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Waste Area Grouping 2 Phase I Task Data Report: Ecological Risk Assessment and White Oak Creek Watershed Screening Ecological Risk Assessment

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FOR THE UNITED STATES
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Energy Systems Environmental Restoration Program

Waste Area Grouping 2 Phase I Task Data Report: Ecological Risk Assessment and White Oak Creek Watershed Screening Ecological Risk Assessment

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PREFACE

This report, *Waste Area Grouping 2 Phase I Task Data Report: Ecological Risk Assessment and White Oak Creek Watershed Screening Ecological Risk Assessment (ORNL/ER–366)*, was prepared in accordance with requirements under the Comprehensive Environmental Response, Compensation, and Liability Act. This work was performed under Work Breakdown Structure 1.4.12.6.1.02.40.08.05 (Activity Data Sheet 3326). Publication of this document meets a project deliverable of May 15, 1996. This document is one of five reports issued in 1996 that provide follow-up information to the *Phase I Remedial Investigation Report for Waste Area Grouping (WAG) 2 at the Oak Ridge National Laboratory*. This report presents results of a screening ecological risk assessment for WAG 2 and, as far as available data permit, for other areas in White Oak Creek watershed. A screening assessment is intended to screen out chemicals that are not hazardous and receptors that are not at risk. A relatively small number of chemicals in water, sediment, and floodplain soils were identified as chemicals of potential ecological concern. However, no receptors could be screened out, and some tributaries and all upland areas except WAG 5 could not be assessed because of lack of data.

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ABBREVIATIONS

BCV	Bear Creek Valley
BMAP	Biological Monitoring and Abatement Program
BWC	Background White Oak Creek
COPECs	chemicals of potential ecological concern
CV	chronic values
DOE	U.S. Department of Energy
DQO	Data Quality Objectives
EPA	U.S. Environmental Protection Agency
ER-L	effects range-low
EWC	White Oak Creek Embayment
FFA	Federal Facility Agreement
FY	fiscal year
HRT	Homogeneous Reactor Test
HQ	hazard quotient
IAEA	International Atomic Energy Agency
ICRP	International Council for Radiation Protection
IHP	Intermediate Holding Pond
LMB	Lower Melton Branch
LOAEL	lowest observed adverse effects level
LOEC	lowest observed effect concentration
LWC	Lower White Oak Creek
MWC	Middle White Oak Creek
NAWQ	National Ambient Water Quality Criteria
NCRP	National Council on Radiation Protection
NOAEL	no observed adverse effects level
NOEC	no observed effect concentration
NPDES	National Pollutant Discharge Elimination System
NWT	NorthwestTributary
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
OU	operable unit
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyl
PRL	probably effect level
RAC	Raccoon Creek
RI	remedial investigation
SAV	secondary acute value
SCV	secondary chronic values
TDEC	Tennessee Department of Environment and Conservation
TEL	threshold effect level
T&E	threatened and endangered
UCL	upper confidence limit
TIE	toxicity identification and evaluation
UCB	upper confidence bound
UMB	Upper Melton Branch
WAG	Waste Area Grouping

WOC	White Oak Creek
WOL	White Oak Lake
WS	West Seep
W4T	WAG 4 Tributary
1WC	WAG 1 White Oak Creek

EXECUTIVE SUMMARY

This report presents an ecological risk assessment for Waste Area Grouping (WAG) 2 based on the data collected in the Phase I remedial investigation (RI). It serves as an update to the WAG 2 screening ecological risk assessment that was performed using historic data. In addition to identifying potential ecological risks in WAG 2 that may require additional data collection, this report serves to determine whether there are ecological risks of sufficient magnitude to require a removal action or some other expedited remedial process.

WAG 2 consists of White Oak Creek (WOC) and its tributaries downstream of the Oak Ridge National Laboratory (ORNL) main plant area, White Oak Lake (WOL), the White Oak Creek Embayment of the Clinch River, associated flood plains, and the associated groundwater. The WOC system drains the WOC watershed, an area of approximately 16.8 km^2 that includes ORNL and associated WAGs. The WOC system has been exposed to contaminants released from ORNL and associated operations since 1943 and continues to receive contaminants from adjacent WAGs.

Since the WAG 2 program was planned, Federal Facility Agreement managers have decided to proceed with a remedial investigation for the WOC watershed. An early requirement of that activity is a screening ecological risk assessment for the watershed to identify hazards and data gaps. Since that assessment was needed at approximately the same time as the WAG 2 screening assessment and since the WAG 2 program is the primary source of data concerning the chemical composition of ambient media other than groundwater for the watershed, the assessments were combined. Therefore, this document used both WAG 2 program data and other data that was available to present a screening assessment of the WOC watershed. WAGs 5 and 6 and the WAG 1 surface impoundments have completed remedial investigations, and their results are not repeated here. Other WAGs have essentially no data concerning surface contamination, and they are scheduled for limited sampling and analysis.

This screening assessment is intended to indicate whether there are credible hazards to ecological endpoints in the WOC watershed due to contamination and whether there are data gaps that need to be filled. The following are conclusions concerning the hazards:

- Radionuclides in WAG 2 do not pose a hazard to fish or aquatic invertebrates. ^{90}Sr in three seeps exceed the screening benchmarks for aquatic life, but these seeps do not support communities of aquatic macroorganisms.
- Radionuclides in WAG 2 pose a hazard to terrestrial plants, wildlife, and soil invertebrates in all four reaches of WOC and in lower Melton Branch. Exposures were highest in the Intermediate Holding Pond (IHP) reach.
- The unfiltered water in all reaches exceeded water quality criteria and other toxicological benchmarks for aquatic life.
- Water from WOC adjacent to and below ORNL was toxic in a fish embryo-larval test but not in the standard subchronic tests.
- The fish community in WOC has low species richness, but this may be due to lack of recovery from past toxicity. However, the riffle invertebrate community that is exposed primarily to chemicals in water like the fish and has flying stages that should allow recovery also has low species richness.

- Sediments from the IHP, Middle WOC (MWC), and Lower Melton Branch (LMB) are contaminated to levels that have been associated with toxic effects at other sites. Contaminants include metals, polycyclic aromatic hydocarbons, and polychlorinated biphenyls.
- The species richness of benthic invertebrates from WAG 2 sediments is lower than in background WOC sediments but not other reference streams. The exception is WOL, which has very low species richness. However, no comparable reference lakes were characterized.
- Chromium and mercury levels in IHP and Lower WOC (LWC) soils and mercury in MWC and LMB soils exceeded levels that were reported to be toxic to earthworms.
- Multiple metals were found in IHP, MWC, LWC, and LMB soils at concentrations that have been reported to be toxic to plants.
- Metals in seep waters in the WAG 4Tributary and West Seep reaches are at concentrations that have been reported to be toxic to plants.
- Hazards to wildlife were found in all WOC reaches. However, the hazards were greatest for mercury among chemicals, for IHP among reaches, and for shrews among species.
- Kingfishers from WOC show elevated levels of contaminants, and the one adult found had tissue levels of mercury and selenium that are indicative of toxicity. Mink from WOC do not appear to be contaminated.

The following are major uncertainties that could be addressed by additional data collection:

- Background concentrations need to be better characterized for the seeps, surface waters, sediments, and floodplain soils.
- The contamination of soil and biota on burial grounds other than WAGs 5 and 6 are unknown.
- The bioavailable concentrations of metals in WOC are unknown.
- The toxicity of soil and sediment in the watershed are unknown.
- The toxicity of water is unknown for some reaches and has been irregular in others.
- The fish communities have not been quantitatively characterized for the smaller tributaries.
- The chemical composition of fish is unknown for some tributaries and reaches.
- Because of the high selenium levels in a kingfisher, selenium levels should be characterized in fish.

In sum, plausible hazards exist in all reaches of WAG 2 and to all ecological endpoints. Radionuclides, organic chemicals, and metals are all implicated. However, none of the risk estimates or observations of the state of the watershed suggest that there is a need for an accelerated response. That is, no threatened or endangered species are at risk and no wetlands or other highly valued populations or ecosystems are experiencing catastrophic effects. Some parts of the watershed are uncharacterized and should be surveyed and sampled before the RI is completed. These results imply that a more complete data set should be assembled and used to prepare a definitive ecological risk assessment for the WOC watershed.

1. INTRODUCTION

1.1 PURPOSE

This is one of five reports issued in 1996 that provide follow-up information to the Phase I Remedial Investigation (RI) Report for Waste Area Grouping (WAG) 2 at Oak Ridge National Laboratory (ORNL). The five reports address areas of concern that could cause immediate risk to public health at the Clinch River and ecological risk within WAG 2 at ORNL. The five reports that complete activities conducted as part of Phase I of the RI for WAG 2 are as follows:

- Waste Area Grouping 2, Phase I Task Data Report: Seep Data Assessment
- Waste Area Grouping 2, Phase I Task Data Report: Tributaries Data Assessment
- Waste Area Grouping 2, Phase I Task Data Report: Ecological Risk Assessment
- Waste Area Grouping 2, Phase I Task Data Report: Human Health Risk Assessment
- Waste Area Grouping 2, Phase I Task Data Report: Sediment Transport Modeling

In December 1990, the "Remedial Investigation Plan for Waste Area Grouping 2 at Oak Ridge National Laboratory" was issued (ORNL). The WAG 2 RI Plan was structured with a short-term component to be conducted while up-gradient WAGs are investigated and remediated, and a long-term component that will complete the RI process for WAG 2 following remediation of up-gradient WAGs. RI activities for the short-term component were initiated with the approval of the U. S. Environmental Protection Agency (EPA), Region IV, and the Tennessee Department of Environment and Conservation (TDEC).

This report presents a screening ecological risk assessment for WAG 2 based on the data collected in the Phase I Remedial Investigation. It serves as an update to the WAG 2 screening ecological risk assessment that was performed using historic data (Suter 1992). In addition to identifying potential ecological risks in WAG 2 that may require additional data collection, it serves to determine whether ecological risks are of sufficient magnitude to require removal or some other expedited remedial process.

1.2 BACKGROUND

WAG 2 consists of White Oak Creek (WOC) and its tributaries downstream of the ORNL main plant area, White Oak Lake, the WOC embayment of the Clinch River and the associated flood plains, and the associated groundwater (Fig. 1.1). The WOC system drains the WOC watershed, an area of approximately 16.8 km^2 that includes ORNL and associated WAGs. The WOC system has been exposed to contaminants released from ORNL and associated operations since 1943 and continues to receive contaminants from adjacent WAGs.

The WAG 2 RI Plan developed in 1990 was not a prototypical RI plan. It was recognized that full implementation of an RI was inappropriate while contaminants continue to enter the system. A phased effort was adopted in response to the need to take initial steps to protect the public and the environment and to characterize and assess risks associated with WAG 2 and the limitations imposed by changing contaminant input.

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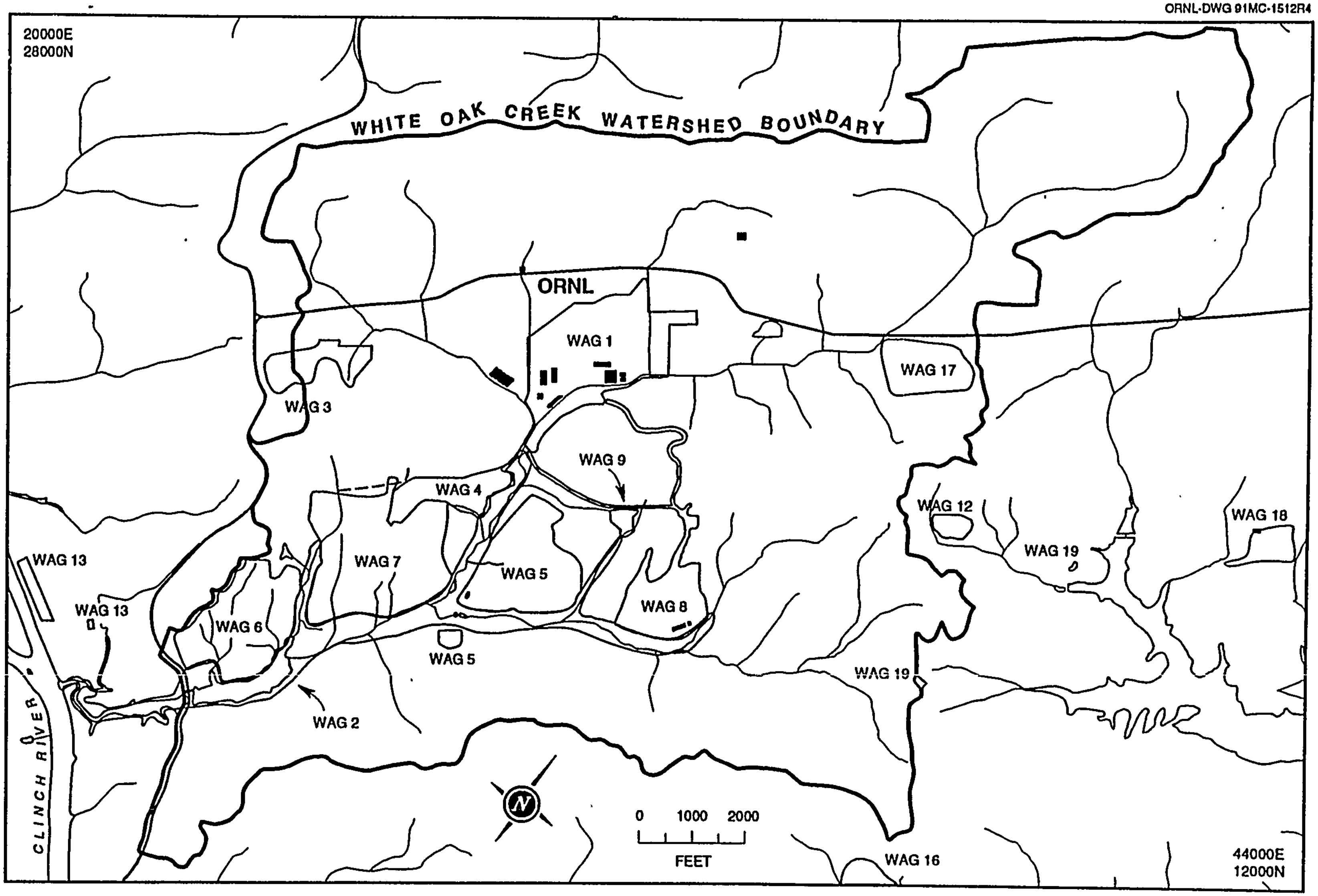

Fig. 1.1. Map of White Oak Creek and its tributaries.

Three phases were initially identified: Phase I was the initial scoping activity to determine the need for early action; Phase II included interim activities during remediation of up-gradient WAGs to evaluate potential changes in the contamination status of WAG 2 that would necessitate revaluation of the need for early action; and Phase III would be completion of the Comprehensive Environmental Response, Compensation, and Liability Act process following remediation of the up-gradient WAGs. Field activities were initiated in fiscal year (FY) 1992 consistent with the RI Plan (ORNL 1990), and a report summarizing Phase I results to date was published in 1993 (DOE).

On June 20 and 21, 1994, a Data Quality Objectives (DQO) Workshop was held with representatives of the Department of Energy (DOE), EPA, and TDEC. The participants defined the nature and boundaries of the problems for the WAG 2 RI, determined decision criteria, and established inputs to be used for characterizing the site for decision-making purposes. During the workshop, the regulators made recommendations that would alter the initial WAG 2 RI plan. Consequently, the Federal Facility Agreement (FFA) managers from EPA, TDEC, and DOE directed that FY 1995 WAG 2 RI activities concentrate on meeting FFA requirements.

The FFA managers also directed that the WAG 2 RI be changed to a two-phase field program by eliminating Phase II activities and transferring needed elements into the newly formed ORNL Environmental Restoration Surface Water Program. A separate FY 1995 WAG 2 RI Work Plan was developed (DOE 1994) to replace previously identified planning and tasking documents. Emphasis was to be on analysis of existing data, data interpretation, and reporting of results. This document reports the results of the ecological risk assessment.

Finally, after the WAG 2 program was planned, the FFA managers decided to proceed with an RI for the WOC watershed. An early requirement of that activity is a screening ecological risk assessment for the watershed to identify hazards and data gaps. Since that assessment was needed at approximately the same time as the WAG 2 screening assessment and since the WAG 2 program is the primary source of data concerning the chemical composition of ambient media other than groundwater for the watershed, the assessments were combined. Therefore, this document used both WAG 2 program data and other data that were available to present a screening assessment of the WOC watershed. WAGs 5 and 6 and the WAG 1 surface impoundments have completed RIs, and their results are not repeated here. Other WAGs have essentially no data concerning surface contamination, and they are scheduled for limited sampling and analysis.

1.3 ORGANIZATION OF THE ECOLOGICAL RISK ASSESSMENT

This screening ecological risk assessment is organized in terms of the standard framework (EPA 1992). After a problem formulation the risks of chemicals to each of the ecological risk assessment endpoints are assessed separately (Sect. 3.7). Each includes an exposure assessment, effects assessment, characterization of risk, and characterization of uncertainty. Finally, ecological risks are summarized. The components of the assessment are explained in the following.

The problem formulation defines the scope and content of the assessment. It describes the site and potential contaminant sources, defines assessment and measurement endpoints, and presents the conceptual model.

Exposure assessment characterizes the distribution in space and time of the concentrations of contaminants to which organisms are exposed. Risk from undetected chemicals was not assessed, as instructed by the FFA. Exposure calculations are performed for each reach.

Effects assessment characterizes the evidence concerning effects of contaminant exposure. The principal 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, bioindicator data that are biochemical and histological indications of the potential mechanisms and causes of effects, and single chemical toxicity data that indicate the toxic effects of the concentrations measured in site media.

Risk characterization is the phase of risk assessment in which the information concerning exposure and the information concerning the potential effects of exposure are integrated to estimate risks (the likelihood of effects given the exposure). Because this is a screening assessment, emphasis is placed on screening the chemical concentrations in ambient media against screening benchmark values. Procedurally, the risk characterization is performed for each assessment endpoint by (1) screening all measured contaminants against toxicological benchmarks and background concentrations if available, (2) considering the implications of other types of data for the hypothesis that a hazard exists that requires further assessment, (3) logically integrating the screening results with the other evidence to determine whether a credible hazard exists to the endpoint, and (4) listing and discussing the major uncertainties in the assessment.

2. ECOLOGICAL PROBLEM FORMULATION

The problem formulation consists of describing the relevant features of the environment, describing the sources of contamination, identifying ecological endpoints, and summarizing that information in terms of a conceptual model of the hazard posed by the contaminants to the endpoint biota.

2.1 ENVIRONMENTAL DESCRIPTION

The environment considered in this assessment is the WOC watershed (henceforth, simply watershed) and in particular WAG 2 (Fig. 2.1). WAG 2 consists of Melton Branch and its floodplain below km 1.5 (just above the discharge from the High Flux Isotope Reactor) and WOC and its floodplain downstream of WAG 1 (km 3.45 at the 7500 Bridge).

The streams and floodplains of the watershed are divided into reaches, which are lengths of stream and associated floodplain that are relatively uniform with respect to exposure and ecology. The reaches within WAG 2 correspond to the areas defined for floodplain soil and sediment characterization (Ford et al. 1996). However, because this is also a preliminary assessment of the watershed, additional reaches are defined beyond the five WAG 2 reaches. These are defined in Table 2.1 and portrayed in Fig. 2.2. One stream that is not in the watershed, Raccoon Creek, is included because it was included in the WAG 2 Phase 1 program and it drains WAG 3 which is an ORNL site.

The reaches are divided into two categories. Mainstem reaches are those on WOC, including WOL, and tributaries are streams that feed WOC and are not ephemeral. Many of the water samples are from springs and small tributaries' sites, which include springs, seeps, and ephemeral tributaries. Each of these sites is associated with the reach to which it drains so that sources of chemicals can be identified, but its data are not averaged with the mainstem or tributary data.

No threatened or endangered species are known to occur in WAG 2, but some state -listed species (e.g., river otter) and federally listed species (e.g., bald eagle) are undergoing range expansions and may use WAG 2 in the future. Threatened or endangered species that are believed to occur on the Oak Ridge Reservation (ORR) are listed by Suter et al. (1996).

2.2 SOURCES

The proximate sources considered in this assessment are the contaminated water, sediment, and soil. The ultimate sources of contaminants are the National Pollutant Discharge Elimination System (NPDES) permitted point discharges at ORNL and releases from wastes in WAGs 1, 3–9, and 17. DOE's operations in the WOC watershed have included waste disposal, spills, and use of chemicals such as pesticides in the environment.

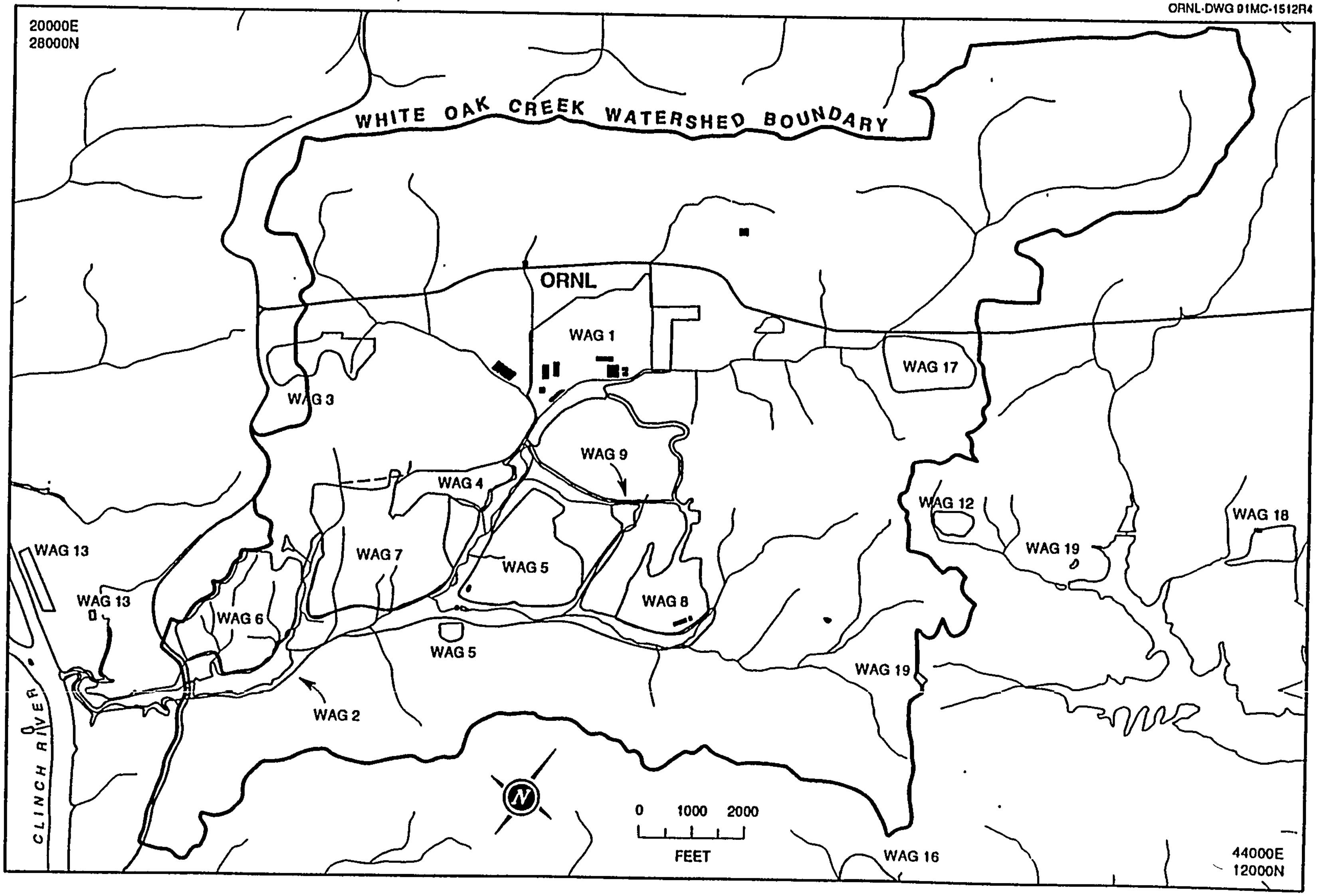

Fig. 2.1. A map of White Oak Creek watershed with anthropogenic features including the waste area groups.

Table 2.1. Definition of reaches in the White Oak Creek watershed

Reach code	Reach description
BWC	Background White Oak Creek (WOC): upstream of WAG 1
1WC	Waste Area Group (WAG) 1 WOC: the stream adjacent to WAG 1 and extending downstream to the 7500 Bridge
IHP	Intermediate Holding Pond: The first WOC reach in WAG 2, extending from the 7500 Bridge to the confluence of the WAG 4 Tributary; it corresponds to the area of the former Intermediate Holding Pond.
MWC	Middle WOC: extends from the WAG 4 Tributary to Melton Branch
LWC	Lower WOC: extends from the Melton Branch to White Oak Lake
WOL	White Oak Lake
EWC	Embayment of WOC: extends from the dam at Route 95 to the coffer dam at the confluence with the Clinch River
UMB	Upper Melton Branch: extends upstream from the Homogeneous Reactor Test (HRT) Tributary
LMB	Lower Melton Branch: extends downstream from the HRT Tributary
1C	First Creek: flows through western ORNL and enters the lower end of the Northwest Tributary
5C	Fifth Creek: flows through central ORNL and enters WOC
NWT	Northwest Tributary: drains the eastern end of WAG 3 and far western ORNL
W4T	WAG 4 Tributary
HRT	Homogeneous Reactor Test Tributary: the tributary of Melton Branch that drains WAG 9
WS	West Seep: a small tributary located between WAGs 6 and 7 that directly enters White Oak Lake
U	Unknown reach: two seeps that are not known to drain into any stream are assigned this designation
RAC	Raccoon Creek: outside the WOC watershed, drains WAG 3

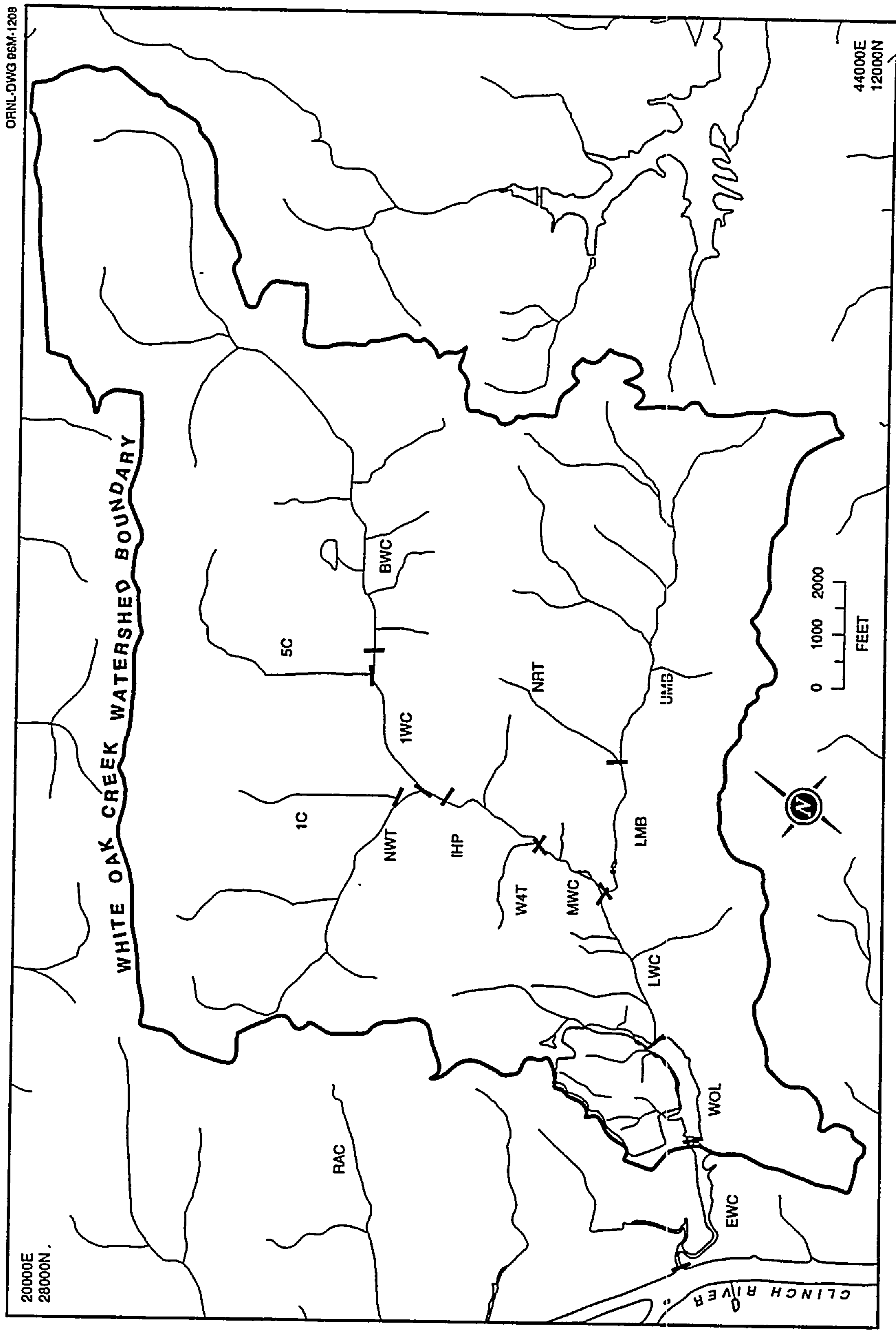

Fig. 2.2. Map of the White Oak Creek watershed showing the stream reaches.

Chemicals of potential ecological concern (COPECs) for ecological risks have been identified in a previous screening assessment for WAG 2 (Blaylock et al. 1992). However, because the large number of COPECs, the revisions that have occurred in the ecological screening benchmarks, the questionable quality of the existing data used by Blaylock et al., and the new contaminant concentration data that have been obtained, chemicals are rescreened against benchmarks for this assessment.

2.3 ECOLOGICAL ENDPOINTS

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

2.3.1 Assessment Endpoints

The following assessment endpoints for aquatic and terrestrial risks have been selected for this assessment:

- reduction in species richness or abundance of fishes or increased frequency of gross pathologies in fish communities resulting from toxicity.
- reduction in species richness or abundance of benthic macroinvertebrate communities resulting from toxicity.
- reduction in abundance or production of earthworms resulting from toxicity.
- reduction in abundance or production of piscivorous wildlife populations (kingfisher and mink) resulting from toxicity.
- reduction in production of terrestrial plant communities resulting from toxicity.
- reduction in abundance or production of terrestrial wildlife populations (short-tailed shrew, white-footed mouse, red fox, red-tailed hawk) resulting from toxicity.

The ecological assessment endpoints have been selected based on DQO meetings that included representatives of DOE, EPA Region IV, and TDEC and the strategy for ecological risk assessment on the ORR, which was also a product of a DQO process (Suter et al. 1995).

The endpoints are chosen on the basis of ecological importance, policy or societal significance, susceptibility to toxic effects, and appropriateness of scale. All of these are self explanatory except for scale. An endpoint has appropriate scale for a site if toxic effects on the site could have a significant effect on the endpoint. For example, there is a distinct plant community on the WOC floodplain so properties of that community have an appropriate scale. However, the WOC watershed supports only a few kingfishers, which form a very small fraction of the biological population to which they belong. Therefore, individual kingfishers have an appropriate scale, but the kingfisher population does not.

The following paragraphs explain the selected endpoints.

- The fish community is considered to be an appropriate endpoint community because it is ecologically and societally important, susceptible, and has a scale appropriate to the site. The societal importance is due to recreational fisheries; the ecological significance comes from the fact that much of the energy flow in temperate streams passes through fishes, and fishes are a major nutrient reservoir in those systems. In addition, the fish species in WAG 2 tend to move within an area smaller than the OU and their movement is restricted by weirs and dams, so the scale is appropriate.
- The benthic invertebrate community is considered an appropriate endpoint community because it is highly susceptible and has a scale appropriate to the site. The high susceptibility is due to the association of these organisms with the sediment, which is the repository of most of the COPECs. In addition, the insects and crustaceans that dominate this community are sensitive to a variety of contaminants. Because these organisms are sedentary for all or most of their lives, their scale is highly appropriate to the scale of the site. Benthic invertebrates may also be ecologically important, but the importance of the invertebrates in the depositional areas is unclear.
- Soil invertebrates are considered to be an appropriate endpoint assemblage because they are susceptible and ecologically important and have an appropriate scale. The susceptibility results from their intimate exposure to contaminated soils including, in the case of earthworms, ingestion. The ecological importance is due to their role in litter degradation, nutrient cycling, and maintaining soil structure. The appropriate scale results from their low mobility; biological populations could occupy an area as small as the reaches defined for this assessment.
- The terrestrial plant community is considered an appropriate endpoint community because it is ecologically important, susceptible, and has a scale appropriate to the site. The ecological significance comes from the fact that the plant community is responsible for primary production. This community is susceptible because it would be directly exposed to the contaminants in the floodplain. Finally, the scale is appropriate because plants are immobile and because a distinct plant community occurs on the floodplain.
- Piscivorous and terrestrial wildlife species are considered appropriate because individuals are potentially sensitive due to food-web magnification of chemicals and because of the known sensitivity of some species (e.g., mink). The watershed has an appropriate scale for small mammals (i.e., mice and shrews) in that it could support a biological population of those species. The watershed clearly does not support biological populations of birds or large (e.g., deer) or medium-sized (e.g., mink) mammals, but the regulators have decided that the organisms occurring in a watershed constitute a population for regulatory purposes.

Threatened and endangered species are not directly addressed in this assessment. This is because the other endpoint species are judged to be as sensitive or more sensitive than the endangered species that may come to use the site. In addition, because this is a screening assessment, it is conservative and affords the extra protection appropriate to T&E species.

Wetlands are assumed to be protected by assessing the risks to plants in the small wetland areas associated with seeps. These wetlands should be more highly exposed than those associated with the streams.

2.3.2 Measurement Endpoints

Three basic types of effects data are potentially available to serve as measurement endpoints: results of biological surveys, toxicity tests performed on media from the CR operable unit (OU), and toxicity test endpoints for chemicals found in the OU. Measurement endpoints are presented in the following paragraphs.

2.3.2.1 Fish

Biological Survey Data. No fish survey data were collected by the WAG 2 program. However, results of Biological Monitoring Abatement Program (BMAP) surveys will be cited as supporting evidence. The BMAP measurement endpoints are assumed to be direct estimates of that assessment endpoint.

Biological Indicators Data. No fish bioindicators data were collected by the WAG 2 program. However, published results of the BMAP biological indicators task will be cited as supporting evidence. Frequencies of gross pathologies are a direct measure of one aspect of the assessment endpoint. Measures of fish fecundity in largemouth bass and bluegill provide an indication of the potential contribution of reproductive toxicity to community effects. Measures of the levels of physiological and histological condition in redbreast sunfish help to confirm that exposures have occurred and may suggest mechanistic connections between exposure and effects on the fish community.

Media Toxicity Data. No fish aqueous toxicity tests were conducted by the WAG 2 program. However, published results of the BMAP tests will be cited as supporting evidence. Test endpoints include reductions in growth and survivorship of larval fathead minnows and in fecundity and survivorship of *Ceriodaphnia dubia (C. dubia)* in 7-day tests of ambient water and reductions in hatching and larval survival and increases in terata in Japanese medaka (*Oryzias latipes*) eggs and larvae exposed to ambient water from shortly after fertilization to 48 hours post-hatch. Responses that are statistically significantly different or are inhibited by 20% or greater relative to control or reference waters are assumed to be indicative of waters that are toxic to fish.

Single Chemical Toxicity Data. Chronic toxicity thresholds for freshwater fish are expressed as chronic EC20s or chronic values (CVs). These test endpoints correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of OU species, and exceeding the CVs and EC20s is assumed to correspond to 20% or greater reductions in abundance, with some uncertainty.

2.3.2.2 Benthic invertebrates

Biological Survey Data. Benthic invertebrate survey data were collected by the WAG 2 program from areas in which fine sediments had been deposited. In addition, results of BMAP surveys of benthic invertebrates in riffles will be cited as supporting evidence. The measurement endpoints for both surveys are assumed to be direct estimates of that assessment endpoint.

Media Toxicity Data. Sediment toxicity tests planned for WAG 2 could not be performed because of concerns for worker safety.

Single Chemical Toxicity Data. Chronic toxicity thresholds for freshwater invertebrates expressed as chronic EC20s or CVs. These test endpoints correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of OU species, and exceeding the CVs and EC20s is assumed to correspond to 20% reductions in population abundance or greater, with some uncertainty. Toxic concentrations in ambient sediments reported by the state of Florida. Two types of values were extracted from that data set: (1) thresholds for modification of benthic invertebrate community properties based on co-occurrence analyses, which are assumed to correspond to the assessment endpoint.; (2) thresholds for lethality in toxicity tests of contaminated sediments, which are also assumed to correspond to the assessment endpoint effect but with greater uncertainty due to the extrapolation to the field.

2.3.2.3 Piscivorous wildlife

Biological Survey Data. Kingfisher reproduction was surveyed in WAG 2 and reference areas. Assuming that the kingfishers in the watershed constitute a population, this is a direct measure of the assessment endpoint for avian piscivores.

Media Toxicity Data. None were performed.

Single Chemical Toxicity Data. Chronic toxicity thresholds for contaminants of concern in birds and mammals gave greater weight to data from long-term feeding studies with wildlife species. Preference was given to tests that included reproductive endpoints. After allometric scaling for the endpoint species, these test endpoints are assumed to correspond to effects on individuals that could result in exceeding the population-level assessment endpoint. An extrapolation must be made to populations if effects on individuals are estimated to occur. In addition, body burdens of a kingfisher were compared to concentrations associated with toxic effects on birds.

2.3.2.4 Terrestrial wildlife

Biological Survey Data. None were performed.

Media Toxicity Data. None were performed.

Single Chemical Toxicity Data. Chronic toxicity thresholds for contaminants of concern in birds and mammals gave greater weight to data from long-term feeding studies with wildlife species. Preference was given to tests that included reproductive endpoints. After allometric scaling for the endpoint species, these test endpoints are assumed to correspond to effects on individuals that could result in exceeding the population-level assessment endpoint. An extrapolation must be made to populations if effects on individuals are estimated to occur.

2.3.2.5 Terrestrial plants

Biological Survey Data. None.

Media Toxicity Data. None.

Single Chemical Toxicity Data. This includes EC20s for growth or production of vascular plants or equivalent chronic toxicity thresholds for contaminants of concern in soil. These test endpoints are assumed to correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of species that would colonize Clinch River dredge spoil; exceeding the test endpoints is assumed to correspond to 20% reductions in abundance or productivity with some uncertainty; and a distinct plant community is assumed to occur on a spoil disposal area.

2.3.2.6 Soil invertebrates

Biological Survey Data. Earthworms were collected by the WAG 2 program in a manner that produces only presence/absence information.

Media Toxicity Data. Earthworm toxicity tests planned for WAG 2 could not be performed because of concerns for worker safety.

Single Chemical Toxicity Data. Chronic toxicity thresholds for earthworms have been obtained from the literature. These test endpoints vary in their relevance, but they are assumed to correspond to the assessment endpoint for this assemblage if they include sublethal responses.

2.3.2.7 Threatened and endangered species

Biological Survey Data. None.

Media Toxicity Data. None.

Single Chemical Toxicity Data. The same chronic toxicity thresholds for contaminants of concern in invertebrates, fish, birds and mammals used as measurement endpoints for other species are used with the T&E species but are interpreted in the risk characterization so as to provide the higher level of protection specified in the assessment endpoint.

2.4 CONCEPTUAL MODELS

Conceptual models are graphical representations of the relationships among sources of contaminants, ambient media, and the endpoint biota. Figure 2.3 shows a conceptual model for the streams and floodplains of the WOC watershed. Figure 2.4 shows a conceptual model for wide-ranging wildlife that utilize the streams and floodplains as well as upland areas and nearby aquatic habitat. The exposure pathways shown are those that are included in the assessment. Note that different benthic invertebrate assemblages are exposed to water and sediments. That is, the invertebrates inhabiting soft sediments in depositional areas are assumed to be exposed to chemicals in those sediments and their pore water with negligible exposure to free water and invertebrates inhabiting the rocky substrates of riffles are assumed to be primarily exposed to chemicals in water. These conceptual models are derived from the generic models developed for the ORR and are discussed in detail in the strategy document for ecological risk assessment on the ORR (Suter et al. 1995).

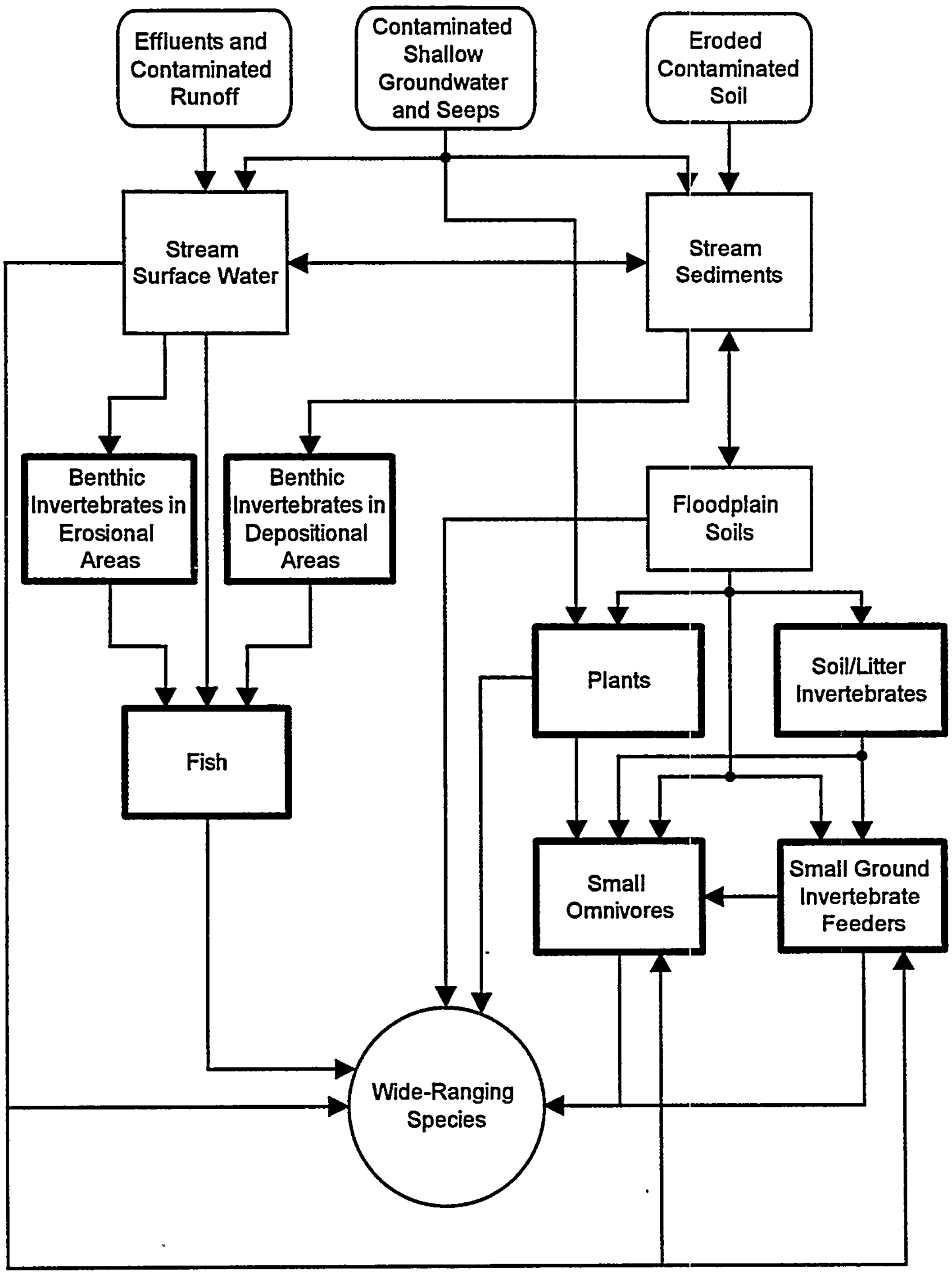

Fig. 2.3. Conceptual model of the mechanisms of transport and exposure by which ecological receptors are exposed to contaminants in White Oak Creek, its tributaries, and its floodplain.

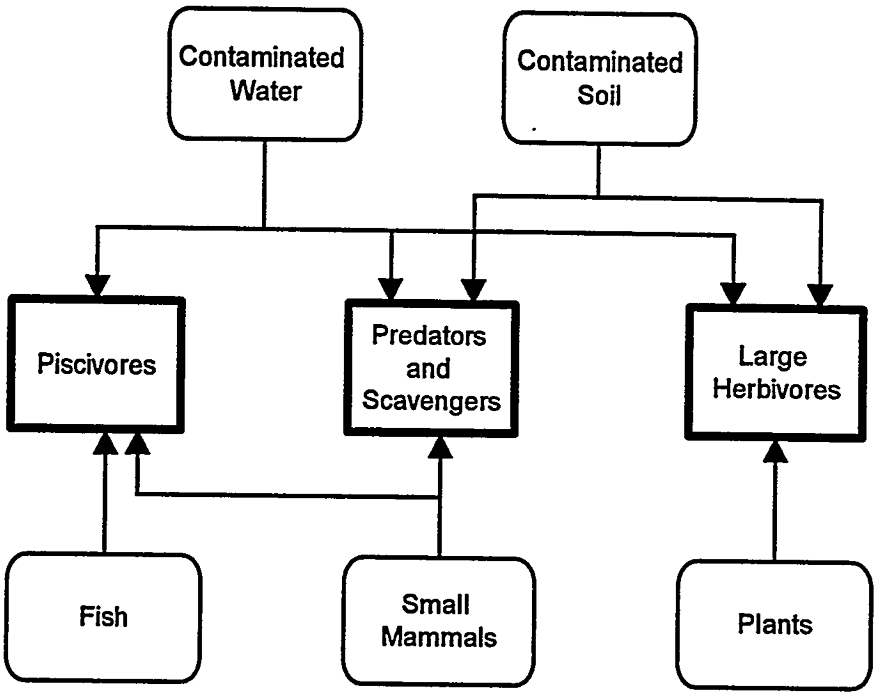

Fig. 2.4. Conceptual model of the mechanisms by which wide-ranging wildlife species are exposed to contaminants in White Oak Creek, its tributaries, and its floodplain.

3. RISKS TO FISH

3.1 EXPOSURE ASSESSMENT FOR FISH

3.1.1 Aqueous Chemical Exposure

Fish are exposed primarily to contaminants in water. Contaminants in water may come from upstream aqueous sources including the permitted outfalls, runoff, and the numerous seeps sampled and analyzed by the WAG 2 RI program. They may also come from internal cycling including exchange of materials between the surface water and contaminated sediments and exchange of contaminants between the biota and the water column. The consensus of the scientific community and of the EPA Office of Water is that aquatic biota should be assumed to be exposed to the dissolved fraction of the chemicals in water because that is the bioavailable form (Prothro 1993). However, EPA Region IV prefers to use total concentrations as conservative estimates of the exposure concentration. Therefore, total concentrations of metals are used in the exposure assessment for fish.

Because water in the OU is likely to be more variable in time than in space because of the rapid replacement of water, the mean water concentration within a subreach is an appropriate estimate of the chronic exposure experienced by fishes. The upper 95% confidence limit on the mean is an appropriately conservative estimate of this exposure for use in the contaminant screening. No distinction was made between storm flow and base flow samples in this screening analysis.

Many of the chemical concentrations in water were below analytical detection limits. Chemicals that were not detected in any sample in a reach were eliminated. If a reach contained some detects and some nondetects for a chemical, the product limit estimator was used to estimate the mean and its variance.

3.1.2 Radiation Exposures

Exposures to radionuclides are expressed as the dose rate received by the organism. The total dose rate is the sum of the internal and external dose rates, which are a function of exposure to the radionuclide and the characteristics of the radiation. Radiation exposures of fish in surface water are likely to be driven by internal exposures, especially for alpha emitters such as U^{234} and Th^{230} (Blaylock et al. 1993). Internal dose rates are based on the concentrations of the radionuclide in the organism, which were estimated by multiplying the water concentration times the biological concentration factors provided in Blaylock et al. (1993). External dose rates are based on the concentrations of the radionuclides in the surrounding water. Because the radiation exposure rate is expressed as absorbed dose per unit time (rads per day), the dose rates for each isotope and pathway can be added to determine the total dose rate.

As with estimates of aqueous chemical exposure, the 95% upper confidence limit (UCL)on the mean is an appropriately conservative estimate of exposure for use in the screening. As with chemical exposures, total concentrations of radionuclides were used in the radiation exposure assessment for fish. This results in conservative estimates of the internal dose and realistic estimates of external dose.

3.2 EFFECTS ASSESSMENT FOR FISH

3.2.1 Single Chemical Aqueous Toxicity

The screening benchmarks for aquatic biota are taken from Suter and Mabrey (1994). Because there are no standard screening benchmarks, sets of alternative benchmarks (described in Table 3.1) were calculated for each chemical. The benchmark preferred by the regulatory agencies is the chronic National Ambient Water Quality Criteria (NAWQC), but they are available for relatively few industrial chemicals. Secondary chronic values (SCV), which are conservative estimates of chronic NAWQC, were calculated for chemicals that do not have NAWQC. Other benchmarks are included to provide greater assurance of detecting all COPECs.

NAWQC that are functions of water hardness are corrected for site-specific conditions. For purposes of screening, we chose conditions that would constitute reasonable maximum toxicity, defined as conditions that would persist for seven days. This was a hardness of approximately 100 mg/L (Blaylock et al. 1992).

3.2.2 Radiological Effects

The recommended acceptable dose rate to natural populations of aquatic biota is 1 rad/d (NCRP 1991). Although there are no standard screening benchmarks for radionuclide concentrations in surface water, Blaylock et al. (1993) provide formulas and exposure factors estimating the dose rates to representative aquatic organisms. These formulas were used to calculate the water concentration that results in a total dose rate of 1 rad per day to a small fish for each radionuclide detected in the water samples from Bear Creek Valley (BCV). The exposure factors used in this assessment are from Blaylock et al. (1993) and are presented in Table 3.2. The screening values include internal and external exposures from the detected isotope and all short-lived daughter products. All major alpha, beta, and gamma emissions for each isotope were included, and the absorbed fraction of each was estimated from the graphs provided in Blaylock et al. (1993). Internal exposures from radionuclides in water were estimated using the biological concentration factors for the detected parent isotopes.

3.2.3 Ambient Water Toxicity

Toxicity tests of WAG 2 water are performed by the ORNL BMAP and are presented in their annual reports. The tests employed include the standard 7-day tests of growth and survival in fathead minnow larvae and fecundity and survival of *Ceriodaphnia dubia* and an early life stage test with Japanese medaka eggs and larvae. The standard tests did not show any aqueous toxicity in the last two years reported, 1992 and 1993. However, the more sensitive medaka embryo-larval test consistently showed toxicity in water collected adjacent to or downstream of ORNL relative to upstream and control waters (Ashwood 1994). Attempts to identify the causal agent determined that mortality was not affected by 0.1 micron filter to eliminate pathogens or by treatment with sodium thiosulfate to eliminate chlorine. However, toxicity was completely eliminated by filtration with activated charcoal, a nonspecific treatment for aqueous contaminants. Finally, in situ toxicity tests were conducted with caged snails (*Elimia clavaeformis*) exposed for seven days. Although snails are benthic invertebrates, they are discussed here because they were exposed to water and not sediment. Snails were killed in lower fifth creek in October 1992 and in WOC adjacent to ORNL in June and October 1992 (Ashwood 1993).

Table 3.1. Descriptions of the ecotoxicological screening benchmarks for aquatic biota

Benchmark	Abbreviation	Description
Acute National Ambient Water Quality Criteria	NAWQC_ACU	Current national criteria for protection of aquatic life from lethal effects in episodic exposures.
Chronic National Ambient Water Quality Criteria	NAWQC_CHR	Current national criteria for protection of aquatic life from lethal and sublethal effects in extended exposures. Criteria for uses of aquatic life (i.e., fish consumption) are not included.
Secondary acute value	S_ACU_V	Values estimated with 80% confidence to not exceed the unknown acute NAWQC. Used when data are inadequate to calculate the acute criterion.
Secondary chronic value	S_CHR_V	Values estimated with 80% confidence to not exceed the unknown chronic NAWQC. Used when data are inadequate to calculate the chronic criterion.
Lowest chronic value for fish	LCV_FISH	The lowest value, from acceptable fish chronic toxicity tests, of the geometric mean of the lowest observed effect concentration (LOEC) and the no observed effect concentration (NOEC).
Lowest chronic value for daphnids	LCV_DAPH	The lowest value, from acceptable daphnid chronic toxicity tests, of the geometric mean of the LOEC and the NOEC.
Lowest chronic value for nondaphnid invertebrates	LCV_ND	The lowest value of the geometric mean of the LOEC and the NOEC from acceptable chronic toxicity tests of nondaphnid invertebrate species.
Lowest plant value	LCV_AQPL	The lowest value from an acceptable daphnid chronic toxicity test of the geometric mean of the LOEC and the NOEC.
Lowest Test EC20 for fish	LTV_FISH	The lowest value, from acceptable fish chronic toxicity tests, of the lowest concentration causing at least a 20% reduction in the weight of young per female or the weight of young per egg.

Table 3.1 (continued)

Benchmark	Abbreviation	Description
Lowest test EC20 for daphnids	LTV_DAPH	The lowest value from an acceptable daphnid chronic toxicity test of the lowest concentration causing at least a 20% reduction in the product of survivorship, growth, and fecundity.

Note: More details are presented by Suter and Mabrey (1994).

Table 3.2. Exposure factors for the radionuclides detected in WAG 2 sediment and surface water and their short-lived progeny

Radionuclide[a]	Kd[b]	Biological Concentration Factor[c]	Emission Energies (MeV)				Absorption Factors			
			Average Alpha	Maximum Beta	Average Beta	Average Gamma	Beta for Small Invertebrates	Beta for Small Fish	Gamma for Small Invertebrates	Gamma for Small Fish
Cesium-137+Barium-137m	1000	2000		1.17e+00	2.52e-01	5.96e-01	0.6	0.99	0.003	0.015
Cobalt-60	45	300		1.48e+00	9.65e-02	2.50e+00	0.5	0.98	0.002	0.01
Hydrogen-3	N/A	1		1.80e-02	5.68e-03		1	1		
Strontium-90 + Yttrium-90	35	60		2.79e+00	1.13e+00	1.69e-06	0.25	0.83	0.4	0.6
Curium-244	2000	30	5.89e+00		8.59e-03	1.70e-03	1	1	0.4	0.6
Thorium-232	150000	100	4.07e+00		1.25e-02	1.33e-03	1	1	0.4	0.6
Radium-228					1.69e-02	4.14e-09	1	1	0.4	0.6
Actinium-228					4.60e-01	9.30e-01	0.95	1	0.003	0.015
Thorium-228	150000	100	5.49e+00		2.05e-02	3.30e-03	1	1	0.4	0.6
Radium-224			5.78e+00							
Radon-220			6.40e+00		8.19e-06	3.85e-04	1	1	0.4	0.6
Polonium-216			6.91e+00		1.61e-07	1.69e-05	1	1	0.4	0.6
Lead-212					1.75e-01	1.48e-01	1	1	0.002	0.012
Bismuth-212			2.22e+00	5.80e-01	4.69e-01	1.85e-01	0.95	1	0.002	0.012
Polonium-212			8.95e+00	2.25e+00			0.3	0.89		
Americium-241	700	30	5.57e+00		5.19e-02	3.24e-02	1	1	0.01	0.05

Table 3.2 (continued)

Radionuclide[a]	Kd[b]	Biological Concentration Factor[c]	Emission Energies (MeV) Average Alpha	Maximum Beta	Average Beta	Average Gamma	Absorption Factors Beta for Small Invertebrates	Beta for Small Fish	Gamma for Small Invertebrates	Gamma for Small Fish
Uranium-233	450	10	4.89e+00		6.08e-03	1.31e-03	1	1	0.4	0.6
Uranium-238	450	10	4.26e+00		1.00e-02	1.36e-03	1	1	0.4	0.6
Thorium-234					5.92e-02	9.34e-03	1	1	0.4	0.6
Protactinium-234m		10		2.29e+00	8.20e-01	1.13e-02	0.3	0.87	0.2	0.5
Plutonium-238	4500	30	5.58e+00		1.06e-02	1.81e-03	1	1	0.4	0.6
Uranium-234	450	10	4.84e+00		1.32e-02	1.73e-03	1	1	0.4	0.6
Thorium-230	150000	100	4.74e+00		1.46e-02	1.55e-03	1	1	0.4	0.6
Uranium-235	450	10	4.47e+00		4.80e-02	1.54e-01	1	1	0.002	0.012
Thorium-231					1.63e-01	2.55e-02	1	1	0.03	0.15

Sources: Soil-water partitioning coefficients are from Baes et al. (1984), biological concentration factors are from IAEA (1982), emission energies are from ICRP (1983), and absorbed fractions were estimated from graphs in Blaylock et al. (1993).

[a] Designations of parent radionuclides and short-lived progeny are taken from Tables A.1 and A.2 in Blaylock et al. (1993). For naturally occurring alpha decay series, the parent is listed first, and progeny are listed immediately below in order of occurrence.

[B] Soil-water partition coefficients are presented only for parent isotopes detected in WAG 2 sediments.

[c] Biological concentration factors are presented only for parent isotopes detected in WAG 2.

3.2.4 Fish Community Survey

Analysis of fish population and community survey data provides a direct measure of impacts of human activities on aquatic ecosystems. The following results are taken from the most recent ORNL BMAP report (Ashwood 1993). The species composition of the fish community in the WOC watershed is stable at 13 species, which is less than other local streams. It is missing two families (Catastomidae and Percidae) and five genera (*Lythruras, Luxilus, Notropis, Noturus,* and *Phoxinus*) that historically occurred in the stream.

3.2.5 Indicators of Fish Reproduction

Reproduction and indicators of reproduction were monitored in redbreast sunfish from 1989 to 1993. Indicators of reproductive condition of female redbreast sunfish improved over the period monitored until they nearly equal background values. However, reproductive success was highly variable with apparent reproductive failure occurring in lower WOC during the last reported year of monitoring. Since this failure was not associated with poor reproductive condition of the female fish prior to spawning, effects on the embryos or larvae were said to be the likely cause (Ashwood 1994).

3.3 RISK CHARACTERIZATION FOR FISH

3.3.1 Single Chemical Aqueous Toxicity

All chemicals detected in stream waters were screened against benchmarks. This was done by dividing the 95% UCL concentration (or maximum if the UCL could not be calculated because the chemical was detected only once) by each of the aqueous screening benchmarks (Table B.1). Chemicals that exceeded benchmarks were eliminated as COPECs if their UCL did not exceed twice the mean background concentration. EPA Region IV recommends this screening criterion. The reader should be aware that this practice in combination with the practice of using total metal concentrations, which is required by EPA Region IV, is potentially anticonservative because the concentration of particulate metal is being doubled along with the bioavailable dissolved fraction. This is potentially important because metals leached from wastes would be likely to be in soluble forms (e.g., chlorides or nitrates) that are potentially toxic, whereas, background concentrations of many metals in unfiltered water are likely to be predominately in the form of relatively insoluble minerals (e.g., sulfides or oxides), which are relatively nontoxic. If contamination doubled the bioavailable metal over background, this would not necessarily double the total metal concentration. The results of the screening are summarized in Table 3.3, and the COPECs for fish and chemicals that exceed NAWQC are in the following paragraphs.

In addition to stream waters, the WAG 2 program sampled waters from seeps and small ephemeral tributaries (Hicks 1996). Because these sites do not support fish communities, they are not assessed for risks to this endpoint. However, it is important to know which of these sources is contributing to risks in the streams. In addition, the state of Tennessee is concerned with whether the seep waters exceed water quality standards; therefore, the UCM or maximum concentrations for all chemicals detected in each seep are compared to their chronic NAWQCs or SCV (Table B.2). The use of the regulatory standard (NAWQC) or an estimate of the standard (SCV) addresses the regulatory concern and provides a means of highlighting those seeps that have high concentrations relative to toxicity without suggesting that the seeps support communities of fish and macroinvertebrates like the streams.

Table 3.3. Results of screening of chemicals that exceed benchmarks in whole or filtered water for chemicals of potential ecological concern (COPECs)

Chemical	COPEC	Reason for Inclusion or Rejection
Aluminum	Yes	Exceeds chronic National Ambient Water Quality Criteria (NAWQC) in all reaches and acute NAWQC in White Oak Lake (WOL) and Raccoon Creek Tributary reach (RAC). The chronic criterion is also exceeded in background water. Upper confidence limit (UCL) concentrations exceed twice background (0.34 mg/L) in 1C, WAG1 White Oak Creek (1WC), Northwest Tributary (NWT), RAC, West Seep (WS), and WOL; and the WOL mean exceeds twice background. It is entirely possible that the elevated Al is due to high particulate matter.
Barium	Yes	Exceeds the secondary chronic values (SCV) in all reaches and the secondary acute value (SAV) in Background White Oak Creek (BWC), Lower White Oak Creek (LWC), Waste Area Group 4 (W4T), and WS, but no other benchmarks. The upper confidence limit (UCL) concentration in BWC and both the mean and UCL concentrations in W4T exceed the twice background concentration (0.004 mg/L).
Berylium	No	Exceeds the LTV daphnids but no other benchmarks in all reaches. That benchmark is apparently below the local background due to the compounding of effects across life stages in its derivation.
Boron	No	The Bo UCL exceeds the LTV for Daphnids in all reaches but no other benchmark. The LTV for Daphnids is much lower than other benchmarks for Bo. It is judged to be anomalous.
Cadmium	Yes	The Cd UCL concentration in W4T exceeds the chronic NAWQC, and the mean exceeds the lowest chronic value (CV) for daphnids. However, none of the concentrations exceed twice background (0.01 mg/L).
Calcium	No	The Ca UCL concentration in NWT and RAC slightly exceeds the CV for *Daphnia* but Ca would occur as nontoxic carbonates in those circumneutral and moderately hard streams rather than the toxic form (chloride) used in the test.
Chromium	Yes	The UCL exceeds the acute and chronic NAWQC, and the mean exceeds the chronic NAWQC in WOL. The WOL mean and the LMB UCL exceed the daphnid lowest CV. However, none of the concentrations exceed twice background (0.02 mg/L).
Cobalt	Yes	Exceeds the SCV in WS and Upper Melton Branch (UMB) and the daphnid LCV in WS in the one sample from each site in which it was detected. However, none of the concentrations exceed twice background (0.04 mg/L).

Table 3.3 (continued)

Chemical	COPEC	Reason for Inclusion or Rejection
Copper	Yes	Mean exceeds the chronic NAWQC in water from LWC, UMB, and W4T and exceeds the lowest CV for daphnids in all reaches where it was detected. However, none of the concentrations exceed twice background (0.023 mg/L).
Iron	No	Means exceeds chronic NAWQC in WS and WOL, and the daphnid LCV all reaches but BWC. However, none of the concentrations exceed twice background (0.81 mg/L).
Lead	Yes	Means in BWC and Intermediate Holding Pond (IHP) and the UCL in WOL exceed the chronic NAWQC. However, none of the concentrations exceed twice background (0.014 mg/L).
Manganese	Yes	Means in LMB, NWT, W4T, and WS exceed the SCV, and those in RAC and WS exceed the SAV and the lowest chronic values (CVs) for fish and daphnids. Mean concentrations exceed twice background (0.21 mg/L) in RAC, W4T, and WS.
Nickel	Yes	Means exceed daphnid and plant LCVs in LMB, W4T, and WS. However, none of the concentrations exceed twice background (0.04 mg/L).
Silver	Yes	Each of the single detected values at BWC, IHP, and W4T exceeds the SCV and lowest CVs for daphnids and fish. Silver was not detected at the background site.
Zinc	Yes	Mean in 1WC and the UCL in UMB exceed the chronic NAWQC. Lowest CVs for daphnids, fish, and plants exceeded in 1WC, IHP, UMB, and WOL. Only the UCL in 1WC exceeds twice background (0.22 mg/L).
Carbon disulfide	Yes	Exceeds the SCV in LMB, M.C., RAC, UMB, and WS. No background for organics.

The following text discusses those chemicals that meet all criteria for COPECs or all criteria except for the exceedence of twice background (Table 3.3). Those metals that exceed NAWQC but not background are retained in spite of the criteria because background needs to be better defined for the watershed and because of the potential problem, discussed above, of background particulate concentrations masking increases in bioavailable metals.

Aluminum. Aluminum is not known to be toxic in circumneutral streams, and one must suspect that the high concentrations are due to particulate aluminum. However, aluminum concentrations in streams clearly exceed background, so it cannot be eliminated. All but five of the 27 seeps exceed the NAWQC for Al, and RS-3a exceeds it by more than a factor of 100.

Barium. The exceedences of the SAV and SCV might be attributable to the conservatism of those benchmarks, which is a result of the poorly defined toxicity of this metal. However, the concentrations in W4T are quite high relative to background as well as benchmarks and therefore pose a credible risk to fish and other aquatic organisms. BTT, SW4-1, and SW4-2 are significant barium sources on W4T, with SW4-2 exceeding the SCV by more than a factor of 100.

Cadmium. Although cadmium does not occur at concentrations exceeding twice background in W4T and in two seeps (RS-3a and SW5-4, neither of which feeds W4T), it exceeds the chronic NAWQC.

Chromium. Although chromium does not occur at concentrations exceeding twice background, it exceeds the acute and chronic NAWQC in WOL and in the following seeps: East Seep, RS-3a, SW7-3, and WAG 4 T2A.

Copper. Although copper does not occur at concentrations exceeding twice background, it exceeds the chronic NAWQC in LWC, UMB, and W4T.

Carbon disulfide. Carbon disulfide was seldom detected in streams. The highest observed concentration, which is from approximately a factor of 100 lower than the lowest observed toxic concentration. The exceedence of the SCV in the screening assessment is attributable to the large safety factors used with minimal data sets. Carbon disulfide exceeds the SCV in 22/33 seeps; but it is a natural constituent of ground water, and the concentrations are not exceptionally high. Hence, there appears to be negligible risk to aquatic biota from carbon disulfide

Manganese. The exceedences of the SCV in six reaches might be attributable to the conservatism of that benchmark, which is a result of the poorly defined toxicity of this metal. However, the concentrations in RAC, W4T, and WS exceed the SAV and chronic toxic concentrations for aquatic life. Seeps SW4-1and SW7-1 are a conspicuous contributors of manganese to the W4T and WS.

Nickel. Nickel exceeded chronically toxic concentrations in lMB, W4T, and WS. Nickel in seep SW4-2, which feeds reach W4T, is much higher than at any other location and exceeds the chronic NAWQC by more than a factor of 50.

Silver. The combination of a detection limit that is barely lower than the lowest CV, the failure to detect silver at the background site, and the small toxicity data set make interpretation of the hazard posed by silver uncertain. However, there is no basis for rejecting it as a COPEC. Silver was detected in only two seeps, RS-3A and SW2-4.

Zinc. The mean concentration of zinc exceeds the chronic NAWQC, and the UCL exceeds twice background in the vicinity of ORNL (reach 1WC). Therefore it is a clear COPEC. Zinc concentrations are not high in seeps except for SW9-2 which exceeds the chronic NAWQC. However, it feeds HRT, which does not have high zinc concentrations. This, and the fact that zinc was not detected above ORNL (reach BWC), suggests that the source of zinc is an effluent or an unidentified seep in the vicinity of ORNL.

Chemicals that were detected in water but for which no benchmarks are available are also of concern. They are silicon and cis-1,2-dichloroethene.

3.3.2 Single Chemical Internal Toxicity

Mercury. Mercury concentrations in largemouth bass, but not sunfish, increased from a mean of 0.11 to 0.57 mg/kg wet weight during the period 1988–1992 for unknown reasons (Ashwood 1994). The maximum concentrations in largemouth bass in the last two reported years (0.70 and 0.71 mg/kg in 1991 and 1992) exceeded the 0.66 mg/kg concentration that corresponded to a CV for fathead minnows but was well below the value for brook trout (5.6 mg/kg). The median concentration for largemouth bass and all concentrations for bluegill and redbreast sunfish were below 0.66 mg/kg.

PCBs. PCBs have been detected in sunfish, largemouth bass, and channel catfish in WOC with the highest concentrations found in channel catfish from the embayment (reach EWC) (Ashwood 1994). However, both mean and maximum concentrations in catfish in 1993 were below the concentration of PCB-1242 that reduced growth and caused liver hypertrophy in channel catfish at a body burden of 14.3 mg/kg (Hansen et al. 1976).

3.3.3 Radiation Risks

The concentrations of detected radionuclides were compared with the screening values for small fish. A hazard quotient (HQ) was calculated for each radionuclide by dividing the 95% UCL concentration by the screening value. The screening value is the concentration resulting in a total dose rate of 1 rad/d from the major emissions of the measured parent radionuclide and all short-lived progeny. At each location the HQs for all detected radionuclides were summed. Radionuclides present a significant risk if the sum of HQs for a site exceeds 1.0. Table B.3 details the exposure estimates (95% UCL), screening values, and HQs for individual radionuclides for all radionuclides detected at each location. Based on these results, radionuclides in BCV surface water appear to present a negligible risk to fish. That is, the individual and sum of HQs were <1.0 at all reaches for which data are available. Seeps and small ephemeral tributaries do not support communities of fish or other aquatic macroorganisms. However, they were also screened to determine whether they were significant sources. Although seeps SW2-6 and -7 and SW5-4 exceeded the screening benchmark for strontium-90, the nearby reaches are far below the benchmark so they are not contributing to a significant risk to aquatic life.

3.3.4 Ambient Water Toxicity

Water from the watershed was found to be lethal to snails and to medaka embryos and larvae. Snails were tested only in the vicinity of ORNL, and toxicity was attributed to chlorine on the basis of correlation. Medaka toxicity was tested in WOC but not its tributaries (Ashwood 1993). Toxicity was not observed above WAG 1 but occurred consistently downstream (Ashwood 1994). In the period 1991–1993 the lowest survival (<10%) was at the WOL weir. This pattern is consistent with an accretion of contaminants in the creek. Since water was not chemically analyzed in conjunction with the toxicity tests and the toxicity identification and evaluation (TIE) was not completed, it is not possible to determine which contaminants may be responsible for the toxicity. However, preliminary TIE results indicate that a chemical other than chlorine was responsible.

In contrast, WOC water was not found to be toxic in the 7-day fathead minnow and *C. dubia* tests. This lack of response relative to the snail test was attributed to the insensitivity of the standard static renewal test to volatile chemicals such as chlorine (Ashwood 1993). The lack of response relative to medaka may be attributed to the inherent sensitivity of medaka or the longer duration of the medaka test and the inclusion of two life stages.

3.3.5 Biological Indicators

Results of the studies of histological, physiological, and reproductive indicators are presented by Ashwood (1994). Although these indicators show improvement, reproductive indicators for redbreast sunfish can not be interpreted as indicative of effects on the fish community endpoint; they are suggestive of inhibited reproduction in individuals of one species.

3.3.6 Biological Surveys

Surveys of the fish community in WOC indicate a significant loss of species and no recent improvement. Although it seems likely that the original loss was due to contaminants, it is not clear that the current contaminant levels would cause the same effect. The dam and numerous weirs inhibit movement of fish and may be preventing recovery of the community which would otherwise occur.

3.3.7 Weight Of Evidence For Fish

The weighing of evidence is performed by asking the following questions concerning each reach in the watershed:

- Is the fish community less species-rich or abundant than would be expected?
- Do individual fish display injuries that are indicative of significant toxic effects?
- Is the water toxic to aquatic organisms?
- Does that water contain chemicals in toxic amounts?
- Do the fish contain chemicals in toxic amounts?
- What factors account for apparent discrepancies in the results?
- What is the likelihood that the fish community is at least 20% less species rich or abundant than it would be in the absence of contamination?

The available evidence for each reach is summarized below. However, some lines of evidence are not available for every reach due to lack of data (Table 3.4).

Table 3.4 Availability of biological data related to the fish community endpoint in the White Oak Creek watershed reaches

Reach	Fish community	Standard tests	Medaka test	Snail test	Redbreast bioindicators	Fish bioaccumulation
EWC						X
WOL			X			X
LWC	X		X		X	X
M.C.	X	X	X		X	X
IHP	X	X	X			
1WC	X	X	X	X		X
BWC	X	X	X	X		
LMB	X	X				X
UMB	X	X				
WS						
W4T						
HRT						
NWT	X	X				X
RAC						
1C	X			X		X
5C	X			X		X

Source: Ashwood 1993, 1994

3.3.7.1 Mainstem reaches

White Oak Creek Embayment (EWC). The WAG 2 program did not sample this reach and biological data are not available. Its water quality would be expected to be similar to WOL but with some dilution from the Clinch River.

White Oak Lake (WOL). Because there are no good reference lakes for WOL, the effects on the fish community of the lake cannot be directly evaluated. WOL effluent water has been lethal to medaka embryos and larvae. WOL water exceeds NAWQC values for Al, Cr, Fe, and Pb as well as levels of Ba, Cu, Mn, and Zn that are toxic to aquatic life.

Lower White Oak Creek (LWC). Reach LWC has a depauperate fish community relative to similar streams; it has experienced reproductive failures of redbreast sunfish, and its waters have been lethal to medaka embryos and larvae. LWC water exceeds chronic NAWQCs for aluminum and copper and contains potentially toxic concentrations for barium and iron.

Middle White Oak Creek (MC). Reach MC has a depauperate fish community relative to similar streams. It has experienced reproductive failures of redbreast sunfish, and its waters have been lethal to medaka embryos and larvae. MC water exceeds the chronic NAWQC for aluminum and has potentially toxic concentrations for Ba, Cu, Fe, and carbon disulfide.

Intermediate Holding Pond (IHP). Reach IHP has a depauperate fish community relative to similar streams. Its waters have been lethal to medaka embryos and larvae. LWC water exceeds the chronic NAWQC for aluminum and lead and has potentially toxic concentrations for Ba, Cu, Fe, Ag, and Zn.

WAG1 White Oak Creek (1WC). Reach 1WC has a depauperate fish community relative to similar streams, and its waters have been lethal to medaka embryos and larvae and to snails. 1WC water exceeds the chronic NAWQC for aluminum and lead and contains potentially toxic concentrations for Ba, Cu, Fe, Ag, and Zn. Chlorinated effluents have been a source of severe episodic toxicity in this reach.

Background White Oak Creek (BWC). Reach BWC waters have not been found to be toxic. BWC water exceeds the chronic NAWQC for aluminum and lead and has potentially toxic concentrations for Ba, Cu, and Ag.

Lower Melton Branch (LMB). Reach LMB has a depauperate fish community relative to similar streams. Reach LMB water exceeds the chronic NAWQC for aluminum and contains potentially toxic concentrations for Ba, Cr, Cu, Fe, Mn, Ni, and carbon disulfide.

Upper Melton Branch (UMB). Reach UMB has a depauperate fish community relative to similar streams. Reach UMB water exceeds the chronic NAWQC for Al, Cu, and Zn as well as containing potentially toxic concentrations for Ba, Cr, Co, Fe, and carbon disulfide.

3.3.7.2 Tributary reaches

West Seep (WS). WS water exceeds the chronic NAWQC for aluminum and iron as well as containing potentially toxic concentrations for Ba, Co, Cu, Mn, No, and carbon disulfide.

WAG 4 Tributary (W4T). W4T water exceeds the chronic NAWQC for Al, Cd, and Cu as well as containing potentially toxic concentrations for Ba, Cr, Fe, Mn, No, and Ag.

Homogeneous Reactor Test Tributary (HRT). HRT water exceeds the chronic NAWQC for aluminum as well as containing potentially toxic concentrations for Cr, Cu and Fe.

Northwest Tributary (NWT). Reach NWT has a depauperate fish community relative to similar streams. NWT water exceeds the chronic NAWQC for aluminum as well as having potentially toxic concentrations for Ba, Cu, Fe, and Mn.

Raccoon Creek (RAC). RAC water exceeds the chronic NAWQC for aluminum as well as containing potentially toxic concentrations for Cu, Fe, Mn, and carbon disulfide.

3.3.8 Summary of Risk Characterization for Fish

Because this is a screening assessment, the goal of the risk characterization is to determine whether risks to fish can be eliminated as an issue or whether there is a credible hazard that should be assessed further. Even if the NAWQC for Al is set aside as inappropriate to this site, credible NAWQCs and CVs are exceeded in nearly all reaches. The toxicity to medaka embryos and larvae provides important supporting evidence that waters in WOC are toxic to fish. The low species richness of the stream is consistent with effects but could have physical causes. Although all of these lines of evidence are consistent with a hazard to fish, the evidence is not conclusive. The high metal concentrations could be nonbioavailable, the medaka could be much more sensitive than any of the fish species native to the stream, and the low species richness could be due to physical barriers to recovery. In particular, it should be noted that three years have passed since the collection of the most recent biological data that were available for this assessment. At that time, the bioindicators data suggested that the condition of fish in WOC was improving and was approaching that of fish in reference streams. It is possible that the fish community has largely recovered and that toxicity would no longer be observed in WOC at this time.

3.4 UNCERTAINTIES CONCERNING RISKS TO FISH

The following issues constitute the major sources of uncertainty in the risk assessment for the fish community.

The bioavailable concentrations of chemicals in water are unknown. In assessments that included analyses of filtered as well as unfiltered water, concentrations of Al and some other metals that were above criteria in the unfiltered water were shown to be associated with particles (i.e., not bioavailable).

The toxicity of water in seven reaches is unknown.

The high observed mortality in medaka embryos and larvae are believed to constitute toxic effects that would result in significant effects on the fish community. However, this test is sensitive and has not been validated against biological survey data at sites where clear toxic effects are occurring and fish community properties are clearly related to toxic effects as has been done with the standard 7-day tests.

4. RISKS TO BENTHIC INVERTEBRATES

4.1 EXPOSURE ASSESSMENT FOR BENTHIC INVERTEBRATES

4.1.1 Chemicals In Sediment

Benthic invertebrates are exposed to contaminants in water or sediments. Those benthic invertebrates that inhabit riffles live on rocks and organic debris and are primarily exposed to contaminants in water. As with fish, exposures of benthic invertebrates to water are conservatively estimated as total concentrations, although actual exposures are best estimated by dissolved phase concentrations. Benthic invertebrates that inhabit the soft sediments of pools are exposed to contaminants in the sediments. Two different expressions of sediment contamination are used in ecological risk assessments: whole sediment concentrations and filtered pore water concentrations. The use of pore water is based on the assumption that chemicals associated with the solid phase are largely unavailable and therefore sediment toxicity can be estimated by measuring or modeling the pore water concentration. This is the approach the EPA uses to calculate sediment quality criteria. Whole sediment concentrations do not account for effects of sediment properties on bioavailability; however, they are required by EPA Region IV and may provide a better estimate of risk for highly particle-associated chemicals, which may not be detectable in pore water.

Benthic invertebrates are exposed to surface sediments and not to deep sediments. In a compromise between realism and the desire to include data for all analytes, a maximum interval of 0 to 50 centimeters was used. Analyses of cores with a depth greater than 50 centimeters were discarded, and core sections other than the surface section were discarded.

Because sediment is likely to be more variable in space than in time (due to its relative immobility) and because the organisms are relatively immobile, it is not appropriate to think of benthic invertebrates as averaging their exposures to sediment over space or time. Therefore, the median sediment concentration is an appropriate measure of the central tendency of the contaminant data. An appropriate conservative estimate of this exposure for use in the contaminant screening is the maximum.

As with water, many of the chemical concentrations in sediments are below analytical detection limits. Chemicals that were not detected in any sample in a reach were eliminated. If a reach contained some detects and some nondetects for a chemical, the product limit estimator was used to estimate the mean, which is reported for reference.

Ford et al. (1996) presented a complete description of the sediment sampling and analysis in WAG 2. Note that sediment samples from the lower W4T were aggregated with the IHP samples and that samples from WS were aggregated with LWC samples for consistency with that report.

4.1.2 Sediment Radiation Exposures

Exposures to radionuclides are expressed as the dose rate received by the organism. As with radiation exposures in surface water, the total dose rate is the sum of the internal and external dose rates, which are a function of exposure to the radionuclide and the characteristics of the radiation. Radiation exposures to benthic invertebrates immersed in sediments are likely to be driven by external exposures to contaminated sediments. The exception is for alpha emitters, such as uranium-234 and thorium-230. Internal dose rates are based on the concentrations of the radionuclide in the organism, which is a function of bioavailabilty. Pore water concentrations are assumed to best estimate the bioavailable fraction in sediments, and standard sediment to benthic invertebrate transfer factors are not available. Therefore, the concentration in benthic invertebrates was estimated by using the measured sediment concentrations and the soil-water partition coefficients (Kd) from Baes et al. (1984) to estimate the sediment pore water concentrations. Concentrations in the organism were then estimated based on the biological concentration factors provided in Blaylock et al. (1993). Because the radiation exposure rate is expressed as absorbed dose per unit time (rads/d), the dose rates for each isotope and pathway can be added to determine the total dose rate.

4.2 EFFECTS ASSESSMENT FOR BENTHIC INVERTEBRATES

4.2.1 Single Chemical Sediment Toxicity

Because there are no standard screening benchmarks and sediment quality criteria for only a few chemicals, sets of alternative sediment benchmarks were derived for each chemical (Table 4.1). Whole sediment concentrations are compared to these alternative benchmarks. The use of multiple benchmarks provides greater assurance of detecting all COPECs. Sediment quality criteria are corrected for site-specific conditions. Sediment benchmarks derived using the equilibrium partitioning method are calculated using location-specific percent organic carbon.

4.2.2.2 Radiological effects

The recommended acceptable dose rate to natural populations of aquatic biota is 1 rad/d (NCRP 1991). Although there are no standard screening benchmarks for radionuclide concentrations in sediments, Blaylock et al. (1993) provide formulas and exposure factors estimating the dose rates to representative aquatic organisms. These formulas were used to calculate the sediment concentration that results in a total dose rate of 1 rad/d to a small invertebrate for each radionuclide detected in the sediment samples from BCV. The exposure factors used in this assessment are from Blaylock et al. (1993) and are presented in Table 3.2. The benchmarks include internal and external exposures from the detected isotope and all short-lived daughter products. All major alpha, beta, and gamma emissions for each isotope were included; and the absorbed fraction of each was estimated from the graphs provided in Blaylock et al. (1993). Conservatism was practiced by assuming the representative invertebrate was immersed in the sediments at all times. Internal exposures from radionuclides in sediments were estimated using the soil-water partition coefficients and biological concentration factors for the detected parent isotopes.

Table 4.1 Descriptions of the ecotoxicological screening benchmarks for benthic biota exposed to contaminated sediments

Benchmark	Abbreviation	Description
Effects Range-Low	ER_L	The tenth percentile of estuarine sediment concentrations reported to be associated with some level of toxic effects.
Effects Range-Median	ER_M	The fiftieth percentile of estuarine sediment concentrations reported to be associated with some level of toxic effects.
Region IV Benchmark	REG_IV	The higher of two values, the EPA Contract Laboratory Program Practical Quantification Limit (CLP PQL) and the Effects Value which is the lower of the ER-L and the Florida NOEL.
Threshold Effect Level	TEL	The geometric mean of the fifteenth percentile of reported concentrations which were associated with some level of effects and the fiftieth percentile of reported concentrations which were associated with no adverse effects. All data are for marine and estuarine sediments.
Probable Effect Level	PEL	The geometric mean of the fiftieth percentile of reported concentrations which were associated with some level of effects and the eighty-fifth percentile of reported concentrations which were associated with no adverse effects. All data are for marine and estuarine sediments.
National Sediment Quality Criteria	EPASQC_A	Sediment quality criteria based on toxicity in water expressed as chronic water quality criteria (recalculated after adding some benthic species) and partitioning of the contaminant between organic matter (1% of sediment by weight) and pore water. Sediment quality criteria were adjusted to the site-specific percent total organic matter content.
Equilibrium Partitioning Benchmark	EQPSQB_A	Benchmarks derived in the same manner as sediment quality criteria except that the expression of aqueous toxicity is the chronic NAWQC or the secondary chronic value, and site-specific percent organic matter is used.

Table 4.1 (continued)

Benchmark	Abbreviation	Description
Ontario Ministry of the Environment Lowest Effect Level	LEL_MOE	Concentrations determined by the Ontario MOE to constitute thresholds for toxic effects in Ontario sediments.
Ontario Ministry of the Environment Severe Effect Level	SEL_MOE	Concentrations determined by the Ontario MOE to constitute thresholds for severe toxic effects in Ontario sediments.
Region V Benchmark for nonpolluted sediments	REG_V_L	A concentration determined to constitute background for sediments in Illinois.
Region V Benchmark for heavily polluted sediments	REG_V_M	A concentration determined to constitute the upper bound of moderately polluted sediments relative to background sediments in Illinois.
Apparent Effect Threshold	AET	A concentration above which toxic effects occurred at all sites in Puget Sound.

Note: More details are presented by Hull and Suter (1994), Long et al. (1995), MacDonald (1994), and Region IV (1994). The last five benchmarks are used only when none of the first seven are available

4.2.2 Invertebrate Community Surveys

4.2.2.1 Sediment communities

Surveys of benthic invertebrates in soft sediments behind weirs and dams were performed in 1995 (Appendix A). The surveys were conducted in large part to determine whether dredging these sediments to maintain pool depth would destroy communities that had any particular ecological value. However, they serve to indicate the effects of sediment contamination more generally including natural pools where sediments accumulate. The survey included two weirs on WOC within WAG 2, two on Melton Branch, and two sites in WOL. These were compared to reference pools behind weirs on upper First Creek, WOC above ORNL, and Clear Creek. The locations are described in Appendix A. The results are somewhat confounded by differences in sediment characteristics among the weirs. The reference weirs tended to have more coarse particulate organic matter (detritus) and less silt than the weirs in WAG 2. This is not surprising given the silt that would be expected to enter the streams from construction and remediation activities at ORNL. The two WOL sites are different from all of the weirs in the depth of water and associated physical characteristics.

The lowest taxonomic richness was found in WOL (~ 4 taxa per sample) ,and the highest was found in background WOC (~ 15 taxa/sample) and upper Clear Creek (~ 12 taxa/sample). The other two reference weirs and the WAG 2 weirs all had (~ 9-11 taxa/sample). Since chironomid midge larvae made up most of the taxa identified, they were tabulated separately and gave the same pattern of relative taxonomic richness. Hence, although taxonomic richness varies by more than a factor of three among sites, the difference could be attributed to physical habitat differences as well as contamination.

4.2.2.2 Riffle communities

Surveys of benthic invertebrates in riffles were performed beginning in 1987 with the last reported results from 1992 (Ashwood 1994). At that time, the taxonomic richness of all invertebrates and particularly of the Ephemeroptera, Plecoptera, and Trichoptera (EPT) was severely reduced at sites on WOC adjacent to and downstream of ORNL. They were also reduced in lower First and Fifth Creeks and in Melton Branch relative to reference sites. However, they were elevated in lower NWT relative to the upstream reference site. These comparisons of upstream and downstream sites are confounded by habitat differences as with the soft sediment communities. However, the differences in substrate are not as great, and because riffle communities have been much more studied than soft sediment communities, it is possible to judge which differences are attributable at least in part to contamination. Therefore, it can be judged with some confidence, that the reduced taxonomic richness was not due simply to stream habitat gradients (Ashwood 1994).

4.3 RISK CHARACTERIZATION FOR BENTHIC INVERTEBRATES

4.3.1 Single Chemical Sediment Toxicity

Screening against benchmarks. Chemicals detected in whole sediments were screened against benchmarks and evaluated as COPECs. An HQ was calculated for each chemical by dividing the maximum concentration by the sediment benchmarks (Table B.5). Chemicals that exceeded any benchmark (e.g., HQ>1) were examined further to determine whether they were

Table 4.2. Results of screening of chemicals that exceed benchmarks in sediments for COPECs

Chemical	COPEC	Reason for Inclusion or Rejection
2-methylnaphthalene	No	Detected only in two reaches and only one and two samples; it barely exceeds the effects range-low (ER-L) in one sample.
Acenaphthene	Yes	Exceeds all benchmarks in Middle White Oak Creek (MWC) and multiple benchmarks in Intermediate Holding Pond (IHP) and Lower White Oak Creek (LWC).
Anthracene	Yes	Exceeds all benchmarks in MWC and multiple benchmarks in IHP and LWC.
Aroclor - 12xx		See PCBs
Arsenic	Yes	Exceeds the threshold effect level (TEL) in all reaches and the ER-L in all but LMB but not the ER-M or PRE in any reach.
Barium	Yes[a]	Exceeds both Region V benchmarks
Benzo(a)anthracene	Yes	Exceeds multiple benchmarks in IHP and MWC
Benzo(a)pyrene	Yes	Exceeds multiple benchmarks in IHP and MWC
Cadmium	Yes	Exceeds multiple benchmarks in IHP and MWC
Chromium	Yes	Exceeds the ER-L in IHP and LMB
Chrysene	Yes	Exceeds multiple benchmarks in IHP and MWC
Copper	Yes	Exceeds multiple benchmarks in IHP, LMB, LWC, and MWC
Dibenz(a,h)anthracene	Yes	Exceeds the ER-L and TEL in IHP and MWC and the PEL in IHP, but detected in only one sample/reach.
Fluoranthene	Yes	Exceeds the ER-L and TEL in IHP and MWC and the TEL in LWC.
Fluorene	Yes	Exceeds the ER-L and PEL in IHP and MWC.
Iron	Yes[a]	Exceeds the Ontario MOE's SEL.
Lead	Yes	Exceeds the ER-L and TEL in IHP and MWC and barely exceeds the Region IV value in LWC and the PEL in MWC.

Table 4.2 (continued)

Chemical	COPEC	Reason for Inclusion or Rejection
2-methylnaphthalene	No	Detected only in two reaches and only one and two samples; it barely exceeds the effects range-low (ER-L) in one sample.
Acenaphthene	Yes	Exceeds all benchmarks in Middle White Oak Creek (MWC) and multiple benchmarks in Intermediate Holding Pond (IHP) and Lower White Oak Creek (LWC).
Anthracene	Yes	Exceeds all benchmarks in MWC and multiple benchmarks in IHP and LWC.
Aroclor - 12xx		See PCBs
Manganese	Yes[a]	Exceeds MOE benchmarks in all reaches.
Mercury	Yes	Exceeds multiple benchmarks in all reaches but particularly high in IHP and MWC.
Nickel	Yes	Exceeds multiple benchmarks in all reaches
PCB, total	Yes	Exceeds multiple benchmarks in all reaches but particularly high in IHP and MWC.
Phenanthrene	Yes	Exceeds multiple benchmarks in IHP, LWC and MWC but particularly high in MWC.
Pyrene	Yes	Exceeds the ER-L in IHP, and MWC
Silver	Yes	Exceeds multiple benchmarks in IHP and MWC.
Zinc	Yes	Exceeds multiple benchmarks in IHP, LMB, and MWC.

[a]These COPECs are based on weak benchmarks which are the only ones available.

credible COPECs (Table 4.2). Only 2-methylnaphthalene could be eliminated without comparison to background and a more detailed evaluation of their distribution among reaches and the relevant effects data. These comparisons will be included in the White Oak Creek watershed RI.

Chemicals that were detected in sediment but for which no benchmarks are available are also of concern. They are aluminum, benzo(p)fluoranthene, benzo(g, h, I) perylene, benzo(k)fluoranthene, beryllium, boron, butylbenzylphthalate, carbazole, cobalt, indeno(1,2,3-cd)pyrene, lithium, methoxychlor, molybdenum, osmium, selenium, strontium, thallium, tin, and hexachlorobenzene.

4.3.2 Sediment Radiation Characterization

The concentrations of detected radionuclides were compared with the screening values for small invertebrates immersed in sediments. An HQ was calculated for each radionuclide by dividing the measured concentration by the screening value. The screening value is the concentration resulting in a total dose rate of 1 rad/d from the major emissions of the measured parent radionuclide and all short-lived progeny. A total dose rate to small invertebrates at each site was calculated by summing the HQs for all detected radionuclides. Radionuclides present a significant risk to benthic invertebrates if the sum of HQs for a site exceeds 1.0. Table B.5 details the detected concentrations, screening values, HQs for individual radionuclides, and the sum of HQs for all radionuclides detected at each site. Based on these results, radionuclides in BCV sediments appear to present a negligible risk to benthic invertebrates. That is, the individual and sum of HQs were considerably <1.0 at all locations.

4.3.3 Benthic Community Surveys

The benthic community survey data cannot be directly related to the chemical analyses because WOL and the community reference sites were not subject to chemical analyses and because differences in the sampling would tend to obscure patterns. The communities in the four pools on WOC and MB within WAG 2 had total taxonomic richnesses that were within the range of reference communities although not as high as the best site on upper WOC (Appendix A). The only indication of a contaminant effect in the soft sediment community is chironomid taxonomic richness (by far the most abundant taxon) in MWC (the most contaminated sediment), which was lower than in reference pools.

4.3.4 Weight of Evidence for Benthic Invertebrates

PCBs (both totals and Arochlor mixtures), 10 polycyclic aromatic hydrocarbons (PAHs), and 12 metals were found at concentrations high enough to indicate a hazard to benthic invertebrates. Sediment toxicity is unknown. Community surveys in depositional areas do not indicate a large effect of contamination except possibly in WOL, but they are confounded by habitat differences. Sediment chemical concentrations at most benthic community survey sites are unknown and were not available for most tributaries. Hence, the sediments are contaminated, but the benthic community is not clearly different from reference communities; or in the case of WOL, the community is different, but contaminant levels are not known.

The communities of riffles have low species richness below ORNL, which is likely to be primarily due to contaminants in water (Chap. 3). The dominance of this mode of exposure is the reason that riffle invertebrate communities are monitored to determine whether water quality goals are being met (Ashwood 1994). However, the feeding habits of some species expose them to the small amount of particulate material that is deposited in such areas so sediment contamination may make a small contribution to this effect.

4.4 UNCERTAINTIES CONCERNING RISKS TO BENTHIC INVERTEBRATES

The following issues are believed to constitute the major sources of uncertainty in the risk assessment for the benthic invertebrate assemblage.

- The benchmarks against which the sediment concentrations are screened are from studies of estuaries and large lakes that may differ considerably in their physical and ecological properties from the steams being assessed.
- The toxicity of the sediments is unknown.
- The chemical concentrations in WOL and in most tributaries are unknown.
- Chemical concentrations in sediments from most of the benthic invertebrate survey sites are unknown.

5. RISKS TO SOIL INVERTEBRATES

This chapter addresses risks from chemical soil invertebrates, which are represented by earthworms. Radiological risks to earthworms are presented in this report along with other terrestrial endpoints.

5.1 EXPOSURE ASSESSMENT FOR SOIL INVERTEBRATES

Earthworms absorb a variety of inorganic and organic soil contaminants through both feeding in soil and litter and burrowing in soil. It is assumed that these organisms represent highly exposed soil invertebrates. It is not possible or necessary to distinguish the exposure of earthworms to solid, liquid, and gaseous phases of the soil. Both in toxicity tests and the field, earthworms integrate exposures to all soil components by all uptake routes.

The bioavailability of the contaminants in soil and decomposing plant litter are controlled by soil, litter, and analyte characteristics; however, current knowledge is not sufficient to allow the prediction of bioavailability based on soil variables. An exception is the association of extremes in pH and metal availability (van Gestel 1992).

It is assumed that the earthworms spend their entire life span in soil with contaminant levels represented by the 0.5 m depth of soil sampled. The assumption is reasonable, though organisms feeding on well-decomposed organic material near the surface may be exposed to different quantities of organic chemicals than worms feeding on litter pulled down into burrows in the subsoil (Curl et al. 1987). Few earthworms are found in waterlogged soils so exposure to seeps is not considered. Because earthworms and other soil invertebrates may spend their entire lives at a single sampling location, the maximum detected contaminant concentrations were screened against toxicity benchmarks, as described in Sect. 5.2.1. These concentrations are used to assess potential negative impacts of contaminated soils on soil invertebrate populations.

5.2 EFFECTS ASSESSMENT FOR SOIL INVERTEBRATES

Toxicity tests and biological surveys were not performed at these sites. Earthworms were sampled, but quantitative surveys were not performed. Thus the only line of evidence in the risk assessment is the screening against benchmarks, described in the following paragraphs. Where benchmarks were not available, risks could not be evaluated.

Contaminant concentrations in soils were compared to toxicological benchmarks for earthworms in order to screen out chemicals that do not constitute a hazard to soil invertebrates and therefore do not require detailed risk characterization. Chemicals that were not screened out because the maximum detected concentrations were greater than benchmarks may potentially pose significant risks to soil invertebrates. References used for the derivation of benchmarks were reports of toxicity tests of individual chemicals in laboratory, greenhouse, or field settings. Tests conducted with natural soils in the laboratory were assumed to represent the exposures of worms in floodplain soils. However, chemicals freshly added to soils in toxicity tests may be more bioavailable than the chemical forms that occur in field soils. Only experiments in which earthworms were exposed to soil (natural or artificial mixture of natural components), soil/litter microcosms, or manure were considered. Twenty percent reduction in growth, reproduction, or activity was used as the threshold for significant effects to be consistent with other screening benchmarks for ecological risk assessment and because the FFA parties adopted this level of effects for ecological endpoints (Suter et al., 1995).

The method used for deriving soil benchmarks are described in Will and Suter (1995a). Screening benchmarks for toxic effects of contaminants present in WAG 2 soils are presented with the hazard quotients in Table B.6. Benchmarks were not available for any of the organic compounds detected in the floodplain, except n-nitrosodiphenylamine.

5.3 RISK CHARACTERIZATION FOR SOIL INVERTEBRATES

The evidence concerning risks to soil invertebrates consists of analyses of inorganic, organic, and radionuclide contaminants in soil. This single line of evidence is summarized in the following text. Toxicity tests and quantitative biological surveys were not available.

5.3.1 Single Chemical Soil Toxicity

Contaminants of potential ecological concern (COPECs) for soil invertebrates in the WAG 2 floodplain were identified by comparing maximum concentrations in soil at each location to the following: (1) twice the mean background for soils from WAG 5 (DOE 1995); and (2) available benchmarks for toxicity to soil invertebrates (Will and Suter 1995a). Maximum detected concentrations in soil are used for screening purposes because earthworms and other soil invertebrates are relatively immobile and can therefore be exposed to the maximum concentration during the course of their lifetimes (Suter et al. 1995). If both background and benchmark criteria were exceeded, the contaminant was identified as a COPEC. he magnitude of exceedence of a benchmark is described by the hazard quotient, the maximum concentration in soil divided by the toxicity benchmark concentration. If background criteria were not available, exceedence of a benchmark was sufficient for the COPEC designation. Benchmarks, maximum concentrations of analytes at each sample location exceeding background for which benchmarks are available, and hazard quotients for the contaminants are given in Table B.6. The major COPECs for soil invertebrates were mercury and chromium. Mercury and chromium are among the few chemicals for which the benchmarks for toxicity to soil invertebrates is lower than the phytotoxicity benchmarks.

Intermediate Holding Pond. The maximum detected concentrations of chromium, mercury, and nickel were 100, 800, and 2 times the benchmarks for toxicity to earthworms.

Lower White Oak Creek. The maximum detected concentrations of chromium and mercury were 200 and 50 times the benchmarks for toxicity to earthworms. Although the maximum detected concentration of zinc is apparently equivalent to the benchmark if a single significant digit is used, the concentration of 190 mg/kg is lower than the benchmark concentration of 200 mg/kg. Also, the mean concentration of zinc in the soils of the IHP reach was about half that of the benchmark concentration. Thus, zinc is not a COPEC for toxicity to soil invertebrates.

Middle White Oak Creek. The maximum detected concentration of mercury was 30 times the benchmark for toxicity to earthworms.

Lower Melton Branch. The maximum detected concentration of mercury was 20 times the benchmark for toxicity to earthworms.

The strength in the literature toxicity line of evidence is in its ability to relate the concentration of a single contaminant in soil to observed effects. The screening is most useful if the chemical species and bioavailability of contaminants in WAG 2 floodplain soils are comparable to those of the chemical in laboratory toxicity tests in other soils. In these soils, the fractions of mercuric chloride, methyl mercury, mercuric sulfide, and some others were not determined. Thus, the hazard quotients for mercury may be quite conservative.

5.3.3 Summary of Risk Characterization for Soil Invertebrates

As in any screening assessment, potential hazard is indicated by the exceedence of toxicological benchmarks. COPECs for soil invertebrates in the WAG 2 floodplain include chromium, mercury and nickel. Further studies, such as toxicity tests (or other measures of bioavailability) and chemical speciation of mercury, would be advisable before concluding that these contaminants constitute hazards to soil invertebrates.

5.4 UNCERTAINTIES CONCERNING RISKS TO SOIL INVERTEBRATES

The following factors create uncertainty in assessing the risk posed by the potential contaminants of concern in soils:

'Bioavailability' of elements. The extraction methods used during sample collection may remove from the soil quantities of elements and compounds greater than those that are extracted from soil as they pass through the gut of the earthworm or those that are adsorbed dermally.

Variable response to contaminants. Various earthworm and other soil invertebrate species and earthworm growth stages tolerate contaminants to different degrees. The literature from which benchmarks were derived is not necessarily based on experiments using earthworm species known to be representative of those occurring in the soils of WAG 2.

Multiple contaminant exposure. Toxicity benchmark concentrations are derived from experiments in which earthworms are exposed to single contaminants. Typical soils of the WAG 2 expose earthworms to multiple contaminants which may not be adequately assessed on the basis of literature-derived benchmarks.

Lack of benchmarks for some metals and most organic compounds. Little research has been conducted on the toxic effects of the 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 the soils.

Chemical speciation. Earthworm toxicity benchmarks for metals are often derived from studies with a single species of a contaminant.

6. RISKS TO PLANTS

This chapter addresses risk from chemicals to plants. Radiological risks to plants are assessed along with other terrestrial endpoints.

6.1 EXPOSURE ASSESSMENT FOR PLANTS

Vegetation growing in the WAG 2 floodplain is exposed to contaminants in soil and potentially in shallow groundwater near seeps and intermittent tributaries.

6.1.1 Soil Exposures

Concentrations of analytes in the soil derived from double acid extractions (nitric and hydrochloric) generally representing the total potential reservoir of contaminants, not the concentrations to which plants are exposed at any one time. Some metals are more readily available for uptake by plants than others, depending on the solubility and species of the compound, interactions with soil constituents (organic matter and clay) and with other contaminants, and mechanisms for bioaccumulation in particular plant species. Organic contaminants may interact strongly with soil organic matter and therefore only be partially available for plant uptake.

Contaminants have vertical distributions in soil that reflect interactions between the soil and the contaminant, controlled by the chemical and physical characteristics of the soil and the quantity and chemical characteristics of the contaminant. The exposure of plant roots to a contaminant in the soil depends on these abiotic factors and the growth characteristics of individual plants, such as rooting depth and density. Deep-rooted trees and shallow-rooted grasses may be exposed to different concentrations of contaminants, depending on the location of the source. In the absence of data on root distribution, it is assumed that the plants are rooted within the zone from which the soil was sampled for analysis.

Two other routes of exposure cannot be addressed because of a lack of data. These are exposure to volatile organic compounds in the air above and in the soil and contaminants in dust particles deposited on above-ground plant parts. These routes are assumed to contribute negligibly to plant exposure in the WAG 2 floodplain.

6.1.2 Soil Solution Exposures

Plants growing in seep areas are exposed to contaminants in the aqueous phase. Numerous seeps, springs, or intermittent tributaries in the WAG 2 floodplain were sampled for contaminants. It is assumed that the roots of plants near the springs may be exposed to water containing contaminants at similar concentrations. Although the discharge in many of these seeps is seasonal and heightened during rainfall, we make the conservative assumption that plants growing at these locations are exposed to contaminants in the water throughout the year. We also assume that the entire root systems of plants growing in seep areas are exposed to contaminants and that this type of exposure is the only significant source for the plants.

Filtered water samples from seeps are better representations than unfiltered samples of concentrations to which plants are exposed. Samples for this assessment were not filtered; thus the data are conservative estimates of exposure.

6.2 EFFECTS ASSESSMENT FOR THE PLANT COMMUNITY

Contaminant concentrations in soils and springs were compared to toxicological benchmarks (Will and Suter 1995b) in order to screen out chemicals that do not constitute a hazard to plants and therefore do not require detailed risk characterization. Tests conducted in natural soils in the laboratory were assumed to be representative of the exposure of plants to contaminants measured in floodplain soils, with the possibility that chemicals freshly added to soils in literature toxicity tests may be more bioavailable than contaminants in WAG 2 soils. Tests conducted in nutrient and mineral solutions are the best available representation of exposures of plants to contaminants in the vicinity of seeps and springs.

Benchmarks were not available for numerous organic contaminants, including polycyclic aromatic hydrocarbons.

6.3 RISK CHARACTERIZATION FOR THE PLANT COMMUNITY

The evidence concerning risks to the plant community consists of analyses of inorganic, organic and radionuclide contaminants in soil. This single line of evidence is summarized in the following text. Toxicity tests and biological surveys were not available.

6.3.1 Single Chemical Soil and Solution Toxicity

COPECs for terrestrial plants in the WAG 2 floodplain were identified by comparing maximum detected site concentrations of soil to the following: (1) twice the mean background for soils from WAG 5 (DOE 1995) and (2) available benchmarks for toxicity to terrestrial plants (Will and Suter 1995b). The screening of maximum concentrations against benchmarks is intended to protect plants that spend their entire lives in a single, highly contaminated location. If both background and benchmark criteria were exceeded, the contaminant was identified as a COPEC. The magnitude of exceedence of a benchmark is described by the hazard quotient, the maximum concentration in soil divided by the toxicity benchmark concentration. If background criteria were not available, exceedence of a benchmark was sufficient for the COPEC designation. Benchmarks for plants growing in soils, maximum concentrations of analytes at each reach exceeding background and for which benchmarks are available, and hazard quotients for the contaminants are given in Table B.7.

COPECs for plants potentially exposed to water from seeps and springs were identified by a similar comparison of contaminant concentrations with background and toxicity benchmarks (Table B.8).

6.3.1.1 Terrestrial plants

The screening of contaminant concentrations in soil against phytotoxicity benchmarks is provided in Table B.7.

Intermediate Holding Pond. The maximum detected concentrations of boron, chromium, lead, lithium, and manganese were 20, 70, 2, 20, and 8 times the phytotoxicity benchmarks (i.e., concentrations causing toxic effects when added to surface soil). Mercury, molybdenum, nickel, silver, and zinc were detected at 300, 3, 10, 8, and 3 times the benchmarks.

Lower White Oak Creek. The maximum detected concentrations of boron, chromium, lithium, manganese, mercury, molybdenum, silver, and zinc were 20, 100, 8, 3, 20, 2, 3, and 4 times the phytotoxicity benchmarks. Lead and selenium were detected at concentrations only slightly above benchmarks (hazard quotient of 1 if appropriate significant figures were used).

Middle White Oak Creek. The maximum detected concentrations of boron, lithium, mercury, molybdenum, selenium, silver, and zinc were 20, 7, 8, 2, 2, 2, and 2 times the phytotoxicity benchmarks.

Lower Melton Branch. The maximum detected concentrations of antimony, boron, lithium, manganese, mercury, and molybdenum were 600, 2, 20, 10, 20, 7, and 4 times benchmarks. Selenium and thallium were detected at concentrations only slightly above benchmarks (hazard quotient of 1 if appropriate significant figures were used).

The strength of the literature toxicity line of evidence is in its ability to relate the concentration of a single contaminant in soil to observed effects. The screening is most useful if the chemical species and bioavailability of contaminants in WAG 2 floodplain soils are comparable to those of the chemicals in laboratory toxicity tests in other soils. In these soils, the fractions of mercuric chloride, methyl mercury, mercuric sulfide, and others were not determined. Thus, the hazard quotients for mercury may be conservative.

6.3.1.2 Plants exposed to seeps and springs

The screening of contaminant concentrations in soil solution against phytotoxicity benchmarks is provided in Table B.8.

Embayment of White Oak Creek. The maximum detected concentration of aluminum at WOCET was 8 times the benchmark for plants in solution.

Homogeneous Reactor Test Tributary. The maximum detected concentration of aluminum in SW9-2 was 7 times the plant benchmark concentration for aluminum in solution.

Intermediate Holding Pond. No contaminants were detected above plant benchmarks in seeps or tributaries.

Lower Melton Branch. The maximum detected concentration of manganese at MID. DRAIN. was slightly above the benchmark for plants in solution. No contaminants were detected above plant benchmarks for the following seeps or tributaries: MBTRIB-3, SW2-5, SW5-2, and SW5-4.

Lower White Oak Creek. The maximum detected concentrations of aluminum from the East Seep, SW7-3, SW7-5, SW7-6, and WCTRIB-1 were 10, 4, 10, 4, and 5 times the plant solution benchmark. No other contaminants were detected above background and benchmarks in seeps in this region.

Middle White Oak Creek. No contaminants were detected above plant benchmarks in seeps or tributaries.

Upper Melton Branch. No contaminants were detected above plant benchmarks in seeps or tributaries.

Unknown reach. The maximum detected concentration of iron in SW2-7 was 2 times the plant solution benchmark.

WAG 4 Tributary. The maximum detected concentrations of arsenic and iron at SW4-1 were 6 and 3 times the plant solution benchmarks. The maximum detected concentration of nickel in SW4-2 was 50 times the plant solution benchmark. Aluminum was detected at 5 times the phytotoxicity benchmark in both WAG 4 MS1 and WAG 4 T2A. Chromium was detected at a concentration slightly above the plant solution benchmark in WAG4 T2A. No contaminants were detected above benchmarks in BTT.

West Seep. The maximum detected concentrations of aluminum and arsenic, chromium, and iron in RS-3A were 90, 20, 2, and slightly greater than 1 times the benchmark for toxicity to plants in solution. Aluminum was detected in RS-3B at up to 7 times the benchmark. Manganese in SW7-1 and aluminum and cobalt in SW7-2 were detected at 2, 3, and 2 times the phytotoxicity benchmark.

Aluminum and iron. Of the contaminants detected at seeps and springs at concentrations above benchmarks, aluminum and iron are unlikely to be of ecological concern. In unfiltered water, both of these contaminants are significantly associated with the particulate fraction, which is wholly or largely unavailable to plants. Also, aluminum and iron are typically found at high background concentrations (twice the mean background is 0.34 mg/kg for Al and 0.81 mg/kg for Fe) in this region.

6.3.2 Summary of Risk Characterization for the Plant Community

As in any screening assessment, potential hazards are indicated by the exceedence of toxicological benchmarks. COPECs for the plant community exposed to soils in the WAG 2 floodplain include: boron, chromium, lead, lithium, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, and zinc. COPECs for the plant community exposed to seeps or springs in the WAG 2 floodplain include: arsenic, chromium, cobalt, manganese, and nickel. Further studies, such as toxicity tests (or other measures of bioavailability) and chemical speciation of mercury, would be advisable before concluding that these contaminants constitute hazards to the plant community.

6.4 UNCERTAINTIES CONCERNING RISKS TO PLANTS

The following factors create uncertainty in assessing the risk posed by the contaminants of potential concern in WAG 2 soils, seeps, and springs.

Bioavailability of elements in soil. The extraction methods used may remove from the soil quantities of chemicals 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. Estimates of exposure are confounded by the concentration- and species-dependent synergistic and antagonistic interactions between metals during uptake by roots and inside plants.

Bioavailability of elements in groundwater and springs. Concentrations of contaminants such as aluminum in whole, unfiltered groundwater are probably higher than the concentrations bioavailable to plants. In unfiltered water samples (i.e., containing soil and organic particles), acidification will bring into solution metals strongly bound to this particulate matter.

Variable response to toxicants. Various plant species and plant growth stages tolerate contaminants to different extents. The literature from which benchmarks were derived is not based on experiments using plants found in ecosystems representative of Bear Creek Valley. It is difficult to extrapolate from largely agricultural crops in early growth stages used in most of the published literature to trees and the varied ground vegetation found at WAG 2.

Multiple contaminant exposure. 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. Typical soils in WAG 2 expose plants to multiple contaminants which may not be adequately assessed on the basis of literature-derived benchmarks.

Lack of benchmarks for some metals and most organic compounds. Little research has been conducted on the phytotoxic effects of the organic compounds. Thus, it is not possible to assess the risk to plant growth posed by some of the analytes found in the Bear Creek Valley soils.

Chemical speciation. The analytical techniques did not generally differentiate among species of chemicals present in the soil which have variable toxicity to plants.

7. RISKS TO WILDLIFE

Risks from radionuclides and nonradionuclides (e.g., organic and inorganic chemicals) were estimated for small mammals, piscivores, and wide-ranging wildlife species within WAG 2. Because the methods for risk estimation differ greatly between radionuclide and nonradionuclide contaminants, each is addressed separately. Risks to wildlife from chemicals are presented in Sect. 7.1; risks from radionuclides to wildlife, terrestrial plants, and earthworms are presented in Sect. 7.2.

7.1 RISKS TO WILDLIFE FROM CHEMICALS

7.1.1 Exposure Assessment

Wildlife may be exposed to contaminants through ingestion of food, soil, and water. In this assessment, exposure through ingestion of food, soil, and water was estimated using exposure models (described in the following paragraphs. For areas where there was insufficient data to assess complete exposure, exposure through water was assessed by comparing unfiltered water concentrations to water benchmarks for wildlife (see Sect 7.1.2.1). Contaminant exposure through ingestion was estimated for small mammals (short-tailed shrew and white-footed mouse), wide-ranging species (white-tailed deer, wild turkey, red-tailed hawk, and red fox) and piscivores (mink and belted kingfisher). Exposure estimates were calculated using soil and soil-biota uptake factors for small mammals, earthworms, and herbaceous, canopy, and browse vegetation. Uptake factors for small mammals and herbaceous, canopy, and browse vegetation were derived from data from 15 locations within the Bear Creek watershed (four within BCOU1 and 11 within the Bear Creek floodplain). Uptake factors for earthworms were derived from data from Bear Creek, WAG 5 and two locations within WAG 2. Exposure to piscivores was assessed using fish data from eight locations in the WOC watershed.

7.1.1.1 Oral Ingestion Exposure Model

Oral exposure to contaminants experienced by wildlife may come from multiple sources. They may consume contaminated food (either plant or animal), drink contaminated water, or ingest soil or sediment. Soil or sediment ingestion may be incidental while foraging or grooming or purposeful to meet nutrient needs. The total oral exposure experienced by an individual is the sum of the exposures attributable to each source and may be described as follows:

$$Etotal \approx Efood + Ewater + Esoil, \qquad \textbf{(1)}$$

where

E_{total} = total exposure from all pathways,
E_{food} = exposure from food consumption,
E_{water} = exposure from water consumption,
E_{soil} = exposure from soil consumption.

For exposure estimates to be useful in the assessment of risk to wildlife, they must be expressed in terms of a body weight-normalized daily dose or mg contaminant per kg body weight per day (mg/kg/d). Exposure estimates expressed in this manner may then be compared to toxicological benchmarks for wildlife, such as those derived by Opresko et al. (1995), or to doses reported in the toxicological literature. Estimation of the daily contaminant dose an individual may receive from a

$$E_j = \sum_{i=1}^{m} p_{ik}\left(\frac{IR_i \ x \ C_{ijk}}{BW}\right) , \qquad (2)$$

particular medium for a particular contaminant may be calculated using the following equation:
where

E_j = total exposure to contaminant (j) (mg/kg/d),
m = total number of ingested media (e.g., food, soil, or water),
IR_i = ingestion rate for medium (I) (kg/d or L/d),
p_{ik} = proportion of type (k) of medium (I) consumed (unitless),
C_{ij} = concentration of contaminant (j) in type (k) of medium (I) (mg/kg or mg/L),
BW = body weight of endpoint species (kg).

Exposure estimates were calculated for all contaminants detected in soil or water from WAG 2. Because wildlife are mobile, their exposure is best represented by the mean contaminant concentration in media. To be conservative, the 95% UCL is used in exposure estimates. These data were used in the initial exposure estimates. Exposure estimates for contaminants that may present a risk to wildlife [based upon comparisons to no observed adverse effects levels (NOAELs) and lowest observed adverse effects levels (LOAELs)] were recalculated using Monte Carlo simulations. (**Note:** because the purpose of the initial exposure estimate is to be conservative and to identify COPECs, the 95% UCL was used regardless of whether or not the value exceeded the maximum observed value. Overestimates of exposure that may occur at the screening level are addressed through the use of Monte Carlo simulation).

7.1.1.2 Piscivore exposures

Exposure of mink and belted kingfisher to mercury and PCBs in fish from WOC was estimated at eight locations within the watershed (and one reference location outside that watershed) as part of the *Preliminary Oak Ridge Reservation-wide Ecological Risk Assessment* (Sample et al. 1995). Estimates for mink and kingfisher are presented in Tables C.1 and C.2 . Additional detail on these exposure estimates may be found in Sample et al. (1995).

7.1.1.3 Soil-biota uptake factors

Contaminant concentrations in biota were not available for WAG 2. To estimate contaminant concentrations in biota, we used uptake factors developed as part of the Bear Creek RI. Summary statistics for the biota uptake factors are presented in Table C.3. Uptake factors for small mammals, canopy, browse, and herbaceous vegetation were developed exclusively with data from the Bear Creek assessment. In addition to the Bear Creek data, data from two samples in WAG 2 and six from WAG 5 were used to develop the earthworm uptake factors. Contaminant concentrations in biota were estimated by multiplying the biota type-specific uptake factor by the soil concentration.

7.1.1.4 Contaminant concentrations in media

Contaminant concentrations in soil and water are needed to estimate exposure. Soil and water data were aggregated into four reaches: IHP, MWC, LWC, and LMB. Definitions of the bounds of these reaches may be found in Table 2.1.

Summary statistics) for contaminants detected in soil and water from WAG 2 are presented in Appendix C. For comparison, estimates of exposure to inorganic analytes were generated for all endpoints using the 2 × mean background soil and water concentrations obtained from the WAG 5 RI.

7.1.1.5 Exposure modeling using point estimates

Initial estimates of exposure of piscivorous wildlife to contaminants were performed for each of the four reaches using point estimates of parameters in the exposure model. Species-specific parameters necessary to estimate exposure using Eq. 1 are listed in Tables C.4 through C.11.

In estimating contaminant exposure experienced by short-tailed shrew, the following assumptions were made:

- body weight = 0.015 kg;
- food consumption = 0.009 kg/d (fresh weight);
- soil consumption = 0.00117 kg/d (dry weight);
- water consumption = 0.033 L/d;
- diet consists 100% of earthworms or soil invertebrates;
- contaminant concentration in earthworms is representative of that in other invertebrate prey.

To estimate contaminant exposure experienced by white-footed mouse, the following assumptions were made:

- body weight = 0.022 kg;
- food consumption = 0.0034 kg/d (fresh weight);
- soil consumption = 0.000068 kg/d (dry weight);
- water consumption = 0.0066 L/d;
- diet consists 50% of earthworms or soil invertebrates and 50% herbaceous plant material;
- contaminant concentration in earthworms is representative of that in other invertebrate prey.

To estimate contaminant exposure experienced by white-tailed deer, the following assumptions were made:

- body weight = 56.5 kg;
- food consumption = 1.74 kg/d (fresh weight);
- soil consumption = 0.0348 kg/d (dry weight);
- water consumption = 3.7 L/d;
- diet consists 33% of browse plants, 33% canopy plants, and 33% herbaceous plant material.

To estimate contaminant exposure experienced by red fox, the following assumptions were made:

- body weight = 4.5 kg;
- food consumption = 0.45 kg/d (fresh weight);

- soil consumption = 0.0126 kg/d (dry weight);
- water consumption = 0.38 L/d;
- diet consists 80.8% of small mammals and birds, 10.4% browse plant material, and 8.8% earthworms or other invertebrates;
- contaminant concentration in small mammals is representative of that in other vertebrate prey;
- contaminant concentration in earthworms is representative of that in other invertebrate prey.

To estimate contaminant exposure experienced by red-tailed hawk, the following assumptions were made:

- body weight = 1.126 kg;
- food consumption = 0.109 kg/d (fresh weight);
- soil consumption = 0 kg/d (dry weight);
- water consumption = 0.064 L/d;
- diet consists 100% of small mammals and other vertebrates;
- contaminant concentration in small mammals is representative of that in other vertebrate prey.

To estimate contaminant exposure experienced by mink, the following assumptions were made:

- body weight = 1 kg;
- food consumption = 0.137 kg/d (fresh weight);
- water consumption = 0.099 L/d;
- diet consists 54.6% of fish or other aquatic prey and 45.4% small mammals;
- contaminant concentration in fish is representative of that in other aquatic prey.

To estimate contaminant exposure experienced by kingfisher, the following assumptions were made:

- body weight = 0.148 kg;
- food consumption = 0.75 kg/d (fresh weight);
- water consumption = 0.016 L/d;
- diet consists 100% of fish.

To estimate contaminant exposure experienced by wild turkey, the following assumptions were made:

- body weight = 5.8 kg;
- food consumption = 0.174 kg/d (fresh weight);
- soil consumption = 0.0162 kg/d (dry weight);
- water consumption = 0.19 L/d;
- diet consists 90.3% of plant material and 9.7% invertebrates;
- contaminant concentration in earthworms is representative of that in other invertebrate prey.

Using Eq. 3 and the assumptions and data described above, we made point estimates of exposure to contaminants within each of the four WAG 2 reaches and the background for each endpoint (Tables C.12 through C.20).

7.1.1.6 Exposure modeling using Monte Carlo simulations

Employing point estimates for the input parameters in the exposure model does not take into account the variation and uncertainty associated with the parameters and therefore may over or under estimate the contaminant exposure that endpoints may receive. In addition, calculating the model using point estimates produces a point estimate of exposure. This estimate provides no information concerning the distribution of exposures or the likelihood that individuals within the watershed will actually experience potentially hazardous exposures. To incorporate the variation in exposure parameters and to provide a better estimate of the potential exposure experienced by wildlife in WAG 2, the exposure model was re-calculated using Monte Carlo simulations.

Monte Carlo simulation is a resampling technique frequently used in uncertainty analysis in risk assessment (Hammonds et al. 1994). In practice, distributions are assigned to input parameters in a model; and the model is recalculated many times to produce a distribution of output parameters (e.g., estimates of contaminant exposure). Each time the model is recalculated, a value is selected from within the distribution assigned for each input parameter. As a result, a distribution of exposure estimates is produced that reflects the variability of the input parameters.

For all endpoints, Monte Carlo simulations were performed for the entire WAG 2 area. The percentiles of the resulting exposure distributions represent the likelihood that an individual within the modeled area will experience a given exposure level. It was assumed that each sampling location contributed equally to the overall mean exposure (i.e., individuals within each watershed do not preferentially forage at any one location within the watershed). While this assumption is not likely to be ecologically correct (foraging effort and therefore exposure is likely to be biased toward those locations with the most abundant food), data were not available to estimate the preferential use at each sampling location.

Simulations were performed for each contaminant where comparison of point estimates of exposure to LOAELs produced HQs≥1 for at least one location (LOAELs are presented in Sect 7.1.2.; Screening of exposure estimates against LOAELs is presented in Sect. 7.1.3.).

Distributions were used for the following parameters in the exposure model: contaminant concentrations in soil and water and soil-biota uptake factors. All distributions were assumed to be normal. Because these wildlife are mobile, the contaminant concentration they are exposed to on a daily basis is best represented by the average concentration instead of the entire distribution. The standard error of the mean was used to describe variation in the average contaminant concentration.

Monte Carlo simulations were performed using the @Risk software. Samples from each distribution were selected using latin hypercube sampling. The number of iterations, or recalculations, of each exposure simulation was determined by the convergence criteria set in the software. Under these criteria, iterations are performed until the between-iteration percent change in the percentiles, mean, and standard deviation is below 1.5% (i.e., the percentile, mean, and standard deviation for the latest iteration is less than 1.5% different than the those from the previous iteration). Using this convergence criterion, from 600 to 2500 model iterations were performed for each exposure estimate. Monte Carlo estimates of contaminant exposures are presented in Table C.19.

7.1.2 Chemical Effects Assessment for Wildlife

7.1.2.1 Single chemical toxicity data

Single chemical toxicity data consist of no observed adverse effects levels (NOAELs) and lowest observed adverse effects levels (LOAELs) of toxicity studies reported in the literature. NOAELs and LOAELs for wildlife endpoints were estimated from these data using the allometric methods outline in Opresko et al. (1995). This methodology for toxicity extrapolation is equivalent to that the EPA uses in their carcinogenicity assessments and Reportable Quantity documents for adjusting from animal data to an equivalent human dose. Using the allometric scaling factor recommended in EPA (1995), the equation for estimating mammalian NOAELs and LOAELs was:

$$NOAEL_w = NOAEL_t \left(\frac{bw_t}{bw_w} \right)^{1/4} \tag{3}$$

where $NOAEL_t$ and $NOAEL_w$ represent NOAELs for a mammalian test species and wildlife species, respectively. Toxicity values for birds were estimated using the scaling factor derived from Mineau (1996) where:

$$NOAEL_w = NOAEL_t \left(\frac{bw_t}{bw_w} \right)^{0} = NOAEL_t\ (1) = NOAEL_t \tag{4}$$

To evaluate the potential risk that contaminants in water may present, water benchmarks were derived according to the methods outline in Opresko et al. (1995). NOAEL's and LOAEL's and water benchmarks were derived for all seven endpoints. Experimental information used to estimate mammalian benchmarks and NOAEL's and LOAEL's for mammalian endpoints are presented in Tables C.20 and C.21. NOAELs and LOAELs for avian endpoints are listed in Table C.22. Water benchmark values for all endpoints are listed in Table C.23.

7.1.2.2 Biological surveys

Mink Survey. Stevens (1995) investigated bioaccumulation of mercury in mink on the ORR in 1993 through 1995. The methods used in the mink survey, while indicating that mink are present on the Reservation, cannot be used to estimate abundance or density on mink on the ORR. A total of four male mink were live-trapped over the course of 6073 trapnights (trapnight=one trap set for 24 h). One juvenile was captured along East Fork Poplar Creek, two adults were captured along Bear Creek, and one adult was captured along WOC. Captured mink were fitted with an intraperitoneal radio transmitter (to monitor movements and home range) and were released. Prior to release, samples of hair were collected for metals analysis. An additional eight roadkill mink (five male and three female) were collected from the ORR and surrounding areas of Roane and Anderson counties. While one roadkill sample (a male) was collected on a bridge over Bear Creek and was assumed to be a resident of Bear Creek, all others were collected off the ORR and were used as referencesResults of metals analysis are presented in Table 7.1.

Radiotelemetry data on home ranges and movements were obtained for 3 mink—one each from the East Fork Poplar Creek, Bear Creek, and WOC watersheds. Mean (± standard deviation) home range for these three individuals was found to be 7.5±3 km of stream. The entire home range of the East Fork Poplar Creek mink was in a highly urbanized area; it included all of upper East Fork inside the Oak Ridge Y-12 Plant and all areas of East Fork upstream of the Oak Ridge Turnpike–Illinois Avenue intersection. The home range of the WOC mink included all of WOC from the headwater tributaries to the Clinch River, including the X-10 facility. This individual was observed to use dens within the X-10 facility and moved through the facility on several occasions.

Table 7.1. Metal concentrations in hair of Mink from the Oak Ridge Reservation and from off-site reference samples[a]

Site	N	Hg	Se	As	Cd	Pb
East Fork Poplar Creek	1	104	0.69	ND[b]	ND	0.33
Bear Creek	3	10.97±3.42	1.88±.1.41	0.15±0.09	0.04±.0.02	0.97±1.28
White Oak Creek	1	8.8	1	ND	ND	0.37
Offsite	7	5.15±3.43	1.11±0.25	0.22±0.31	0.04±0.02	0.7±0.31

[a]Mean ± standard deviation mg/kg dry weight
[b]ND = Not Detected

Belted Kingfisher Survey. A field monitoring effort (Baron and Ashwood 1996) was initiated in 1994 to evaluate population parameters and contaminant bioaccumulation by belted kingfisher on the ORR. Areas surveyed included: WOC, WOL, WOL embayment, Melton Branch, Poplar Creek, portions of East Fork Poplar Creek (within the Y-12 Plant to downstream of Lake Reality and approximately 1 mile east of the confluence of Poplar Creek), and portions of Bear Creek.

Methods. Nest burrows were monitored for nesting activity. If activity were observed, samples of feathers and eggshells were collected. In addition to specimens collected from the burrows, three carcasses of adult kingfishers were found on the ORR (two from East Fork Poplar Creek and one from WOC). These carcasses were necropsied, and organs were extracted and analyzed for metals and radionuclides. Additional detail concerning methods are reported in Baron and Ashwood (1996).

Results. During April-July of 1994, a total of 27 potential kingfisher burrows were identified on the ORR, 11 of which contained swallow nests. Twenty-five of these burrows were found on the Clinch River. One kingfisher burrow, containing a single unhatched kingfisher egg, was found on WOC (downstream of WOC km 3.5).

One active burrow, containing a clutch of six to seven eggs, was found on the Clinch River. This burrow was later abandoned with no sign of the eggs or the parents. Another burrow, containing six nestlings was located on the Clinch River approximately 12 miles upstream of all DOE contaminant outfalls. It was, therefore, considered uncontaminated. Three weeks following the initial observation of this burrow, three nestlings had fledged and three had died. Feathers were collected and analyzed. Results of residue analysis for eggshells and feathers from nestlings are presented in Table C.24; results for adult carcasses are presented in Table C.25.

Nestling feathers collected from the burrow on the Clinch River, upstream of ORR outfalls (Table C.24), contained relatively low levels of metal and radioactive contaminants. Feathers from the carcasses of three fledglings accumulated similar concentrations of As, Cd, Pb, Se, and Hg. Mercury concentrations in feathers were approximately 1 mg/kg. Mercury concentrations found in fish downstream of the nesting site are approximately 0.04 ± 0.01mg/kg (Peterson et al. 1994). Thus, biomagnification is occurring in kingfishers foraging in up-gradient areas of the ORR. However, these feather concentrations are much lower than those found in adult kingfishers on the ORR. While selenium concentrations in nestling feathers appear high, they are similar to selenium levels in adult kingfishers (Table C.25) and mink and raccoons collected at reference locations (Ashwood et al. 1994). The fourth feather sample presented in Table C.24 (CRU) represents a mixture of feathers retrieved from the three nestlings. This sample was analyzed to provide additional information on the variability of chemical concentrations within the feathers.

A burrow on the Clinch River contained fragments of egg shells and fish vertebrae from regurgitant. Analysis of the egg shells indicated that minimal metal contamination was present (CRD, Table C.24). Another burrow on WOC contained an unhatched kingfisher egg (WOC, Table C.24). Metal concentrations in this egg were similar to that for the Clinch River egg, except for cesium-137. The presence of this radionuclide in the egg indicates that the parent kingfisher bioaccumulated cesium-137 from foraging within WOC or a nearby surface impoundment (cesium-137 is a typical contaminant of this stream and the impoundments).

In at least one kingfisher from the ORR, ^{137}Cs, Cd, Pb, Se, and Hg were each detected (Table C.25). Arsenic was analyzed for but was not detected. Feathers of adult kingfishers contained elevated mercury (Table C.25) relative to feathers from the nestlings (Table C.24). The greatest burdens of Hg, Se, Pb, and Cs-137 were observed in the bird from the WOC watershed (bird 3; Table C.25). In contrast, cadmium levels were higher in the birds from East Fork Poplar Creek (birds 1 and 2) than in the WOC bird (Table C.25).

7.1.3 Characterization Chemical Risks to Wildlife

Risk Characterization integrates the results of the exposure assessment (Sect. C.1.1) and effects assessment (Sect. C.1.2) to estimate risks (the likelihood of effects given the exposure) based on each line of evidence and then applies a weight of evidence inference logic to determine the best estimate of risk to each assessment endpoint. In an ideal risk assessment, there are three lines of evidence: literature-derived single chemical toxicity data (which indicate the toxic effects of the concentrations measured in site media); biological surveys of the affected system (these indicate the actual state of the receiving environment); and toxicity tests with ambient media (these indicate the toxic effects of the concentrations measured in site media). With the exception of the biosurvey data for mink and belted kingfisher (Sect 7.1.2.2), only one line of evidence was available to assess risk to wildlife in WAG 2— that is single chemical toxicity data.

7.1.3.1 Single chemical toxicity data

Exposure estimates generated by the exposure model (see Sect. 7.1.1) produced by both point estimates of parameter values and Monte Carlo simulation represent exposure at the individual level. The exposure estimates using point estimates of parameter values at each reach are used to identify COPECs and locations that contribute significantly to risk. In contrast, the WAG 2-wide exposure distributions generated by Monte Carlo simulation represent the likelihood that an individual within the WAG will experience a particular exposure.

Two types of single chemical toxicity data are available with which to evaluate wildlife contaminant exposure: NOAELs and LOAELs. NOAELs are used to screen exposure estimates generated from point estimates of exposure parameters; if the estimate is greater than the NOAEL, adverse effects are possible and additional evaluation is necessary. LOAELs are compared to the exposure distribution generated by the Monte Carlo simulation. If the LOAEL is lower than the 80th percentile of the exposure distribution, there is a >20% likelihood that individuals within the modeled location are experiencing contaminant exposures that are likely to produce adverse effects. By combining measured or literature-derived population density data with the likelihood or probability of exceeding the LOAEL, the magnitude of population-level impacts may be estimated.

Screening Point Estimates of Exposure. To determine if the contaminant exposures experienced by wildlife in each reach and throughout WAG 2 are potentially hazardous, the dietary contaminant exposure estimates (generated using point estimates of parameter values; see Tables C.1 and C.2, and C.12 through C.18) were compared to estimated NOAELs and LOAELs for these species (Tables C.21 and C.22). To quantify the magnitude of hazard, an HQ was calculated, where HQ = exposure/NOAEL or LOAEL. HQs greater than 1 indicate that individuals may be experiencing exposures that are in excess of NOAELs or LOAELs. While exceeding the NOAEL suggests that adverse effects are possible, exceeding the LOAEL suggests that adverse effects are likely. HQs for all endpoints are presented along with the point estimates of exposure in Tables C.1 and C.2, and C.12 through C.18

While exposure of shrews (Table C.12), mice (Table C.13), deer (Table C.14), fox (Table C.15), and turkey (Table C.17) exceeded both NOAELs and LOAELs for aluminum in all four reaches, all WAG 2 estimates for each endpoint were less than that at the background. Because exposures to aluminum were less than background and background exposure is assumed to be nonhazardous, aluminum was dropped as a contaminant of concern for wildlife.

Mercury was the dominant contaminant presenting risks to wildlife in WAG 2. Mercury exposure in WAG 2 exceeded NOAELs, LOAELs, and background for all endpoints within all reaches (except for deer, where LOAELs were exceeded only within the IHP area, and turkey, where LOAELs were not exceeded in the LMB area). The next most important contaminants presenting risks were PCBs and selenium, which presented risks to shrews (Table C.12), mice (Table C.13), and fox (Table C.15). In general, few other contaminants presented significant risks. With the exception of short-tailed shrews, only one to at most four contaminants were identified as presenting risks to any endpoint. In the case of shrews, seven contaminants contributed significant risks (Table C.12).

The spatial distribution of contamination and potential risks to wildlife in WAG 2 are illustrated in Figs. 7.1 through 7.9. These figures display the sum of the LOAEL-based HQs (e.g., sum of toxic units or ΣTUs) for those contaminants where at least one LOAEL-based HQ>1 was obtained. For all endpoint species, the greatest risks were identified in the mainstem of WOC. The highest ΣTUs are in the IHP area. ΣTUs decline with increasing distance downstream (IHP>MWC>LWC). Risks in the LMB were consistently lower than that in all three WOC reaches. The spatial patterns of contaminant risk were comparable for all endpoint species.

Screening Monte Carlo Simulation Estimates of Exposure. To incorporate the variation present in the parameters employed in the exposure model, Monte Carlo simulations were performed for the exposure estimates of each species to analytes where at least one LOAEL-based HQ>1 was observed. For all endpoints, simulations were performed only at the WAG 2-wide level. The mean, standard deviation, and 80th percentile of the simulated exposures are presented in Table C.19.

By superimposing NOAEL and LOAEL values on the exposure distributions generated from the Monte Carlo simulation, the likelihood of an individual experiencing potentially hazardous exposures can be estimated, and the magnitude of risk may be determined. Interpretation of the comparison of exposure distributions to NOAELs and LOAELs is described in Table 7.2.

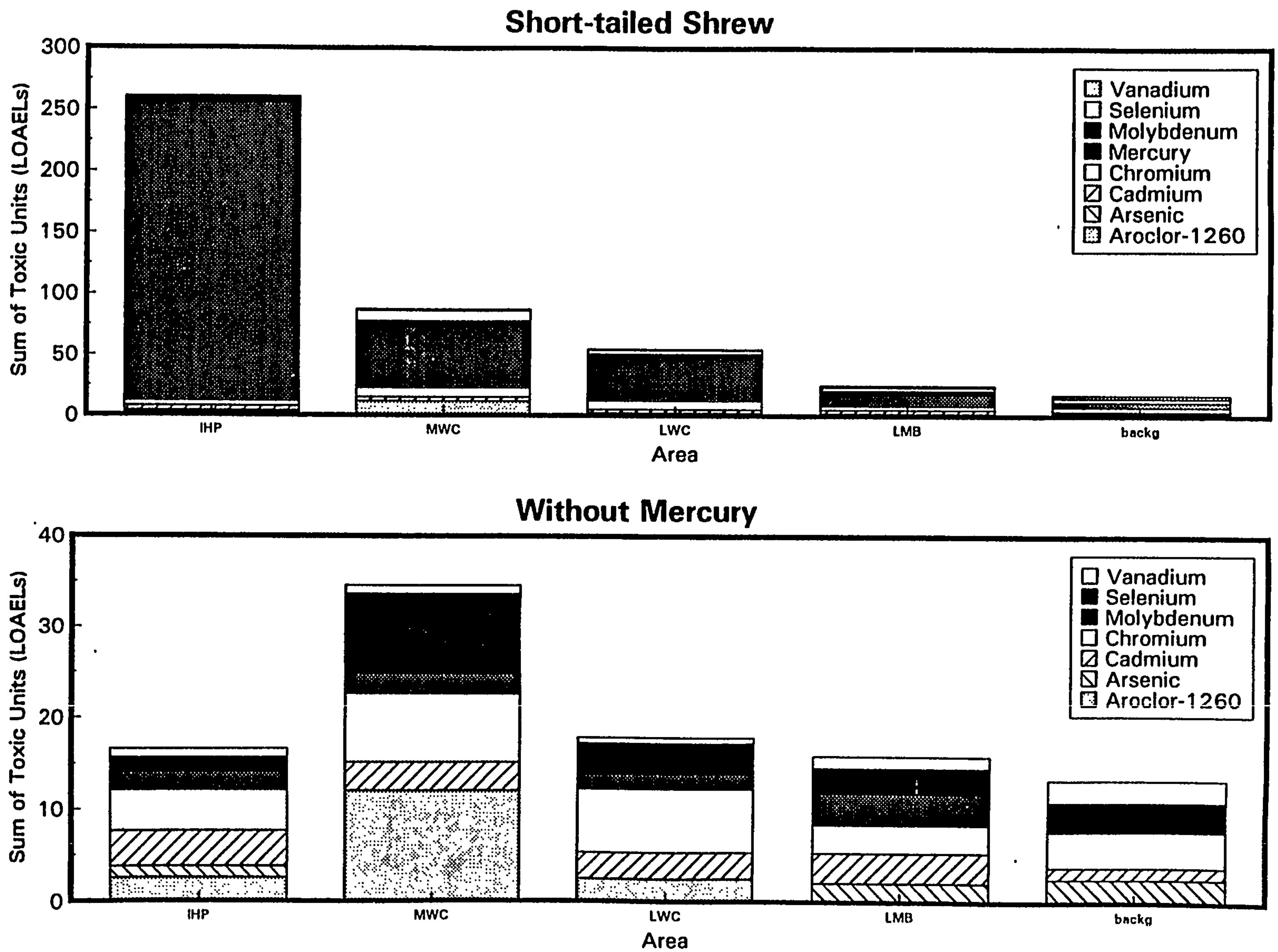

Fig. 7.1. Summary of lowest observed adverse effects level-based toxic units for short-tailed shrews in WAG 2.

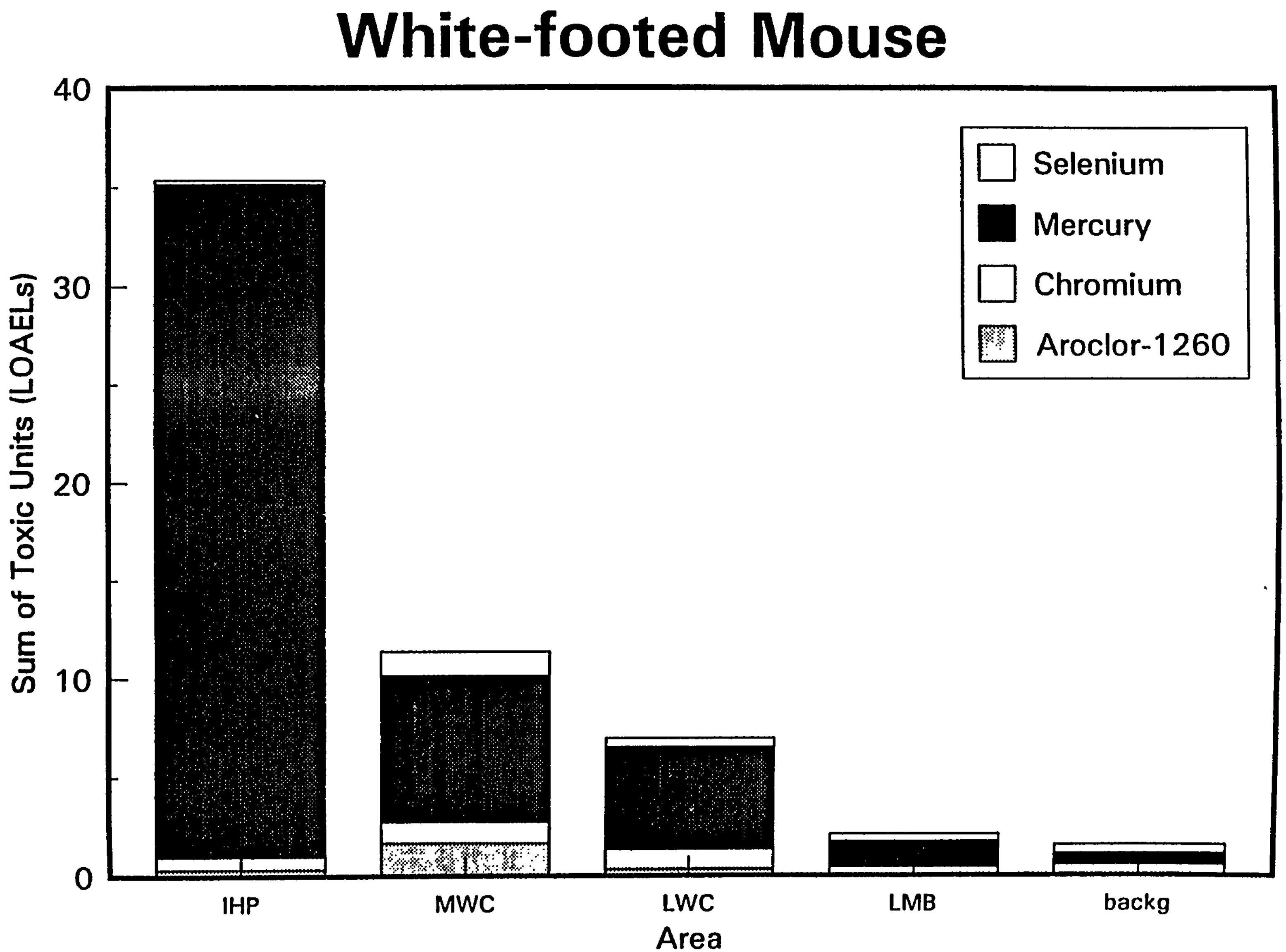

Fig. 7.2. Summary of lowest observed adverse effects level-based toxic units for white-footed mice in WAG 2.

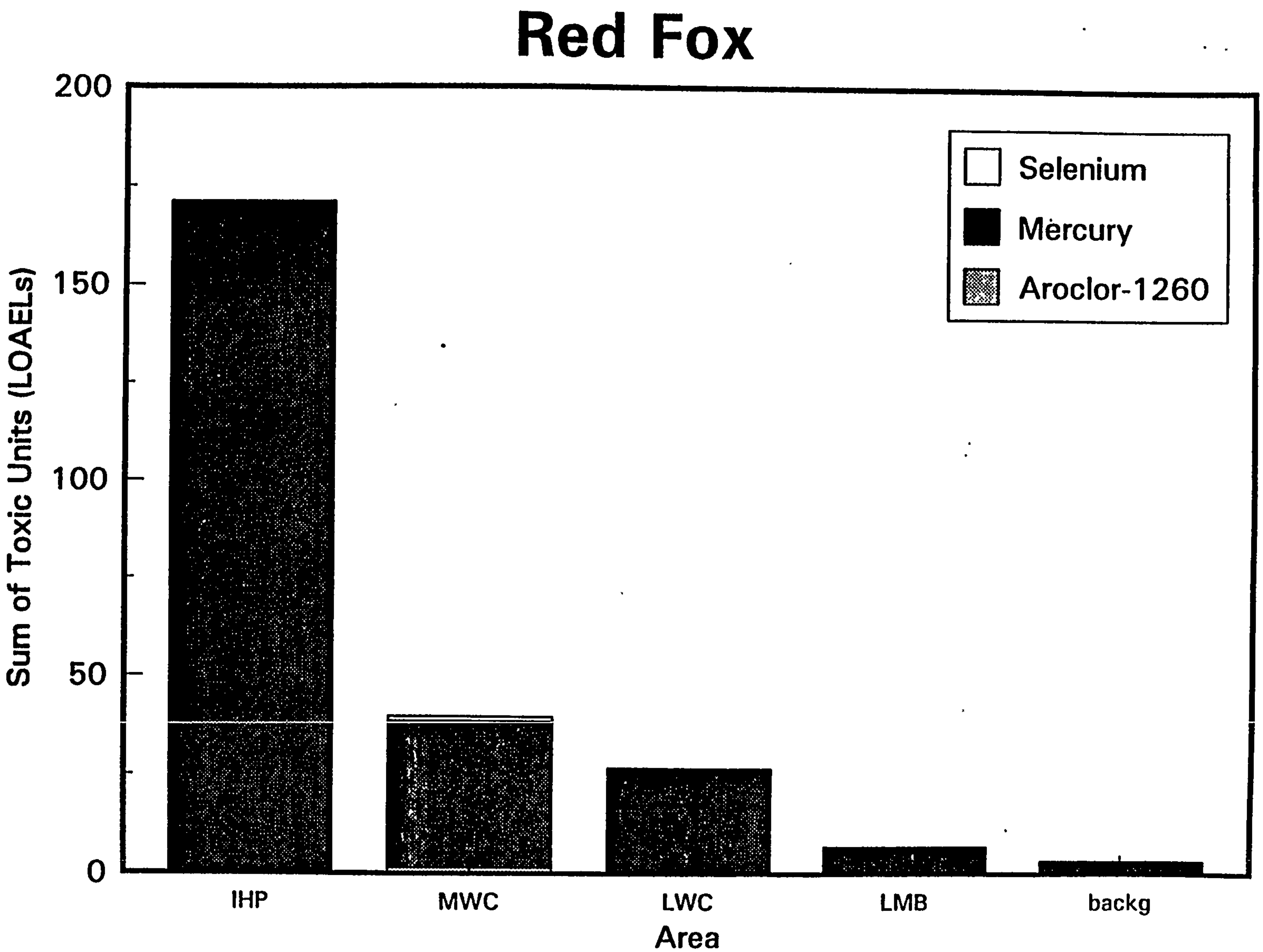

Fig. 7.3. Summary of lowest observed adverse effects level-based toxic units for red fox in WAG 2.

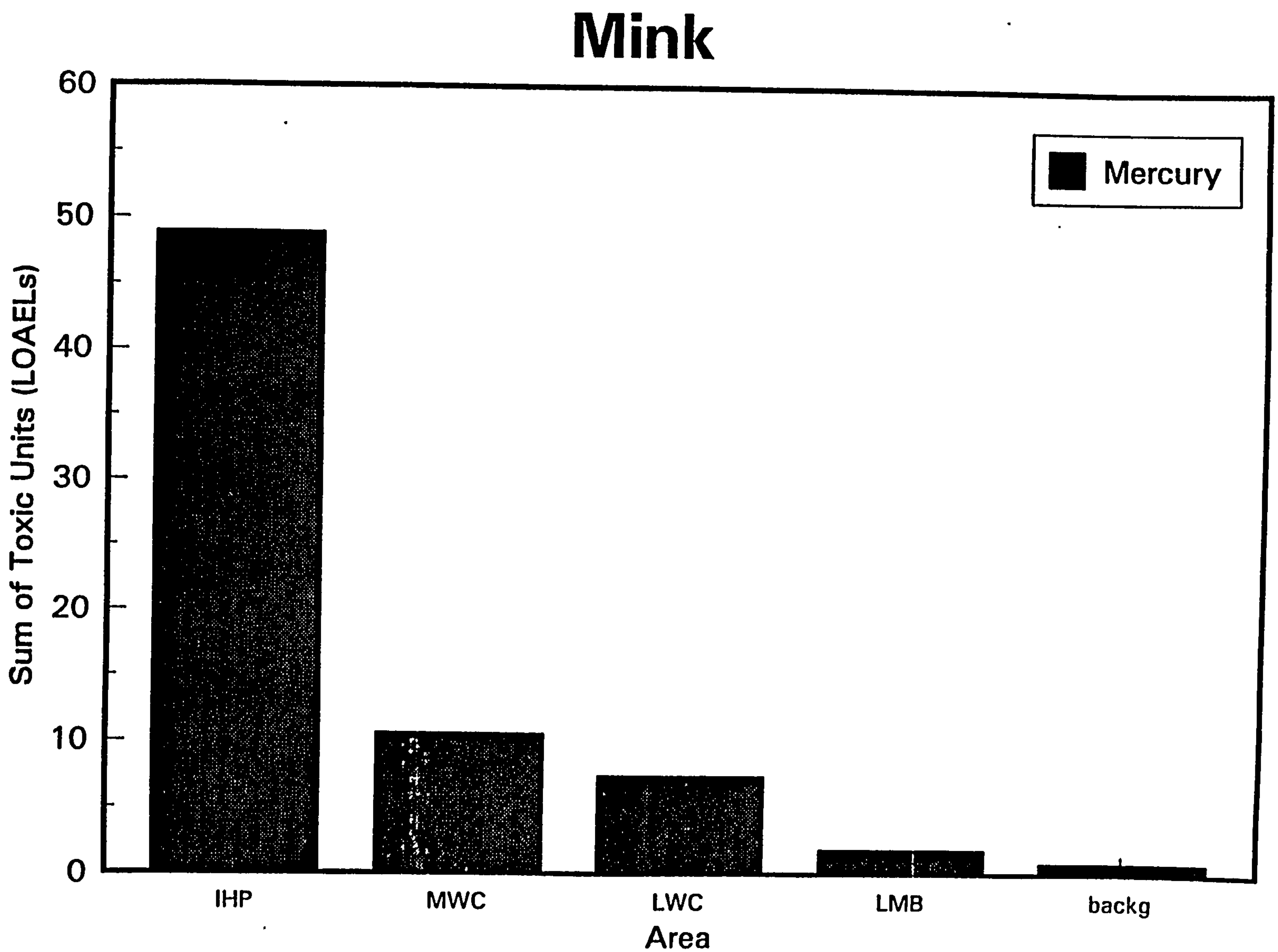

Fig. 7.4 Summary of lowest observed adverse effects level-based toxic units for mink in WAG 2.

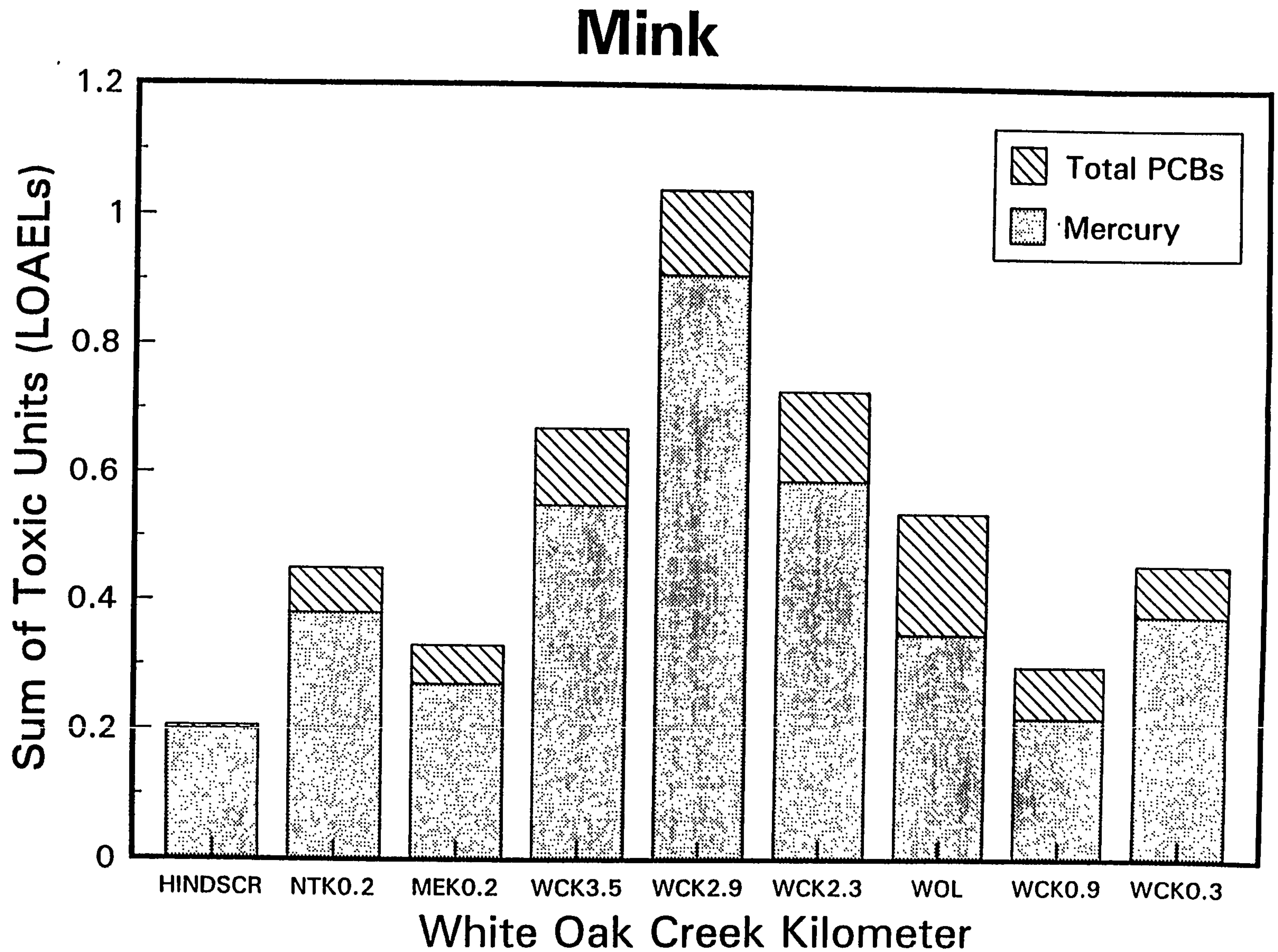

Fig. 7.5. Summary of lowest observed adverse effects level-based toxic units for mink in WAG 2 (exposure through ingestion of fish only).

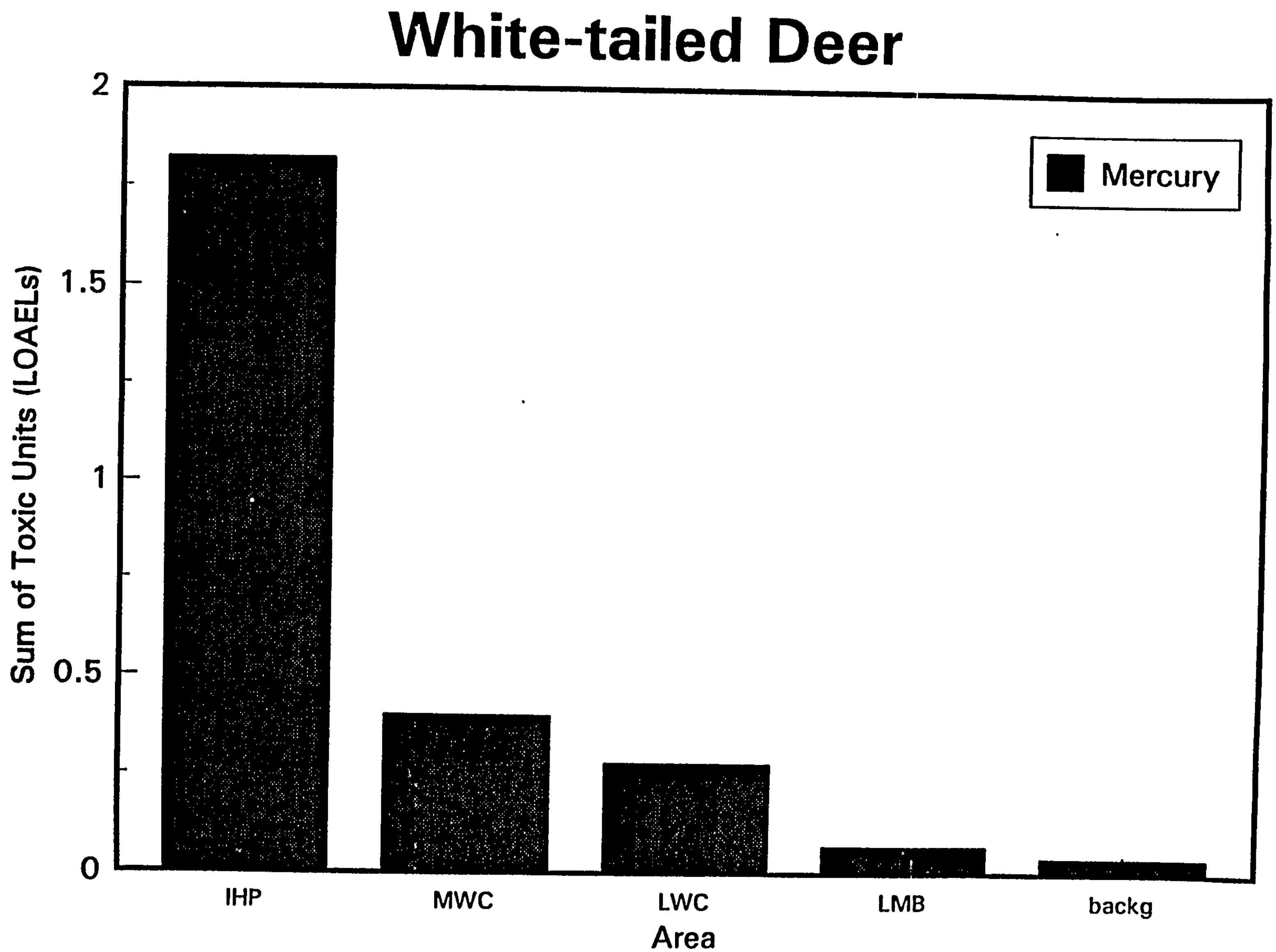

Fig. 7.6. Summary of lowest observed adverse effects level-based toxic units for white-tailed deer in WAG 2.

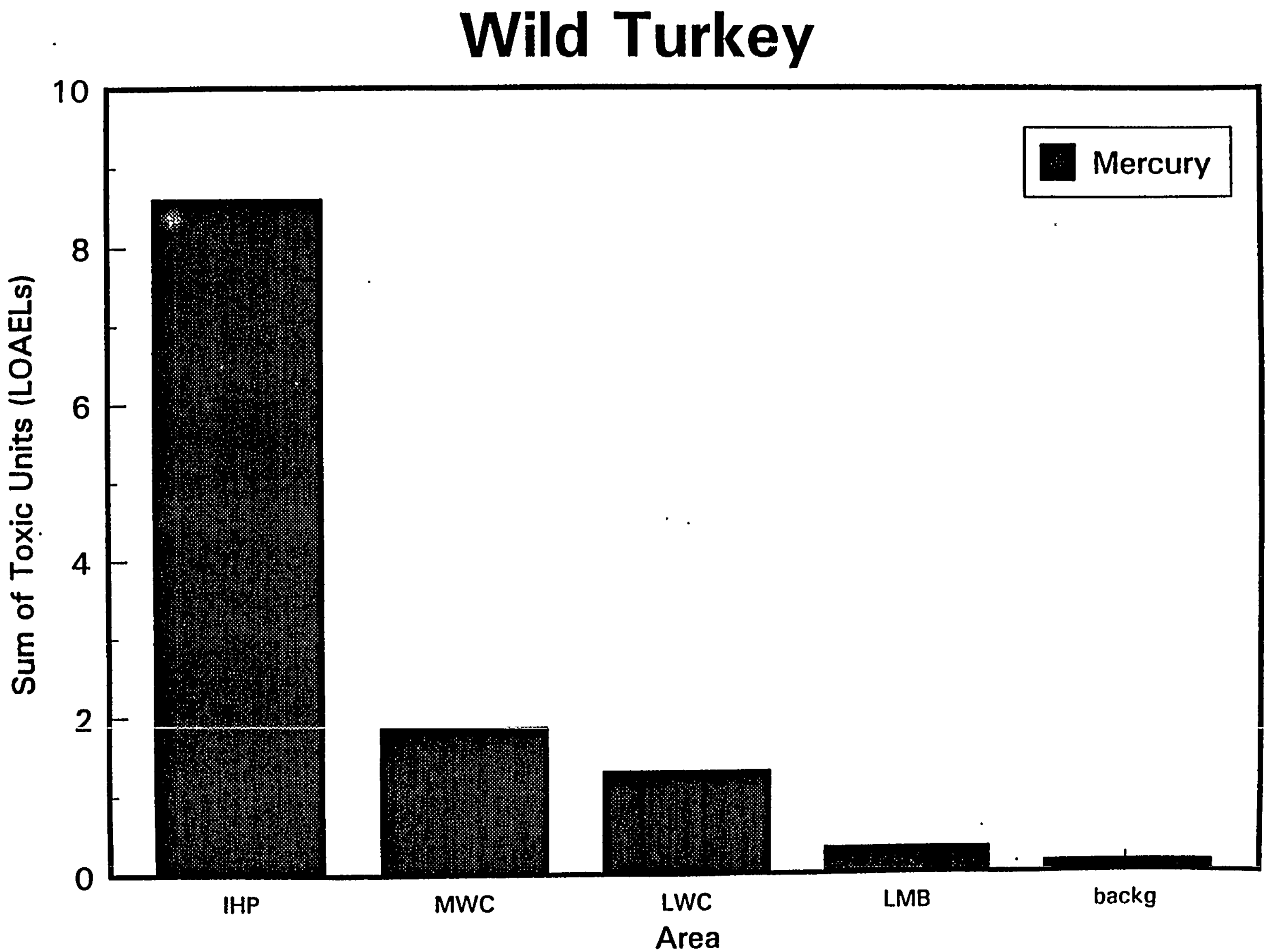

Fig. 7.7. Summary of lowest observed adverse effects level-based toxic units for wild turkey in WAG 2.

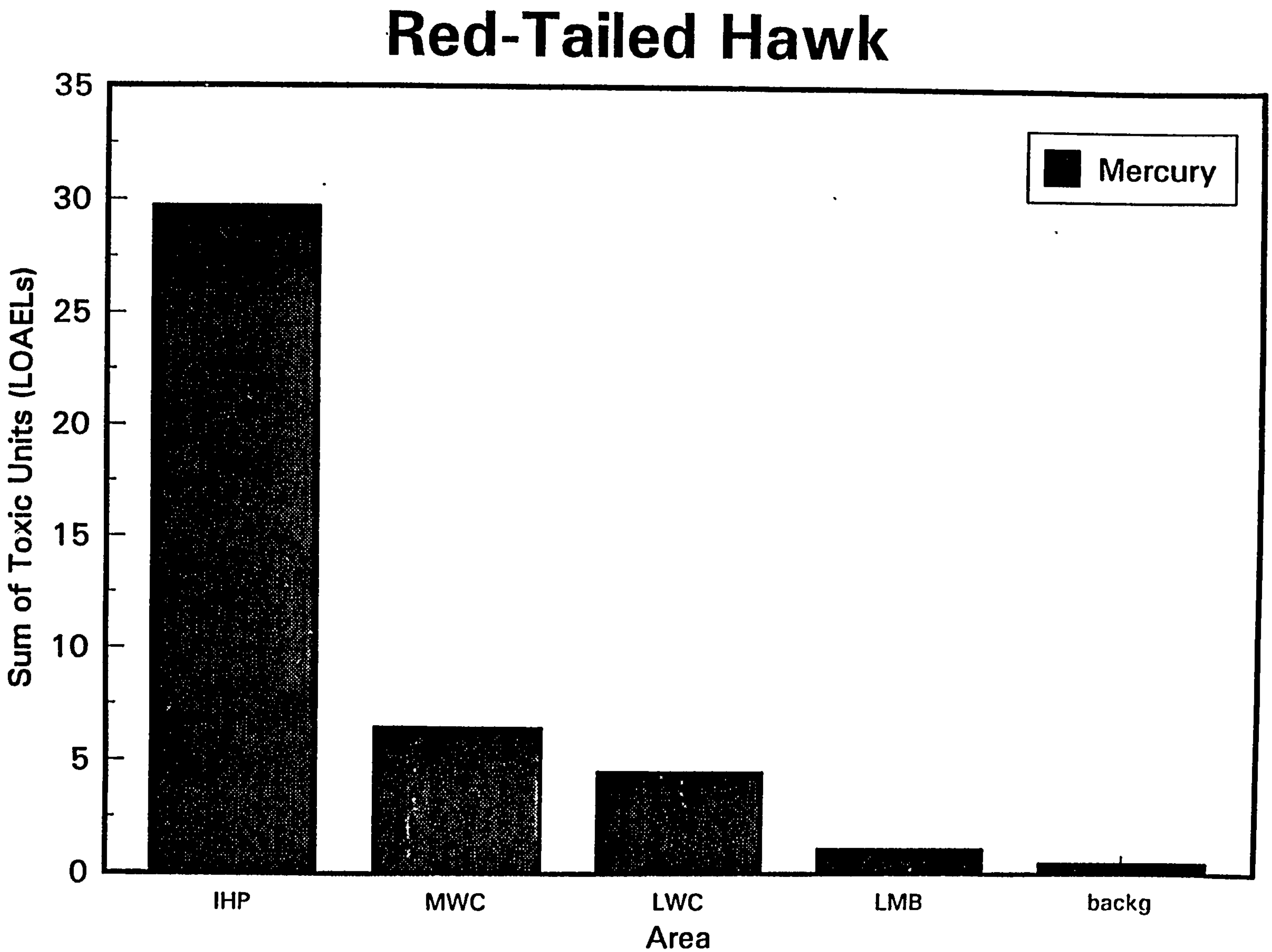

Fig. 7.8. Summary of lowest observed adverse effects level-based toxic units for red-tailed hawks in WAG 2.

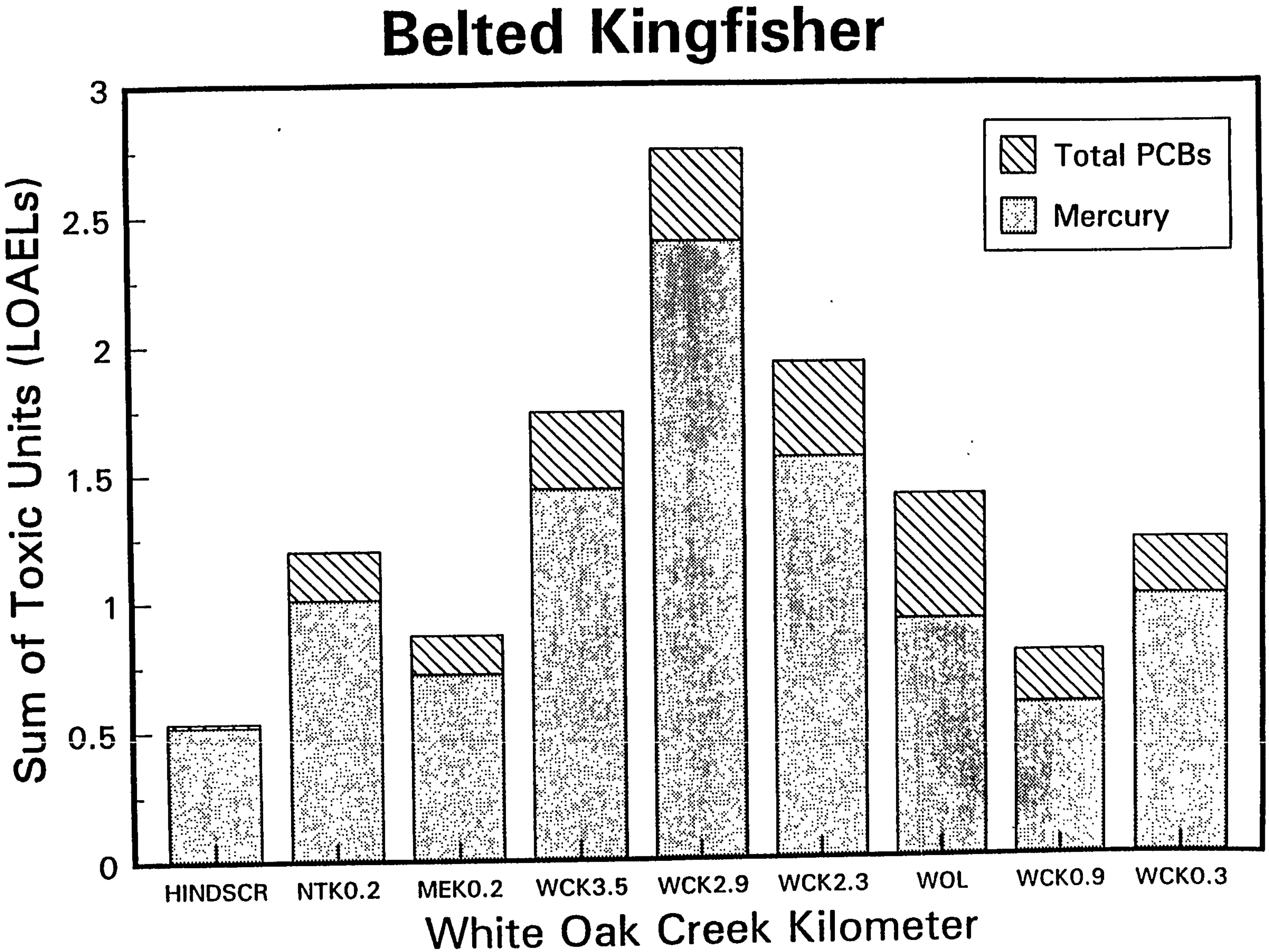

Fig. 7.9. Summary of lowest observed adverse effects level-based toxic units for belted kingfisher in WAG 2 (exposure through ingestion of fish only).

Table 7.2. Comparison of exposure distributions

Comparison	Meaning	Risk-based Interpretation
NOAEL>80th percentile of exposure distribution	less than 20% of exposures> NOAEL	Individual- and population-level adverse effects are highly unlikely
NOAEL<80th percentile<LOAEL	More than 20% of exposures > NOAEL, but less than 20% of exposures> LOAEL	Individuals experiencing exposures at the high end of the distribution may experience adverse effects, but those effects are unlikely to significantly contribute to effects on the Oak Ridge Reservation (ORR) population.
LOAEL<80th percentile of exposure distribution	More than 20% of exposures> LOAEL	Effects on some individuals are likely and they may contribute significantly to effects on the ORR population.

Note: NOAEL=no observed adverse effects level; LOAEL=lowest observed adverse effects level

To evaluate the likelihood and magnitude of population-level effects on wildlife, literature-derived population density data (expressed as number of individuals/ ha or km of stream) were combined with hectares of suitable habitat (within WAG 2) to estimate the number of individuals of each endpoint species expected to be present in the watershed. For the terrestrial species (shrew, mice, deer, fox, turkey and hawk), habitat preferences follow those reported in the *Preliminary Reservation-wide Ecological Risk* Assessment (Sample et al. 1995). These habitat preferences were compared to the habitat types identified in WAG 2 in Washington-Allen et al. (1995). Because streams are the preferred habitats for mink and kingfisher, the length of White Oak Creek and its tributaries was assumed to represent suitable habitat. The estimated abundance of wildlife endpoint species is reported in Table C.26.

The number of individuals within WAG 2 likely to experience exposures >LOAELs can be estimated using cumulative binomial probability functions (Dowdy and Wearden 1983). Binomial probability functions are estimated using the following equation:

$$b(y;n;p) = \binom{n}{y} p^{y} (1-p)^{n-y} , \qquad (5)$$

where

y = the number (or percent) of individuals experiencing exposures >LOAEL,
n = total number (or percent) of individuals within the watershed,
p = probability of experiencing an exposure in excess of the LOAEL,
b (y; n; p) = probability of y individuals out of a total of n, experiencing an exposure >LOAEL, given the probability of exceeding the LOAEL=p.

Solving Eq. 8 for y=0 to y=n generates a cumulative binomial probability distribution, which can be used to estimate the number of individuals within WAG 2 who are likely to experience adverse effects.

Binomial probability distributions were generated only for contaminant-endpoint combinations where the percent of the exposure distribution exceeding the LOAEL was 20% to 80% (these values are reported in Table C.19). If the percent of the exposure distribution exceeding the LOAEL was <20%, it was assumed that no individuals within the area of interest were experiencing adverse effects. Conversely, if the percent of the exposure distribution exceeding the LOAEL was >80%, it was assumed that all individuals within the area of interest were experiencing adverse effects. Exposure estimates for six contaminant-endpoint combinations met the 20% to 80% exceedance criterion: PCB, cadmium, and selenium exposure to shrews and mercury exposure to fox, kingfisher, and turkey. Figures 7.10 through 7.15 graphically display the cumulative binomial probability distributions for each contaminant-endpoint combination. The total numbers of individuals for each endpoint species estimated to be experiencing adverse effects within WAG 2 are summarized in Table C.27.

Based on the Monte Carlo analysis, comparison to background exposure estimates, and binomial distribution analysis, the following conclusions may be made:

- Because <20% of the WAG 2 populations are estimated to be experiencing exposures >LOAEL, the following contaminants do not present significant risks:

Endpoint	Analytes
Short-tailed shrew	As, Ba, Cu, Mn, Mo, Ni, V, and Zn
White-footed mouse	Aroclor-1260, Cr, and Se
Red fox	Aroclor-1260 and Se
Mink	Hg (exposure through fish consumption only)
White-tailed deer	Hg

- Because >20% of the WAG 2 shrew population is estimated to be experiencing exposures >LOAEL, Aroclor-1260, cadmium, chromium, mercury, and selenium present a significant risk to shrews in WAG 2;

- Because >20% of the populations are estimated to be experiencing exposures >LOAEL, mercury presents a significant risk to white-footed mice, red fox, mink, belted kingfisher, red-tailed hawk, and wild turkey in WAG 2.

Screening Point Estimates of Exposure: Surface Water. To evaluate the potential risk that contaminants in surface water present to wildlife in reaches and seeps where total exposure was not estimated (due to lack of surface soil data), the 95% UCLs for concentrations in unfiltered water were compared to NOAEL and LOAEL water benchmarks for all species. HQs (water concentration/benchmark value) were calculated for all species. Comparisons of reach concentrations to NOAEL benchmarks are presented in Table C.28. NOAEL-based screening benchmarks were not exceeded within any reach, indicating that exposure to surface water within these WOC watershed reaches does not contribute to risk. These screening results indicated that exposure to seep water within the WOC watershed does not contribute to risk.

Comparisons of seep concentrations to NOAEL and LOAEL benchmarks are presented in Tables C.29 and C.30. Only one contaminant (aluminum) at one seep (RS-3A) exceeded NOAEL benchmarks (Table C.29); the LOAEL benchmarks, however were not exceeded (Table C.30). At all other seeps for all other analytes, NOAEL-based benchmarks were not exceeded.

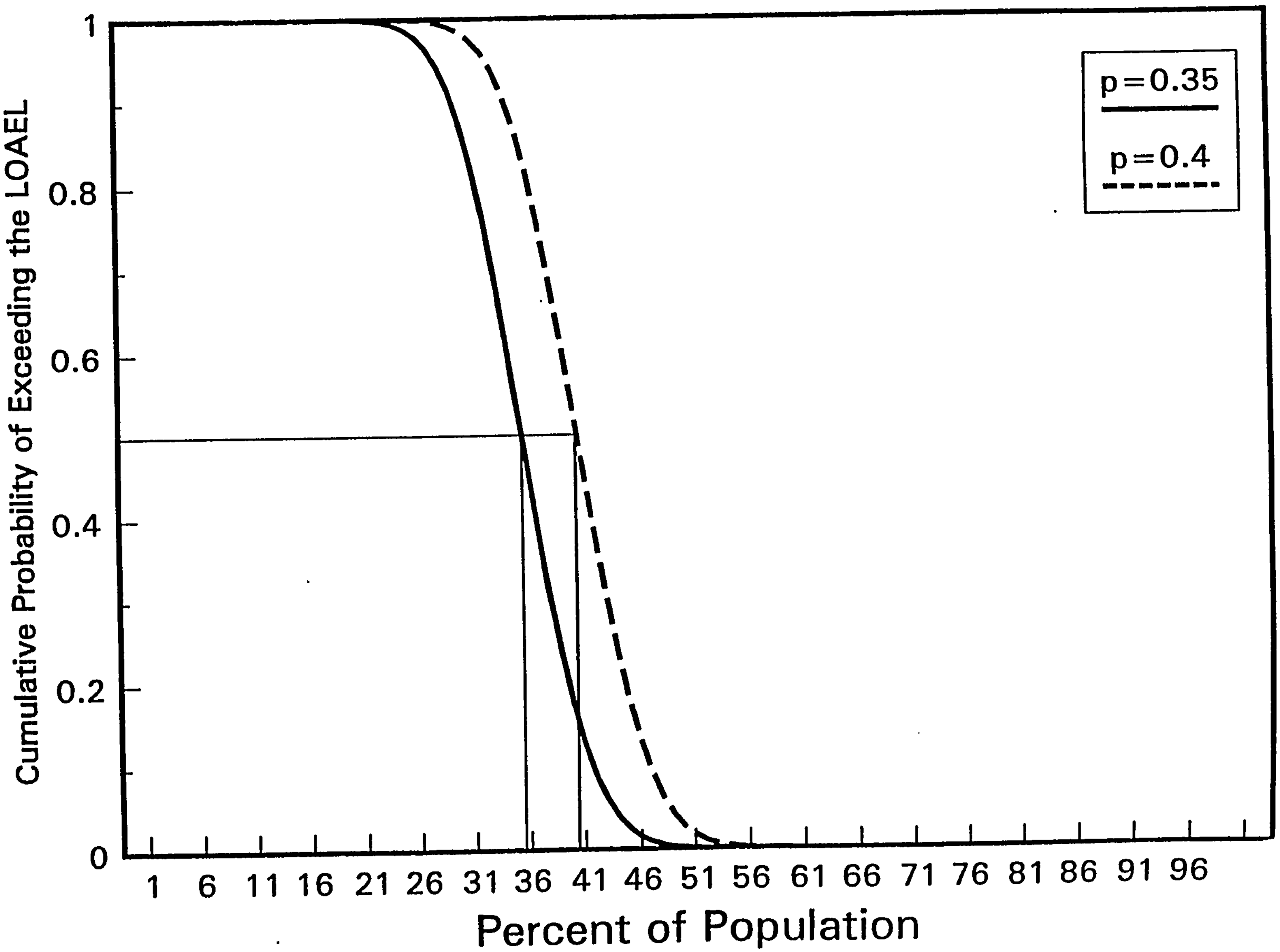

Fig. 7.10. Cumulative binomial probability of short-tailed shrews experiencing exposure to Aroclor-1260 in WAG 2 in excess of the lowest observed adverse effects level.

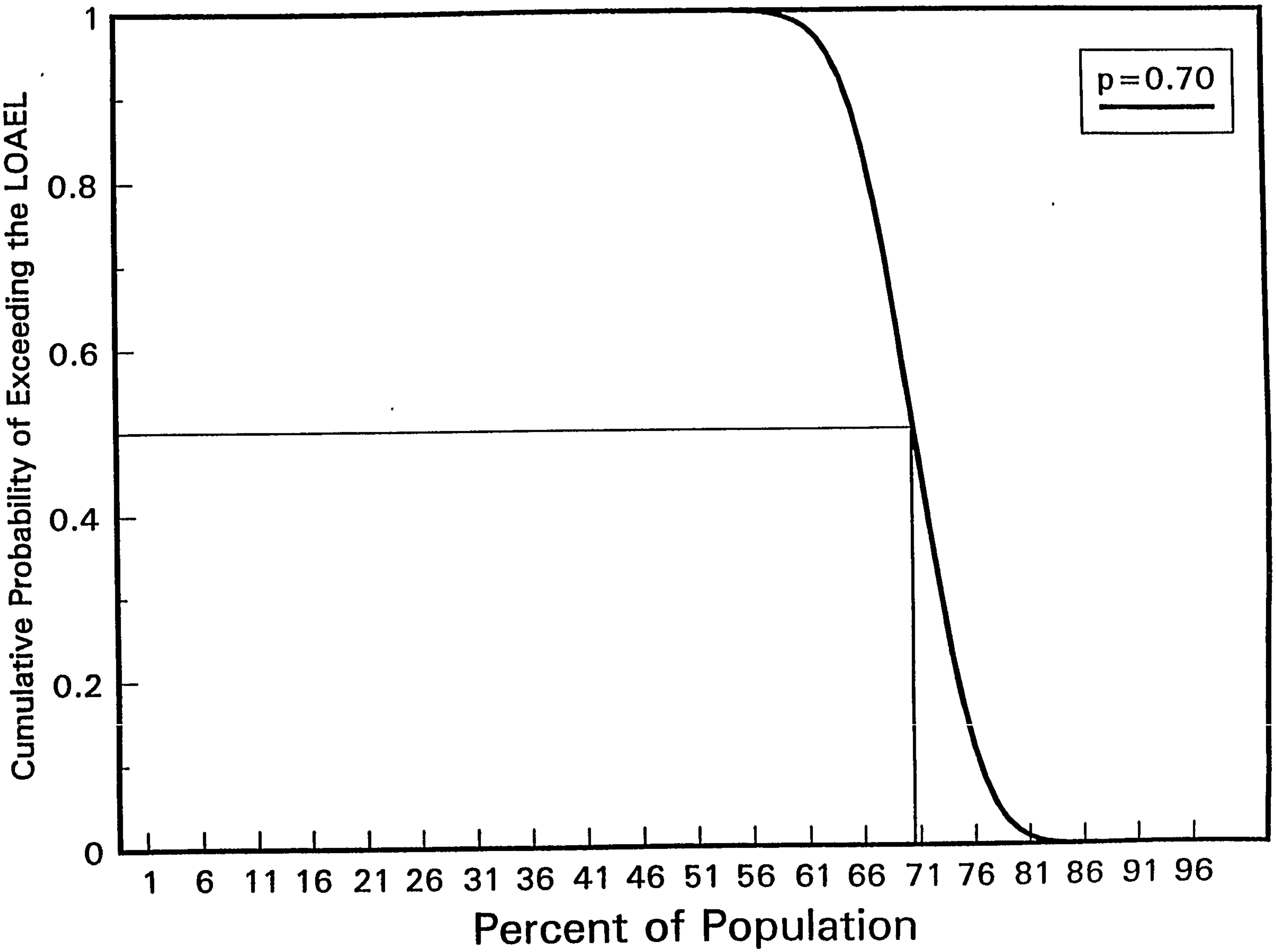

Fig. 7.11. Cumulative binomial probability of short-tailed shrews experiencing exposure to cadmium in WAG 2 in excess of the lowest observed adverse. effects level.

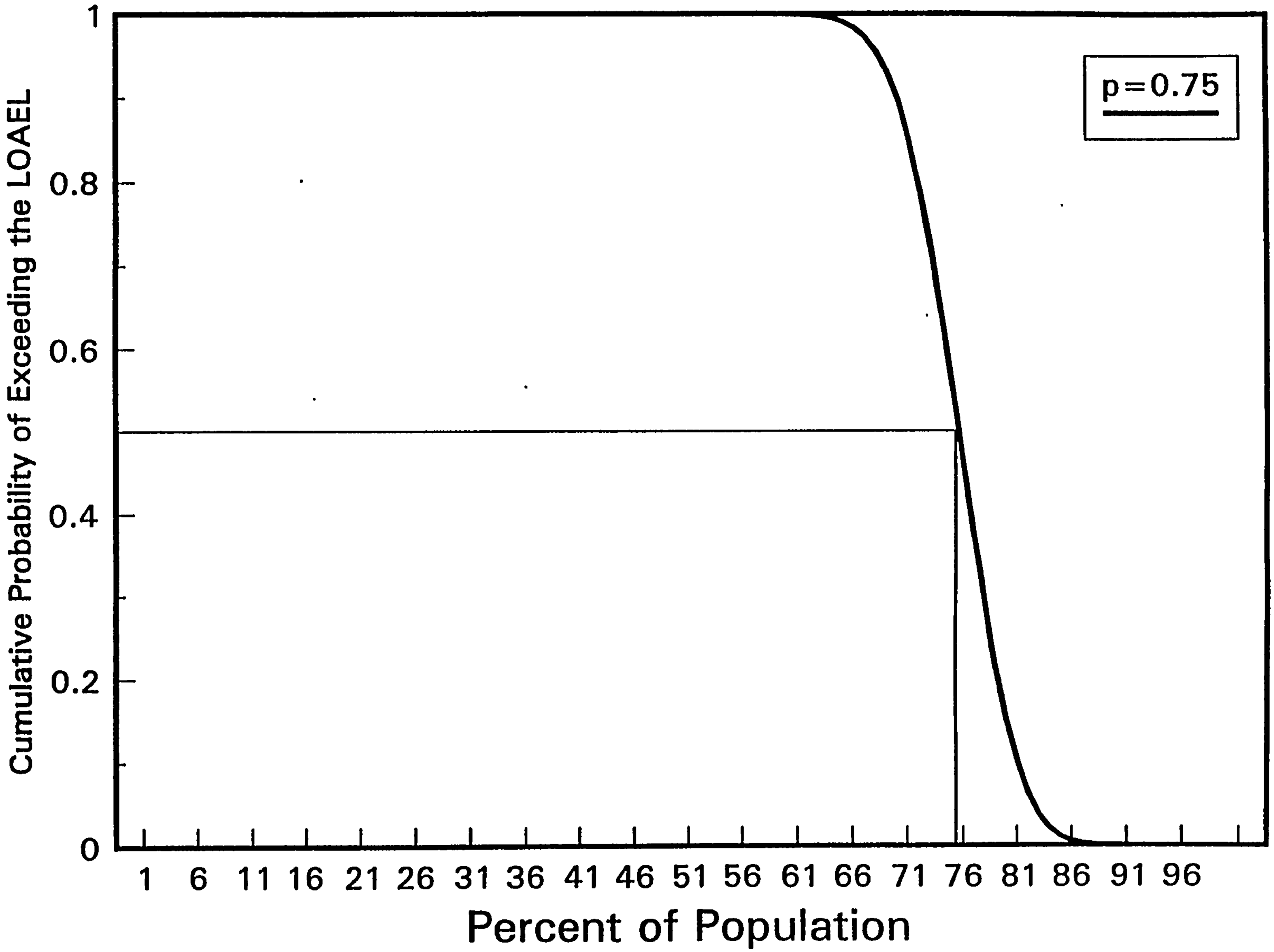

Fig. 7.12. Cumulative binomial probability of short-tailed shrews experiencing exposure to selenium in WAG 2 in excess of the lowest observed adverse effects level.

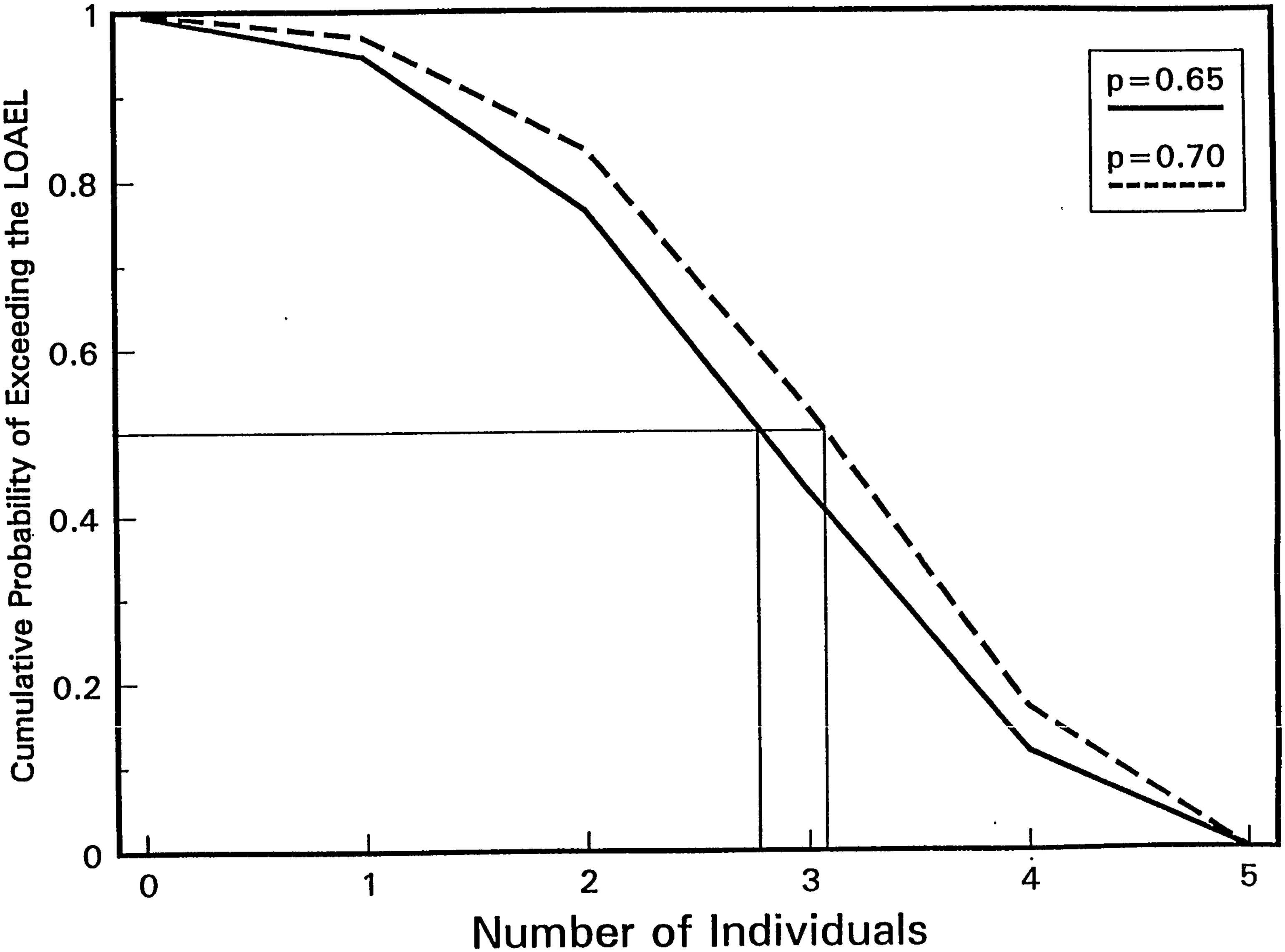

Fig. 7.13. Cumulative binomial probability of red fox experiencing exposure to mercury in WAG 2 in excess of the lowest observed adverse effects level.

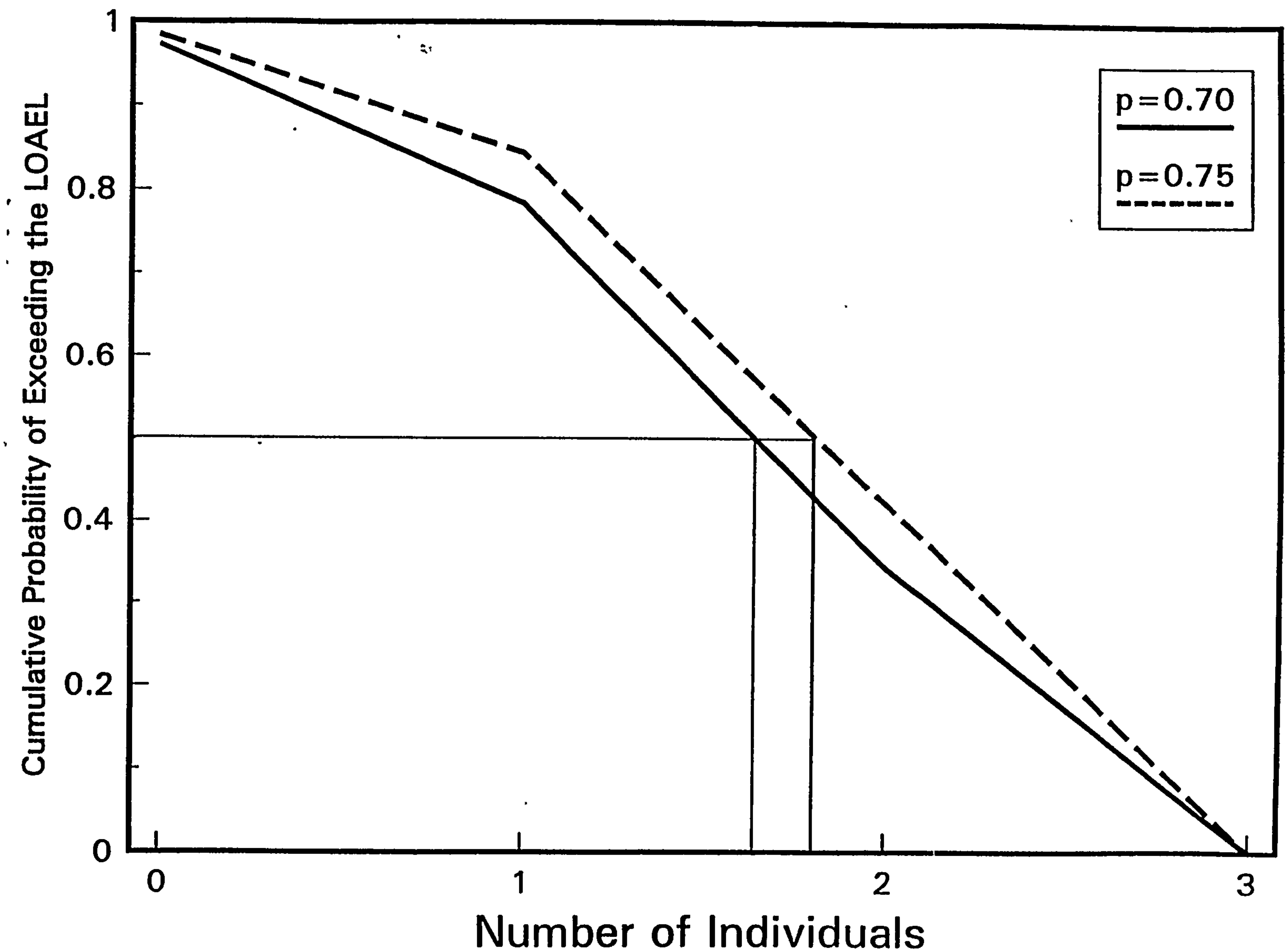

Fig. 7.14. Cumulative binomial probability of belted kingfisher experiencing exposure to mercury in WAG 2 in excess of the lowest observed adverse effects level.

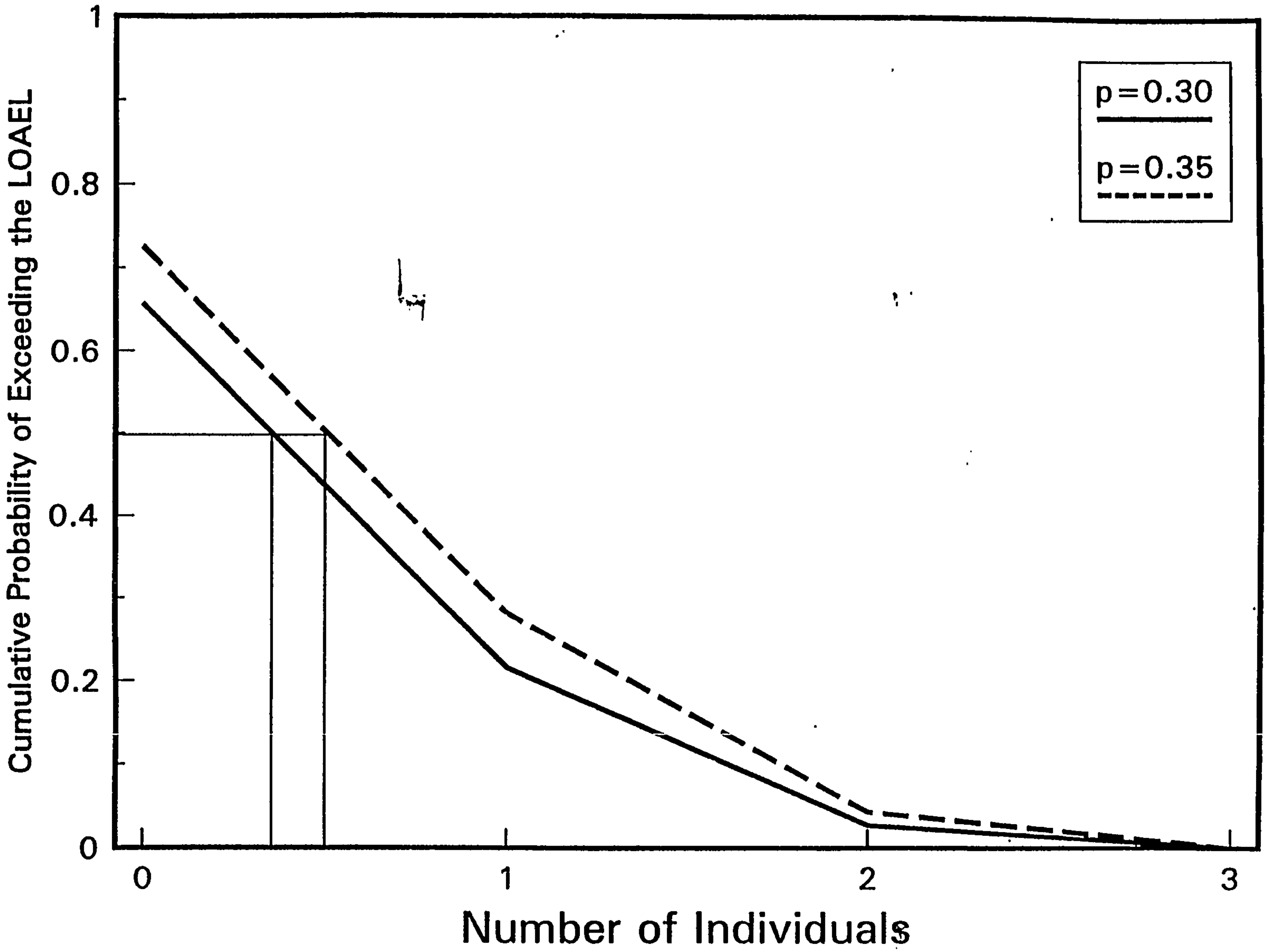

Fig. 7.15. Cumulative binomial probability of wild turkey experiencing exposure to mercury in WAG 2 in excess of the lowest observed adverse effects level.

7.1.3.2 Effects of retained contaminants

Cadmium. The mammalian NOAELs and LOAELs for cadmium were derived from a study of mice fed cadmium for multiple generations. (Schroeder and Mitchner 1971). Consumption of 2.518 mg/kg-d cadmium impaired reproduction and caused the exposed line to die out before the third generation. Only one dose level was tested. The study was considered to represent a chronic exposure; therefore, a subchronic-chronic correction factor was not employed. A NOAEL was estimated using a LOAEL-LOAEL correction factor of 0.1. The 0.25 mg/kg-d exposure was considered to be a chronic NOAEL; the 2.5 mg/kg-d exposure was considered to be a chronic LOAEL. Based on the results of Schroeder and Mitchner (1971), shrews experiencing exposure ≥ LOAEL are likely to display impaired reproduction.

Chromium. The mammalian NOAEL for chromium was based on a study of rats fed Cr^{+6} in water for one year (Mackenzie et al. 1958). No adverse effects were observed at the highest dose of 3.28 mg/kg-d. The study was considered to represent a chronic exposure; therefore, a subchronic-chronic correction factor was not employed. The 3.28 mg/kg-d exposure was considered to be a chronic NOAEL. The mammalian LOAEL for chromium was based on a study of rats fed Cr^{+6} in water for 3 months (Steven et al. 1976). Mortality significantly increased among rats consuming 131.4 mg/kg-d. The study was considered to represent a subchronic exposure, therefore a subchronic-chronic correction factor was employed. The 13.14 mg/kg-d exposure was considered to be a chronic LOAEL. Based on the results of Steven et al. (1976), shrews experiencing exposure ≥ LOAEL are likely to display increased mortality.

Mercury. For the purposes of this assessment, it is assumed that 100% of the mercury to which wildlife is exposed consists of methyl mercury.

Both the avian NOAEL and LOAEL are based upon a study of mallard ducks fed methyl mercury for three generations (Heinz 1979). The study was considered to represent a chronic exposure, and a subchronic-chronic correction factor was not employed. The only dose level administered, 0.064 mg/kg-d, caused hens to lay fewer eggs, lay more eggs outside of the nest box, and produce fewer ducklings. This dose level was considered to be an LOAEL. Because an experimental NOAEL was not established, the NOAEL was estimated using an LOAEL-NOAEL correction factor of 0.1. Based on the results of Heinz (1979), kingfisher, red-tailed hawks, and turkey experiencing exposure ≥ LOAEL are likely to display impaired reproduction.

The fox and mink NOAELs and LOAELs for mercury were derived from a study of mink fed methyl mercury for 93 d (Wobeser et al. 1976). While consumption of 0.247 mg/kg-d methyl mercury resulted in significant mortality, weight loss, and behavioral impairment, no effects were observed at the 0.15 mg/kg-d exposure level. The 0.15 mg/kg-d exposure was considered to be an NOAEL and the 0.247 mg/kg-d exposure was considered to be an LOAEL. Because the study was subchronic in duration (<1 yr), a subchronic-chronic correction factor was applied (NOAEL=0.015, LOAEL=0.025). Based on the results of Wobeser et al. (1976), shrews, mice, fox, and mink experiencing exposure ≥ LOAEL are likely to display increased mortality, weight loss, and behavioral impairment.

PCBs. The mammalian NOAEL and LOAEL for PCBs were derived from a study of mink fed Aroclor 1254 for 4.5 mo. (Aulerich and Ringer 1977). While consumption of 0.69 mg/kg-d of Aroclor 1254 reduced kit survivorship, no effects were observed at the 0.14 mg/kg-d exposure level. The 0.14 mg/kg-d exposure was considered to be a chronic NOAEL; the 0.69 mg/kg-d exposure was considered to be a chronic LOAEL. Based on the results of Aulerich and Ringer (1977), shrews experiencing exposure ≥ LOAEL are likely to display reduced survivorship of young.

Selenium. The mammalian NOAELs and LOAELs for selenium were derived from a study of mice fed selenium for three generations. (Schroeder and Mitchner 1971). Consumption of 0.76 mg/kg-d selenium resulted in reduced reproductive success, and increased incidence of runts and failure to breed. Only one dose level was tested. The study was considered to represent a chronic exposure; therefore, a subchronic-chronic correction factor was not employed. An NOAEL was estimated using an LOAEL-LOAEL correction factor of 0.1. The 0.076 mg/kg-d exposure was considered to be a chronic NOAEL; the 0.76 mg/kg-d exposure was considered to be a chronic LOAEL. Based on the results of Schroeder and Mitchner (1971), shrews experiencing exposure ≥ LOAEL are likely to display impaired reproduction.

7.1.3.3 Biological surveys

Mink Survey. Results of the mink survey (see Sect. 7.1.2.2.1) indicate that mink are present on the ORR and within WAG 2, have large home ranges, and do not avoid the industrial facilities on the ORR. The methods employed in the study do not allow numbers or density of mink to be determined. Concentrations of metals in hair of the single mink collected from WAG 2 were comparable to that from mink collected offsite.

Kingfisher Survey. Results of the kingfisher survey (see Sect. 7.1.2.2.2) indicate that contaminants are being accumulated by both juveniles and adult birds. While contaminants in eggshells and nestling feathers indicate exposure, there is insufficient information to evaluate the toxicological significance of this contamination.

The toxicological significance of the tissue concentrations in adult kingfisher was evaluated by comparison of burdens and effects levels reported in other bird species. This comparison suggests that it is unlikely that cadmium or lead in the kingfisher from WOC contribute significantly to risk. Leach et al. (1979) observed a 50% reduction in egg production among chickens consuming a diet containing 48 mg/kg cadmium. Cadmium concentrations in the livers and kidneys of these birds were 100 mg/kg and 40 mg/kg. Cadmium concentrations in healthy birds from unpolluted areas ranged from 0.1 to 32 mg/kg in liver and 0.3 to 137 mg/kg in kidney (Furness 1996). In comparison, cadmium concentrations in the kidney (1.53 mg/kg) and liver (0.9 mg/kg) of the kingfisher collected from the WOC watershed were significantly less than concentrations associated with reproductive impairment and at the low end of the ranges observed among healthy birds from unpolluted areas. Lead concentrations in the kidney (0.42 mg/kg) and liver (0.4 mg/kg) of the WOC kingfisher were approximately one order of magnitude lower than the minimal level at which overt toxicity is observed in birds (3 to 6 mg/kg)(Frenson 1996), suggesting that lead accumulation is unlikely to be contributing to risks to kingfishers.

In contrast to cadmium and lead, selenium and mercury burdens may present a hazard to WOC kingfishers. The concentration of selenium in the liver of Bird 3 (from WOC; 7.5 mg/kg) is less than the 10 mg/kg toxicity threshold recommend by Heinz (1996), but greater than the 3 mg/kg reproductive impairment threshold, suggesting the potential for reproductive impairment. Mercury concentrations of 49 to 125 mg/kg in kidney and 4.6 to 91 mg/kg in liver have been reported for free-living birds found dead or dying (Thompson 1996). Nephrotoxicity and kidney lesions occur in birds at mercury concentrations in kidney of 5 to 13 mg/kg (Nicholson and Osborn 1983). While mercury concentrations in the kidney (26.8 mg/kg) and liver (17.6 mg/kg) of the WOC kingfisher were generally lower than concentrations associated with mortality, the kidney concentration exceeds nephrotoxic levels, suggesting that mercury accumulation may be causing kidney damage to White Oak Creek kingfishers.

7.1.3.4 Weight of evidence

Short-tailed Shrews. One line of evidence, literature toxicity data, was available to evaluate potential risk to short-tailed shrews in WAG 2. Point estimates of exposure indicated that 18 contaminants exceeded NOAELs with 16 also exceeding LOAELs (Table C.12). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that Aroclor-1260, cadmium, chromium, mercury, and selenium present significant risks to the shrew population in WAG 2.

White-footed Mice. One line of evidence, literature toxicity data, was available to evaluate potential risk to white-footed mice in WAG 2. Point estimates of exposure indicated that ten contaminants exceeded NOAELs with five also exceeding LOAELs (Table C.13). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that only mercury presents a significant risk to the mouse population in WAG 2.

White-tailed Deer. One line of evidence, literature toxicity data, was available to evaluate potential risk to white-tailed deer in WAG 2. Point estimates of exposure indicated that seven contaminants exceeded NOAELs with two also exceeding LOAELs (Table C.14). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that no contaminant presents a significant risk to the deer population in WAG 2.

Red Fox. One line of evidence, literature toxicity data, was available to evaluate potential risk to red fox in WAG 2. Point estimates of exposure indicated that 12 contaminants exceeded NOAELs with 3 also exceeding LOAELs (Table C.15). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that only mercury presents a significant risk to the fox population in WAG 2.

Red-tailed Hawk. One line of evidence, literature toxicity data, was available to evaluate potential risk to red-tailed hawk in WAG 2. Point estimates of exposure indicated that three contaminants exceeded NOAELs with one also exceeding LOAELs (Table C.16). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that only mercury presents a significant risk to the hawk population in WAG 2.

Wild Turkey. One line of evidence, literature toxicity data, was available to evaluate potential risk to wild turkey in WAG 2. Point estimates of exposure indicated that three contaminants exceeded NOAELs with two also exceeding LOAELs (Table C.17). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that only mercury presents a significant risk to the turkey population in WAG 2.

Mink. Two lines of evidence, biological survey data and literature toxicity data, were available to evaluate potential risks to mink within WAG 2. The biological survey data indicate that mink are present within WAG 2; but due to the sampling methods employed, estimates of the abundance of the mink population cannot be made from these data. Residue analysis indicates that mink in WAG 2 have contaminant concentrations in hair similar to that in mink from offsite.

Point estimates of exposure indicated that four contaminants exceeded NOAELs with two also exceeding LOAELs (Tables C.1 and C.18). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that only mercury presents a significant risk to the mink population in WAG 2.

Belted Kingfisher. Two lines of evidence, biological survey data and literature toxicity data, were available to evaluate potential risks to belted kingfisher within WAG 2. The biological survey data indicate that kingfisher are present and may be reproducing within WAG 2;, but due to the sampling methods employed, estimates of the abundance of the kingfisher population cannot be made from these data. Limited residue analysis indicates that kingfisher in WAG 2 have selenium and mercury burdens in liver and kidney that may impair reproduction (selenium) or be nephrotoxic (mercury).

Point estimates of exposure indicated that mercury exceeded NOAELs and LOAELs, while PCBs exceeded NOAELs only (Table C.2). Monte Carlo simulation of exposure and comparison of these estimates to NOAELs and LOAELs (Table C.19) and calculation of binomial probability distributions (Table C.27) suggest that only mercury presents a significant risk to the kingfisher population in WAG 2.

The limited mercury body burden data corroborate the results of exposure assessment for mercury; therefore, the weight of evidence suggests that mercury presents risks to kingfisher within WAG 2. Because fish from WOC were not analyzed for selenium, the literature toxicity line of evidence is unavailable. The weight of evidence for selenium, therefore, is based solely on the body burden data, which suggest that selenium may present a risk to kingfisher within WAG 2.

7.1.4 Uncertainties Concerning Risks to Wildlife

Limited Sample Sizes. Because contaminant body burden data for belted kingfisher in WOC watershed are limited to one observation, conclusions based upon this data point should be viewed with caution. This single observation may not be representative of body burdens present in other kingfisher from WOC.

Bioavailability of Contaminants. Bioavailability of contaminants was assumed to be comparable between soil and water from WAG 2 and the diets used in the literature toxicity tests. Because bioavailability may not be comparable, exposure estimates based upon the contaminant concentrations may either under- or overestimate the actual contaminant exposure experienced.

Extrapolation from Published Toxicity Data. While published toxicity studies are available for mink, there are no published data for the other endpoints. To estimate toxicity of contaminants at the site, it was necessary to extrapolate from studies performed on test species (i.e., mallard ducks, ring-necked pheasant, and rats). While it was assumed that toxicity could be estimated as a function of body size, the accuracy of the estimate is not known. For example, hawks or kingfisher may be more or less sensitive to contaminants than ducks or pheasants due to factors other than metabolic rate.

Additional extrapolation uncertainty exists for those contaminants for which data consisted of only LOAELs or tests were subchronic in duration. For either case, an uncertainty factor of 10 was employed to estimate NOAELs or chronic data. The uncertainty factor of 10 may either over- or underestimate the actual LOAEL-NOAEL or subchronic-chronic relationship.

Toxicity of PCBs to piscivorous wildlife was evaluated using toxicity data from studies on Aroclor 1254. Because toxicity of PCB congeners can vary dramatically, the applicability of data for Aroclor 1254 is unknown.

Variable Food Consumption. While food consumption by wildlife was assumed to be similar to that reported for the same or related species in other locations, the validity of this assumption cannot be determined. Food consumption by wildlife in WAG 2 may be greater or less than that reported in the literature, resulting in either an increase or decrease in contaminant exposure.

Single Contaminant Tests vs Exposure to Multiple Contaminants in the Field. While wildlife in WAG 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, single contaminant studies may overestimate their toxic potential. Similarly, for those contaminants that interact additively or synergistically, single contaminant studies may underestimate their toxic potential.

Inorganic Forms or Species Present in the Environment. Toxicity of metal species varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. For example, asenic (III) and methyl mercury are more toxic than arsenic (V) and inorganic mercury, respectively. The available data on the contaminant concentrations in media do not report which species or form of contaminant was observed. Because benchmarks used for comparison represented the more toxic species/forms of the metals (particularly for arsenic and mercury), if the less toxic species/form of the metal were actually present in media from WAG 2, potential toxicity at the sites may be overestimated.

Uptake Factors. Soil to biota uptake factors specific to WAG 2 were unavailable. Therefore, it was assumed that the uptake factors developed as part of the Bear Creek assessment were applicable. Due to the differing geologies and histories between WAG 2 and Bear Creek, the Bear Creek uptake factors may over- or underestimate the actual biota concentrations in WAG 2.

Contaminant Concentrations in Unanalyzed Food Types. Uptake factors were not available for all food types consumed by the endpoint species. It was assumed that the uptake factors for food types for which we had data were representative of that for those without data. Due to the different life histories of among food types, contaminant burdens are likely to differ from the measured data. Therefore, an assumption of comparability among food types may either over- or underestimate exposure.

Monte Carlo Simulation. In performing Monte Carlo simulations, distributions must be assigned to parameters. Because wildlife are mobile, the mean of the contaminant concentration is likely to best represent their exposure. or this report, the contaminant concentrations in fish were assumed to be normally distributed.

7.2 RISKS TO PLANTS AND EARTHWORMS FROM EXPOSURE TO RADIONUCLIDES

7.2.1 Exposure Assessment

Native biota may receive external radiation exposure from radionuclides in air, water, soil, sediment, and other biota, such as vegetation or earthworms. Organisms also receive internal radiation exposure from radionuclides ingested via food and water and from radionuclides absorbed through the skin and respiratory organs (IAEA 1976, Blaylock and Trabalka 1978). Evaluation of the resulting radiation doses received by biota requires quantitative information on the radionuclides to which they are exposed. In all cases the radiation source must be known in terms of the quantity of each specific radionuclide and the corresponding energy released per disintegration. Special corrections for factors that may affect exposures are applied when appropriate.

Radiation dose rates (mrad/d) were calculated for plants, earthworms, and representative terrestrial and semiaquatic wildlife species in the WOC watershed using methodology adapted from Blaylock et al. (1993), Baker and Soldat (1992), and DOE (1995b). The methodology was modified to address site-specific conditions and receptors. Current condition dose rates from internal exposures via ingestion of food and soil and inhalation of dust were evaluated, as were dose rates from external exposures via soil.

The representative terrestrial and semiaquatic wildlife selected as endpoints for the radiological assessment were the same as those for the chemical data assessment: short-tailed shrew, white-footed mouse, wild turkey, red fox, white-tailed deer, red-tailed hawk, mink, and belted kingfisher (see Sect. 7.1). Life history parameters used in the radiological assessment were identical to those used for the chemical data assessment (Table C.4-C.11). In addition, it was necessary to develop species-specific values for fraction of time spent above and below ground, underwater, or at the water's surface. A mink was assumed to spend 50% of its time above ground, 20% below ground (in burrows), 20% swimming at the water's surface, and, conservatively, 10% of its time immersed in water. Kingfishers were assumed to split their time equally between perching above water or soil and roosting in burrows. The short-tailed shrew, white-footed mouse, and red fox were all assumed to spend 75% of their time above ground and 25% below the soil surface in dens or burrows. White-tailed deer and wild turkeys spend 100% of their time above ground and were assigned a value of 1. Red-tailed hawks also spend 100% of their time above ground, but much of this is flying or perched high in trees; thus, it was assumed that hawks would only be exposed to external radiation 10% of the time.

7.2.1.1 Exposure models

Terrestrial receptors may receive both external and internal doses of radiation. Internal exposures are a result of ingestion of contaminated food, soil, or water or inhalation of contaminated soil. Receptors on the ground surface receive external exposures from contaminated surface soil via direct radiation. Both above-ground and below-ground exposures are possible, depending on the habits of the receptor. Baker and Soldat (1992) provide general equations for estimating dose rates to wildlife. An adaptation of the Blaylock et al. (1993) methodology was used in estimating radiation dose rates in the *Remedial Investigation for Waste Area Grouping 5 at Oak Ridge National Laboratory* (DOE 1995a). The Blaylock et al. methodology was further modified for the current assessment to account for the availability of site-specific uptake data. The general methodology and the equations specific to each exposure route used in estimation of dose rates for WOC wildlife are described in this subsection. Equations used in this assessment estimate the daily dose rates from current conditions. Dose rates from alpha, beta, and gamma emissions (only beta and gamma for external exposures) were calculated for each radionuclide, including the dose rates from all short-lived daughter products. Dose rates from each radionuclide were then summed over all exposure routes and all radionuclides to arrive at the overall dose rate received for each receptor at each site. Data on radionuclide activities in White Oak Creek water were not available in time for evaluation in this report. The models for estimating dose rates from exposure to radionuclides in water are provided in this section, and the analysis will be performed when the data become available. Mink and kingfisher are the only wildlife receptors likely to experience significant water-related exposures.

External Exposures: Direct radiation from soil. The equation for estimating external dose rates (mrad/d) for terrestrial receptors exposed to contaminated soil uses dose coefficients was published by Eckerman and Ryman (1993). Dose rate reduction factors are used to account for the fraction of time the receptor spends above or below ground. Dose coefficients assume the source region is a smooth plane (Eckerman and Ryman 1993), but this is rarely the case in a terrestrial habitat. A representative average dose reduction factor for ground roughness is 0.7 (Eckerman and Ryman 1993). The equation for dose from external exposures for an organism above ground is written as follows:

Above soil exposures

$$D_{abovegrd} = F_{above} \ F_{ruf} \sum C_{soil,i} \ DF_{grd,i} \ CFb \ , \qquad (6)$$

where

$D_{above\ grd}$ = external dose rate to receptor from above ground exposures to contaminated soil (mrad/d),

F_{above} = dose rate reduction factor accounting for the fraction of time the receptor spends above ground (unitless),

F_{ruf} = dose rate reduction factor accounting for ground roughness (unitless). [representative average of 0.7 (Eckerman and Ryman 1993) used for this assessment],

$C_{soil,i}$ = activity of radionuclide *I* in surface soil (pCi/g),

$DF_{grd,i}$ = dose coefficient for radionuclide *I* in surface soil (Table III.5, Eckerman and Ryman 1993) (Sv/s per Bq/m^3),

CFb = conversion factor to change Sv/s per Bq/m^3 to mrad g/pCi d equals 5.12×10^{14}.

Dose from alpha radiation is not a concern for external sources as alpha radiation lacks penetrating power. The effective dose coefficients from Eckerman and Ryman (1993) incorporate both beta and gamma emissions. Radionuclide-specific parameters are provided in Table C.31. The lower of the 95% UCL and the maximum detected concentration in surface soil were used in estimating the dose rate from external exposures for each radionuclide.

Below-ground exposures are calculated similarly. However, the ground roughness dose reduction factor is not used for below-ground exposure, and the exposure fraction is adjusted to reflect the fraction of time the receptor spends below ground. This mode of exposure is treated as immersion in a continuous soil medium. To account for exposures from above and below, a factor of 2 is included. Because wild turkey, white-tailed deer, and red-tailed hawks do not go below ground, they do not receive a dose via this exposure route. The equation for below-ground external exposures is written as follows:

Below ground exposures—terrestrial wildlife

$$D_{belowgrd} = 2 \ F_{below} \sum C_{soil,i} \ DF_{grd,i} \ CFb \ , \qquad (7)$$

where

$D_{below\ grd}$ = external dose rate to receptor in burrow from contaminated soil (mrad/d),

2 = dose addition factor to account for receptor immersed in soil (unitless),

F_{below} = dose rate reduction factor accounting for the fraction of time the receptor spends below ground (unitless).

All other parameters are as defined for above-ground exposures.

External exposures for plants and earthworms are calculated using similar equations. Earthworms are assumed to spend 100% of their time below ground. Therefore, the F_{below} term in the below ground equation is set to 1. In addition, the dose coefficient is replaced with the energy for emission of beta or gamma particles for each radionuclide; and the CFb conversion factor is replaced with a conversion factor appropriate for converting MeV to g mrad/pCi d (0.0512).

Below-ground—Earthworms and Plants

$$D_{belowgrd} = 2 \sum C_{soil,i} \; \epsilon_i \; CFa \; , \tag{8}$$

where

$D_{below\,grd}$ = external dose rate to plant or earthworm from contaminated soil (mrad/d),
2 = dose addition factor to account for receptor immersed in soil (unitless),
ϵ_i = energy for β or λ emissions by nuclide *I* (MeV/nt),
Cfa = conversion factor to go from MeV/nt to g mrad/pCi d (5.12×10^{-2}).

All other parameters are as defined for above-ground exposures.

Plants receive external exposures to both above- and below-ground parts. The below-ground model is identical to that for earthworms. While the above-ground parts of plants receive a lower dose rate from soil than below-ground parts, the models used in this assessment estimate only the dose rate for below-ground parts.

External Exposures: Direct radiation from water. The equation for estimating external dose rates (mrad/d) for terrestrial receptors exposed to radionuclides in water uses dose coefficients published by Eckerman and Ryman (1993) and is similar to the equation used for soil. Dose rate reduction factors are used to account for the fraction of time the receptor spends underwater or at the water's surface. The equation for dose from external exposures for an organism immersed in water is written as follows:

Underwater exposures

$$D_{underwater} = F_{underwater} \sum C_{water,i} \; \frac{1}{WD} \; DF_{water,i} \; CFw \; , \tag{9}$$

where

$D_{underwater}$ = external dose rate to receptor from underwater exposures to contaminated soil (mrad/d)
$F_{underwater}$ = dose rate reduction factor accounting for the fraction of time the receptor spends underwater (unitless)
$C_{water,i}$ = activity of radionuclide *I* in water (pCi/L)
WD = density of water (1000 g/L)
$Df_{water,i}$ = dose coefficient for radionuclide *I* for organism immersed in water (Eckerman and Ryman 1993) (Sv m^3/Bq s)

CFw = conversion factor to change Sv m^3/Bq s to mrad g/pCi d equals 3.20×10^{14}.

Dose from alpha radiation is not a concern for external sources as alpha radiation lacks penetrating power. The effective dose coefficients from Eckerman and Ryman (1993) incorporate both beta and gamma emissions. Radionuclide-specific parameters are provided in Table C.31. The mink is the only representative wildlife species likely to spend time under water.

Exposures at the water surface are calculated similarly. However, the exposure fraction is adjusted to reflect the fraction of time the receptor spends at the water surface. Because organisms at the water's surface are exposed from below instead of from above and below, a dose reduction factor of 0.5 is applied. Mink were assumed to spend 20% of their time swimming at the water surface, and kingfishers were assumed to spend 33% of their time perched close to the water surface. Other receptors were not expected to spend significant time in water, so external water exposures were not addressed. The equation for external exposures at the water surface is written:

Water surface exposures

$$D_{surface} = F_{surface} \ 0.5 \sum C_{water,i} \ \frac{1}{WD} \ DF_{water,i} \ CFw \ , \tag{10}$$

where

$D_{surface}$ = external dose rate to receptor from exposures at the water surface (mrad/d),

$F_{surface}$ = dose rate reduction factor accounting for the fraction of time the receptor spends at the water surface (unitless),

0.5 = dose reduction factor to account for the difference between being immersed in water and being at the water surface (unitless).

All other parameters are the same as those defined for underwater exposures.

Internal Exposures: Ingestion. Wildlife receptors may receive internal radiation doses after ingesting contaminated prey, soil, or water or after inhaling contaminated dust. Blaylock et al. (1993) provide an equation for estimating the internal dose to fish contaminated with radionuclides. This equation can be modified to address plants, earthworms, and consumers eating a variety of prey types, ingesting soil, and drinking water.
The equation is

$$D = \sum QF \ C_{tissue} \ E_i \ CFa \ , \tag{11}$$

where

D = internal dose rate received after ingestion of contaminated prey, soil, and water, (mrad/d) or uptake of radionuclides from soil (for plants and invertebrates).

QF = quality factor to account for the greater biological effectiveness of α particles (20 for α; 1 for β and λ emissions; unitless),

C_{tissue} = activity (pCi/g) of radionuclide *I* in tissue of organism,

ϵ_i = energy for α, β, or λ emissions by nuclide *I* (MeV/nt),

Cfa = Conversion factor to go from MeV/nt to g mrad/pCi d. (5.12×10^{-2}).

The primary modifications involve estimating the concentration in the consumer species and because absorbed energy fractions are generally unavailable for terrestrial wildlife, conservatively assuming that absorbed energies equaled 1 for all radionuclides and all receptors. Estimation of tissue concentrations was conducted using soil-to-tissue ratios derived from data collected in the Bear Creek watershed for three plant types (browse, canopy vegetation, and herbaceous vegetation) and small rodents; soil-to-tissue ratios for earthworms were derived from data collected at WAG 2, WAG 5, and the Bear Creek watershed. These data are likely to be more relevant to the WOC watershed than are more general literature-derived bioaccumulation factors. When site-specific uptake factors were unavailable for specific radionuclides, values were derived from those for related isotopes with measured values or from measured values of the elemental form of the radionuclide.

Dose from internal exposures was calculated for alpha, beta, and gamma energies of each radionuclide. Energies were obtained from Eckerman and Ryman (1993) and are provided in Table C.31. Because different types of radiation differ in their relative biological effectiveness per unit of absorbed dose, a quality factor derived from data on humans is normally applied (NCRP 1987). A quality factor of 1 is used for beta and gamma radiation and 20 for alpha radiation (Blaylock et al. 1993).

The lower of the 95% UCL and the maximum detected concentration in surface soil was used in exposure calculations. Radiation energies (MeV) were obtained from Eckerman and Ryman (1993) for all radionuclides and their short-lived daughter products.

When measured soil-to-tissue uptake factors were available, tissue concentrations for plants, earthworms, short-tailed shrews, and white-footed mice were estimated by multiplying the appropriate uptake factor by soil activity levels. When site-specific data were lacking, it was necessary to use transfer factors from available literature. For plants and earthworms the literature-derived values were soil-to-tissue transfer factors, so estimation of tissue concentrations was performed as described previously. However, for mammals and birds, literature values used were bioaccumulation factors defined as the ratio of activity level in receptor tissue divided by the activity level in the receptor's food. Therefore, when site-specific data were lacking, tissue concentrations were estimated as follows:

$$C_{tissue} = BAF_i \; (C_{ij} \;\; P_j + C_{soil} \;\; P_{soil} + C_{water} \;\; \frac{1}{WD}) \; , \qquad (12)$$

where

C_{tissue}	=	concentration of radionuclide I in receptor tissue (pCi/g),
BAF_i	=	bioaccumulation factor for radionuclide I. (pCi/g tissue over pCi/g food),
C_{ij}	=	activity (pCi/g) of radionuclide I in prey type j,
P_j	=	proportion of prey type j in receptor's diet (unitless),
C_{soil}	=	activity (pCi/g) of radionuclide I in surface soil,
P_{soil}	=	proportion of soil in receptor diet. (unitless),
C_{water}	=	activity (pCi/L) of radionuclide I in surface water,
WD	=	density of water (1000 g/L).

It was assumed that uptake of radionuclides from ingested food, water, and soil was similar. For predators which consume small mammals, a small mammal tissue concentration is needed to estimate the dose rate to the predator. In order to simplify the exposure models, the white-footed mouse was used as the generic prey species. Therefore, small tissue concentrations used as input to the red fox, red-tailed hawk, and mink dose rate models were based on white-footed mouse tissue concentrations calculated as described previously. Uptake factors used in the radiological assessment are provided in Table C.32.

Concentrations in fish will be estimated by multiplying water concentrations by fish bioconcentration factors obtained from Blaylock et al. (1993) when data on radionuclide activity in water become available.

Internal Exposures: Inhalation. Wildlife species using burrows receive an additional internal dose from inhalation of dust originating from contaminated soil. Intake of radionuclide *I* by inhalation is estimated as follows (DOE 1995b):

$$D_{inh} = QF \quad F_{below} \sum C_{soil,i} \quad A \quad \frac{1}{AD} \quad \epsilon_i \; CFa \; , \qquad (13)$$

where
D_{inh} = internal dose rate from inhalation of contaminated soil (mrad/d),
F_{exp} = dose reduction factor for fraction of time receptor spends below ground (unitless),
A = mass of respirable dust per volume of air breathed (0.1 g/m^3; DOE 1995b),
AD = air density (1200 g/m^3; Eckerman and Ryman 1993),
ϵ_i = α, β, or γ radiation energies for radionuclide *I* (MeV/nt),
CFa = conversion factor to go from MeV/nt to mrad g/pCi d (5.12 x 10^{-2}).

Healy (1980) suggests that 0.0001 g/m^3 would be a conservative value when addressing human exposures to dust. Because burrowing animals are likely to spend a greater portion of their time in a confined space (burrow) than humans and are physically closer to the soil surface, an air mass loading of 0.1 g/m^3 was selected as a conservative estimate of the mass of respirable dust (A) to which these animals may be exposed.

Total internal exposures were obtained by adding ingestion and inhalation dose rates over all radionuclides, including all short-lived daughter products. In addition, the dose rate to a kingfisher collected near ORNL Building 4505 was estimated based on a measured body burden of 91.27 pCi/g Cs-137 (Table C.25).

7.2.2 Effects Levels for Radionuclides

The discharge of radioactive waste into the environment results in long-term, low dose exposure to organisms. In most cases, acute mortality can be discounted. Any potential increase in morbidity and mortality that might result from the exposure to chronic irradiation above background is unlikely to be detected because of natural fluctuations in the size of populations. The International Atomic Energy Agency (IAEA) and the Atomic Energy Act (1992) recommend limiting the dose for terrestrial organisms to 100 mrad/d (IAEA 1992). Species-specific effects data were not available, so 100 mrad/d was selected as the threshold dose for all the representative wildlife receptors. This level of exposure to the maximally exposed individual is thought to be protective of the overall receptor population (IAEA 1992, DOE 1995a).

7.2.3 Risk Characterization for Terrestrial Receptors Exposed to Radionuclides

Potential risks to plants, invertebrates, and selected terrestrial and semiaquatic wildlife receptors from exposures to radionuclides were estimated for each of four reaches along WOC by comparing estimated dose rates to the recommended dose rate limit of 100 mrad/d. Dose rates were summed for all radionuclides over all exposure routes to obtain the overall dose in mrad/d received by each receptor at each site. Tables C.33 through C.36 present overall exposures from soil for each receptor by site and identify the radionuclides contributing the majority of the dose rate. Water-related dose rates were not estimated for this assessment.

Possible radiation effects are anticipated for plants, earthworms, and wildlife receptors frequenting all four reaches of WOC as the overall dose rates in each reach exceed the effects threshold of 100 mrad/d. Dose rates were generally highest for all receptors at the IHP, lower in LWC and MWC WOC, and high in LMB. Watershed-wide population level effects are likely considering herbaceous plant, earthworm, short-tailed shrew, and white-footed mouse dose rates exceeded the dose rate limit in all reaches. The primary contributors to dose rates in the IHP, MWC, and LWC were ^{239,240}Pu, ^{241}Am, and ^{137}Cs; and ^{90}Sr, ^{239,240}Pu, ^{241}Am, and ^{137}Cs were the primary contributors in LMB.

Intermediate Holding Pond (IHP). Potential radiation effects are expected for terrestrial plant, invertebrate, and wildlife receptors at the IHP as dose rates exceeded the effects threshold of 100 mrad/d for all receptors except the white-tailed deer and belted kingfisher (Table C.33). The kingfisher dose rate estimate was based only on external exposures to radionuclides in soil; adding likely internal exposures from ingestion of water and fish may result in a dose rate exceeding 100 mrad/d. Plants, earthworms, shrews, and mice exceeded the limit by more than a factor of 10 (ranging from 21.77 for mice to 43.4 for plants). Dose rates for turkey, fox, hawk, and mink were lower (1.5 to 3.26 times the effects threshold).

Plutonium-239/240, Americium-241, and Cesium-137 were the primary contributors to estimated dose rates for plants, earthworms, shrews, and mice (Table C.33). Plutonium-239/240 accounted for greater than 40% of the dose for these receptors. Cesium-137 was the only significant contributor to dose rates for turkey, deer, fox, hawk, mink, and kingfisher, accounting for greater than 91% of the dose rate for these receptors.

Middle White Oak Creek (MWC). Potential radiation effects are expected for terrestrial plant, invertebrate, and wildlife receptors in the vicinity of MWC as dose rates exceeded the effects threshold of 100 mrad/d for plants, earthworms, shrews, and mice (Table C.34). Dose rates for turkey, deer, fox, hawk, mink, and kingfisher were all well below the effects threshold (ranging from 3 to 17 mrad/d). Plants, earthworms, shrews, and mice exceeded the limit by more than a factor of 7 (ranging from 7.2 times the threshold for mice to 14 times for plants).

Plutonium-239/240 and Americium-241 were the primary contributors to estimated dose rates for plants, earthworms, shrews, and mice (Table C.34). Plutonium-239/240 accounted for greater than 62% of the dose for these receptors. Cesium-137 was the only significant contributor to dose rates for turkey, deer, fox, hawk, mink, and kingfisher, accounting for greater than 53% of the relatively low dose rates for these receptors.

Lower White Oak Creek (LWC). Potential radiation effects are expected for terrestrial plant, invertebrate, and wildlife receptors in the vicinity of LWC as dose rates exceeded the effects threshold of 100 mrad/d for plants, earthworms, shrews, and mice (Table C.35). Dose rates for turkey, deer, fox, hawk, mink, and kingfisher were all below the effects threshold (ranging from 10 to 49 mrad/d). Plants, earthworms, shrews, and mice exceeded the limit by more than a factor of 7 (ranging from 7.2 times the threshold for mice to 13.5 times for plants).

Plutonium-239/240 and Americium-241 were the primary contributors to estimated dose rates for plants, earthworms, shrews, and mice (Table C.35). Plutonium-239/240 accounted for greater than 51% of the dose for these receptors. Cesium-137 was the only significant contributor to dose rates for turkey, deer, fox, hawk, mink, and kingfisher, accounting for greater than 84% of the dose rates for these receptors.

Lower Melton Branch (LMB). Potential radiation effects are expected for terrestrial plant, invertebrate, and wildlife receptors at the LMB as dose rates exceeded the effects threshold of 100 mrad/d for all receptors except the belted kingfisher (Table C.36). The kingfisher dose rate estimate was based only on external exposures to radionuclides in soil; adding likely internal exposures from ingestion of water and fish may result in a dose rate exceeding 100 mrad/d. Plants, earthworms, fox, and hawk exceeded the limit by more than a factor of 10 (ranging from 10.87 for the hawk to 30.5 for earthworms). Dose rates for shrews, mice, turkeys, deer, and mink were lower (2.75 to 6.22 times the effects threshold).

Plutonium-239/240 (66.6%) and Cesium-137 (17.1%) were the primary contributors to the estimated dose rate for plants. Strontium-90 and Curium-244 accounted for over 80% of the estimated dose rate for shrews and mice (Table C.36). Strontium-90 was the only significant contributor to dose rates for earthworms, turkey, deer, fox, hawk, and mink, accounting for greater than 92% of the dose rate for these receptors.

Kingfisher Tissue Data. In addition to evaluation of potential radionuclide exposures in the four reaches of WOC, the internal dose rate for a single kingfisher collected near Building 4505 at Oak Ridge National Laboratory, upstream of WOC, was estimated based on a measured body burden of 91.27 pCi/g of Cesium-137 (Table C.25) (Baron and Ashwood 1996). The model for internal ingestion exposures described in Sec. 7.2.1.1 was used to estimate a dose rate of 3.8 mrad/d for this kingfisher, a rate well below the effects threshold of 100 mrad/d. Baron and Ashwood suggest that the high levels of Cesium-137 in this kingfisher indicate its foraging territory includes WOC or nearby surface impoundments. If the Cs-137 body burden found in the kingfisher from the Building 4505 area is representative of body burdens in kingfishers along WOC, the overall dose rates for kingfishers along the creek may be below 100 mrad/d. Cs-137 was the primary contributor to external exposures from radionuclides in soil for kingfishers along all four reaches of WOC, and the highest dose rate (at the IHP) was estimated at 64 mrad/d (Tables C.33-C.36).

7.7.4 Uncertainties in the Radiological Risk Assessment

There are a number of areas of uncertainty in the estimation of risks to wildlife from exposure to radionuclides at WOC. It is believed that the methodology used in this assessment overestimates the dose rates that would be received. Whereas some of the information needed to implement the methodology is well known, much is unknown or unspecified statistically. A conservative but reasonable approach to model assumptions and radiological exposure scenarios was adopted for this assessment in order to avoid underestimating risks to biota.

- It was conservatively assumed that energy absorption by the wildlife receptors evaluated in this assessment was 100%. While larger organisms are likely to absorb a greater fraction of energy than smaller organisms, it is unlikely that actual absorption would be as high as 100%. Thus, radiation dose rates may be lower than those reported here.
- The dose coefficients obtained from Eckerman and Ryman (1993) used to estimate dose rates from external exposures were developed for application in determining dose rates to humans. These dose coefficients were applied directly for wildlife receptors, but the actual dose

coefficients for wildlife, given differences in size, behavior, and general morphology, may be greater or lesser than those developed for humans. However, similar values have not been developed specifically for wildlife receptors.

- Uptake factors from soil to earthworms were not available for many radionuclides. If uptake factors were unavailable, the higher of the plant or mammal uptake factors was used for earthworms. It is unknown whether actual earthworm uptake factors for these radionuclides would be higher or lower than those used, but use of the larger of the plant and mammal values was a conservative approach.

- Mammal soil-to-tissue uptake factors were calculated from measured data on small rodents. While these clearly apply directly to white-footed mice, they were also used for the other small mammalian receptor, the short-tailed shrew. Short-tailed shrew diets differ somewhat from that of white-footed mice as shrews eat a higher proportion of invertebrates. However, it was assumed that site-specific data on mice were more applicable to shrews than literature-derived bioaccumulation factors.

- Uptake factors were not available for birds such as the red-tailed hawk. Radionuclide activities in red-tailed hawk tissues were estimated using mammal uptake factors assuming that uptake would be similar for hawks and mammals. It is unknown whether actual bird uptake factors would be higher or lower than those used here.

- Literature-derived bioaccumulation factors were used to estimate radionuclide activities in mammal and bird tissues. While these values are believed to be representative of bioaccumulation likely to observed on site, actual values may be higher or lower than used in this assessment.

- Dietary information for the wildlife receptors evaluated here was obtained from available literature. Diets of given species in the WOC watershed may vary somewhat from those reported in the literature from other areas.

- Species-specific time budget information is lacking, so it was necessary to make a number of assumptions regarding the fraction of time each receptor spends above or below ground, underwater, or at the water's surface. The fractions used in this assessment were selected based on general knowledge of species behavior patterns, but actual fractions may be higher or lower.

- The air mass loading factor of 0.1 g/m^3 used in estimating exposures from inhalation of radionuclide-contaminated dust was selected as a conservative value. Healy (1980) suggested 0.0001 g/m^3 would be a conservative value for estimating human exposures from inhalation of dust.

8. SUMMARY AND CONCLUSIONS

This screening assessment is intended to indicate whether there are credible hazards to ecological endpoints in the WOC watershed due to contamination and whether there are data gaps that need to be filled. The following are conclusions concerning the hazards:

- Radionuclides in WAG 2 do not pose a hazard to fish or aquatic invertebrates. Strontium-90 in three seeps exceeds the screening benchmarks for aquatic life, but these seeps do not support communities of aquatic macroorganisms.

- Radionuclides in WAG 2 pose a hazard to terrestrial plants, wildlife, and soil invertebrates in all four reaches of WOC and in lower MB. Exposures were highest in the IHP reach.

- The unfiltered water in all reaches exceeded water quality criteria and other toxicological benchmarks for aquatic life.

- Water from WOC adjacent to and below ORNL was toxic in a fish embryo-larval test but not the standard subchronic tests.

- The fish community in WOC has low species richness, but this may be due to lack of recovery from past toxicity. However, the riffle invertebrate community, which is exposed primarily to chemicals in water like the fish and which has flying stages which should allow recovery, also has low species richness.

- Sediments from the IHP, MWC, and LMB are contaminated to levels that have been associated with toxic effects at other sites. Contaminants include metals, PAHs, and PCBs.

- The species richness of benthic invertebrates from WAG 2 sediments is lower than in background WOC sediments but not other reference streams. The exception is WOL, which has very low species richness; but no comparable reference lakes were characterized.

- Chromium and mercury levels in IHP and LWC soils and mercury in MWC and LMB soils exceeded levels that were reported to be toxic to earthworms.

- Multiple metals were found in IHP, MWC, LWC, and LMB soils at concentrations that have been reported to be toxic to plants.

- Metals in seep waters in the W4T and WS reaches are at concentrations that have been reported to be toxic to plants.

- Hazards to wildlife were found in all WOC reaches. However, the hazards were greatest for mercury among chemicals, for IHP among reaches, and for shrews among species.

- Kingfishers from WOC show elevated levels of contaminants, and the one adult found had tissue levels of mercury and selenium that are indicative of toxicity. Mink from WOC do not appear to be contaminated.

The following are major uncertainties that could be addressed by additional data collection:

- Background concentrations need to be better characterized for the seeps, surface waters, sediments, and floodplain soils.
- The contamination of soil and biota on the burial grounds other than WAGs 5 and 6 are unknown.
- The bioavailable concentrations of metals in WOC watershed are unknown.
- The toxicity of soil and sediment in the watershed are unknown.
- The toxicity of water is unknown for some reaches and has been irregular in others (Table 3.4).
- The fish communities have not been quantitatively characterized for the smaller tributaries (Table 3.4).
- The chemical composition of fish is unknown for some tributaries and reaches (Table 3.4).
- Because of the high Se levels in a kingfisher, Se levels should be characterized in fish.

In sum, plausible hazards exist in all reaches of WAG 2 and to all ecological endpoints. Radionuclides, organic chemicals, and metals are all implicated. However, none of the risk estimates or observations of the state of the watershed suggest that there is a need for an accelerated response. That is, no threatened or endangered species are at risk and no wetlands or other highly valued populations or ecosystems are experiencing catastrophic effects. Some parts of the watershed are uncharacterized and should be surveyed and sampled before the RI is completed. These results imply that a more complete data set should be assembled and used to prepare a definitive ecological risk assessment for the WOC watershed.

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