg, V, and Zn are found in ash and soil in concentrations high enough to cause reduced microbial growth and reduction in enzyme activities involved in organic matter breakdown and nutrient cycling. |
| Weight-of-evidence | + | Growth and functioning of microbial populations are likely to have been depressed by at least 20% in all three areas. |

[a] + indicates that the evidence is consistent with the occurrence of the endpoint effect.

**Table 6.68. Hazard quotients for estimated total exposure received by white-footed mice[a]**

| | Hazard quotients | | | |
|---|---|---|---|---|
| Contaminant | FCAP | Sluice Channel Area | Upper McCoy Branch | Background |
| Aluminum | 17.25 | 10.77 | 15.15 | 7.89 |
| Arsenic | 3.41 | 3.23 | 1.02 | 0.91 |
| Barium | 0.58 | 0.54 | 0.73 | 0.15 |
| Beryllium | 0.0053 | 0.0019 | 0.0035 | 0.00093 |
| Cadmium | 0.48 | 0.14 | 0.028 | 0.11 |
| Chromium | 0.079 | 0.046 | 0.0046 | 0.047 |
| Copper | 0.0028 | 0.0028 | 0.0024 | 0.00029 |
| Fluoride | 0 | 0 | 0.013 | 0 |
| Lead | 0.0095 | 0.0089 | 0.0021 | 0.011 |
| Manganese | 0.0075 | 0.0086 | 0.023 | 0.06 |
| Mercury | 0.1 | 0.11 | 0.017 | 0.17 |
| Nickel | 0.00097 | 0.0011 | 0.00095 | 0.00023 |
| Niobium | 0 | 0 | 0.00089 | 0 |
| Nitrate | 0 | 0 | 0.0062 | 0 |
| Selenium | 30.89 | 2.64 | 0.16 | 1.32 |
| Thallium | 3.13 | 3 | 0 | 3 |
| Vanadium | 0.13 | 0.074 | 0.11 | 0.055 |
| Zinc | 0.0043 | 0.0027 | 0.0049 | 0.0034 |
| bis(2-Ethylhexyl)Phthalate | 0.00028 | 0 | 0.0044 | 0.00055 |
| Chloroform | 0 | 0 | 0 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0 |

[a]Only contaminants for which NOAELs have been derived are listed.

**Table 6.69. Hazard quotients for estimated total exposure received by short-tailed shrew[a]**

| Contaminant | Hazard quotients | | | |
|---|---|---|---|---|
| | FCAP | Sluice Channel Area | Upper McCoy Branch | Background |
| Aluminum | 706.36 | 454.46 | 638.21 | 332 |
| Arsenic | 85.88 | 104.22 | 38.9 | 3.78 |
| Barium | 24.38 | 22.8 | 30.34 | 5.86 |
| Beryllium | 0.23 | 0.082 | 0.15 | 0.039 |
| Cadmium | 0 | 0 | 1.19 | 0.59 |
| Chromium | 0.29 | 0.2 | 0.19 | 0.23 |
| Copper | 0.12 | 0.12 | 0.1 | 0.012 |
| Fluoride | 0 | 0 | 0.012 | 0 |
| Lead | 0.1 | 0.098 | 0.088 | 0.18 |
| Manganese | 0.16 | 0.36 | 0.92 | 2.49 |
| Mercury | 0.81 | 0.7 | 0.67 | 0.35 |
| Nickel | 0.041 | 0.046 | 0.04 | 0.0097 |
| Niobium | 0 | 0 | 0.00083 | 0 |
| Nitrate | 0 | 0 | 0.0058 | 0 |
| Selenium | 16.49 | 5.24 | 6.83 | 1.15 |
| Thallium | 5.56 | 0 | 0 | 0 |
| Vanadium | 5.57 | 3.11 | 4.75 | 2.3 |
| Zinc | 0.15 | 0.11 | 0.2 | 0.14 |
| bis(2-Ethylhexyl)Phthalate | 0.012 | 0 | 0 | 0.00051 |
| Chloroform | 0.00001 | 0 | 0 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0 |

[a] Only contaminants for which NOAELs have been derived are listed.

**Table 6.70. Summary of risk characterization for small mammals**

| Evidence | Result[1] | Explanation |
|---|---|---|
| Biological surveys | ± | Shrews were absent from the FCAP and only one was collected from the Sluice Channel Area; only two shrews were collected from the reference site; rodents appeared to be unaffected |
| Body burdens | - | Body burdens measured at CR OU 2 are less than those observed in other studies to have no effect |
| Media toxicity tests | NA | These tests are generally not performed for small mammals and are therefore unavailable |
| Literature toxicity data | + | Al, As, Ba, Cd, Se, Tl and V in ash, food, and water sufficient to cause reduced survival or reproduction |
| Weight-of-evidence | + | Abundance of shrews and possibly other small mammals is likely to have been depressed by at least 20% on the FCAP, Sluice Channel Area, or UMB |

[a] NA indicates data are not available.
+ indicates that the evidence is consistent with the occurence of the endpoint effect.
- indicates that the evidence is inconsistent with the occurence of the endpoint effect.
± indicates that the evidence is too ambiguous or provides insufficiant data to interpret.

**Table 6.71. Hazard quotients for estimated total exposure received by white-tailed deer[a]**

| | Hazard quotients: absolute maximum exposure | | | | | Hazard quotients: maximum proportional exposure | | |
|---|---|---|---|---|---|---|---|---|
| Contaminant | FCAP (ash for sodium)[b] | FCAP | Sluice Channel Area | Upper McCoy Branch | Background | FCAP | Sluice Channel Area | Upper McCoy Branch |
| Aluminum | 586.28 | 83.10 | 51.41 | 72.36 | 37.71 | 8.48 | 5.24 | 7.38 |
| Arsenic | 72.87 | 16.68 | 15.81 | 5.13 | 4.45 | 1.70 | 1.61 | 0.52 |
| Barium | 20.72 | 2.87 | 2.64 | 3.58 | 0.76 | 0.29 | 0.27 | 0.36 |
| Beryllium | 0.19 | 0.026 | 0.0095 | 0.017 | 0.0046 | 0.0027 | 0.0010 | 0.0017 |
| Cadmium | 0 | 2.38 | 0.72 | 0.14 | 0.57 | 0.24 | 0.073 | 0.015 |
| Chromium | 0.24 | 0.38 | 0.22 | 0.022 | 0.23 | 0.039 | 0.022 | 0.0023 |
| Copper | 0.10 | 0.014 | 0.014 | 0.012 | 0.0014 | 0.0014 | 0.0014 | 0.0012 |
| Fluoride | 0 | 0 | 0 | 0.084 | 0 | 0 | 0 | 0.0085 |
| Lead | 0.088 | 0.046 | 0.043 | 0.010 | 0.053 | 0.0047 | 0.0044 | 0.0010 |
| Manganese | 0.14 | 0.042 | 0.042 | 0.11 | 0.29 | 0.0043 | 0.0043 | 0.012 |
| Mercury | 0.69 | 0.49 | 0.54 | 0.082 | 0.83 | 0.8 | 0.057 | 0.0068 |
| Nickel | 0.035 | 0.0048 | 0.00537 | 0.0046 | 0.0011 | 0.00048 | 0.00054 | 0.00047 |
| Niobium | 0 | 0 | 0 | 0.0056 | 0 | 0 | 0 | 0.00053 |
| Nitrate | 0 | 0 | 0 | 0.039 | 0 | 0 | 0 | 0.0040 |
| Selenium | 13.78 | 150.72 | 12.89 | 0.78 | 6.43 | 15.33 | 1.31 | 0.078 |
| Thallium | 4.74 | 15.15 | 14.5 | 0 | 14.5 | 1.54 | 1.48 | 0 |
| Vanadium | 5.20 | 0.72 | 0.40 | 0.60 | 0.30 | 0.074 | 0.04 | 0.062 |
| Zinc | 0.13 | 0.022 | 0.013 | 0.024 | 0.017 | 0.0022 | 0.0013 | 0.0024 |
| bis(2-Ethylhexyl)Phthalate | 0.05 | 0.0015 | 0 | 0.029 | 0.0036 | 0.00018 | 0 | 0.0029 |
| Chloroform | 0.00001 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0 | 0.00002 | 0 | 0 | 0 |

[a] Only contaminants for which NOAELs have been derived are listed.
[b] HI for contaminant exposure through consumption of ash only. For total hazard to deer at the FCAP, HIs for ash for sodium and for FCAP (columns 2 and 3) should be added.

**Table 6.72. Hazard quotients for estimated total exposure received by red fox[a]**

| | Hazard quotients: absolute maximum exposure | | | | Hazard quotients: maximum proportional exposure | | |
|---|---|---|---|---|---|---|---|
| Contaminant | FCAP | Sluice Channel Area | Upper McCoy Branch | Background | FCAP | Sluice Channel Area | Upper McCoy Branch |
| Aluminum | 26.32 | 17.65 | 22.11 | 11.54 | 1.55 | 0.93 | 1.31 |
| Arsenic | 4.17 | 4.25 | 1.69 | 0.48 | 0.25 | 0.25 | 0.099 |
| Barium | 0.86 | 0.80 | 1.07 | 0.25 | 0.05 | 0.046 | 0.063 |
| Beryllium | 0.0076 | 0.0028 | 0.0050 | 0.0013 | 0.00047 | 0.00014 | 0.00028 |
| Cadmium | 0.091 | 0.13 | 0.041 | 0.12 | 0.0055 | 0.0076 | 0.0024 |
| Chromium | 14.95 | 1.31 | 0.066 | 3.80 | 0.093 | 0.076 | 0.0039 |
| Copper | 0.0041 | 0.0042 | 0.0035 | 0.00045 | 0.00024 | 0.00024 | 0.00021 |
| Fluoride | 0 | 0 | 0.049 | 0 | 0 | 0 | 0.0029 |
| Lead | 0.14 | 0.037 | 0.0031 | 0.065 | 0.0081 | 0.0022 | 0.00018 |
| Manganese | 0.019 | 0.026 | 0.035 | 0.087 | 0.0011 | 0.00072 | 0.0021 |
| Mercury | 1.18 | 5.32 | 0.052 | 5.61 | 0.070 | 0.31 | 0.0022 |
| Nickel | 0.0014 | 0.00165 | 0.0014 | 0.00033 | 0.00008 | 0.00009 | 0.00008 |
| Niobium | 0 | 0 | 0.0033 | 0 | 0 | 0 | 0.0002 |
| Nitrate | 0 | 0 | 0.023 | 0 | 0 | 0 | 0.0013 |
| Selenium | 29.99 | 13.30 | 0.23 | 9.05 | 1.75 | 0.78 | 0.013 |
| Thallium | 4.67 | 4.25 | 0 | 4.25 | 0.27 | 0.25 | 0 |
| Vanadium | 0.20 | 0.11 | 0.16 | 0.081 | 0.012 | 0.0062 | 0.0095 |
| Zinc | 0.0075 | 0.0062 | 0.0072 | 0.0050 | 0.00044 | 0.00023 | 0.00042 |
| bis(2-Ethylhexyl)Phthalate | 0.0004 | 0 | 0.016 | 0.002 | 0.00004 | 0 | 0.00092 |
| Chloroform | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0.00001 | 0 | 0 | 0 |

[a] Only contaminants for which NOAELs have been derived are listed.

**Table 6.73. Hazard quotients for estimated total exposure received by red-tailed hawk[a]**

| Contaminant | Hazard quotients: absolute maximum exposure | | | | Hazard quotients: maximum proportional exposure | | |
|---|---|---|---|---|---|---|---|
| | FCAP | Sluice Channel Area | Upper McCoy Branch | Background | FCAP | Sluice Channel Area | Upper McCoy Branch |
| Aluminum | 0.0078 | 0 | 0.00042 | 0.00037 | 0.00018 | 0 | 0.00001 |
| Arsenic | 4.22 | 2.30 | 0.13 | 1.20 | 0.094 | 0.052 | 0.003 |
| Cadmium | 0.011 | 0.016 | 0 | 0.012 | 0.00025 | 0.00035 | 0 |
| Chromium | 1.73 | 1.43 | 0 | 4.33 | 0.039 | 0.032 | 0 |
| Copper | 0.00018 | 0 | 0.00013 | 0.0001 | 0 | 0 | 0 |
| Lead | 0.62 | 0.16 | 0.00003 | 0.28 | 0.014 | 0.0035 | 0 |
| Mercury | 5.49 | 25.8 | 0.0016 | 27.36 | 0.12 | 0.58 | 0 |
| Nickel | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Selenium | 20.96 | 9.35 | 0 | 6.42 | 0.47 | 0.21 | 0 |
| Vanadium | 0.00009 | 0 | 0 | 0.00003 | 0 | 0 | 0 |
| Zinc | 0.00368 | 0 | 0.00058 | 0.00032 | 0.00008 | 0 | 0.00001 |
| bis(2-Ethylhexyl)Phthalate | 0 | 0 | 0 | 0.006 | 0 | 0 | 0.0011 |
| Di-N-butylphthalate | 0 | 0 | 0 | 0.011 | 0 | 0 | 0 |

[a] Only contaminants for which NOAELs have been derived are listed.

Table 6.74. Summary of risk characterization for reservation-wide endpoint species

| Evidence | | Result[a] | Explanation |
|---|---|---|---|
| Biological surveys | | NA | Surveys for wide-ranging species not performed<br><br>Incidental observations indicate that white-tailed deer use Chestnut Ridge OU 2<br><br>Red fox and red-tailed hawks are common throughout the Oak Ridge Reservation, therefore it is likely that they use Chestnut Ridge OU 2 |
| Media toxicity tests | | NA | Test not performed for wide-ranging wildlife species |
| Literature toxicity data | | | |
| White-tailed deer | Maximum[b] | + | Al, As, Ba, Cd, Se, and Tl in ash, food, and water sufficient to cause reduced survival or reproduction |
| | Proportional[c] | + | Al, As, Se, and Tl in ash, food, and water sufficient to cause reduced survival or reproduction |
| Red fox | Maximum | + | Al, As, Ba, Cr, Hg, Se, and Tl in ash, food, and water sufficient to cause reduced survival or reproduction |
| | Proportional | + | Al and Se in ash, food, and water sufficient to cause reduced survival or reproduction |
| Red-tailed hawk | Maximum | + | As, Cr, Hg, and Se in ash, food, and water sufficient to cause reduced survival or reproduction |
| | Proportional | - | Contaminants in ash, food, and water not sufficient to cause reduced survival or reproduction |
| Weight-of-evidence | | | |
| White-tailed deer | | + | survival or reproduction of individual white-tailed deer that use Chestnut Ridge OU 2 is likely to be depressed under either exposure scenario |
| Red fox | | + | survival or reproduction of individual red fox that use Chestnut Ridge OU 2 is likely to be depressed under either exposure scenario |
| Red-tailed hawk | | +<br><br>and<br>- | survival or reproduction of individual red-tailed hawks that use Chestnut Ridge OU 2 is likely to be depressed under the maximum exposure scenario<br><br>survival or reproduction of individual red-tailed hawks that use Chestnut Ridge OU 2 is ***not*** likely to be depressed under proportional exposure scenario |

[a] NA indicates data are not available.
+ indicates that the evidence is consistent with the occurence of the endpoint effect.
- indicates that the evidence is inconsistent with the occurence of the endpoint effect.
± indicates that the evidence is too ambiguous to interpret.
[b] Literature toxicity data compared to absolute maximum exposure estimate.
[c] Literature toxicity data compared to maximum proportional exposure estimate.

# 7. PRELIMINARY REMEDIATION GOALS

## 7.1 INTRODUCTION

Section 300.430(e) of the NCP calls for the development of PRGs before the onset of the FS process. PRGs are chemical-specific, medium-specific numerical concentration limits that address COCs at the site and all pathways found to be of concern during the RI process. PRGs are provided to the remedial design staff as long-term targets to use during the design and comparison of remedial action alternatives. Ideally, these goals will comply with the ARARs and leave residual risks at the site that are fully protective of human health and the environment as required by the NCP.

Remedial Action Objectives (RAOs) are site-specific, quantitative targets that define the extent of cleanup required to achieve a CERCLA response action. RAOs address COCs, media of concern, potential exposure pathways, and remediation goals. These objectives are used by engineers as design criteria during the alternative development and selection process.

## 7.2 CHESTNUT RIDGE OU 2 RAOs

The general remediation goals for the Chestnut Ridge OU 2 site are to protect human health and the environment and comply with the ARARs. The following are RAOs that will attain these goals.

- Control leaching of contaminants from ash to surface water above TDEC ambient water quality standards.
- Minimize direct contact by humans and biota with the ash.
- Control potential future failure of the dam.
- Preserve the local habitat in the long term.

The primary migration pathway resulting in risk is leaching of contaminants from the ash to surface waters. Direct access to the ash and contaminated soil also presents an environmental and human health risk. Additionally, significant failure of the dam would result in a loss of aquatic and terrestrial habitat through the release of ash. However, because of the ecological setting of Chestnut Ridge OU 2, the RAOs must be balanced against one another. Meeting inflexible RAOs could cause more damage to the environment than current or future risk posed by the contaminants present.

For example, the first objective developed above is to control leaching of contaminants. Currently, the levels of contaminants in the surface waters of UMB exceed some TDEC standards (see Table 4.2). These standards are designed to protect the aquatic environment and are ARARs for this site. However, the removal or containment of the ash, contaminated soils, and sediments to achieve ambient water quality standards, would destroy habitats at the site and may compromise the purpose of the standards—to protect aquatic organisms.

Other elements of the objectives include preventing dam failure and the further release of ash and preserving the local habitat. There are no quantitative levels/objectives associated with controlling dam failure or preserving the local habitat. The PRGs and the corresponding alternative actions are more fully developed in the FS report.

Elements of the objectives include preventing dam failure and the further release of ash; controlling exposure to the ash, soil, and surface water; and minimizing leaching of contaminants to the surface waters. The PRGs and the corresponding alternative actions are more fully developed in the FS report.

## 7.3 RISK-BASED PRGs

There are very few chemical-specific ARARs identified for Chestnut Ridge OU 2. Therefore, the results of the ERAs and HHRAs provide the PRGs in both quantitative and qualitative terms. PRGs have been developed for COCs identified as having unacceptable carcinogenic and/or noncarcinogenic/systemic effects to human health (i.e., a risk greater than 1.0E-04 and/or an HI $\geq$ 1.0) and are presented in Table 7.1. For comparison, the representative concentration of the COCs in the various media at the site are presented in Table 7.2. Specific PRGs have not been developed for reduction of ecological risk. The COCs likely to produce a reduction of >20% in selected ERA endpoint populations are contained in Chap. 6, Tables 6.39, 6.44, 6.62, 6.64, 6.67, 6.70, and 6.74. Because specific numerical toxicity values are not available for all contaminants and species of concern, the goals are constructed in qualitative terms. Federal and state ambient water quality criteria are available for consideration as ARARs for evaluation of ecological risk reduction.

Potential health effects resulting from exposures to radioactive and chemical contaminants are divided into two categories—carcinogenic and noncarcinogenic effects. For carcinogens, EPA had identified a target range for incremental risks of $1 \times 10^{-6}$ to $1 \times 10^{-4}$ (or 1 in 1 million to 1 in 10,000) to limit the possibility that an individual could develop cancer from exposures to residual contaminants at an NPL site (EPA 1990). The range is used as a point of reference for the risk estimates presented in this analysis. For noncarcinogenic contaminants, EPA has identified a hazard index, which equals 1 as the level of concern for potential adverse health effects associated with site exposures (EPA 1989a). The risk conditions under which remedial action is required are as follows:

| | HI < 1 | HI > 1 |
|---|---|---|
| Risk < $1 \times 10^{-6}$ | No | Yes |
| $1 \times 10^{-6}$ < Risk < $1 \times 10^{-4}$ | Maybe | Yes |
| Risk > $1 \times 10^{-4}$ | Yes | Yes |

Although the upper end of the target range is generally used to make a risk management decision to determine whether a remedial action is warranted, EPA does not consider $1 \times 10^{-4}$ a discrete limit [i.e., risks above that level may be considered acceptable on the basis of site-specific conditions (EPA 1991)]. For example, the presence of $^{228}$Th in the ash at OU 2 represents a somewhat unique situation when compared with typical CERCLA sites. The $^{228}$Th is naturally occurring in eastern U.S. coal ash and the ash is not a regulated hazardous waste (i.e., disposal off site would not be required to address risks due to exposure or ingestion of the ash.

**Table 7.1 Preliminary remediation goals for COCs identified for Chestnut Ridge OU 2**

| COC | Medium | PRG at HI = 1[a] | PRG at risk = $10^{-6}$ | ARAR | Units of measure |
|---|---|---|---|---|---|
| Arsenic | ash/soil/sediment | 82 | - | 50[b] | mg/kg |
| Beryllium | ash/soil/sediment | 1400 | 0.15 | - | mg/kg |
| Cadmium | ash/soil/sediment | 270 | - | 5.0[b] | mg/kg |
| Manganese | ash/soil/sediment | 38,000 | - | - | mg/kg |
| Mercury | ash/soil/sediment | 82 | - | 1.0[b] | mg/kg |
| Thorium-228 | ash/soil/sediment | - | 0.0074 | - | pCi/g |
| Arsenic | water | 0.011 | - | 0.05[c](190)[d] | mg/L |
| Beryllium | water | 0.180 | 0.00002 | 0.004[c] | mg/L |
| Cadmium | water | 0.018 | - | 0.005[c] | mg/L |
| Manganese | water | 0.180 | - | 0.05[e] | mg/L |
| Mercury | water | 0.011 | - | 0.002[c](0.012)[d] | mg/L |
| Thorium-228 | water | - | 0.87 | 15[f] | pCi/L |

[a]All risk-based PRGs are calculated from the residential scenario. The water PRGs are from the ingestion pathway only; the nonradionuclide soil PRGs are also from the ingestion pathway only, while the radionuclide soil PRG is from the ingestion plus external exposure pathways.

[b]NOAA action levels for COCs in sediment/soil/ash. Overall apparent effects threshold value.

[c]MCLs from Federal (40 CFR 141) and State (TDEC 1200-5-1) regulations.

[d]TDEC water quality criteria for protection of freshwater organisms. Four-day average concentration not to be exceeded more than once every 3 years. Arsenic value is for Arsenic (III).

[e]Secondary MCL, no primary MCL/MCLG available.

[f]Gross alpha particle activity as pCi/L.

**Table 7.2 Representative concentrations[a] of COCs in Chestnut Ridge OU 2 media**

| Medium | Units of measure | Arsenic | Beryllium | Cadmium | Maganese | Mercury | Thorium-228 |
|---|---|---|---|---|---|---|---|
| SCA ash | mg/kg | 159 | 1.60 | - | 344 | - | 1.10[b] |
| FCAP ash | mg/kg | 131 | 4.39 | - | 152 | - | 2.17 |
| UMB ash | mg/kg | 59.2 | 2.90 | 3.67 | 876 | 0.581 | 9.20 |
| SCA soil | mg/kg | 51.2 | - | - | - | - | 0.840 |
| UMB soil | mg/kg | 27.1 | 0.919 | 7.48 | - | - | - |
| FCAP sediment | mg/kg | 76.60 | 2.10 | - | - | - | 0.730 |
| UMB sediment | mg/kg | 2,300 | 4.70 | - | 13,700 | - | 4.30 |
| FCAP sediment | mg/L | 0.002 | - | - | 1.70 | - | 0.22[c] |
| UMB surface water | mg/L | 0.075 | - | - | 0.514 | - | 0.041 |
| OU 2 groundwater | mg/L | 0.005 | 0.0014 | 0.0048 | 1.00 | - | 0.121 |

[a]The representative concentration is the smallest of two values: the maximum detected concentration versus the upper 95% confidence limit on the mean.
[b]Radionuclides in soil, sediment, and ash in pCi/g.
[c]Radionuclides in water in pCi/L.

# 8. SUMMARY

The Y-12 Plant is a major facility on the DOE ORR in Anderson County, Tennessee. An RI/FS is being conducted at the site in response to CERCLA, as amended by SARA. This RI Report addresses contamination in Chestnut Ridge OU 2 at the Y-12 Plant, which consists of the FCAP, Sluice Channel Area, and UMB.

The specific objectives for the RI are enumerated in Chap. 1. To provide data to address these objectives, environmental sampling programs were conducted as a part of the RI/FS. Additional data collected as part of previous studies of the ecological impacts of the facility were also evaluated. Sampling and analysis and the collection of other site data are described in Chap. 2. The data were used to characterize the three zones that comprise the OU, as well as the environmental setting.

A description of the site history and current conditions is presented in Chap. 3. This chapter contains sufficient information on which to make the determinations required by NEPA. The nature and extent of contamination in Chestnut Ridge OU 2 and surrounding environmental media, including underlying native soils, groundwater, drainage sediments, surface water, and biota, were characterized as described below.

Coal ash samples were collected from UMB during the Phase 1 effort and from the FCAP and the Sluice Channel Area during the Phase 2 effort. Also, the quantity and distribution of the coal ash were determined by measuring the depth and extent of the ash along surveyed transects. The Phase 1 effort established transects perpendicular to UMB. The Phase 2 effort established transects perpendicular to the dam in the FCAP zone and measured the depth of the ash in nine boreholes.

Results of chemical analyses of the Chestnut Ridge OU 2 coal ash indicated that it is typical of coal ash from the combustion of eastern United States coals. Analytical results of the Chestnut Ridge OU 2 coal ash are within the range of eastern United States coal ash for all the parameters reported. A comparison of FCAP ash to eastern U.S. coal ash is made in Table 3.6.

It is estimated from the measurements of the ash depth and areal extent that there are ~236,000 $yd^3$ of coal ash present in the FCAP, 6000 $yd^3$ in the Sluice Channel Area, and 4300 $yd^3$ in UMB. The total ash volume is estimated to be 246,300 $yd^3$.

Surface waters were characterized by sampling and analysis in Phase 1 and 2 of the RI. The Phase 1 results showed a clear trend of reduced contaminant concentrations with progression downstream. The UMB Phase 2 surface water data indicate that reference concentrations were exceeded for six metals and two organic compounds. Radiological data for UMB and the reference streams were primarily nondetections with the UMB data not exceeding the reference location data. A trend of reduced contaminant concentrations was observed in Phase 2 data also.

Sediment samples were collected and analyzed in both the Phase 1 and Phase 2 investigations. Al, Ba, Fe, Mn, K, and Na in the sediments of Chestnut Ridge OU 2 were well above the reference sediment results. These high levels of these metals are typically associated with coal and coal ash. The $^{238}U$ and $^{232}Th$ results for the on-site sediments were elevated

relative to the reference sediments results. These are the most common isotopes of uranium and thorium and, being typical trace elements of coal and coal ash, are expected to occur in elevated concentrations, along with the other heavy metals in the on-site sediments. Four organic compounds also were detected in the sediments: 4-methylphenol, butylbenzylphthalate, di-n-butylphthalate, and 2-butaznone.

Both phases of the Chestnut Ridge OU 2 RI characterized groundwater quality through sampling of monitoring wells and laboratory analysis of the samples.

The Phase 1 data suggest that former activities at Chestnut Ridge OU 2 may have had an impact on the groundwater, particularly with respect to metals, but, in general, the concentrations are quite low and typically do not exceed primary and secondary drinking water standards. Concentrations of parameters in Phase 1 were generally higher in samples from the overburden wells than in samples from the bedrock wells. Phase 1 included installation of monitoring wells.

No additional monitoring wells were installed during Phase 2. Many of the same wells sampled in Phase 1 were included in the Phase 2 investigation. The results from Phase 2 were similar to the Phase 1 groundwater results in that although metals, organics, and radionuclides were found in groundwater samples from the study area, they are present at levels below federal MCLs; the Tennessee SMCL for manganese was exceeded in four wells. In general, the presence and extent of contaminants in groundwater at Chestnut Ridge OU 2 are limited.

Soils beneath the ash in the Sluice Channel Area, FCAP, and UMB zones were sampled and analyzed to ascertain whether contaminants are leaching from the ash into the soils.

The analytical results of the Phase 1 soils beneath the ash samples show that all of the metals, except mercury and uranium, exceeded the reference soil mean in one or more of the samples. Iron is the only contaminant that exceeded reference soil mean in every sample. Arsenic was significantly elevated in 8 of the 12 samples. The lower inorganic concentrations relative to the soils data for the upper two zones indicate that the ash has been subjected to a leaching process before migrating into the UMB zone.

The results for FCAP and Sluice Channel Area soils beneath the ash from Phase 2 are similar. The mean of the FCAP and Sluice Channel Area sample results for each of the metals, except mercury and uranium, exceeded the reference soil mean. Individual results for AL, As, and Fe exceeded the reference soil mean in all samples. Three organic compounds were detected at trace levels: bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 2-methylnaphthalene, and 4-nitrophenol. The soil contaminants identified in Phase 2 were similar to the results from Phase 1 and confirm these findings with respect to the entire OU.

Biological monitoring data include media toxicity (aquatic, soil/sediment/ash), benthic macroinvertebrates survey of UMB, ash invertebrate survey, and heavy metal bioaccumulation in vegetation and small mammals.

The fathead minnow tests provide no strong evidence for acute or chronic toxicity of water at any of the tested sites. The results of the *Ceriodaphnia* tests also provided no strong evidence for toxicity. The snail test results provide no evidence for acute toxicity either from the water or sediments of UMB. It is possible that longer exposures to these media would

reveal snail behavioral differences that might be related to the presence of contaminants. The two earthworm tests showed that FCAP ash may be problematic for soil organisms.

The analyses of the water quality factors showed that the UMB sites that were tested for toxicity could be distinguished from one another statistically based on factors such as conductivity, alkalinity, and hardness. However, these analyses do not provide evidence for or against the presence of toxicants. Additionally, since no strong evidence for toxicity was detected, no relationships could be established between toxicity and measurements of pH, conductivity, alkalinity, or hardness.

Data from the two benthic macroinvertebrate surveys conducted in July 1992 and March 1993 exhibited no evidence of strong impact at either UMB site.

Chromium and selenium were the metals found in highest concentrations in plant tissues from sites at the FCAP. Concentrations of selenium in plants on the FCAP are high enough to warrant further study of possible toxicity effects on small mammals and soil invertebrates. Average concentrations of As, Se, and Pb found in small mammals collected at Chestnut Ridge OU 2 are higher than concentrations of those metals found in small mammals from the reference site. Concentrations and tolerance limits exceed levels that may pose a toxicological risk to the small mammals and their predators.

Chapter 4 presents the ARARs identified for Chestnut Ridge OU 2. The ARARs address chemical-specific and location-specific requirements. Action-specific ARARs will be determined once remedial alternatives have been developed in the FS process. The principal ARARs identified pertain to the ash being a nonhazardous solid waste, ambient water quality criteria, and wetlands protection.

Evaluation of potential contaminant migration pathways (soil, ash, sediment, surface water, and groundwater) are presented in Chaps. 5 and 6. The baseline HHRA, described in Chap. 5, addresses potential risks to human health associated with contaminants identified at the site. Risks are quantified for two land use scenarios:

- trespasser/hunter-current land use with passive land use controls and
- residential-future land use without access controls, assuming reasonable maximum exposure.

The baseline ERA is presented in Chap. 6. Ecological risks were characterized for aquatic biota, sediment-associated biota, terrestrial plants, soil invertebrates, soil microbes, and wildlife.

PRGs are presented in Chap. 7. With the exception of the ambient water quality criteria for surface waters, very few chemical-specific ARARs were identified for Chestnut Ridge OU 2. Therefore, the risk assessments were used to provide PRGs. Information relating to the PRGs (COPCs, ARARs, HHRA, and ERA) was communicated among the RI/FS team during the concurrent development of the RI and FS reports.

Sections 8.1 and 8.2 provide summaries of the nature and extent of contamination, contaminant migration, baseline HHRA, and baseline ERA.

## 8.1 NATURE AND EXTENT OF CONTAMINATION

Organic compounds were detected in the Chestnut Ridge OU 2 media at low concentrations and low frequencies, with bis(2-ethylhexyl)phthalate, diethylphthalate, di-n-butylphthalate, and phenol being the most common. Bis(2-ethylhexyl)phthalate is a common laboratory contaminant and is not believed to be associated with Chestnut Ridge OU 2 operations. However, no statistical test/analysis allowed bis(2-ethylhexyl)phthalate to be eliminated from the list of COPCs.

A number of inorganic constituents were detected in the soil, ash, surface water, sediments, and groundwater above background concentrations. The most frequent detections were for Al, As, Fe, Mg, and Zn.

As, Be, $^{228}$Th, and Mn are the most important waste-related constituents of Chestnut Ridge OU 2. Analysis of the underlying native soils indicated that inorganic contaminants have migrated from the waste into the soils. Elevated levels of the COPCs were also measured in the surface waters, sediments, spring waters, and groundwaters.

Groundwater data from Chestnut Ridge OU 2 suggest that former activities at the site have had some impact on the groundwater, but the impact is quite limited. Data from both phases included only one MCL exceedence; a duplicate sample from GW-676 exceeded the gross alpha MCL of 15 pCi/L. In general, the presence and extent of organics, metals, and radionuclides in groundwater at OU 2 is quite limited.

Contaminant migration from Chestnut Ridge OU 2 is limited. Transport of contaminants off-site is limited because of the inclusion of UMB within Chestnut Ridge OU 2. The surface water data indicated that most contaminants are settling out of the water column before exiting Chestnut Ridge OU 2. Sediment transport is a potential pathway for contaminant migration. The groundwater data did indicate groundwater to be a limited contaminant migration pathway. The ERA did identify potential transport of contaminants off-site by wildlife.

## 8.2 RISK ASSESSMENT

For the Chestnut Ridge OU 2 HHRA, two risk scenarios were evaluated. Elevated risks (carcinogenic greater than $10^{-4}$ or HI equal to or greater than 1.0) associated with the scenarios and their respective pathways derive from the following contaminants:

- Trespasser scenario
  - ash external exposure: $^{228}$Th
- Residential scenario
  - ash external exposure: $^{228}$Th
  - ash incidental ingestion: As
  - soil beneath the ash external exposure: $^{228}$Th
  - sediment incidental ingestion: As

- surface (pond) water ingestion: As, Mn
- sediment external exposure: $^{228}$Th
- groundwater ingestion: Be, Mn
- homegrown garden vegetable/fruit ingestion: As, Cd, Mn, Hg

The cumulative risk or HI by media for the two scenarios evaluated are shown in Tables 8.1 and 8.2. In summary, the pathways of exposure, and thus the risks to human health, are eliminated for al media and contaminants, except $^{228}$Th in ash, with the scenario that assumes access controls are maintained.

The ecological risks characterization indicates the potential for adverse effects from the COCs at the following endpoints:

- Fish—Al, Cr, Cu, Fe Mn, and Zn concentrations in the waters of the FCAP ponds and UMB.
- Benthic macroinvertebrates—as in FCAP sediments and Al, Cr, Cu, Fe, Pb, Mn, Zn concentrations in FCAP surface waters. Ba, Mn, and Ni in UMB sediments and Al, As, Cu, Fe, Pb, Hg, and Zn concentrations in UMB surface waters.
- Plants—Al, Sb, As, Ba, Cd, Cr, Co, Cu, Fe, Mn, Hg, Ni, Se, Ag, V, and Zn concentrations in ash soil and shallow groundwater in the FCAP, Sluice Channel Area, and UMB.
- Soil invertebrates—Cd, Cr, Cu, Hg, and Tl concentrations in the ash and soil of the FCAP and Sluice Channel Area.
- Soil microbes and microbial processes—Al, Cd, Cr, Cu, Fe, Mn, V, and Zn concentrations in ash and soil of the entire OU 2.
- Small mammals—Al, As, Ba, Cd, Se, Tl, and V concentrations in ash, food, and water in the entire OU 2.
- Reservation-wide endpoints—metals concentrations exposures for white-tailed deer (Al, As, Ba, Cd, Se, and Tl), red fox (Al, As, Ba, Cr, Hg, Se, and TL) and red-tailed hawk (As, Cr, Hg, and Se) in ash, food, and water in the entire OU 2.

The ERA for the Chestnut Ridge OU 2 determined that all lines of evidence available for fish, benthic macroinvertebrates, plants, soil invertebrates, small mammals, and wide-ranging mammals indicate the potential for adverse effects from the current inorganic contaminant levels at the site. The ecological risks associated with the future scenario are identical to the current risk scenario, given that a catastrophic failure of the dam does not occur.

**Table 8.1 Cumulative risk for Chestnut Ridge OU 2—trespasser scenario**

| Analyte[a] | Representative concentration[b] (pCi/g) | External exposure risk |
|---|---|---|
| SCA ash | | |
| Thorium-228 | 1.10E+00 | 7.1E-06 |
| Total pathway risk | | 7.2E-06[c] |
| FCAP ash | | |
| Thorium-228 | 2.17E+00 | 1.4E-05 |
| Total pathway risk | | 1.4E.05[c] |
| UMB ash | | |
| Thorium-228 | 9.20E+00 | 5.9E-05 |
| Total pathway risk | | 6.0E-05[c] |

[a]Only analytes that exceed $10^{-6}$ risk.
[b]The representative concentration is the smallest of two values—the maximum detected concentration versus the upper 95% confidence limit on the mean.
[c]The sum of the risks from all radionuclides for this pathway.

**Table 8.2 Cumulative risk and HI for Chestnut Ridge OU 2—residential scenario**

| Medium | Cumulative risk* | HI* |
|---|---|---|
| Sluice Channel Area ash | 1.7E-04 | 2.1E+00 |
| FCAP ash | 3.4E-04 | 1.8E+00 |
| UMB ash | 1.3E-03 | 9.2E-01 |
| Sluice Channel Area soil | 1.1E-04 | 6.4E-01 |
| UMB soil | 9.5E-06 | 3.9E-01 |
| FCAP sediment | 1.2E-04 | 1.0E+00 |
| UMB sediment | 6.3E-04 | 2.9E+01 |
| FCAP surface water | 9.8E-07 | 1.8E+01 |
| UMB surface water | 5.4E-05 | 1.8E+01 |
| Groundwater | 1.6E-04 | 1.2E+01 |

*This cumulative risk/HI does not include ingestion of leafy vegetables.

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