RISK CHARACTERIZATION REPORT FOR THE HWIR 99 MULTIMEDIA, MULTIPATHWAY, AND MULTIRECEPTOR RISK ASSESSMENT (3MRA)

Work Assignment Managers and Technical Direction:	Stephen M. Kroner David A. Cozzie U.S. Environmental Protection Agency Office of Solid Waste Washington, DC 20460
Prepared by:	Center for Environmental Analysis Research Triangle Institute 3040 Cornwallis Road Research Triangle Park, NC 27709-2194 Under Contract No. 68-W-98-085

U.S. Environmental Protection Agency
Office of Solid Waste
Washington, DC 20460

October 1999

ACKNOWLEDGMENTS

This document is a compendium of other background documents produced by the entire HWIR Team. That Team consists of individuals from the following organizations: EPA Office of Solid Waste, EPA Office of Research and Development, Research Triangle Institute, Pacific Northwest National Laboratories, and HydroGeoLogic, Inc.

DISCLAIMER

The work presented in this document has been funded by the U.S. EPA. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by the Agency.

Table of Contents

Figures vii

Tables vii

1.0 Introduction 1-1

2.0 HWIR Overview 2-1
- 2.1 Assessment Strategy 2-1
  - 2.1.1 Human Receptors and Exposure Pathways 2-3
  - 2.1.2 Ecological Receptors 2-6
- 2.2 Input Data 2-7
- 2.3 The 3MRA Model 2-9
- 2.4 Risk Protection Criteria 2-10
  - 2.4.1 Risk Level 2-10
  - 2.4.2 Hazard Quotient (HQ) 2-10
  - 2.4.3 Ecological Hazard Quotient 2-10
  - 2.4.4 Population Percentile 2-11
  - 2.4.5 Probability of Protection 2-11

3.0 Characterization of Potential Risk 3-1
- 3.1 Results 3-3
- 3.2 Wells Located in the Plume 3-3
- 3.3 System Testing Capabilities 3-4

4.0 Uncertainty and Limitations 4-1
- 4.1 Scenario Uncertainty 4-1
  - 4.1.1 Waste Management Units 4-1
  - 4.1.2 Spatial Scales (Area of Interest) 4-2
  - 4.1.3 Site Locations 4-3
  - 4.1.4 Receptors 4-6
    - 4.1.4.1 Human Receptors 4-6
    - 4.1.4.2 Ecological Receptors 4-6
  - 4.1.5 Temporal Scale 4-8
  - 4.1.6 Multimedia Modeling 4-9
  - 4.1.7 Exposure Pathways 4-10
  - 4.1.8 Endpoints 4-11
  - 4.1.9 Risk Metrics 4-12
- 4.2 Data Variability/Uncertainty 4-12
  - 4.2.1 Human Health Effects/Benchmarks 4-13
    - 4.2.1.1 Methods and Data Sources 4-16
    - 4.2.1.2 Assumptions 4-18
    - 4.2.1.3 Uncertainties and Limitations 4-19

Table of Contents (continued)

4.2.2 Ecological Benchmarks and Chemical Stressor Concentrations Limits (CSCLs) 4-20
4.2.2.1 Methods and Data Sources 4-20
4.2.2.2 Assumptions and Uncertainties 4-21
4.2.3 Facility, Waste Management Unit, and Waste Property Data 4-28
4.2.3.1 Methods and Data Sources 4-29
4.2.3.2 Assumptions 4-31
4.2.3.3 Uncertainties and Limitations 4-34
4.2.4 Meteorological Data 4-35
4.2.4.1 Methods and Data Sources 4-35
4.2.4.2 Assumption 4-36
4.2.4.3 Uncertainties and Limitations 4-37
4.2.5 Watershed and Waterbody Data 4-38
4.2.5.1 Methods and Data Sources 4-38
4.2.5.1 Assumptions 4-40
4.2.5.3 Uncertainties and Limitations 4-41
4.2.6 Soil Data 4-45
4.2.6.1 Methods and Data Sources 4-45
4.2.6.2 Assumptions 4-47
4.2.6.3 Uncertainties and Limitations 4-47
4.2.7 Farm Food Chain / Terrestrial Food Web Data 4-48
4.2.7.1 Methods and Data 4-48
4.2.7.2 Assumptions 4-49
4.2.7.3 Uncertainty and Limitations 4-50
4.2.8 Aquatic Food Web Data 4-52
4.2.8.1 Methods and Data Sources 4-53
4.2.8.2 Assumptions. 4-54
4.2.8.3 Uncertainties and Limitations 4-55
4.2.9 Human Exposure Factors 4-57
4.9.2.1 Methods and Data Sources 4-58
4.2.9.2 Assumptions. 4-60
4.2.9.3 Uncertainties and Limitations 4-61
4.2.10 Human Receptor Data 4-63
4.2.10.1 Methods and Data Sources. 4-64
4.2.10.2 Assumptions. 4-66
4.2.10.3 Uncertainties and Limitations. 4-67
4.2.11 Ecological Exposure Factors 4-69
4.2.11.1 Methods and Data Sources 4-69
4.2.11.2 Assumptions 4-70
4.2.11.3 Uncertainties and Limitations 4-72
4.2.12 Ecological Receptors and Habitats 4-73
4.2.12.1 Methods and Data Sources 4-74

Table of Contents (continued)

4.2.12.2 Assumptions. 4-75
4.2.12.3 Uncertainties and Limitations. 4-79
4.3 Model Uncertainty 4-80
4.3.1 NonWastewater Sources (LF, LAU, WP) Modules 4-80
4.3.1.1 Spatial and Temporal Scales 4-81
4.3.1.2 Key Assumptions 4-84
4.3.1.3 Methodologies.. 4-85
4.3.1.4 Limitations/Uncertainties 4-89
4.3.2 Wastewater Sources (SI, AT) Modules 4-92
4.3.2.1 Spatial and Temporal Scales. 4-92
4.3.2.2 Key Assumptions 4-93
4.3.2.3 Methodologies 4-93
4.3.2.4 Limitations/Uncertainties 4-98
4.3.3 Air Module 4-100
4.3.3.1 Spatial and Temporal Scales. 4-100
4.3.3.2 Key Assumptions. 4-101
4.3.3.3 Methodologies. 4-101
4.3.3.4 Limitations/Uncertainties 4-102
4.3.4 Watershed Module 4-102
4.3.4.1 Spatial and Temporal Scales 4-102
4.3.4.2 Key Assumptions 4-103
4.3.4.3 Methodologies 4-104
4.3.4.4 Limitations/Uncertainties. 4-105
4.3.5 Vadose Module 4-105
4.3.5.1 Spatial and Temporal Scales 4-105
4.3.5.2 Key Assumptions. 4-105
4.3.5.3 Methodologies 4-105
4.3.5.4 Limitations/Uncertainties 4-106
4.3.6 Saturated Zone Module 4-106
4.3.6.1 Spatial and Temporal Scales. 4-107
4.3.6.2 Key Assumptions 4-107
4.3.6.3 Methodologies. 4-107
4.3.6.4 Limitations/Uncertainties 4-108
4.3.7 Surface Water Module 4-109
4.3.7.1 Spatial and Temporal Scales 4-109
4.3.7.2 Key Assumptions 4-109
4.3.7.3 Methodologies. 4-109
4.3.7.4 Limitations/Uncertainties 4-110
4.3.8 Terrestrial Food Web Module 4-111
4.3.8.1 Spatial and Temporal Scales 4-111
4.3.8.2 Key Assumptions. 4-111

Table of Contents (continued)

4.3.8.3 Methodologies 4-111
4.3.8.4 Limitations/Uncertainties 4-113
4.3.9 Farm Food Chain Module 4-113
4.3.9.1 Spatial and Temporal Scales 4-113
4.3.9.2 Key Assumptions 4-113
4.3.9.3 Methodologies 4-114
4.3.9.4 Limitations/Uncertainties 4-115
4.3.10 Aquatic Food Web Module 4-116
4.3.10.1 Spatial and Temporal Scales 4-116
4.3.10.2 Key Assumptions 4-116
4.3.10.3 Methodologies 4-117
4.3.10.4 Limitations/Uncertainties 4-120
4.3.11 Human Exposure Module 4-121
4.3.11.1 Spatial and Temporal Scales 4-121
4.3.11.2 Key Assumptions 4-122
4.3.11.3 Methodologies 4-122
4.3.11.4 Limitations/Uncertainties 4-123
4.3.12 Human Risk Module 4-125
4.3.12.1 Spatial and Temporal Scales 4-125
4.3.12.2 Key Assumptions 4-125
4.3.12.3 Methodologies 4-125
4.3.12.4 Limitations/Uncertainties 4-126
4.3.13 Ecological Exposure Module 4-127
4.3.13.1 Spatial and Temporal Scales 4-127
4.3.13.2 Key Assumptions 4-127
4.3.13.3 Methodologies 4-128
4.3.13.4 Limitations/Uncertainties 4-130
4.3.14 Ecological Risk Module 4-131
4.3.14.1 Spatial and Temporal Scales 4-131
4.3.14.2 Key Assumptions 4-132
4.3.14.3 Methodologies 4-132
4.3.14.4 Limitations/Uncertainties 4-133
4.4 System/Technology Uncertainty 4-134

5.0 References 5-1

Appendix Table of Results A-1

FIGURES

4-1 Area of interest (AOI) for HWIR 3MRA 4-29
4-2 Overview of data collection methodology for watersheds and waterbodies 4-39
4-3 Overview of HWIR soil data collection methodology 4-46
4-4 Human and farm receptor GIS process flow chart 4-65
4-5 3MRA release, media, uptake, exposure and risk modules 4-81
4-6 Local watershed containing WMU 4-82
4-7a Local watershed 4-82
4-7b Cross section view 4-83
4-8 Runoff quality conceptual model 4-87
4-9 Schematic of general model construct for tanks and surface impoundments 4-94
4-10 Conceptual model for SI infiltration algorithm 4-98
4-11 Flowchart of infiltration algorithm 4-99
4-12 Example of simplified food web for lake habitat (Gobas, 1993) 4-117

TABLES

2-1 HWIR Receptor Types and Exposure Pathways 2-4
2-2 Assessment Endpoints Considered for the HWIR Ecological Assessment 2-7
2-3 Risk Protection Combinations Evaluated for HWIR Risk Assessment 2-12
2-4 HWIR Exemption Level Categories 2-12

3-1 Chemical Concentrations (C_w) for Risk Group 1 3-2
3-2 Risk Group 1 - Risk to Subpopulations at Human Health Landfill Chemical Concentrations (C_w) 3-3
3-3 Risk Group 1 - Risk by Exposure Pathway at Human Health Landfill Chemical Concentrations (C_w) 3-3
3-4 Percent of Wells Inside and Outside the Plumes of Contamination Across all Sites 3-4

4-1 Data Types Used by HWIR 3MRA Component Modules 4-14
4-2 HWIR Data Collection Approach, by Data Type 4-16
4-3 Recptor Benchmarks and CSCLS Based on No-Effects Data 4-26
4-4 Meteorological Data File Access, by HWIR 3MRA Model Module 4-35
4-5 Source Data for Watershed and Waterbody Delineations 4-40
4-6 Input Parameters and Data Sources: HWIR Human Exposure Factors 4-59
4-7 Primary Human Receptor Data Sets, Date, and Scale 4-66
4-8 Matrix of Biota Types in Food Webs of Representative Aquatic Habitats 4-68

1.0 Introduction

In 1995, the U.S. Environmental Protection Agency (EPA) issued its policy statement (U.S. EPA, 1995a) and associated guidance for risk characterization (U.S. EPA, 1995b), which described the types of information to be presented in a risk assessment document and how the information is to be presented. A guiding principle in the policy statement is the need for the risk assessments to exhibit "clarity, transparency, reasonableness, and consistency" across Agency programs and guidance. Of particular importance is the need to highlight both the confidence and uncertainty associated with the risk assessment.

The purpose of this report is to provide an overview of the HWIR risk assessment methodology, a discussion of the risk assessment results, and a detailed discussion of the uncertainties and limitations of the study design, data, and models used. The HWIR risk assessment is a complex national assessment. We start with an overview of the assessment, which is written in a less technical style than the corresponding technical background documents from which the information is taken. The overview is meant to provide a general understanding of what is being assessed and the major components of the assessment. The results section presents the various outputs of the assessment so as to be understandable to those who will use the results in making decisions as well as those who may be impacted by the decisions (e.g., industry, general public). Because these results must be understood in light of various uncertainties and limitations of the assessment, we discuss in some detail the uncertainties and limitations associated with the study design (scenario uncertainty), the data, the scientific models, and the modeling system.

For an assessment of the magnitude of HWIR 99, there are many ways to present and provide a critical perspective on the uncertainties and limitations of the results. This is a multimedia, multiple exposure pathway and multiple receptor risk assessment (3MRA), which required considerable investment in both the development of the modeling system and in the databases used by the system. The system continues to be refined and to go through quality assurance checks. However, we believe that there are several major strengths readily apparent in the development of the 3MRA Model and associated components, the data collection approach selected to implement the regional site-based approach, and the testing and quality assurance process followed during both the developmental and implementation phases of the assessment to ensure the accuracy and usefulness of the information produced.

2.0 HWIR Overview

The goal of the HWIR99 risk assessment is to identify wastes currently listed as hazardous that could be eligible for exemption from hazardous waste management requirements. The HWIR risk assessment estimates chemical-specific potential risks to human and ecological receptors living within a radius of two kilometers of industrial nonhazardous waste sites that could manage HWIR-exempted wastes. We used these risk estimates, along with other information, to identify the chemical-specific concentrations for exempted waste that would be protective of human health and the environment at selected sets of risk protection criteria.

The risk assessment developed for the HWIR99 rulemaking is an integrated, multimedia, multiple exposure pathway, and multiple receptor risk assessment (3MRA) that evaluates impacts to human and ecological receptors. The national scale assessment evaluates risks that may occur from the long-term, multimedia release of a chemical from waste management facilities typically expected to handle exempted waste.

This assessment does not consider short-term catastrophic events. The assessment does consider the potential risk or hazard (HQ) of a chemical that will be managed in a waste unit throughout its operational life. We assume landfills and wastepiles have waste added annually for 30 years, and land application units have waste applied quarterly for 40 years; and surface impoundments and tanks have continuous loadings for 50 years. We designed the assessment to provide flexibility in producing a distribution of risk outputs that describe the range of individual risks across the nation from potential exposures to HWIR-exempted waste. We conducted the assessment with a newly developed risk assessment model that had three principal components: (1) the assessment strategy, (2) the input data for the modules (e.g., environmental setting, chemical, and meteorological data), and (3) the 3MRA Model, which includes the chemical release, fate, exposure, and risk modules.

2.1 Assessment Strategy

The 3MRA assessment strategy (U.S. EPA, 1999b) evaluates multiple exposure pathway risks to human and ecological receptors at a statistically representative sample of waste management units (WMUs) and environmental settings to estimate the distribution of risk nationally. It is a forward-calculating approach that begins with selected concentrations of a chemical in waste and estimates the associated hazards and risks to human and ecological receptors. By evaluating a range of waste concentrations managed in a statistical sample of WMUs and using a probabilistic approach to select many of the input parameters, we were able to identify chemical-specific concentrations in waste that match our risk protection criteria (i.e., risk level, hazard quotients, population protection, and probability of site being protected). The

results are intended to represent national distributions of receptor impacts near the waste management units typically expected to manage exempted waste.

Within each type of waste management unit, we sought to maintain mass balance. We begin with a total mass of chemical and partition the mass to volatile, liquid, and sorbed phases. Mass released via each phase is no longer available for partitioning to and release through other phases. The partitioning algorithms and media coefficients that we used are described in the two technical background documents for the modules for the sources (U.S. EPA, 1999t and 1999u) and module verifications are described in U.S EPA (1999ad and 1999ae). The data used in partitioning include a range of Kds representing various waste forms that are intended to reflect the full range of waste leachability including wastes that have been fully treated and those that are untreated.

We are presenting an approach in the HWIR 3MRA model to address the physical relationship between waste concentrations and leachate concentrations, and mass limitations in the leachate. In the 3MRA model we start with a specified concentration of a chemical constituent and the total mass in a waste management unit, partition the constituent in the waste unit into various environmental media. The partitioning takes into consideration the physical and chemical characteristics of the chemical and the characteristics of the media. The relationship in the model, between the concentration of a chemical constituent in the waste and its concentration in the leachate, depends on these physical and chemical characteristics. The initial chemical mass in the waste management unit depletes with time due to the partitioning, degradation and transport. The 3MRA model assumes the initial mass to be finite and then depletes. The concentration of a chemical constituent in a downgradient well is initially zero, gradually reaches a maximum and then declines as the mass released from the waste management unit passes the receptor well area. The details of the partitioning of the chemical mass based on the relationships between the waste and the leachate depend on the physical characteristics of the chemical constituent and the environment. For example, the relationship for organic chemical depends on the fraction of organic carbon in the waste and other factors. For metals, the relationship depends on the pH, the presence of other organic and organic species, temperature, and other factors. This is further described in the various waste management units being modeled in the 3MRA model for HWIR99 (U.S. EPA, 1999t and 1999u).

We assessed the potential human health and ecological impacts at 201 nonhazardous industrial waste management sites. Since some of these sites have more than one type of WMU, there are 419 unique combinations of WMU and environmental setting referred to as settings throughout this document. The sites were randomly selected to be representative of the management sites found in EPA's *Screening Survey of Industrial Subtitle D Establishments* (Westat, 1987). We selected the 201 sites from a survey of approximately 2,850 facilities representing a total population of nearly 150,000 facilities that had one or more of four types of waste management units (landfill, wastepile, land application unit, and surface impoundment). The methodology for the selection of the 201 sites is explained in U.S. EPA (1999s).

The risk assessment is designed to produce chemical-specific distributions of cancer risks or hazards to humans and ecological receptors living in the vicinity of industrial waste sites that could manage HWIR exempted wastes throughout their operating life. For each site and waste

concentration, the model can generate risks for each receptor location and then sums the number of receptors that fall within a specified risk range (bin) to get the distribution of risks for the population at each site. We can use the distribution of risks for a setting to determine whether a setting is protected based on the percentage of the population protected, a specified cancer risk or hazard level, and the initial concentration in waste. The model then uses these data to generate a percentile distribution based on the number settings protected at a specified risk level for each waste concentration to generate the national distribution.

These results are evaluated over a 10,000 year period of exposure. This time frame applies mainly to the groundwater pathway, since receptors are exposed to chemicals via other pathways much sooner. Evaluating peak doses over this time horizon allows the model to capture the slow movement of certain chemicals through the subsurface. Although the time frame for such travel may be long, such contamination could be a serious problem when the chemical reaches the receptor wells. Particularly for chemicals that do not degrade, it is important to determine the magnitude of risk that would be experienced once the contamination does reach a drinking water well. The selection of a long time frame assumes that peaks are more likely to be considered in the assessment.

2.1.1 Human Receptors and Exposure Pathways

At each setting we modeled four human receptor types: residents, home gardeners, farmers (beef and dairy), and recreational fishers. Some of these receptor types overlap; a resident, gardener, or farmer could each also be a recreational fisher, and the farmer could be a beef farmer, dairy farmer, or both. However, for this analysis, each of the receptor types is mutually exclusive so that the total population across all receptor types at a site equals the total population within the study area at the site. For each receptor type, we evaluated exposures to four age cohorts: ages 1 to 5, ages 6 to 11, ages 12 to 19, and older than age 19. These are aggregated into three groups for this assessment: less than 1 year old, ages 1 to 11, and 12 years and older. In addition, the 3MRA model has the ability to evaluate infants (less than 1 year old) as a separate receptor type for 2,3,7,8-TCDD, a dioxin chemical that can be transferred to infants through breast milk. However, 2,3,78-TCDD has not been evaluated in this analysis at this time.

Receptors are estimated to be exposed to chemicals present in ambient air (both vapors and particulates), soils, groundwater, fruits and vegetables, beef and dairy products, and fish. Table 2-1 summarizes the exposure pathways for each receptor type. Annual exposures are chemical- and environmental-setting-specific and are estimated to occur for up to 10,000 years. The source modules simulate release of a chemical until the concentration in the WMU decreases to 1 percent of the maximum or until 200 years have been simulated; however, media concentrations are simulated until the chemical concentration in a particular medium (e.g., groundwater) decreases to 1 percent of the maximum for that medium or until 10,000 years has been simulated. Hence, modeling simulations can range from a few hundred years for chemicals that move quickly in the environment, are not persistent, and do not bioaccumulate to 10,000 years for the most persistent and least mobile chemicals such as some metals. As shown in Table 2-1, not all individual classified in a particular receptor type are exposed to the some pathways. For example, some fraction or subset (up to 1) of each receptor type may be exposed to contaminated groundwater used for drinking and showering as described below.

Table 2-1. HWIR Receptor Types and Exposure Pathways

	Resident	Home Gardener	Farmer	Fisher	Infants
Inhalation	X	X	X	X	
Soil ingestion	X	X	X	X	
Groundwater ingestion	X (subset)	X (subset)	X (subset)	X (subset)	
Inhalation during showering	X (subset)	X (subset)	X (subset)	X (subset)	
Fruit and vegetable ingestion		X	X	X (subset)	
Beef and/or milk ingestion			X	X (subset)	
Fish ingestion				X	
Breast milk ingestion					X

Residents breathe contaminated air and ingest contaminated soil (as an incidental contamination of hands or foods). A subset of residents have private drinking water wells and are exposed to contaminated groundwater through both direct drinking water ingestion and inhalation through showering. Those on public water supply are assumed to have treated water that meets all drinking water standards. We used the 1990 U.S. Census block survey data to estimate the number and ages of residents within 2 kilometers of each of the 201 sites evaluated in the assessment.

Home gardeners are residents who are also exposed to contaminated homegrown fruits and vegetables. We estimated which percentage of the entire population within 2 kilometers of the waste management unit are home gardeners based on national data presented in EPA's *Exposure Factors Handbook (EFH)* (U.S. EPA, 1997d).

Farmers are exposed through inhalation of ambient air, inhalation of shower air, ingestion of groundwater, ingestion of soil, and ingestion of fruits and vegetables. In addition, beef farmers are exposed through ingestion of beef, and dairy farmers are exposed through ingestion of milk. We estimated the numbers and types of farms and farmers within the 2-kilometer area of interest from a combination of the 1990 Census data (U.S. Bureau of the Census, 1990), Geographic Information Retrieval and Analysis System (GIRAS) land use data, and county-level Census agricultural data (U.S. EPA, 1994a). We averaged 1987 and 1992 Census of Agriculture data to approximate 1990 (for consistency with the population census).

A percentage of residents, home gardeners, and farmers are classified as recreational fishers. Recreational fishers have the same exposures as either the resident, the home gardener or the farmer but are also exposed through fish ingestion. The number of recreational fishers was

estimated from the 1990 Census data (U.S. Bureau of the Census, 1990) and state-level information from the U.S. Fish and Wildlife Service National Wildlife Survey (U.S. FWS, 1991).

Infants are assumed to be exposed through mother's contaminated breast milk. For infant exposure through breast milk, the maternal exposure through all pathways was summed. The mother is assumed to be an adult (as opposed to a teenager) for the purpose of calculating maternal dose in the infant breast milk pathway. The current methodology for infant exposure applies only to dioxin and dioxin-like chemicals.

For each of the receptor types, we estimated carcinogenic risks assuming a 9-year exposure duration. Nine years is the median residence duration of the distribution for all ages as reported in the *Exposure Factors Handbook* (U.S. EPA, 1997d). Thus, the average exposure is calculated over every 9-year period in the model simulation. For example, a 100-year model simulation would have 91 9-year averages calculated. Aging of cohorts into subsequent cohort age classes, and their differing exposures, is included. This accounts for children aging from one cohort to the next over a 9-year exposure period.

For each receptor location, human risk is estimated using aggregated pathway exposures, when appropriate, to add exposures across pathways. The aging of a cohort into the subsequent cohort age category(s), and the resulting differences in exposure, is included in this moving average calculation. For noncancer risk calculations, exposure is assumed to vary annually; we did not use a longer averaging period. Therefore, a single high year of maximum exposure would not be "diluted" by a multi-year averaging period. The exposure and risk methodologies are described in the *Background Document for the Human Exposure Module for the HWIR99 3MRA Model* (U.S.EPA, 1999aj) and *Background Document for the Human Risk Module for the HWIR99 3MRA Model* (U.S.EPA, 1999ak).

We estimated exposures for residential receptors (residents and home gardeners) at a single location in each of the census blocks in the 2-km study area, and for farmers at a single farm in each of the census block groups in the 2-km study area. Recreational fisher exposures are calculated and averaged across up to three randomly selected waterbodies over the entire study area. The random selection of waterbodies is made once for recreational fishers who are residential receptors, and once for recreational fishers who are farmers. We assumed that human receptors both reside and work at the receptor location identified for them during site characterization. This assumption may overestimate or underestimate exposure to an unknown degree and bias because it is possible that individuals may reside at the identified location within the study area but commute to work areas outside of the study area or could commute to more highly contaminated areas within the study area.

For each receptor type, we estimated only the incremental exposures, risks, and hazard quotients for a chemical. We did not consider background exposures from natural or other man-made sources. For cancer risks, we assumed that lifetime exposure risks are in direct proportion to the fraction of a lifetime actually exposed (that is, 350 of 365 days per year (15 days away per year) for each year of the exposure duration. We did not consider additive, synergistic, or antagonistic effects among multiple chemicals. In addition, we did not consider age-specific differences in exposure responses; that is, we did not vary cancer slope factors with cohort age.

2.1.2 Ecological Receptors

We defined several ecological assessment endpoints to evaluate, based on the management goal of protecting terrestrial and aquatic ecosystems from HWIR-exempted waste. The assessment endpoints that we chose to evaluate are shown in Table 2-2. These endpoints represent the general trophic levels within a food web and are broad enough to characterize the functionality and trophic level interactions within most habitats. In addition, these assessment endpoints generally capture the significant biota of most habitats.

Our first step in selecting ecological receptors was to identify the habitats that may exist near a site. We collected GIRAS land use maps, National Wetland Inventory maps, and National Wildlife Refuge maps to plot the types of land uses around the sample sites. We then delineated habitats within 2 kilometers of the waste management unit to identify the types of uses around the site. We identified subclasses of terrestrial habitats, aquatic habitats, and wetlands based on the regional location of the site. A detailed description of the subclasses considered is found in *Ecological Exposure Module: Background and Implementation for the HWIR99 Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) Model* (U.S. EPA, 1999an). We then used the habitat description and regional location to identify receptors for each site-based habitat.

The second step in the process was to assign receptors to habitats identified at each site. Based on the ecological assessment endpoints, we sought to capture the range of organisms that may reside in a specific habitat and represent the functions and trophic levels typically present in that habitat. Thus, we modeled a suite of receptors that represent various trophic levels within terrestrial, aquatic, and wetlands habitats. The receptors we evaluated were: soil communities, terrestrial plant communities, mammalian populations, and avian populations for terrestrial habitats; and sediment communities, aquatic plant communities, aquatic communities, amphibian populations, mammalian populations, and avian populations for aquatic habitats. For wetlands, we assigned groups of these aquatic and terrestrial receptors based on the type of wetlands present at a site. In an effort to make the assessment site-based, we used information on the location of the site to identify the receptors that may occupy different functions or trophic levels. The list of receptors by habitat is found in U.S. EPA (1999an). The ecological risk methodologies are described fully in *Ecological Risk Module: Background and Implementation for the 3MRA Model for the HWIR99* (U.S. EPA,1999ao).

We estimated ecological receptor exposures based on simulated contaminant concentrations in the various environmental media and food items, pathway-specific ingestion rates, and receptor type-specific body weights. An inhalation pathway was not considered for ecological receptors. The methodologies and equations used for exposure estimates are fully described in the technical background document (U.S. EPA, 1999an).

Table 2-2. Assessment Endpoints Considered for the HWIR Ecological Assessment

Ecological Significance	Assessment Endpoint	Receptors	Characteristic(s)	Measure of Effect
• Upper trophic level consumers • Top recipients of bioaccumulative chemicals • Represent species with large foraging ranges • Represent species with longer life spans	Viable mammalian wildlife populations	Deer mouse, meadow vole, red fox	Reproductive and developmental success	Chronic or subchronic NOAEL(s) or LOAEL(s) for developmental and reproductive effects
	Viable avian wildlife populations	Red-tailed hawk, northern bobwhite	Reproductive and developmental success	Chronic or subchronic NOAEL(s) or LOAEL(s) for developmental and reproductive effects
• Species represent unique habitat niches (partially aquatic and terrestrial) • Some species are sensitive to contaminant exposure	Viable amphibian and reptile wildlife populations (“herps”)	Frog, newt, snake, turtle	Reproductive and developmental success	Chronic or subchronic NOAEL(s) or LOAEL(s) for developmental and reproductive effects
• Represent base food web in terrestrial systems • Habitat vital to decomposers and soil aerators • Proper soil community function related to nutrient cycling	Sustainable soil community structure and function	Nematodes, soils mites, springtails, annelids, arthropods	Growth, survival, and reproductive success	95% of species below no effects concentration at 50th percentile confidence interval
• Primary producers of energy in ecosystems • Act as food base for herbivores • Able to sequester some contaminants • Can act as vectors to bioaccumulation • Constitute a large fraction of the earth’s biomass	Maintain primary terrestrial producers (plant community)	Soy beans, alfalfa, rye grass	Growth, yield, germination	10th percentile from LOEC data distribution
• Highly exposed receptors from constant contact with contaminated media • Act as vectors to transfer contaminants to terrestrial species	Sustainable aquatic community structure and function	Fish (salmonids), aquatic invertebrates (daphnids)	Growth, survival, reproductive success	Ambient water quality criteria (NAWQC) for aquatic life (95% species protection)
• Provide habitat for reproductive lifestages (eggs, larval forms) • Habitat for key invertebrate species • Act to process nutrients and decompose organic matter	Sustainable benthic community structure and function	Protozoa, flat worms, ostracods	Growth, survival, reproductive success	10th percentile from LOEC data distribution
• Primary producers of energy in the aquatic system. • Base food source in the aquatic system • Can act to sequester contaminants from the water column • Act as substrate for other organisms in the water column (periphyton)	Maintain primary aquatic producers (algal & plant community)	Algae and vascular aquatic plants	Growth, mortality, biomass, root length	EC_{20} for algae; lowest LOEC for aquatic plants

2.2 Input Data

The 3MRA Model requires over 700 input parameters covering a wide range of general data categories including: WMU characteristics; meteorological data, surface water, and watershed characteristics; soil properties; aquifer properties; food chain or food web characteristics; human and ecological exposure factors; types and locations of human and ecological receptors and habitats surrounding the WMU; and chemical-specific properties and toxicity values. The values for specific parameters were either point estimates, distributions, or assumed values. We implemented the assessment on a national scale but based the analysis on a

regional, site-based approach. In this approach, site-based data were used when available as inputs to the model. When site-based data were not available, data collected on a regional level, followed by data collected on a national level, were used for the evaluation. Detailed documentation regarding what data were collected, where the data were obtained, how the data were collected and processed, and issues and uncertainties associated with the data collected for the database of the 3MRA Model is provided in a series of documents (U.S. EPA, 1999d through r). The uncertainty and limitations concerning the data that were collected for the HWIR risk analysis are discussed in Section 4.2 of this document.

We collected a large amount of data to better describe and model plausible exposure scenarios from chemical-specific releases from the waste management units. Examples of the types of data collected to identify site-based characteristics included facility locations and the physical and environmental characteristics of the sites and surrounding areas (e.g., land use, human receptor locations, and ecological habitats). Examples of regional data we collected were: meteorological data, soils characteristics, aquifer data, and types of ecological receptors. Data collected and available at the national level included human exposure factors, ecological exposure factors, human health toxicity values, and ecological toxicity values.

We used measured, calculated, and estimated chemical-specific data to generate all relevant chemical-specific thermodynamic and kinetic data for the HWIR assessment. The lack of reliable measured thermodynamic data necessitated the use of data generated by computational methods. The SPARC (System Performs Automated Reasoning in Chemistry) model, which is a computational method based on fundamental chemical structure theory, was the primary tool for calculating the thermodynamic constants. The process of assembling kinetic constants for degradation pathways (hydrolysis, anaerobic biodegradation, and aerobic biodegradation) focused on finding, evaluating, and summarizing measured data. Due to the complex nature of biodegradation processes, only measured kinetic constants for a select group of high-volume chemicals were used in the HWIR chemical database. We grouped these kinetic data according to reaction conditions (i.e., pH, temperature, and redox conditions) (U.S. EPA, 1999ai).

We used several types of human health toxicity values to describe the toxicological dose-responses for the chemicals evaluated. For human health effects, the toxicity values included: cancer slope factors (CSFs) in units of $(mg/kg\text{-}d)^{-1}$ for oral exposure to carcinogenic chemicals, reference doses (RfDs) in units of mg/kg-d for oral exposure to noncarcinogenic chemicals, inhalation CSFs derived from unit risk factors (URFs) in units of $(mg/kg\text{-}d)^{-1}$ for inhalation exposure to carcinogenic chemicals, and reference concentrations (RfCs) in units of mg/m^3 for inhalation exposure to noncarcinogenic chemicals. Some constituents may have only toxicity value such as an oral CSF; others may have both oral and inhalation values for cancer or noncancer; others may have a full set of oral and inhalation toxicity values for both carcinogenic and non-carcinogenic endpoints.

There are a number of sources available for toxicity values that attempt to determine the most sensitive health effects associated with the constituents and express the relationship between dose and effect quantitatively. We established an order of preference for the sources of health toxicity values as follows (from most preferred to least preferred): (1) the Integrated Risk Information System (IRIS) online database of verified health benchmarks (U.S. EPA 1998g);

(2) the Health Effects Assessment Summary Tables (HEAST; U.S. EPA 1997e); and (3) EPA's National Center for Environmental Assessment (NCEA) provisional values.

The data used to develop the ecological benchmarks were gathered from peer-reviewed literature and EPA-developed criteria (e.g., Ambient Water Quality Criteria). The data sources for the ecological benchmarks developed for each of the chemicals are available in technical background document (U.S. EPA, 1999p).

We developed two types of ecological toxicity values for this analysis. The first values are population-level values and are expressed as an applied dose in milligrams per kilogram per day. The ecological benchmarks are relevant to mammals, birds, amphibians, and reptiles. The second set of toxicity values are chemical stressor concentration limits (CSCLs) that are expressed in media concentrations (e.g., mg/L). These are community-level benchmarks that are relevant for terrestrial and aquatic plants, aquatic organisms, benthos, and soil organisms. The methodology for the development of these benchmarks is described in *Data Requirements and Confidence Levels for the Development of Ecological Toxicity Values to Support HWIR Exemption Levels* (U.S. EPA, 1999as).

2.3 The 3MRA Model

The 3MRA Model automates the assessment strategy. The model consists of 18 media-specific pollutant fate, transport, exposure, and risk modules; 6 data processors to manage the information transfer within the system; and 3 databases that contain the data required to estimate risk.

Unlike previous HWIR risk assessment efforts (57 FR 21450 and 60 FR 66344), which considered groundwater and nongroundwater pathways separately, the 3MRA Model evaluates simultaneous exposures across multiple media and pathways to estimate the resulting health and environmental effects. For example, instead of looking at the risks of a person drinking contaminated groundwater, breathing contaminated air, and eating contaminated food separately and at potentially different points in time, we estimated the risk from the simultaneous exposure from multiple pathways, where appropriate, across time.

The 3MRA Model incorporates the following interacting modules:

- Source modules, which estimate the simultaneous chemical mass losses to the different media and maintain chemical mass balance of the releases from the waste management unit into the environment

- Fate/transport modules, which receive calculated releases from waste management units and distribute the mass through each of the media to determine the chemical concentrations in air, groundwater, soil, and surface water across space and time

- Food chain modules, which receive the outputs from the fate and transport modules and estimate the uptake of chemicals in various plants and animals

- Exposure modules, which use the media concentrations from the fate and transport modules to determine exposure to human and ecological receptors from inhalation (for humans only), direct contact (for ecological receptors only), and ingestion (for both receptor types)

- Risk module, which predicts the risk/hazard quotient for each receptor of concern.

Each of these modules is discussed in detail in Section 4.0 of this document and the corresponding background documents.

2.4 Risk Protection Criteria

The HWIR assessment strategy uses five different risk protection criteria to generate exemption levels: (1) risk level, (2) human health hazard quotient (HQ), (3) ecological hazard quotient, (4) population percentile, and (5) probability of protection. By setting a value for each of these criteria, we identified the chemical-specific waste concentrations that are protective at those values. Each of these risk criteria is explained in more detail below.

2.4.1 Risk Level

Risk level refers to a person's increased chance of developing cancer over a lifetime due to potential exposure to a specific chemical. A risk of $1x10^{-6}$ translates as an increased chance of one in a million of developing cancer during a lifetime. EPA generally sets regulations at risk levels between 10^{-6} and 10^{-4} (in other words, from one in a million to one in ten thousand increased chance of developing cancer during a lifetime). In the RCRA hazardous waste listing program, a 10^{-6} risk is usually the presumptive "no list" level, while 10^{-5} is often the presumptive list level, and was the level chosen in setting the toxicity characteristic (TC). For HWIR, we evaluated the exemption levels that result from both 10^{-6} and 10^{-5} risk levels.

2.4.2 Hazard Quotient (HQ)

The HQ refers to the likelihood that exposure to a specific chemical would result in a noncancer health problem (e.g., neurological effects). The hazard quotient is developed by dividing the estimated exposure to a chemical by the RfD or RfC (the highest dose or concentration that might be considered safe). An HQ of 1 or lower indicates that the given exposure is unlikely to result in adverse health effects. Some programs, such as the drinking water program, set the HQ target at less than 1 to provide a safety factor against exposure to a chemical from other sources. For example, we have used 20% of the RfD in setting drinking water standards (see, for example, 57 FR 31776) and 25% of the RfD in setting standards for Boilers and Industrial Furnaces (BIFs) (56 FR 7134). For HWIR, we evaluated the exemption levels that result from both an HQ of 0.1 and an HQ of 1.

2.4.3 Ecological Hazard Quotient

The ecological hazard quotient is analogous to the human health HQ, except that the estimated exposure is compared with an ecological toxicity value rather than the human health

RfD or RfC. For this analysis, we developed two types of toxicity values: (1) an ecological benchmark that is analogous to the human health HQ using an RfD; and (2) a chemical stressor concentration limit that is analogous to the human health HQ using an RfC. In developing ecological benchmarks for this risk assessment, we used the geometric mean between a no observed effects level (NOEL) and a lowest observed effects level (LOEL). (Human health reference doses are based on NOELs.) The ecological hazard quotient protects ecological health at the population or community level and, therefore, focuses on reproductive and developmental effects rather than the mortality of individual organisms. This approach is similar to the approach used for developing Ambient Water Quality Criteria, where the assumption is that most, but not all, of the aquatic species and animals are protected (Stephan et al., 1985) . For HWIR, we evaluated the exemption levels that result from ecological hazard quotients of both 1 and 10.

2.4.4 Population Percentile

The population percentile is the percentage of the population protected at the specified risk levels and hazard quotients for a single environmental setting. A setting is a specific unit at a specific site and is defined by combining site-based information (such as unit size and unit placement) with variable environmental information (such as rainfall and exposure rates) generated from regional and national data. For HWIR, we evaluated the exemption levels that result from population protection percentiles of 99 percent and 95 percent. A small fraction of the population is unprotected, but that does not mean that these individuals have extremely high risk or hazard quotients. As we conduct analyses in the future, we will take a close look at those who are left unprotected.

2.4.5 Probability of Protection

The probability of protection is defined as the percentage of settings (i.e., 419 WMU/site location combinations) that meet the population percentile criteria. These distributions reflect the uncertainty and the variability of the model and underlying data required by the model. We generally describe a probability of protection as "high end" when it focuses on individual risk to those people at the upper end of the distribution, generally above the 90th percentile. For HWIR, we evaluated the exemption levels that result from both 95 percent and 90 percent probabilities of protection.

By evaluating different values for each risk protection criteria, we generated potential HWIR exemption levels for four different risk protection groups (see Table 2-3) . The risk protection groups are two-dimensional in nature. With respect to the Group 2 criteria, for example, the interpretations for cancer and noncancer risks, respectively, are:

- 99 percent of the population are subject to cancer risks of less than 10^{-6} across 90 percent of the environmental settings
- 99 percent of the population experience exposure levels below the HQ of 1 across 90 percent of the environmental settings.

The combinations in Table 2-3 capture a range of protection levels, from most protective (Group 1) to least protective (Group 4). These groups are not an exhaustive look at all possible combinations of potential risk protection criteria; we could choose a different combination altogether. These groups were chosen to help bound the possible values.

We grouped the unit-specific results to construct HWIR exemption levels for each waste category as shown in Table 2-4. As Table 2-4 suggests, HWIR exemption levels for liquids are derived from releases evaluated at surface impoundments and tanks. Exemption levels for semisolids are based on releases evaluated at surface impoundments, tanks, and land application units. Solids use risk-based numbers based on the releases evaluated at waste piles and landfills.

Table 2-3. Risk Protection Combinations Evaluated for HWIR Risk Assessment

	Group 1 (most protective)	**Group 2**	**Group 3**	**Group 4 (least protective)**
Risk level	10^{-6}	10^{-6}	10^{-5}	10^{-5}
Human health HQ	0.1	1	1	1
Ecological HQ	1	1	1	10
Population percentile	99	99	99	95
Probability of protection	95	90	90	90

Table 2-4. HWIR Exemption Level Categories

	Liquids (TSS < 1percent) (mg/L)	**Semisolids (1percent≤TSS≤30percent) (mg/kg)**	**Solids (TSS > 30percent) (mg/kg)**
Surface impoundment	Evaluated	Evaluated	
Tank	Evaluated	Evaluated	
Land application unit		Evaluated	
Waste pile			Evaluated
Landfill			Evaluated

The exemption levels for each waste form were determined for each waste management unit by selecting the lowest (most stringent) chemical concentration from the units evaluated. For example, the liquid exemption level is based on the lower of the surface impoundment and

tank results. In developing the semisolid numbers, we converted the surface impoundment and tank results, which are in milligrams per liter, to milligrams per kilogram based on an assumed density of 1 kg/L (the density of water).

These categories of waste forms group the wastes that are expected to be managed in similar ways. Realistically, some waste forms will not be managed in certain management units. For example, it is unlikely that a true solid would be managed in a tank system or that a true liquid would be managed in a landfill. The liquid and solid definitions distinguish wastes that are clearly liquid and clearly solid from the rest of the waste universe. Creating separate exemption levels for these two waste forms should not affect the protectiveness of the exemption and may allow for more appropriate exemption levels and greater regulatory relief.

3.0 Characterization of Potential Risk

The results of this analysis are chemical-specific distributions of cancer risks or hazards to human health and ecological receptors living in the vicinity of industrial waste sites that may manage HWIR-exempted wastes. At this time, only chemical-specific results are available for acrylonitrile managed in industrial sub-title D landfills. In addition, results are not available for ecological receptors exposed to acrylonitrile because of the lack of ecological toxicity values. Additional chemicals and waste management unit combinations will become available after updates are made to the 3MRA model to correct certain errors apparent in the current version.

For each site and waste concentration, the model generated risks for each receptor location and then summed the number of receptors that fall within a specified risk range (bin) to get the distribution of risks for the population at each site. We used the distribution of risks at a site to determine whether a site was protected based on the percentage of the population protected, a specified cancer risk or hazard level, and the initial concentration in waste as described in Section 2.4. The model then used these data to generate a percentile distribution based on the number of sites protected at a specified risk level for each waste concentration to generate the national distribution. In addition, we generated results that allow the flexibility to query the results based on several risk descriptors. The risk descriptors for the human health risk and ecological risk are discussed below.

For the human health assessment, the model calculated the aggregate risk or hazard from multiple exposure pathways that occur simultaneously at the receptor location to generate the distribution of individual risks. For carcinogenic effects, we chose seven risk bins ranging from less than $1x10^{-8}$ to greater than $1x10^{-4}$ to generate the distribution. For human health hazard quotients, we chose four hazard bins ranging from less than 0.1 to greater than 10. We generated results for three distance rings, including within 500 meters, within 1,000 meters, and within 2,000 meters. Results are available for 12 exposure pathways, including total ingestion and inhalation combined, total ingestion, total inhalation, total groundwater ingestion and shower inhalation combined, air inhalation, shower inhalation, groundwater ingestion, soil ingestion, fruit and vegetable ingestion, beef ingestion, dairy ingestion, and fish ingestion. In addition, the results are available for five receptor types: all receptors, residents, gardeners, farmers, and fishers. Finally, the results can be queried by three age cohorts: all ages, children 11 and under, and children and adults 12 and over.

For the ecological assessment, the model will calculate impacts to ecological receptors using the same general methodology, but we evaluated impacts to populations or communities of ecological receptors rather than to individuals. For each site, the model generates a distribution of hazard quotients by receptor and sorted the receptors into one of four hazard bins, ranging from less than 0.1 to greater than 10. The model will use the receptor results to evaluate impacts

to several attributes of habitats, including three habitat groups (terrestrial, aquatic, and wetlands), 11 habitat types (e.g., forest, lake, river), nine receptor groups (e.g., mammals, aquatic biota, terrestrial plants), and five trophic levels (e.g., producers, top predators). The model generates results for each of the attributes by three distance categories: within 1,000 meters, between 1,000 and 2,000 meters, and within 2,000 meters. In addition, the model will generate results for the evaluation of some combinations of these attributes, including impacts by habitat group and trophic level and by habitat group and receptor group.

We have not completed final internal and independent testing of the software system, nor have we completed peer review of the system, modules, and data. Therefore, the use and interpretation of the results is limited at this time. The results should be viewed as representing the capabilities of the model with respect to the types of information that the model can produce. The numbers are likely to change after additional diagnostic and final testing of the software system.

3.1 Results

We are presenting the results for acrylonitrile managed in industrial Subtitle D landfills in order to show the outputs of the model and the various ways the output data can be presented. Results for this chemical is shown in the Appendix. The format of the tables used to present these results is shown below. (Because results are available only for acrylonitrile managed in landfills, we do not present values for each of the template tables.)

As described in Section 2.4, we have defined four levels of protection, Group I as most protective to Group 4 as least protective. Table 3-1 provides a template for Chemical Concentration (C_w) for each waste type and for human health effects and eco effects. This table will be available for each level of protection and for each distance evaluated (i.e., 500_M, 1000_M, and 2000_M).

Table 3-1. Chemical Concentrations (C_W) for Risk Group 1

Chemical Name	Liquid (ppm)			Semi-solid (ppm)			Solid (ppm)		
	HH	Eco	Lowest (HH, Eco)	HH	Eco	Lowest (HH, Eco)	HH	Eco	Lowest (HH, Eco)

Liquid = Lowest C_w Between Surface Impoundment and Aerated Tank
Semi-Solid = Lowest C_w Between LAU, Surface Impoundment, and Aerated Tank
Solid - Lowest C_w Between Landfill and Wastepile

Tables 3-2 and 3-3 provide templates for risk or hazard to human subpopulation and by exposure pathway, respectively, for the human health based C_ws shown in Table 3-1. Table 3-2 can be generated for each protectiveness group, each distance, and each waste type. Table 3-3 can be generated for each receptor type, as well as each protectiveness group, distance, and waste type. These tables for acrylonitrile are presented in the Appendix.

Table 3-2. Risk Group 1 - Risk to Subpopulations at Human Health Landfill Chemical Concentrations (C_w)

Chemical Name	Infants	1-12 years	12+ years	Farmer	Gardener	Fisher	Resident

Table 3-3. Risk Group 1 - Risk by Exposure Pathway at Human Health Landfill Chemical Concentrations (C_w)

Chemical Name	Inhalation Air	Shower Air	Ingestion Ground water	Ingestion Soil	Ingestion Beef	Ingestion Milk	Ingestion Fish	Ingestion Fruits and Vegetables

3.2 Wells Located in the Plume

Under this site-based approach, the distributions of risks or hazards for acrylonitrile presented in the Appendix include all of the receptors that are exposed through one or more exposure pathways as well as any receptors not exposed. For example, the distributions present the risk and hazard estimated for all receptors using groundwater at a site for drinking or showering. This includes receptors using groundwater from both wells located within the contaminated plume and the receptors outside of the plume. The receptors located outside the contaminated plume have no risk or hazard through the groundwater pathway.

We have also designed the model to have the capability to estimate risk and hazard to only those receptors that are exposed to a chemical through one or more pathways. With respect to receptors using groundwater for drinking or showering, if we were to implement this capability, the distributions would reflect only the risk and hazard to the receptors located within

the groundwater plume. The receptors using groundwater as a source of drinking or showering and located outside of the plume would not be included in the distribution of risk and hazard in this additional analysis.

The number of wells within the ground water plume will vary significantly by site, by chemical, and by waste management unit. Some chemical and waste management combinational had no wells within the groundwater plume.

The extent of a plume depends on the concentration and mass of a chemical constituent in the waste management unit, physical and chemical properties of the waste, characteristics of the waste management unit, site hydrogeological characteristics and the site climate. Because these are variable factors, the plume of a contaminant is not constant. We estimated the number of wells inside a contaminant plume for acrylonitrile at a site by first estimating the extent of the plume at that site. The plume extent is characterized by approximate stream surfaces that separate the fluid emanating from the waste management unit and the ambient ground-water flow field, and the transverse dispersion normal to the stream surfaces.

For a given distance from the source (or the waste management unit), the lateral extent of the plume is defined as a cross-section normal to the flow field where the receptor well concentration has the probability of more than 99.74 percent of being greater than 0.001 of the maximum concentration at the center of the plume at that longitudinal distance from the waste management unit. We estimated the extend of the plume based on the assumption that the groundwater flow field is steady-state. The details of the derivation of the plume's extent are described in Appendix D of the background document for the vadose zone and aquifer modules (U.S. EPA, 1999aa). Table 3-4 shows the estimates for the number of wells within the 2-km radius of the WMU (i.e., landfill) across all sites and the percent of these wells inside and outside the plumes of contamination. We show the results of acrylonitrile to illustrate the data generated by the 3MRA model.

Table 3-4. Percent of Wells Inside and Outside the Plumes of Contamination Across all Sites in the Vicinity of Industrial Subtitle D Landfills.

	Landfill		
	Total	**Percent In**	**Percent Out**
Acrylonitrile	3,130	20.2	79.8

3.3 System Testing Capabilities

Although extensive testing and debugging has significantly increased the percentage of chemical/site/WMU combinations that have been successfully executed by 3MRA, there remain some combinations that are still problematic. For the set of results for acrylonitrile presented in the Appendix greater than 95 percent of the environmental settings ran successfully.

Some errors have continued to occur for a few settings and chemical combinations in recent versions of the 3MRA system. The 3MRA system has been designed to identify errors and inconsistencies. The following system checks have been incorporated into the 3MRA model.

Range checks. As each 3MRA module reads its input variables (parameters and outputs of other modules), the values of those variables are checked against realistic upper and lower bounds. For example, concentrations must be nonnegative. If a value exceeds these bounds, an error is generated. Examples of such errors and the associated input variables that have occurred in recent model runs are

- Surface Water–total suspended solids concentrations (WBNTSSWater), associated number of years in the time series (WBNTSSWaterNy), and year associated with total chemical concentration in the water column (WBNConcWaterTotYr).
- Chemical Properties Processor–hydrolysis rate constant (ChemHydRate)
- Watershed–chemical load in runoff/erosion to waterbody (SWLoadChemR)
- Saturated Zone–chemical flux passing to surface water (AquRchMassFlux)
- Landfill–chemical flux in leachate (LeachFlux).

Checks for missing parameter or input values. A similar error is generated when a module tries to read a parameter value or modeled input and no value exists. Examples of these errors and the related variables that have occurred in recent model runs are

- Aquatic Food Web–number of chemicals simulated (WBNNumChem)
- Surface Water–upstream chemical concentration (C_upstream).

Solubility Limit Checks. None of the 3MRA modules are designed to simulate chemical fate and transport at concentrations that violate solubility limits. Any violation of this limitation is related to the input chemical concentration and the extent to which the chemical can concentrate in media over time.

Checks for missing Input Files. Each 3MRA module reads its module-specific input parameters from a separate input file (a “ssf” file) created by the Site Simulation Processor from the several input databases. If that file is missing, the module generates an error.

Other checks. Finally, when a module generates an error, an error message is typically written in accordance with an internal “error trap,” thus documenting the nature of the error, for example, Exit Level Processor “missing” sites (counter error). Nonetheless, some undocumented errors remain for which no error traps exist and which will require diagnosis to determine the specific cause. Such errors are relatively few.

4.0 Uncertainty and Limitations

4.1 Scenario Uncertainty

The term "scenario" is used to define a number of different elements in risk assessment, including (but not limited to) the waste management scenario, the exposure scenario, and the risk protection scenario. Taken together, these elements make up the conceptual framework for the analysis; that is, they define what scenarios are evaluated in the risk modeling efforts. Elements that are outside of this definition are excluded from the conceptual framework. For example, if combustion is not considered as a source (which it is not in this analysis), then the final results necessarily do not include risks resulting from combustion. None of the individual modules constituting the larger system are at fault; any limitations/uncertainties presented are simply a limitation of the overall conceptual risk model's architecture. In this context, the scenario describes the what, where, who, when, and how of the analysis. Scenario uncertainty is used here to denote the uncertainty in chemical-specific exit levels that results from the assumptions that are implicit in the overall conceptual risk model. For the purposes of this discussion, it is useful to think of these elements in terms of the following basic questions:

- What types of waste management units are included?
- What is the spatial scale of interest?
- Where are the sites that are modeled?
- Who are the receptors potentially exposed?
- When is exposure assumed to occur (i.e., temporal scale)?
- How does exposure occur (i.e., exposure pathways)?
- How do constituents affect the receptors (e.g., cancer endpoints)?
- How are risks estimated and characterized (e.g., risk metrics)?

The underlying assumptions and limitations/uncertainties relevant to each of these questions are discussed below. In general, the discussions are pertinent to exit criteria generated for both human and ecological receptors. However, some assumptions and limitations/uncertainties are unique to the ecological risk assessment. In these instances, we have noted that the text applies only to ecological risks.

4.1.1 Waste Management Units

Five types of waste management units (WMUs) are considered in the HWIR analysis: surface impoundment, aerated tank, land application unit (LAU), wastepile, and landfill. The HWIR exemption levels are derived for each of the WMU types.

Assumptions

- WMUs are assumed to have limited management controls.
- The geometry of WMUs is assumed to be square.
- Only one WMU is assumed to be present at a setting.
- Inclusion of other WMU types would not change the analysis.

Limitations/Uncertainties

Limited Management Controls. The source modules assume only limited controls for mitigation of environmental releases. For example, the Landfill Module assumes no daily cover, but only a permanent cover once a landfill cell is filled (after 1 year). The permanent cover is assumed to be a soil-like material that is permeable, which allows continual contaminant loss and infiltration. In addition, no liner or leachate collection system is assumed to exist. To the extent that facilities managing exempted wastes will have more rigorous controls than these minimum controls assumed, the risks posed by these facilities will have been overestimated by an unknown amount.

WMU Geometry. The actual geometric configuration of a WMU can significantly alter the spatial distribution of contamination at a given site and affect the risks relative to the assumed, square configuration. For example, a rectangular LAU oriented with its long dimension parallel to the prevailing wind will have a longer fetch from which wind erosion can occur than would a square LAU, with higher associated particulate emissions. The converse, of course, is also true, so this assumption presents uncertainty but no particular bias.

Single WMU. The assumption of a single WMU when multiple WMUs are present will result in an under- or overestimation in contaminant amount, distribution, and risk. When multiple WMUs of the same type were present, a single WMU sized with the average area of the multiple WMUs was assumed. This assumption may under- or overestimate the total releases within the area of interest (AOI). If all multiple WMUs, in fact, exist within the AOI, the assumption will underestimate the releases by approximately the ratio of the total area to the average area. This underestimate may result in underprediction of exposure and risk. Conversely, if the multiple AOIs are in fact distributed over a larger area than the AOI, the placement of a single WMU of average area within the AOI may overestimate the release actually occurring within the AOI with overprediction of exposure and risk. For multiple WMUs of different types, each WMU is simulated, but in a separate simulation under the assumption that it exists alone at the site. This introduces a bias toward underestimation of unknown magnitude of cumulative releases, exposures, and risks within the AOI.

4.1.2 Spatial Scales (Area of Interest)

The geometry of each WMU is assumed to be square. The spatial scale of the total AOI to be simulated is defined as a 2-km radius from a circle that circumscribes the WMU square. Because the size of the units varies, the total AOI at a given site is determined by the WMU type and size of the WMU at that site. Thus, for each site, the spatial scale is a function of the WMU type and size.

Assumptions

- The 2-km radius is sufficient to capture all significant exposures.
- The appropriate unit for evaluating ecological risk is at the level of a habitat, as defined by geography and land use boundaries.

Limitations/Uncertainties

Area of Interest Defined by 2-km Radius. The 2-km assumption could create biased results in either direction. We conducted a sensitivity analysis for air concentration and deposition for each of these land-based WMUs. The 2-km AOI was based on data showing a steep decline in air concentration and deposition rates within the first kilometer of the WMU boundary. If significant exposure occurs beyond the 2-km distance, then the resulting risk distributions will obviously not include the full complement of receptors at risk. Conversely, if significant exposure does not extend to the full 2-km distance, then "spatial dilution" of risk could occur; that is, the risk distribution percentiles will be affected by inclusion of receptors that have essentially no exposure or risk, thus driving those that are into lower percentiles of the total receptor population.

Spatial Resolution for Ecological Risk. Defining the spatial character of the ecological assessment in terms of the habitat simplifies the analysis in that apportionment of animals across multiple habitats is not required. It also increases the resolution possible at a site in that (1) exposure concentrations need not be averaged over the entire area of interest and (2) the spatial connection between predator and prey can be maintained. As an added benefit, issues concerning the carrying capacity of a given study area do not need to be addressed, that is, it is not necessary to determine how many populations may exist at a given site. One limitation of this approach is that many wildlife species tend to forage across different types of habitats (e.g., stream, forest) and do not recognize the artificial boundaries imposed by land use patterns. Another limitation is that linkages between different habitats and receptors cannot be evaluated; each habitat is essentially a spatial unit that is independent from other habitats. Hence, there is uncertainty in conceptualizing the spatial character for the ecological risk assessment in terms of habitats.

4.1.3 Site Locations

To represent facilities across the nation likely to receive exempted wastes, the HWIR 3MRA assessed the potential human health and ecological impacts at 201 individual nonhazardous industrial waste management sites. EPA randomly selected the 201 sites from about 2,850 establishments with onsite waste management included in EPA's *Screening Survey of Industrial Subtitle D Establishments* (Westat, 1987). EPA designed and implemented this survey as a representative sample of a total population of nearly 150,000 industrial facilities, 12,000 of which were estimated by the survey to have onsite waste management units.[1] The

[1] Onsite WMUs addressed by the survey included landfills, wastepiles, land application units, and surface impoundments. Other WMU types were not addressed.

survey was stratified first across 17 industry groups selected by EPA to represent the population of facilities managing nonhazardous industrial wastes and then by facility size (small, medium, and large), as represented by employment statistics. Because small and medium facilities were sampled, the survey results include weights attached to each facility. Large facilities have a weight of 1 because the survey contacted all large facilities. Additional information on the Industrial D survey design and results may be found in Westat (1987), Clickner and Craig (1987), and Clickner (1988).

The size of the sample for the HWIR assessment was set at 200 and was not determined because of statistical criteria, but rather for logistical and resource considerations associated with model run time and the data collection effort. Because it was important to represent each of the 17 industry groups, EPA randomly selected the sites independently from each industry group, with the proportion drawn from each industry set at its proportion in the original survey. Because sample size was rounded for each sector, the total sample size was 201 rather than 200. The small sample size did not allow stratification by establishment size. Additional details on methodology for the selection of the 201 sites are provided in U.S. EPA (1999s).

As mentioned above, survey data for the small and medium-sized establishments included sample weights that ranged up to 10 for medium facilities and above 25 for a few small facilities. The goal for the HWIR 3MRA is to estimate the potential risk to human and ecological receptors living in the vicinity of industrial waste sites that could manage HWIR-exempted waste. Although the weights could have been used as defined, EPA did not consider these weights in the HWIR risk assessment. EPA was concerned that large weights for some of the sites could overly influence the results (U.S. EPA, 1999s).

Assumptions. Several assumptions are inherent in the selection of sites to be modeled in the HWIR 3MRA.

- The snapshot of Industrial D facilities in the 1985 Industrial D Screening Survey is representative of present and future locations for WMUs likely to receive exempted wastes.

- The sample population of 201 sites adequately represents the Industrial D survey sites in terms of industries likely to handle exempted wastes, WMU characteristics, population characteristics, and environmental conditions (e.g., meteorological data, hydrology, ecological habitats).

- Ignoring the information provided by the sample weights and not stratifying by establishment size does not significantly bias the sample in terms of human or ecological risks.

Limitations and Uncertainties. There are several known concerns that accompany use of the Industrial D survey data for estimating national risk. Most of these uncertainties center around whether the 201-site sample is representative of the locations of current and future exempted facilities that could manage exempted wastes.

- The national database was developed from a 1985 industry list and may not reflect current and future locations. However, in developing the best locations for each facility, 181 of the 201 facilities were matched to establishments with EPAIDs in EPA's Envirofacts database. This suggests that many of the facilities are still in existence. However, the assessment assumes the locations remain static over the modeling time frame and does not consider additional or fewer locations over time. In addition, the facility location effort (described in U.S. EPA, 1999d) reviewed the locations of practically all of the 201 facilities to ensure that they were reasonably located in terms of land use, population data, and the most current EPA locations available for each site.

- One type of WMU, aerated tanks, was not considered in the Subtitle D Survey, but was required to be evaluated in HWIR99. Simulated results for aerated tanks are based on the assumption that any site that has a surface impoundment handles liquid wastes and could have an aerated tank as well.

- Some of the 17 industrial sectors addressed by the Industrial D survey may be unlikely to generate or receive exempted hazardous waste, but were analyzed as if they do.

- Commercial industrial waste management facilities are not part of the sampling frame and the population at risk near these facilities is not evaluated. It is not known how significant these uncertainties are in terms of risks to receptors.

- Because the 201-site sample size was not statistically based, there is uncertainty with respect to its representativeness of the location and WMU characteristics of sites in the Industrial D survey. However, initial analysis suggests that the sample does adequately represent both the geographic locations (i.e., the majority of the site locations are found in the Northeast, Southeast, and Mid-Atlantic regions of the country) and the WMU types and characteristics in the overall survey (U.S. EPA, 1999s).

- The geographic representativeness, in terms of environmental conditions that affect the human and ecological risk, is uncertain as the original survey was not designed from the risk assessment perspective. However, the sites do appear to be well spread across the country, with more than 80 percent of Bailey's ecological provinces being represented by the site locations. EPA believes that these locations are adequate in terms of spanning, nationally, human population and environmental conditions associated with industrial waste management. However, a comprehensive, quantitative assessment of this uncertainty has not been conducted.

The Industrial D survey, although over 10 years old, represents the largest consistent set of data available on facility locations and nonhazardous WMU dimensions. In relying on these data to represent potential facilities that may manage and dispose HWIR-exempt waste, EPA recognizes the limitations listed above, including the likelihood that at some of the 201 facilities

may have had WMU additions or closures, or may no longer exist as Industrial D waste disposal facilities. However, EPA considers using these locations and the associated WMU, land use, and population data preferable to developing and evaluating hypothetical exposure scenarios.

4.1.4 Receptors

4.1.4.1 <u>Human Receptors</u>. Three "core" human receptors were selected to evaluate human health risks: resident, home gardener, and farmer. The core receptors include both children and adults. However, no subsistence behavior (i.e., subsistence farmer, subsistence fisher) is modeled as a separate receptor type. Each of these core receptors may also be designated as a fisher; consequently, the receptor categories should not be regarded as mutually exclusive. For example, a receptor could be a resident who raises fruits/vegetables in a home garden and also fishes in freshwater habitats. However, in developing population risk estimates, the receptor categories are mutually exclusive so as to preclude double counting of people.

Assumptions

- The set of receptors defined for the analysis includes those behaviors that place humans at most exposure and risk.
- The behaviors that define exposures are independent for each receptor and across receptors. For example, it is assumed that there is no correlation between the amount of beef and vegetables that the farmer eats. Similarly, a fisher at the high end of the distribution for fish intake may also be at the high end for beef intake.
- Current estimates of receptor populations are sufficient for the analysis.

Limitations/Uncertainties

Snapshot of Population. The 1990 Census data were used to identify numbers of human receptors and their distribution within the AOI. Exposure and risk calculations using this fixed population base were then generated for, in some cases, thousands of years. The assessment do not account for future population increases and/or spatial redistributions occur (through land use changes, for example). However, because risk results are presented as population percentiles, not absolute numbers, the results will be biased only to the extent that differential population increases or changes in spatial distributions will occur among the 201 sites. If all sites change equally, the overall statistics would not be affected.

4.1.4.2 <u>Ecological Receptors</u>. The suite of ecological receptors selected for the HWIR analysis includes wildlife species found in terrestrial, freshwater aquatic, and wetlands habitats. These receptors include mammals, birds, amphibians, reptiles, terrestrial plants, soil fauna, aquatic plants, algae, benthic fauna, and aquatic life. Taken together, these receptors are intended to capture the key structural and functional elements of "healthy" ecological habitats (e.g., primary producers, decomposers, herbivores, top predators).

Assumptions

- It is possible to produce meaningful estimates of ecological risk by estimating risks to a suite of ecological receptors, including species, population, and communities.

- Only receptors in freshwater aquatic habitats were considered; estuarine and marine systems were not included.

- The suite of ecological receptors characterized for this analysis (e.g., mammals, birds, soil fauna) is adequate to address variability in feeding behavior and other receptor attributes that determine exposure.

- Receptors associated with the representative habitats are assumed to be present in those habitats; that is, all sites are assumed to be of sufficient quality to support wildlife.

Limitations/Uncertainties

Ecological Risk Can Be Equated with Risks to Receptors. Ecosystems are enormously complex, and our understanding of even simple community dynamics is limited. In addition, data on chemical stressors are available almost exclusively for individual organisms; that is, they consider adverse effects on a single organism rather than on an assemblage of organisms. Therefore, we select ecological receptors that are the focus of the risk assessment. In using these receptors to predict ecological risks, there is great uncertainty in assuming that we can predict all significant risks to a given area (made up of various habitats) by evaluating exposures and risks to individual receptors.

Exclusion of Estuarine and Marine Systems. Because the model construct developed to perform the multimedia modeling simulation was not designed to handle the complex environmental chemistry of salt and brackish waters, these habitats and the receptors associated with them have not been addressed. As a result, it is not known whether the exit criteria developed for freshwater habitats will be sufficiently protective of estuarine and marine systems. In short, the risk estimates to freshwater systems do not necessarily provide the basis to consider risks to other aquatic systems. Hence, this limitation introduces an unknown degree of uncertainty.

Representativeness of Ecological Receptors. Assuming that receptor risks are a useful tool in estimating ecological risks, there is considerable uncertainty in using a simple suite of receptors to cover all possible exposures. In selecting receptors for each habitat, it is important to acknowledge the uncertainty in assuming that all critical exposure pathways and receptors are represented when, in fact, the risk estimates are only relevant to the types of receptors for which we have data (e.g., body weight, dietary preferences).

Assumption of Habitat Quality. There is considerable uncertainty in assuming that all representative habitats can sustain the ecological receptors assigned to them. The habitats are

delineated using GIS coverages of land uses and the geography of the site (e.g., the presence of a pond). All surface waters are presumed to support an aquatic community and associated wildlife, and all terrestrial areas designated as habitat are presumed to support a variety of wildlife. The presence or absence of receptors at these sites has not been verified and, as a result, the risk estimates are predicated on the assumption of habitat quality and would tend to overestimate risk.

4.1.5 Temporal Scale

Exposure may occur during any part of the model simulation, which, depending on the constituent of interest, may extend from 30 to 10,000 years. The source modules simulate release of a chemical until the concentration in the WMU is 1 percent of the maximum or until 200 years has been simulated. The media concentrations are modeled annually until the chemical concentration in a particular medium (e.g., ground water) decreases to 1 percent of the maximum for that medium or until 10,000 years has been simulated. There are no changes in land use or the location or density of receptors during the modeling period. The exposure duration for humans may occur during any continuous period in the simulation for 1 year (for noncarcinogens) to 9 years (for carcinogens).

Assumptions

- It is appropriate to simulate transport of constituents for possibly thousands of years by using current meteorological and hydrological conditions.
- The duration of exposure for carcinogens is fixed at 9 years.
- The duration of exposure for noncarcinogens is fixed at 1 year.
- For ecological exposures, the duration for exposure to all receptors is fixed at 1 year.

Limitations/Uncertainties

Simulation Duration. Any simulation that attempts to forecast conditions into the future based on current conditions is highly uncertain. The uncertainty increases with the forecast horizon.

Exposure Duration. A 9-year exposure duration for carcinogens was used to reflect the current average residence time at a given location across all populations (farm and residential). This exposure duration was not varied randomly but held fixed. To the extent that actual exposure durations in some instances on either side of this average could disproportionately affect risks, the resulting risk distributions under this assumption involve uncertainty. For example, a site surrounded by predominatly rural residents may entail much longer average residence times than 9 years. (Rural populations typically reside at the same address longer than do urban residents.) If increased exposure duration puts many of these residents over the risk thresholds, then final risk distributions will be biased low. Alternatively, a 1-year exposure duration was used for noncarcinogens. The single year of highest exposure across the modeling

period is selected to evaluate noncancer exposures. A 1-year exposure duration for all constituents was used for ecological receptors to reflect the smallest meaningful time increment that could be produced by the modeling system (i.e., the system is based on annualized average concentrations, although many modules maintain the capability to implement smaller time steps). There is substantial uncertainty in using annual averages since chemical stressor spikes that are of high ecological signficance may be averaged out over the year, resulting in relatively low risk estimates. In contrast, the use of annual averages may overestimate risks to longer-lived species.

4.1.6 Multimedia Modeling

Assumptions

- Media are unidirectionally coupled (i.e., forward) only.
- Exposures occur from a single constituent for the length of the simulation; synergistic/antagonistic effects of multiple constituents are not considered.
- Exposures from other sources (e.g., background) are not considered; exposures to daughter products are not considered.

Limitations/Uncertainties

Modeling Approach Does Not Fully Integrate Environmental Media. In nature, interactions among environmental media are completely and instantaneously coupled in both feed-forward and feedback directions. For example, wind erosion may suspend contaminated, surficial soil from a WMU, which will subsequently be deposited—and possibly resuspended—elsewhere. The HWIR99 modeling approach considers only the deposition, not the resuspension. As another example, a stream might be a gaining stream in some reaches, where it may receive contaminated groundwater flows, and a losing stream in other reaches, where it might be a source of contaminated loads to groundwater. The HWIR99 modeling approach considers only transfers of loads from groundwater to surface water. As a practical matter, it becomes quite difficult computationally to have these models iterate back and forth among each other, alternating between sending outputs to another medium and receiving inputs from that medium. (Fugacity models accomplish this complete coupling, but they do so often at a significant credibility cost of sacrificing internal structure (i.e., they view each medium as a zero-dimensional, completely mixed "compartment"). This sacrifice was considered unacceptable for HWIR99.) The implications of this incomplete coupling, which constitute limitations and uncertainties of the HWIR99 model, are several:

- Failure to consider secondary emissions/sources (such as the examples cited above).
- Necessity of setting medium-specific "boundary conditions," which may be somewhat artificial, at media interfaces. (In a truly holistic model, "boundaries" would not exist between media.)

- Necessity of simulating one medium for a fixed time period (1 year) and then simulating another — rather than simulating multiple media simultaneously at a single time step.

Single Chemical Simulation. Given that the mixture of constituents that could make up a potentially exempt waste is unknown, the assessment was designed to look at chemicals independently. Therefore, the effects of transformation products ("daughter products") on the risk estimates are not known. In addition, the contribution to risk from other contaminant sources is not considered in the simulation. Rather, we evaluated the incremental risk/hazard that would result from exposures to exempt wastes containing the constituent. The uncertainty introduced by this assumption likely tends to underestimate the total risk due to constituent release and exposure.

4.1.7 Exposure Pathways

Exposures occur through transport of a single constituent released from a WMU to a contact medium (e.g., food, soil, air). Other sources of chemical contamination are not considered. Three basic exposure pathways are evaluated. For human and ecological receptors, the ingestion of contaminated food items such as fruits, vegetables, beef, milk, fish, and/or media subject to the chemical release and transport from a WMU was evaluated. For human receptors, two inhalation pathways were also considered: (1) the direct inhalation of chemical vapor and particulates released from a WMU, and (2) the indirect inhalation of chemical vapor during showers from contaminated groundwater.

Assumptions

- The exposure pathways considered include those pathways that present maximum exposures.

- Not all pathways are evaluated (e.g., dermal exposures, subsurface vapors entering a residence, inhalation for ecoreceptors).

- For ecological risk, inhalation exposures do not pose substantial ecological risks at levels that are protective of human receptors.

Limitations/Uncertainties

Selection of Exposure Pathways. There is some uncertainty associated with the assumption that all of the significant exposure pathways have been represented in the analysis. For example, dermal exposures were not addressed. The impact on the results for excluding various pathways depends, to a large degree, on the particular chemical. For example, if the skin is highly vulnerable to a certain chemical, then dermal pathways may be as important as several of the pathways included in the analysis, such as inhalation during showering. Although we cannot quantify the effects of not modeling every possible pathway, the pathways included in the analysis are assumed to be the most important for most chemicals.

Inhalation Exposures Not Significant for Ecological Receptors. The significance of ecological risks from inhalation relevant to inhalation risks in humans is not known with certainty but would increase exposure/hazard by an unknown amount. In general, data to evaluate inhalation hazards for wildlife are not adequate.

4.1.8 Endpoints

Cancer and noncancer endpoints were evaluated for human receptors and noncancer endpoints were evaluated for ecological receptors. For human receptors, the noncancer endpoints include a wide range of critical effects depending on the constituent of concern. Where appropriate, constituent-specific exposures from inhalation and ingestion were summed to estimate the total applied dose. The endpoints for cancer and noncancer effects in adult receptors were adopted for child receptors. For ecological receptors, endpoints relevant to population sustainability (e.g., reproductive fitness) and community stability (e.g., survival, growth, abundance) were chosen.

Assumptions

- Endpoints for cancer and noncancer effects predominantly derived for adults are relevant as well for child receptors.
- Endpoints on individual organisms can be used to infer risks to higher levels of biological organization, including the population and community levels.
- The most appropriate endpoints for population sustainability are reproductive and developmental.

Limitations/Uncertainties

Relevance of Endpoints and Benchmarks to Children and Adults. The appropriateness of using the same cancer slope factors and reference doses for both adults and children is uncertain. We estimate the risk of developing cancer from the estimated lifetime average daily dose and the slope of the dose-response curve. A cancer factor is derived from either human or animal data and is taken as the upper bound on the slope of the dose-response curve in the low-dose region, expressed as a lifetime excess cancer risk per unit exposure. However, individuals exposed to carcinogens in the first few years of life may be at increased risk of developing cancer. We modified the exposure factors for children to account for differences between adult and child receptors (e.g., body weight, exposure duration). We did not adjust the cancer slope factors to account for age-specific differences in exposure assumptions (e.g., body weight). However, we recognize that significant uncertainties exist regarding the estimation of lifetime cancer risks in children. Methodologies for estimating environmental threats to children's health are relatively new. They are currently being debated within the scientific community and will continue to evolve. The underlying assumption in our assessment that cancer risks for children can be calculated the same as cancer risks for adults has not been peer reviewed.

Noncancer effects in children is also an area of uncertainty. Noncancer reference doses and reference concentrations for children are based on comparing childhood exposure, for which we have age-specific data, with adult toxicity measures, where adequate age-specific dose-response data are lacking. This mismatch results in a large amount of uncertainty in the estimation of hazard quotients for children. This would sometimes result in an overestimation of children's risk and sometimes in an underestimation. This issue is still under investigation in the scientific community and no consensus has been reached.

Inference of Risks to Higher Levels of Organization. There is uncertainty associated with selecting endpoints for individual organisms and using those results to infer risks to wildlife populations and communities. In short, the measures of effect generally reflect toxicity to individual organisms while the assessment endpoints represent ecological values that go beyond the individual receptor. This is currently a limitation in the state-of-the-science and limits our ability to interpret ecological risk results. For example, we are unable to determine what fraction of a wildlife population will sustain an adverse effect and what the severity of that effect might be.

Appropriateness of the Ecological Endpoints. The endpoints for wildlife populations were almost exclusively limited to reproductive and developmental studies. Although these endpoints have been recognized by EPA as relevant to population sustainability, they are not always the critical effect associated with a chemical stressor. The assumption that other effects that may occur at lower environmental concentrations are not significant with respect to the population sustainability is uncertain. Studies regarding this question are inconclusive and, therefore, there is some uncertainty in using only reproductive and developmental studies to address the assessment endpoint of population sustainability.

4.1.9 Risk Metrics

For human cancer risks, the CSF is used to estimate the probability of cancer expressed as a lifetime excess cancer risk per unit exposure. For human and ecological receptor noncancer effects, the HQ represents the threshold risk of noncancer effects.

Assumptions

- Risk metrics are time-invariant.
- The HQ is an appropriate measure of noncancer effects for humans and ecological receptors.
- The HQ based on low levels of effect (e.g., 10 percent effect) is sufficiently protective of ecological receptors.

Limitations/Uncertainties

Risk Time-Discounting. The risks and HQs generated are assumed to be constant over the simulation horizon, which may cover up to 10,000 years. That is, the exposure incurred at any future date will lead to the same risk or HQ that it does today. This assumption is highly uncertain.

Appropriateness of the HQ. The hazard quotient has been widely accepted as an appropriate risk metric for noncancer effects. However, the concept of a noncancer threshold for all critical effects is associated with some uncertainty because the severity of noncancer effects varies widely. For the purposes of ecological risk, this metric is associated with considerable uncertainty in that it cannot be used to determine what percentage of a given population will sustain a particular effect. Similarly, the HQ approach for community-based risks (by inference) provides little insight into the actual ecological effects that will occur. In this application, the HQ cannot be used to determine the ecological significance of the effect; it can be used only as a bright line for adverse ecological effects.

Protectiveness of Low Effect HQ. Because HWIR99 goes well beyond a screening-level analysis, the hazard quotients reflect de minimis levels of effect to various ecological receptors. The degree of protection that may be inferred from this approach is uncertain. However, the uncertainty inherent in using HQs for no effects levels is likely to be associated with far greater uncertainty. The high level of protectiveness and the statistical issues concerning the use of no effects levels were considered sufficient to justify the calculation of HQs that reflect de minimis ecological risks.

4.2 Data Variability/Uncertainty

The 18 component modules of the HWIR 3MRA model require a large number of input variables (over 700) in a variety of categories, including facility and WMU characteristics, waste properties, meteorological data, surface water and watershed layout and characteristics, soil (vadose zone) properties, hydrogeologic variables, food chain or food web characteristics, human and ecological exposure factors, and the types and locations of human and ecological receptors and habitats surrounding a WMU. In addition, for each chemical to be assessed under HWIR, the models require chemical properties, bio-uptake factors, and human and ecological health benchmarks. Table 4-1 shows module-specific data used for the primary HWIR data collection categories. These categories provide the structure for the following subsections discussing data variability and uncertainty by category.

In addition, measured, calculated, and estimated chemical-specific data were used to generate all relevant chemical-specific thermodynamic and kinetic data for the HWIR assessment. The lack of reliable measured thermodynamic data necessitated the use of data generated by computational methods. The SPARC (System Performs Automated Reasoning in Chemistry) model, which is a computational method based on fundamental chemical structure theory, was the primary tool for calculating the thermodynamic constants. The process of assembling kinetic constants for degradation pathways (hydrolysis, anaerobic biodegradation,

Table 4-1. Data Types Used by HWIR 3MRA Component Modules

	Data Type												
Section:	**4.2.1**	**4.2.2**	**4.2.3**		**4.2.4**	**4.2.5**	**4.2.6**	**4.2.7**	**4.2.8**	**4.2.9**	**4.2.10**	**4.2.11**	**4.2.12**
Module	**Human Health Bench-mark[a]**	**Ecological Bench-mark**	**WMU**	**Waste Property**	**Meteor-ologic**	**Watershed & Surface Water**	**Soil Property**	**Terrestrial Food Web/ Farm Food Chain**	**Aquatic Foodweb**	**Human Exposure Factor**	**Human Receptor Type & Location**	**Ecological Exposure Factor**	**Eco-Receptor/ Habitat Type & Location**
Source modules													
Aerated tank			N	N	R								
Landfill			S, N	N	R		S						
LAU			S, N	N	R	S	S						
Surf. impoundment			S, N	N	R		S						
Wastepile			S, N	N	R	S	S						
Fate and transport modules													
Air			S		R						S		S
Watershed			S		R	S	S						
Surface water			S		R	S,R,N							
Vadose zone			S				S						
Aquifer			S			S					S		
Aquatic food web			S			S,R			N				S,R
Farm food chain			S				S	N			S,R		
Terres. food web			S				S	N					S,R
Exposure and risk modules													
Eco. exposure			S			S,R	S					R,N	S,R
Human exposure			S			S	S			N	S,R,N		
Eco risk		N											S,R
Human risk	N										S,R,N		

[a] Chemical-specific.
S = site-based data; R = regional data; N = national data; blanks = not applicable to module.

and aerobic biodegradation) focused on finding, evaluating, and summarizing measured data. Due to the complex nature of biodegradation processes, only a limited amount of measured kinetic constants were available for chemicals and are included in the HWIR chemical database. These kinetic data were grouped according to reaction conditions (i.e., pH, temperature, and redox conditions). However, because the rate constants for metabolism are unavailable for most constituents given the general paucity of data on metabolic rate constants in fish, the metabolic rate constant was set to a default zero until data can be developed for a larger universe of hydrophobic organic chemicals.

Although EPA has implemented the HWIR risk assessment on a national scale, the analysis is based on a regional site-based approach, where risks are evaluated at a number of representative individual nonhazardous industrial waste management sites across the country. To provide data consistent with this approach, the HWIR data collection effort centered around the collection of site-based data for industrial nonhazardous waste management sites representative of those that could manage HWIR-exempted wastes. National and regional data were also collected as necessary to complement the site-based data set where site-specific data were not readily available. Direct measures of facility/site characteristics (e.g., depth to ground water, aquifer thickness, hydrologlic conductivity) were not done at any of the sites. We account for these limitations by making assumptions that we believe are protective of human health and the environment.

Table 4-2 shows the collection approach (site-based, regional, or national) for each data category discussed in this section. In general, the uncertainty as to whether a variable represents conditions at a particular site is greatest if it is collected nationally and least if it is collected site-specifically. The data collected and passed to the modules reflects this, with nationally collected variables (e.g., human exposure factors) often being represented by distributions and site-specific variables (e.g., WMU dimensions) often being fixed. In other words, distributions are used to represent the more uncertain variables. For example, soil texture is fixed for each watershed by the predominant soil type across a watershed, while soil hydrologic properties, which are more uncertain, were developed as national distributions correlated with soil texture.

EPA believes that this site-based data collection approach has captured, to the greatest extent possible, the nationwide variability and distribution of potential risks from the management of exempted wastes in units in which they will typically be managed and that uncertainty in these risk estimates has been captured as well. EPA stresses that, although site-based data are the basis for this analysis, and these data are determined (to the extent possible) by where each site is located, this is not intended as a site-specific analysis. The site-based data are intended to provide a representative data set for a national assessment and may not be accurate in representations of conditions at specific sites.

4.2.1 Human Health Effects/Benchmarks

Human health benchmarks are required by the human health risk module of the 3MRA modeling system to evaluate human health effects that may result from exposure to constituents modeled in the HWIR analysis. Benchmarks are used in the human health risk module to evaluate exposure through the following pathways:

Table 4-2. HWIR Data Collection Approach, by Data Type

Data Type	Data Collection Approach		
	Site-Based	Regional	National
Waste management unit (WMU)	■		■
Waste property			■
Meteorological		■	
Watershed and waterbody	■	■	■
Soil property	■		■
Hydrogeologic		■	■
Farm food chain and terrestrial food web		■	■
Aquatic food web		■	■
Human exposure factors			■
Ecological exposure factors		■	■
Chemical-specific (chemical properties,[a] biouptake factors, health benchmarks)			■
Human receptor type and location	■	■	■
Ecological receptor and habitat type and location	■	■	

[a] Addressed under separate HWIR data collection and documentation efforts.

- Ingestion of water
- Ingestion of food products (i.e., beef, milk, fruits, and vegetables)
- Ingestion of fish
- Ingestion of soil
- Inhalation of ambient air
- Inhalation of air during showering.

The human health benchmarks used in the HWIR analysis are: oral RfDs, inhalation RfCs, oral CSFs, and inhalation CSFs. RfDs and RfCs are used to evaluate potential noncancer risks and CSFs are used to evaluate potential cancer risks. The benchmarks are chemical-specific. Human health benchmarks are applied nationally and do not vary by site or region. The human health benchmarks do not vary between receptors (i.e., residents, home gardeners, farmers, and recreational fishers) or age groups.

4.2.1.1 <u>Methods and Data Sources</u>. Only human health benchmarks derived by EPA were used in the HWIR analysis. The following EPA sources (in hierarchical order) provided the human health benchmarks used for constituents in the HWIR analysis:

- Integrated Risk Information System (IRIS) (U.S. EPA, 1999at)
- Health Effects Assessment Summary Tables (HEAST) (U.S. EPA, 1997e)
- *U.S. EPA TEF Values* and *Provisional Guidance for Quantitative Risk Assessment of Polycyclic Aromatic Hydrocarbons* (U.S. EPA, 1998g, 1993a)
- Superfund Technical Support Center
- Various other EPA criteria documents.

IRIS is an EPA database containing information on human health effects, RfDs, RfCs, cancer classifications, oral CSFs, and oral and inhalation unit risk factors (URFs). IRIS is the official repository of EPA-wide consensus human health risk information (U.S. EPA, 1999at). HEAST is a listing of provisional noncancer and cancer benchmarks (RfDs, RfCs, CSFs, and URFs) derived by EPA (U.S. EPA, 1997e). Although the human health benchmarks in HEAST have undergone review, they are not recognized as Agency-wide consensus information.

The Superfund Technical Support Center (U.S. EPA, National Center for Environmental Assessment) derives provisional RfCs, RfDs, CSFs, and URFs for certain chemicals. These health benchmarks can be found in NCEA risk assessment issue papers. These values have not undergone EPA's formal review process and, therefore, do not represent verified EPA benchmarks. EPA also may derive health benchmark values in other risk assessment documents such as Health Assessment Documents (HADs), Health Effect Assessments (HEAs), Health and Environmental Effects Profiles (HEEPs), Health and Environmental Effects Documents (HEEDs), Drinking Water Criteria Documents (DWCDs), and Ambient Water Quality Criteria Documents (AWQCDs). Evaluations of potential carcinogenicity of chemicals in support of reportable quantity adjustments have been published by EPA's Carcinogen Assessment Group (CAG) and may include cancer potency factor estimates. Health benchmark values identified in these EPA documents are not recognized as EPA-wide consensus information, however.

EPA has established toxicity equivalency factors (TEFs) to calculate CSFs for some dioxin-like compounds (U.S. EPA, 1998g). TEFs have been established for a number of polychlorinated dibenzodioxins, polychlorinated dibenzofurans, and polychlorinated biphenyl (PCB) congeners thought to have dioxin-like toxicity. TEFs for several polycyclic aromatic hydrocarbons (PAHs) have also been established (U.S. EPA, 1993a). For the HWIR analysis, some CSFs were calculated using EPA-derived TEFs.

Some inhalation CSFs were calculated from inhalation URFs. For the HWIR analysis, if inhalation CSFs were not available, then they were converted from URFs using the following equation:

$$\text{inhal CSF (mg/kg-d)}^{-1} = \text{inhal URF } (\mu\text{g/m}^3)^{-1} \times 70 \text{ kg} \times 1{,}000\ \mu\text{g/mg} \div 20\ \text{m}^3\text{/d} \qquad (4\text{-}1)$$

where

70 kg	=	default adult human body weight
20 m^3/d	=	default adult human daily rate of inhalation
1,000 µg	=	1 mg.

4.2.1.2 <u>Assumptions</u>. The RfD and RfC are used to estimate a level of environmental exposure at or below which no adverse noncancer effects are expected to occur. Most RfDs and RfCs used in the HWIR analysis were developed by EPA using the standard "threshold" no-observed-adverse-effect-level/lowest-observed-adverse-effect-level (NOAEL/LOAEL) approach. In the NOAEL/LOAEL approach, the RfD (or RfC) is based on the best available study; the critical study identifies the most sensitive effect in the most sensitive species, demonstrates dose-response, and identifies a NOAEL and/or LOAEL. The NOAEL or LOAEL is then adjusted for exposure duration and a human equivalent dose (in mg/kg-d) (or concentration in mg/m^3) is calculated. A series of standard uncertainty factors are then applied. In addition, a modifying factor may be applied to account for additional uncertainties. The RfD (or RfC) is derived by using the following formula:

$$RfD = NOAEL/(UF \times MF) \quad . \tag{4-2}$$

Cancer dose-response assessment usually involves the extrapolation from the relatively high doses administered to experimental animals to the lower levels expected for human exposure. Extrapolation is usually performed by fitting a mathematical model to the experimental data and then extending the model (or a bound on the risks it predicts) from the observed range down toward risks expected at low exposure. The linearized multistage model is used in the absence of adequate information to the contrary. The slope of the line in the low-dose region provides the average estimate for the CSF; however, as a protective measure, the 95 percent upper confidence limit on the slope is used as the CSF. When animal data are used, the human equivalent dose is calculated based on the assumption that different species are equally sensitive to the effects of a toxin if they absorb the same dose per unit of body surface area (U.S. EPA, 1999at).

For the HWIR analysis, it was assumed that exposure is of chronic duration. It was also assumed that the noncancer and/or cancer health response in children would be the same as the health response in adults; therefore, the same human health benchmarks were used for both children and adults in the HWIR analysis.

RfCs and inhalation CSFs were used to evaluate risk from the inhalation exposure pathway. For most constituents, a single RfD and/or oral CSF was applied for the ingestion of food, fish, soil, and water. However, for a few HWIR constituents, it was necessary to use ingestion exposure pathway-specific RfDs and oral CSFs. In addition, no human health benchmarks are available for lead.

RfDs are used for individual ingestion exposure pathways: foods other than fish (i.e., fruits, vegetables, beef, and milk), fish, soil, and water. The distinction between RfDs for food and fish results from the availability of different RfDs for mercury for fish and for foods other

than fish (methylmercury and mercuric chloride, respectively). The distinction between RfDs for food and water results from the derivation of source-specific RfDs for cadmium; the fraction of ingested cadmium that is absorbed varies with the source (e.g., food vs. drinking water), and it was necessary to account for this difference in absorption when EPA determined the RfD.

Oral CSFs were used to evaluate cancer risk from food (i.e., all foods including fish) and soil and water ingestion exposure pathways. The distinction between food and soil and water results from the availability of different toxicity values for PCBs for food and soil and water. The cancer potency of PCB mixtures is determined using a tiered approach that depends on the information available; for oral exposures, the upper-bound CSF for high risk and persistence was used for food and soil ingestion (2 per mg/kg-d) and the upper-bound CSF for low risk and persistence (0.4 per mg/kg-d) was used for water ingestion.

4.2.1.3 <u>Uncertainties and Limitations</u>. Uncertainties generally associated with human health benchmarks are discussed in detail in EPA's *Proposed Guidelines for Carcinogenic Risk Assessment* (U.S. EPA, 1996d), *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994c), and IRIS (U.S. EPA, 1999at). EPA defines the RfD (or RfC) as "an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime" (U.S. EPA, 1994c, 1999at). RfDs and RfCs are based on an assumption of lifetime exposure and may not be appropriate when applied to less-than-lifetime exposure situations (U.S. EPA, 1999at). The CSF is an upper-bound estimate of the human cancer risk per milligram of chemical per kilogram body weight per day. The unit risk, which is calculated from the slope factor, is an estimate in terms of the risk per µg/L drinking water or the risk per $\mu g/m^3$ air concentration (U.S. EPA, 1999at).

Uncertainty and variability in the toxicological and epidemiological data from which RfDs and RfCs were derived are accounted for by applying uncertainty factors of 1, 3, or 10 for each of the following: extrapolation from animals to humans, extrapolation from LOAELs to NOAELs, and extrapolation from subchronic to chronic data as well as those used to account for sensitive subpopulations. These uncertainties are discussed further in the source documents (e.g., IRIS) of the individual human health benchmarks.

Human health benchmarks in IRIS have been subjected to rigorous internal and external reviews and represent EPA-wide consensus human health risk information. The health benchmarks in HEAST have not undergone as rigorous a review as those in IRIS and do not represent EPA-wide consensus information. Provisional human health benchmarks identified in other EPA documents (e.g., HEA or HEEP) or derived by the Superfund Technical Support Center have not undergone EPA's formal review process and, therefore, do not represent verified EPA benchmarks.

In addition to the strength of the review process, confidence in human health benchmarks also can be associated with the date the benchmarks were derived (or last reviewed). Additional toxicological data become available over time and, subsequently, the database becomes more complete. Methodologies evolve, with improvements in existing methods (e.g., EPA's 1994 RfC

methodology) and the development of new health benchmark practices (e.g., benchmark dose methodology).

Human health data are incomplete. EPA-derived human health benchmarks are not available for all potential HWIR constituents. In addition, many constituents are lacking benchmarks for both exposure pathways (i.e., oral health benchmark available but no inhalation benchmark, or vice versa). Currently, the HWIR analysis is limited to using only EPA-derived human health benchmarks (e.g., IRIS, HEAST, EPA TEFs). However, alternative human health benchmarks are available from the Agency for Toxic Substances and Disease Registry (ATSDR) and the California Environmental Protection Agency (CalEPA). The ATSDR Minimal Risk Levels (MRLs) are substance-specific health guidance levels for noncarcinogenic endpoints. MRLs are derived for acute, intermediate, and chronic exposure durations for oral and inhalation routes of exposure. CalEPA has developed unit risk factors, cancer potency factors, and chronic inhalation reference exposure levels (RELs) for chemicals regulated under California's Hot Spots Air Toxics program (CalEPA, 1995, 1997, 1999a, 1999b). Additional TEFs from alternative sources are also available (Ahlborg et al., 1994; CalEPA, 1994; Nisbet and LaGoy, 1992). ATSDR and CalEPA human health benchmarks have been subject to expert scientific and public review. Alternative benchmarks, such as ATSDR MRLs and CalEPA RELs and cancer potency factors, offer a valuable opportunity to model both exposure pathways (i.e., oral and inhalation) and/or additional HWIR constituents. EPA has underway an effort to evaluate the supporting toxicology data used by other federal agencies in order to expand both the number of constituents that have at least one benchmark and increase the number that have multiple benchmarks.

4.2.2 Ecological Benchmarks and Chemical Stressor Concentrations Limits (CSCLs)

Ecological benchmarks (EBs) and chemical stressor concentration limits (CSCLs) are required to predict risks to ecological receptors. Risks are defined in terms of the hazard quotient and represent the ratio of the applied dose to the EB, or the concentration to the CSCL, respectively. The EBs, in units of daily dose (mg/kg-d), are used to evaluate risks to wildlife species populations exposed through the ingestion of contaminated plants and prey, as well as contaminated media. The EBs were generally based on endpoints relevant to the reproductive fitness and developmental success of wildlife. The CSCLs, in units of concentration (e.g., mg/L), are used to evaluate risks to communities of organisms in terrestrial (e.g., soil and plant communities) and aquatic habitats (e.g., sediment and surface water communities). In addition, CSCLs were developed for selected receptor groups such as amphibians to reflect specific exposure routes of concern, in this case, direct contact with contaminated surface water.

4.2.2.1 Methods and Data Sources. The methods applied to develop benchmarks and CSCLs for this analysis reflect current guidance on ecological risk assessment within EPA, as well as the general state-of-the-science for risk applications of chemical stressors. In developing the chemical-specific benchmark/CSCL database used for HWIR, the major source of ecotoxicity data was the primary literature. Secondary data sources included documents and databases developed by the EPA, other federal agencies (e.g., the National Oceanographic and Atmospheric Administration, [NOAA]), and research facilities (e.g., Oak Ridge National Laboratories). The methods used to identify and derive benchmarks and CSCLs vary across receptor type and

medium of concern. Section 14 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999p) details these methods and data sources.

4.2.2.2 Assumptions and Uncertainties. The assumptions and uncertainties associated with the application and development of the ecological benchmark and CSCL database center around three key issues: (1) the relationship between assessment endpoints and measures of effect, (2) the quality of the ecotoxicological data, and (3) the extrapolation methods used to derive receptor-specific benchmarks and CSCLs from the ecotoxicity data.

Relationship Between Assessment Endpoints and Measures of Effect

In developing ecological benchmarks and CSCLs for the HWIR analysis, it is crucial to establish the relationship between the assessment endpoints (i.e., the ecological values to be protected) and the measures of effect (i.e., the ecotoxicity data used to support benchmarks and CSCLs). Because the HWIR assessment is predictive, there is no mechanism to verify that assessment endpoints are, in fact, protected by the measures of effect selected to support benchmark/CSCL development. Consequently, there is uncertainty in applying the benchmarks/CSCLs to evaluate risks to the assessment endpoints chosen for HWIR. These uncertainties are discussed below along with key assumptions that are implicit in developing benchmarks and CSCLs (e.g., 95 percent protection level for aquatic biota).

Mammals and Birds

- *Assessment Endpoint*: *Maintain viable mammalian and avian wildlife populations.* The attribute to be protected was the reproductive and developmental success of representative species.

- *Measure of Effect*: *A de minimis threshold for developmental and reproductive toxicity in mammalian and avian wildlife species.* The threshold was calculated as the geometric mean of the NOAEL and LOAEL, frequently referred to as the maximum acceptable toxicant concentration (MATC). Implicit in this calculation is the assumption that the toxicological sensitivity is lognormal.

An important source of uncertainty is the benchmark calculation of the MATC. EPA determined that this threshold value was appropriate given the assessment endpoints for a national assessment. The rationale for this determination is based on two assertions: (1) the MATC is above a no effects level and, therefore, is associated with some finite level of risk (i.e., the regulations would not be based on no effects), and (2) the MATC is conservative in that it does not allow for the level of effect associated with a low effects level (often estimated at roughly 20 percent of the population). In making the assumption that a threshold for effects on individual organisms can be used to predict the potential risks to populations, there is considerable uncertainty regarding the “true” effects on wildlife populations that can only be addressed through simulations with population-level models. These models are designed to address various elements of population dynamics, such as predator prey interactions, carrying capacity of the habitat, immigration and emigration, and initial population size (to name but a few attributes of population-level models). Nevertheless, because the MATC is assumed to be

below the level of effect distinguishable from natural population variability, it is assumed that the benchmark provides a reasonably conservative level of protection to wildlife populations.

A second important source of uncertainty is in the selection of study endpoints; in essence, how well an MATC based on reproductive/developmental effects translates into maintenance of wildlife populations. Although the selection of reproductive and developmental effects is consistent with current EPA guidance on ecological risk assessment, it is not possible to demonstrate with certainty that these are the critical endpoints of concern. For example, neurological effects that impact behavior may occur at a threshold value below the MATC. Depending on the nature and severity of the effect (e.g., inability to avoid predators), a constituent may affect enough organisms that relatively few reach reproductive maturity. As a result, the overall impacts on the wildlife population may be greater by an unknown amount than those inferred from endpoints on reproductive fitness and developmental effects. The implications for receptors with relatively large home ranges (e.g., wolves) are difficult to interpret. For these receptors, the study area may only impact a single reproducing pair of animals and the endpoint of interest may be crucial in determining: (1) whether adverse effects occur, and (2) what the ecological significance of those effects might be.

CSCLs for the Freshwater and Soil Communities

- *Assessment Endpoint: Maintain sustainable community structure and function.* The attributes to be protected were growth, survival, and reproductive success of species that represent key functional roles in the community.

- *Measure of Effect: Concentration in soil or surface water, respectively, based on ecotoxicity studies on endpoints that include lethality, fecundity, growth, and survival.* The CSCLs for the freshwater and soil communities were typically derived at a 95 percent protection level using both no effects and low effects data, as appropriate. When available, the Ambient Water Quality Criteria for chronic effects were chosen as the freshwater CSCLs.

The CSCLs derived for the soil and freshwater organisms are intended to ensure protection of critical structures and functions of the respective communities. However, the CSCLs are not *true* community-level measures of effect in that they do not consider the complex interactions among community members. Consequently, there is uncertainty inherent in inferring risk estimates for the community from a statistical interpretation of data on individual organisms.

In addition, there has been some criticism of the method used to develop the community-based soil CSCL because it does not incorporate microbial populations as a taxa category. There is some question about the endpoint and level of effect that would result in ecologically significant no- or low-adverse effects to microorganisms (e.g., LOEC, EC_{25}, or EC_{50} for nitrification). There is no doubt that microbial communities are critical to the continued functioning of soil communities, and work is ongoing to consider how to appropriately include this receptor group in the species requirements. Their absence in the current method generates some uncertainty that the measure of effect provides protection to this key receptor.

Ecotoxicity data on earthworms were used to derive CSCLs for the soil community if no other suitable data were identified (i.e., if the data set were limited exclusively to earthworms). However, earthworms represent only one element of a healthy soil community and there is large uncertainty in applying these data to evaluate risks to the entire community. Earthworms play an important role in the soil community (e.g., soil fertility, nutrient release, aeration, food source for predators) and, because ecotoxicity data are relatively abundant, are valuable as indicator species.

Benthic Community

- Assessment Endpoint: *Maintain sustainable community structure and function*. The attributes of the benthic community to be protected included the growth, survival, and reproductive success of benthic biota.

- Measure of Effect: *Concentration in sediment based on ecotoxicity studies on endpoints that include lethality, fecundity, growth, and survival*. The CSCLs for the sediment community were typically derived at a 95 percent protection level using both no effects and low effects data, as appropriate. As with the freshwater CSCLs, the Ambient Water Quality Criteria for chronic effects were used to support the sediment CSCLs. In addition, field data on the toxicity of metals to sediment communities in salt water were also used to develop sediment CSCLs and include a number of "true" community-level effects (e.g., abundance of sediment biota).

The CSCLs for metals were derived from field studies conducted in marine ecosystems. These data were based on effects to marine biota and reflect the surface water chemistry and equilibria characteristic of marine systems. It is implicitly assumed that: (1) the relative sensitivity of species in saltwater sediments is similar to those in freshwater sediments, and (2) the behavior of metals (e.g., bioavailability) in the marine environment is not significantly different from the freshwater environment. However, comparisons made between freshwater and marine CSCLs developed using analogous methods indicate that there is not a significant difference between the effects seen across these systems at low exposure levels (Smith et al., 1996). The methods used to collect field data do not fully support a definitive cause-effect relationship because they do not account for other stressors that may impact the sediment community (i.e., temperature, predation). The CSCLs for nonionizing organic constituents were developed based on EPA guidelines for sediment community criteria.

Terrestrial Plant Community

- Assessment Endpoint: *Maintain structure and function of terrestrial plant community*. The attributes to be protected included growth and survival of terrestrial plants.

- Measure of Effect: *Soil concentrations related to growth, yield, seedling emergence germination endpoints*. The low effects data on phytotoxicity were rank ordered and the plant CSCL was estimated as the 10th percentile value.

The endpoints for plants were limited to low effects concentrations for growth and yield parameters such as seed germination, seedling emergence, and vegetative vigor. It is unclear,

however, if the selected measure of effect actually represents a biologically significant effect to populations of wild plants since terrestrial plant communities are quite robust and can shift to more tolerant plant species and still maintain an adequate prey base for herbivores. There is added uncertainty because most of the ecotoxicity data identified were based on studies using agricultural varieties of plants. Further, other effects such as RNA synthesis or respiration may be more sensitive indicators of potentially significant risks to plants. Substantial uncertainty will be associated with evaluating risks to the plant community until studies are available to determine: (1) the most sensitive, biologically significant endpoint for plants (e.g., seed germination, early growth), and (2) the effects level at which the effect should be considered significant in terms of plant population growth and survival.

Algae/Aquatic Plants

- <u>*Assessment Endpoint*</u>: *Maintain primary producers in freshwater systems, including both algal and vascular aquatic plant communities*. The attribute to be protected for this taxa was the growth and biomass.

- <u>*Measure of Effect*</u>: *Surface water concentrations related to gross measures of "health" (e.g., biomass) for the algal community and a variety of endpoints for aquatic plants (e.g., number of fronds, root number, plant number, root length).* For algae, the EC_{20} was selected as an adequate threshold for adverse effects and, because of the paucity of data, the lowest LOEC for endpoints of interest was chosen for vascular aquatic plants.

Algae and aquatic plants not only provide a food source for aquatic biota, but also provide needed structure and habitat for many aquatic species. Because the assessment endpoint includes the functional contribution of primary aquatic producers to aquatic ecosystems, there is uncertainty in applying low effects concentrations to this receptor group to protect its value to the ecosystem as a whole. Nevertheless, the design of the HWIR analysis goes well beyond screening, and, therefore, use of no effects data was considered inappropriate.

Data availability on algae far exceeded the data identified for vascular aquatic plants. As a result, the CSCL for aquatic producers generally reflects ecotoxicity studies on algae. Because little is known about the relative sensitivity between algae and aquatic plants, representing this receptor group with algal data introduces additional uncertainty in the risk estimates for freshwater systems. There are significant differences in uptake, transport, and biochemical processes between algae and aquatic plants and, therefore, uncertainty in determining how well the measures of effect act to maintain communities of primary producers in aquatic systems.

Herpetofauna

- <u>*Assessment Endpoint*</u>: *Maintain viable amphibian and reptile populations ("herps").* The attribute to be protected was the survival and developmental success of these receptors.

- *Measure of Effect*: The measure of effect selected to meet the assessment endpoint was the acute LC_{50}s for lethality and survival and developmental effects resulting from early life stage exposures.

As indicated by the assessment endpoint, this surface water CSCL was designed to protect both reptile and amphibian species. One uncertainty identified in data collection efforts was the lack of reptilian ecotoxicity data. There is significant uncertainty that the data identified to derive the measure of effect will, in fact, provide protection to reptiles. This data gap limits the overall characterization of risk of this receptor taxa.

There is additional uncertainty that the measure of effect used for herpetofauna adequately represents a level of protection suitable to the assessment. The CSCL represents a relatively severe effect that includes lethality to 50 percent of the population; this level of effect far exceeds other CSCLs developed for the HWIR analysis and should be considered as nonconservative. Therefore, there is substantial uncertainty applying this measure of effect given the goal of maintaining viable herp populations. Acute data were used to develop this CSCL because so few chronic data were available. There potentially could be dramatic impacts to other more sensitive endpoints such as reproduction at these surface water CSCLs; moreover, herpetofauna appear to be under considerable stress from other, as yet unidentified, factors.

The CSCLs for herpetofauna cannot be categorized as protective given the nature and magnitude of potential effects. Additional analysis of the risk results for this receptor group may be warranted for exit criteria associated with herp HQs close to the target value of 1. The uncertainty associated with the protection of this species needs special attention when the risk results are interpreted.

Data Quality Issues

In addition to uncertainties associated with the application of benchmarks/CSCLs, there are uncertainties inherent in the development of benchmarks/CSCLs (e.g., extrapolating from dose-response data; interspecies scaling of benchmarks). The quality and quantity of data used to develop the benchmarks and CSCLs vary greatly across the receptor groups. Thus, there are uncertainties inherent in deriving benchmarks and CSCLs using such a wide range of ecotoxicological data. The key data quality issues are reviewed here.

No-Effect Concentrations Used When Low-Effect Data Were Unavailable

For some of the receptor taxa, only a no-effects concentration was identified through literature searches. When no-effects data were available but low-effects data were not, the no-effects level was used. As a result, some benchmarks are more conservative than others. This was the case for the receptor chemical combinations shown in Table 4-3.

Table 4-3. Receptor Benchmarks and CSCLs Based on No-Effects Data

Constituent	Receptor Taxa
Lead	Soil community
Cadmium	Soil community
Mercury (divalent)	Mammal
Bis(2-ethylhexyl) phthalate	Bird

Statistically Based Soil Community CSCLs

The community-based soil CSCL was developed to assess the potential effects to multiple trophic elements and various key taxa. This method was only applied for a limited number of metals in the analysis; however, there are several unresolved issues in its application.

- A variety of endpoints were aggregated to derive a geometric mean for the NOEC data for each of the eight representative soil species data requirements. The geometric mean of multiple values representing several measurement endpoints has statistical meaning but not well-defined biological significance. The loss of the true stressor-response relationship in this approach makes interpreting the ecological significance of CSCL exceedances difficult.

- The issue of bioavailability is particularly important in assessing the impacts to the soil community, particularly for metals that exist in multiple ionic forms in the environment. The toxicity and mobility of a metal can be highly influenced by the local environmental chemistry of the soil matrix. Characterizing the bioavailability of metals in different environmental conditions is crucial in establishing CSCLs that are useful across different soil matrices. Bioavailability was not accounted for in this methodology.

- Although soil invertebrates may be classified according to ecological function (e.g., trophic level, feeding habits), few studies were identified that supported the assumption that taxonomically related soil invertebrates have toxicologically similar responses to chemical stressors (e.g., Neuhauser et al., 1986). In addition, many species of soil invertebrates were excluded that occur only in specialized micro environments such as dung piles, carrion, and rotting wood (i.e., niche organisms). As a result, species were selected to represent a range of trophic levels and functions in the community (rather than selecting the "most sensitive" species). This community-based approach assumes that, if key components in the soil community are protected, community structure and function will not be adversely affected. However, this approach has not been validated in field or mesocosm studies, and there are more than 100,000 species of invertebrates (excluding protozoa) per square decimeter in forest, meadow, and arable soils (Eijsackers, 1994). The development of a more generalized soil community reduced the resolution of the potential impacts to this community.

Statistical versus Biological Endpoints

There is some uncertainty associated with using the MATC as a benchmark because it is derived from **statistically relevant** endpoints rather than biologically relevant endpoints. Because the NOAELs and LOAELs are generated using hypothesis-testing statistics, the quantification of these effects levels depends on the size, design, and variability of an experiment (e.g., range-finding test or definitive test). Because the MATC is a geometric mean of two statistically derived toxicity endpoints, uncertainty is generated by establishing this value as a "protective" benchmark. In some cases, low effects and no effects concentrations derived using hypothesis testing result in chronic benchmarks for aquatic organisms (e.g., MATCs) that result in greater than 50 percent mortality (Stephan et al., 1985). For example, the MATC has been shown to correspond to fairly high levels of effect. Data from 176 tests on 93 chemicals with 18 species indicated that average reductions in reproductive endpoints at the MATC were 20 percent for parental survival, 42 percent for fecundity, and 35 percent for an integrative weight/egg parameter (Suter and Rosen, 1987). The uncertainty associated with the biological relevance and conservatism of these doses needs to be considered in the characterization of the risk results.

Research and Analysis of Data Supporting Benchmarks and CSCLs

A complete and exhaustive primary literature search has been completed for almost half of the constituents considered in this analysis. For other constituents, secondary sources have been consulted to identify appropriate benchmarks and CSCLs. The different levels of data review and analysis were the result of changes made to the list of constituents after primary literature searches were completed. Rather than conducting primary literature searches for the newly added constituents, compendia were used to identify preliminary benchmarks and CSCLs to support the risk estimates. The limited literature review given to these chemicals (as shown in Table 14-2 of U.S. EPA, 1999p), magnifies the uncertainties in developing benchmarks and CSCLs.

Extrapolation

Uncertainty Factors Applied

Finally, additional uncertainty was introduced into the analysis through developing the measures of effect by applying uncertainty factors to convert LOAELs for mammalian and avian benchmarks to NOAELs. When only LOAELs were available, the LOAEL was divided by 10 to estimate the NOAEL. This is not an uncommon procedure when only a low-effects concentration is available; however, EPA has recently assessed the accuracy of uncertainty factors and reported that, in many cases, the difference can be less than 10. Applying a factor of 10 may create added conservatism in the MATCs generated for mammals and birds (Abt, 1995). For algae and aquatic plants, a similar uncertainty factor of 5 was applied to convert EC_{50} data to estimate a low effects concentration.

Allometric Scaling

For mammals, differences in interspecies uncertainty were indirectly addressed through the use of the species-scaling equation. This method is used by EPA in carcinogenicity

assessments when extrapolating from rats to humans. Wildlife toxicologists commonly scale dose to body weight without incorporating the exponential factor. There is continued disagreement among experts whether the application of scaling factors is appropriate in ecological risk because this method may not account for physiological/biochemical differences in species sensitivity. Applying this method to species demonstrating different sensitivities across chemical classes introduces some uncertainty in the analysis. Allometric scaling was not applied to avian receptors because a recent study indicated that scaling benchmark doses for birds may not be protective of small-bodied avian receptors (Mineau et al., 1996).

Lab-to-Field Extrapolation

The toxicological benchmarks for ecological receptors were developed assuming that effects that are observed in laboratory test species are applicable to wildlife species under similar field conditions. As a result, there were no laboratory-to-field extrapolation factors applied to account for the additional stress that may be encountered under field conditions (e.g., cold or drought). Van Straalen and Denneman (1989) and Stephan et al. (1985) examined arguments both for and against a laboratory-to-field extrapolation factor and concluded that laboratory-to-field extrapolation factors were not necessary; i.e., criteria derived with laboratory data should protect soil fauna in the field. However, other authors have suggested that laboratory species tend to be more homogeneous and have narrower tolerance distributions than their field counterparts and that the distribution of the target population of species is likely to have a different shape and scale relative to the laboratory species (Smith and Cairns, 1993; Suter et al., 1983; Seegert et al., 1985). As a result, the distribution of the endpoint will be narrower for the laboratory species. In addition, Smith and Cairns (1993) point out that local adaptation to conditions may make an individual species more or less tolerant to a chemical stressor.

4.2.3 Facility, Waste Management Unit, and Waste Property Data

The HWIR 3MRA assessment is designed to provide EPA information, together with other information, to set exemption levels for hazardous waste constituents; wastes with constituents below these exit levels would be exempt from hazardous waste regulations. To represent facilities across the United States that are likely to receive exempted wastes, EPA selected nonhazardous industrial waste management sites from its 1985 Screening Survey of Industrial Subtitle D Establishments (Westat, 1987). This survey was designed to represent a total population of nearly 150,000 establishments generating nonhazardous industrial wastes. A total of 201 sites were selected from the 2,850 facilities (out of a total of 15,844 surveyed facilities) that had one or more of four types of waste management units: landfill, wastepile, land application unit (LAU), and surface impoundment. Although over 10 years old, the survey represents the largest consistent set of data available on Industrial D facility location and WMU dimensions.

EPA selected the sample of 201 facilities to represent the types, dimensions, and geographic locations of WMUs at which exempt waste could be currently disposed of. The locations of these facilities determine the sites where the 3MRA is implemented and the survey data determine the types and dimensions of WMUs that are present at a particular site. At each site, the area of interest (AOI) for the HWIR 3MRA is the WMU area plus the area encompassed

by a 2-km radius extending from the corner of the square WMU (Figure 4-1). The AOI is the spatial area within which data are collected and risks are estimated. EPA defined this radius based on air and groundwater modeling results, which show risk levels flattening out at and beyond 2 km from the edge of the WMU.

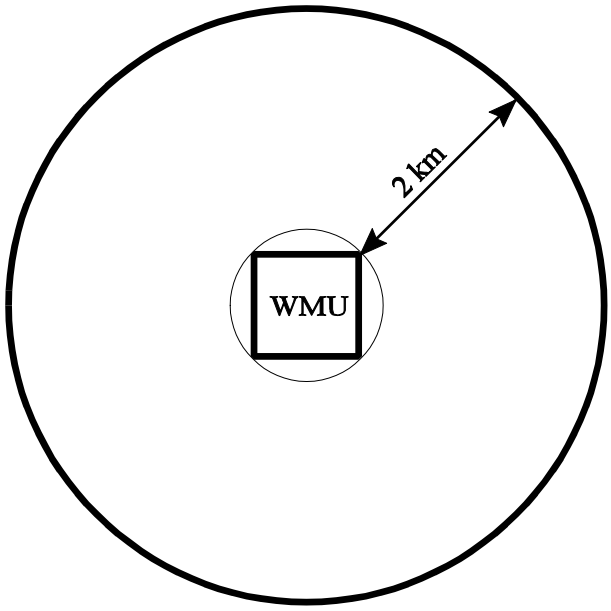

Figure 4-1. Area of interest (AOI) for HWIR 3MRA

Because of difficulties in modeling multiple units at a single site, where there are multiple WMU types at an Industrial D site, the data collection effort assigned multiple AOIs and multiple settings. HWIR defines a setting as a unique WMU type/site combination. Thus, a site with a landfill and a wastepile would constitute two settings. Settings are the basic spatial modeling unit for the 3MRA analysis, with each model realization being run at a single WMU/site setting. There are 419 unique WMU/site settings across the 201 sites selected for the HWIR risk analysis, including 282 unique Industrial D WMU/site combinations (56 landfills, 61 wastepiles, 28 LAUs, 137 surface impoundments), plus 137 aerated tank settings (aerated tanks are placed at each Industrial D facility with a surface impoundment).

WMU input data were used explicitly by seven of these modules: the five source models, the air model, and the vadose zone model. These WMU inputs describe the size and operation of the five land-based WMU types to be modeled as sources of contamination in the HWIR risk assessment: landfills, surface impoundments, wastepiles, land application units (LAUs), and aerated tanks. Waste property data were used by the five HWIR source modules only.

4.2.3.1 <u>Methods and Data Sources</u>. Three approaches were used for collecting facility, WMU, and waste property data: site-specific, site-based, and national. Site-specific data on facility location and WMU area, capacity, and waste loading rates for landfills, surface impoundments, wastepiles, and LAUs were obtained from the Industrial D Screening Survey (Westat, 1987). Size-related WMU variables, which were derived from these Industrial D data and using best professional judgment, are referred to as site-based data. All other WMU inputs for these WMU types, along with waste property data, were developed on a national basis as either distributions or fixed values, depending on potential variability and model sensitivity.

The Industrial D Screening Survey does not include aerated tank data, and comprehensive, national data on nonhazardous aerated waste treatment tanks were not available. A national database on tank designs from a survey of hazardous waste treatment, storage,

disposal, and recycling (TSDR) facilities (U.S. EPA, 1987), supplemented with other available data (U.S. EPA, 1999e), was used to provide national data on aerated tanks, assuming that tank wastewater treatment tank design and operation is similar for hazardous and nonhazardous wastes. Aerated tanks were placed at all Industrial D facilities with surface impoundments to ensure that tanks were placed at locations where liquid wastes are managed.

The location of each of the 201 Industrial D facility was initially based on the Industrial D address and zip code and further refined by obtaining best EPA locations from the Envirofacts database. Each location was then inspected using maps with respect to the land use surrounding the site to ensure that each facility was in a plausible location (in some cases it was necessary to move a WMU out of a waterbody or from a residential to a nearby industrial land use type). Once the facility location was fixed, a square WMU was placed at the facility centroid and a 2-km radius was drawn from the corner of the square to form the AOI (as shown in Figure 4-1). The WMU and AOI boundaries served as the boundary for all site-based data collection and one of the basic spatial layers of the GIS-based collection effort, as described in Section 2 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999d), which also provides details on how facilities were located.

Prior to use in HWIR, the Industrial D data were screened for reasonableness, and questionable data were replaced based on correlations in the remaining data (U.S. EPA, 1999e). Following screening and replacement, the general approach for WMU data collection was to develop model facility designs based on standard industry practices and scale the designs to the unit sizes extracted from the Industrial D data or from the TSDR data for aerated tanks. These designs are general descriptions of the key unit features that determine the parameter values of interest for the source models. Based on these designs, the parameter values were estimated, either as fixed values or value ranges. The following steps describe this approach:

1. ***Collect information to define typical WMU designs.*** Initially, information was collected from the Industrial D and TSDR databases and from the literature to define the typical designs for each WMU. For example, aerated tank design features (relating to unit depth, size and number of impellers, and aeration method) vary depending on unit size, flow rate of waste through the unit, and materials being treated. The collected information was then used to determine the number and general characteristics of the model facilities for each type of WMU.

2. ***Collect detailed data for each WMU model design.*** Once the general WMU model facility designs were reviewed, available information was collected from literature about the specific parameter values for each design. In general, multiple sources were consulted for each parameter to identify and characterize typical ranges for parameter estimates. When no information was available in the literature for a parameter, engineering calculations were made, where possible, based on other aspects of facility design. As a final option, engineering judgment was used as a basis for developing data.

Section 3.0 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999e) provides additional detail about WMU data collection, methods, and sources.

Because of a lack of information on Industrial D waste characteristics, waste property data were represented by national distributions based on best engineering judgment. Waste property data passed to the model are described in Section 16 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999r).

4.2.3.2 Assumptions. The Industrial D Screening Survey provided only WMU dimensions (area and capacity) and the TSDR survey data provided only tank capacity, throughput, and treatment type. The following assumptions were used to adapt these data to populate the HWIR facility, WMU, and waste property variables.

General Assumptions

- The locations and dimensions of WMUs in the 1985 Industrial D Screening Survey are representative of future waste management practices for exempted wastes.
- The center of the WMU is at the best location determined for each facility. For a particular site, a single WMU type is modeled to define each setting, with other types of WMUs for the same site being placed at exactly the same location.
- An average Industrial D WMU is assumed. For example, if the Industrial D data indicate that there are several WMUs of a type at a facility, one unit is modeled with the WMU area for that type set equal to the total WMU area in the database divided by the number of units (Industrial D data do not provide unit-specific dimensions).
- All WMUs are assumed to be square in shape (length = width).
- In terms of quality of Industrial D data, WMU area data are more reliable than total capacity data, which are more reliable than annual waste quantity data.

Landfill Assumptions

- Only one type of landfill is used for disposal of waste (i.e., there are no significant differences in the design of landfills depending on size or purpose).
- A landfill is filled at 30 years, after which it is closed with wastes in place.
- Landfills are excavated below ground surface.
- The landfill is capped with soil cover to establish a vegetative cover after a cell is filled (on an annual basis).
- The landfill cover soil is obtained from soil at or very near the facility and could be soil excavated to construct the landfill itself (i.e., soil properties for the vadose zone directly underlying the landfill are used for cover soil properties).

- Landfills are unlined.

Wastepile Assumptions

- Wastepiles are temporary units used for storing or accumulating waste prior to final treatment or disposal.
- Wastepiles are composed of solid waste materials that are dumped into a pile and may subsequently be moved or spread.
- A wastepile is operated for 30 years, after which the wastes are removed.
- Waste is delivered by dump truck to the unit location and deposited to create a pile of uniform height throughout the area of the unit.
- 1985 Industrial D waste quantity data are representative and applicable to all years of operation.
- The pile is refreshed at least once every 5 years, although a greater refresh frequency may occur for large Industrial D 1985 waste quantities and relatively small wastepile areas.
- The wastepile is placed directly on native soil, with no compaction or liner underneath.
- There is no cover (engineered or otherwise) and no control practices are employed to limit water erosion or volatile emissions from the pile.

Land Application Unit Assumptions

- Land application (or treatment) involves the application of wastes to an agricultural plot of land in either a liquid or semisolid form, tilling the wastes into the soil, and treatment through the biological degradation of the hazardous constituents in the soil zone.
- The HWIR LAU is a single plot design where wastes are spread uniformly over the entire unit area.
- The LAU is operated for 40 years, after which it is closed with all applied wastes remaining in place.
- 1985 Industrial D waste quantity data are representative and applicable to all years of LAU operation.

- A 30-m buffer zone is present between the LAU and the nearest downslope waterbody.

- Beyond standard agricultural soil erosion control practices (e.g., contour plowing, cover crops), no dust, leachate, or runoff controls are in place.

- Waste applications do not alter soil hydrologic properties, bulk density, organic carbon content, or pH.

Surface Impoundment Assumptions

- A surface impoundment is an excavation or diked area typically used for the treatment, storage, or disposal of liquids or sludges containing free liquids.

- Surface impoundments operate for 50 years, and all waste is removed from the unit when it is closed.

- Surface impoundment releases are limited to percolation into underlying soils and volatile emissions to the atmosphere. Planned or accidental releases to surface water are not modeled.

- Liquids and solids typically separate in a surface impoundment by gravity settling.

- Liquids from surface impoundments are removed by draining, evaporation, or flow through an outlet structure. Accumulated solids are removed by periodic dredging during impoundment operation or at the time of closure.

- Surface impoundments are not lined and there are no volatile emission controls.

Aerated Tank Assumptions

- An aerated tank is present at every HWIR facility that has a surface impoundment. This is based on the assumption that surface impoundments are an indication that a facility manages liquid wastes and that tanks are likely to be present at such facilities. Tanks cannot be larger in area than the average surface impoundment area at a facility.

- The HWIR aerated tanks are treatment tanks that are open to the atmosphere and have some level of agitation of material contained in the tank.

- Aerated tanks have an economic lifetime of 20 years. The HWIR analysis assumes 2.5 economic lifetimes so that a tank is present at a facility for the same length of time as a surface impoundment (i.e., tanks are no longer present after 50 years).

- Tank releases are limited to volatile emissions to the atmosphere. No spillage release through infiltration or overland transport occurs.

- Aerated tanks for treating hazardous waste (as represented by the 1987 TSDR survey) are representative in size and design of treatment tanks for nonhazardous wastes.

- Two levels of agitation (or aeration) are assumed. Highly aerated tanks actively mix the liquid surface for the purpose of aeration (transferring oxygen from the atmosphere into the liquid). Low aeration tanks have convective currents and some degree of turbulence that increases volatile losses, but they are not designed specifically to enhance air-liquid mass transfer.

- Aerated tanks can be elevated (bottom of tank above ground level), on the ground (bottom of tank at ground level), or in-ground (bottom of tank below ground level). As the tank volume increases, so does the likelihood that the tank is on or in the ground.

Waste Property Assumptions

- With the exception of total porosity, which is assumed to be greater than field capacity, which is greater than wilting point, waste properties are not correlated.

4.2.3.3 Uncertainties and Limitations. Although many of the specific assumptions listed under Section 4.2.3.2 may not be so at certain Industrial D sites, they are based on best available data and information is not available to check their validity at this time. The primary areas of uncertainty with respect to WMU data are (1) whether the 1985 Screening Survey data are appropriate for characterizing Industrial D units that may receive exempted wastes and (2) limitations associated with using the hazardous waste TSDR tank data to represent nonhazardous tank designs.

Age/Accuracy of Industrial D Data. For the nationally collected data, one issue of concern was the availability of recent data on actual Industrial D units (the Industrial D data are more than 10 years old). Although significant data can be found on Subtitle C waste management units (e.g., permit records, design requirements), recent Industrial D data are not compiled or readily available. In spite of its age, the Industrial D Screening Survey represents the largest consistent set of data available on facility locations and WMU dimensions. The 201-facility sample was selected from the survey to represent the types and geographic locations of WMUs at which exempted waste could be disposed of. At some of the 201 facilities, there probably have been WMU additions or closures since the survey was conducted. EPA considers this approach of basing the assessment on actual WMU, land use, and population data, however, to be preferable to developing and evaluating hypothetical exposure scenarios.

Another issue that was identified during use of the Industrial D data was whether to revisit the methodology used to screen out questionable entries in the Industrial D database. For consistency, EPA decided to adapt the methodology from previous EPA Composite Model for Leachate Migration with Transformation Products (EPACMTP) modeling efforts, as described in U.S. EPA (1999e).

Underrepresentation of Highly Aerated Tanks. The tank database appears to under-represent highly aerated tanks. This is probably due to a disproportionate number of aerated biological treatment systems being operated at facilities that process only onsite waste. This underrepresentation introduces some uncertainty into the analysis, the result of which is that risks from highly aerated tanks may be underestimated.

Representativeness of Aerated Tank Dataset. The TSDR data used to develop model aerated tank designs are based on hazardous waste tanks. However, EPA does not believe that there should be a substantive difference between basic tank dimensions for hazardous and nonhazardous wastes. Other, non-TSDR data were derived using engineering relationships that are applicable to aerated tanks for general wastewater treatment, hazardous or nonhazardous.

4.2.4 Meteorological Data

Of the 18 media-specific pollutant release, fate, transport, exposure, and risk modules in the 3MRA model, meteorological data are used directly by the source, air, watershed, and waterbody modules. Although 3MRA model risk estimates are long-term estimates, the meteorological data time scales differ by component module, with shorter time scales necessary to accurately estimate release or fate and transport in media sensitive to fluctuations in meteorological data. For example, the surface impoundment and tank modules use monthly data to capture temperature extremes that can impact volatilization. The wastepile, land application unit, and watershed modules use daily data to accurately estimate precipitation-driven runoff and erosion events.

During system execution, the HWIR 3MRA modeling system calls meteorological data for the appropriate meteorological station directly as separate American Standard Code for Information Interchange (ASCII) files containing hourly, daily, monthly, annual, and long-term meteorological data. Table 4-4 shows which of the 3MRA modules access which file types.

4.2.4.1 <u>Methods and Data Sources</u>. Meteorological data were collected regionally by meteorological station, with each of the 201 Industrial D sites modeled for HWIR assigned to the nearest station with similar weather conditions and adequate weather data for the analysis. In

Table 4-4. Meteorological Data File Access, by HWIR 3MRA Model Module

	Meteorological Data Files				
HWIR 3MRA Module	**Hourly**	**Daily**	**Monthly**	**Annual**	**Long-Term**
Air	●				●
Wastepile		●	●	●	●
Land application unit		●	●	●	●
Surface impoundment			●		
Aerated tank			●		
Landfill		●		●	●
Watershed		●	●		●
Surface water					●

making these assignments, EPA considered all available data from 218 meteorological stations across the United States to find the best data for each site. This process resulted in 99 meteorological stations being matched to the 201 Industrial D sites. This is a considerable improvement from the 29 meteorological stations used to represent the nation in the 1995 HWIR proposal and was made possible by several automated programs developed to reduce the effort required to prepare the data files needed by the 3MRA model. Details about these programs can be found in Section 4.0 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999f).

Most meteorological data were extracted from Solar and Meteorological Surface Observation Network (SAMSON; U.S. DOC and U.S. DOE, 1993) hourly data files and converted as necessary to daily time series, monthly time series, annual time series, and long-term averages. Because SAMSON precipitation data were inadequate, precipitation data were obtained from cooperative station daily summaries (NCDC, 1995), with SAMSON data used to help allocate these daily data to hourly time series. Mixing heights were obtained from upper air station data. Land use data also were required in the vicinity of each meteorological station to derive air model inputs such as Bowen ratio, surface roughness height, minimum Monin-Obukhov length, noontime albedo, and the fraction of net radiation absorbed by the ground.

Data quality control review uncovered, along with precipitation data problems, significant portions of missing data within the various meteorological data time series required for the HWIR analysis. Programs were written to automatically find and correct these data gaps, within technical constraints established by Atkinson and Lee (1992).

4.2.4.2 <u>Assumptions</u>. Several assumptions were necessary to complete accurate data processing of meteorologic data. Significant assumptions include:

- Weather at the meteorological station is representative of weather at the site.
- The available 10 to 30 years of station data are adequate to represent future meteorological conditions at each site.
- Roughness heights are equivalent at the meteorological station and the site; otherwise, separate meteorological files would be required for each HWIR site (201 versus 99 files).
- Anthropogenic heat flux is zero at all meteorological stations. This parameter can usually be neglected outside of highly urbanized locations, and data were not available to assume a value.
- A default anemometer height of 6.1 m is appropriate for stations for which data were not available.
- Simple linear interpolation is adequate for filling in from one to five missing meteorological data points.

- Although SAMSON precipitation records cannot be used directly, they are adequate for allocating daily precipitation to hourly data files.

4.2.4.3 <u>Uncertainties and Limitations</u>. The HWIR 3MRA meteorological data set represents a very comprehensive set of meteorological data. All available meteorological data were used for the effort to ensure that the most applicable data were used for each of the 201 Industrial D sites modeled for HWIR. In addition, extensive quality control (QC) was conducted to ensure that the data were accurate and complete and to identify and correct data gaps.

The great volume of data (five data files for 99 meteorological stations) required extensive automated data processing to compile and calculate. Most of the issues and uncertainties associated with this effort have to do with the assumptions and simplifications necessary to write the automation programs or uncertainties associated with replacing missing data. In summary, in practically every case, these uncertainties were not likely to have introduced any significant consistent bias to the data or the analysis. One advantage of automated processing is that, unlike subjective methods based on professional judgment, there is not the potential bias associated with different meteorological judgments.

***Precipitation Data*.** Inaccurate hourly precipitation data in the SAMSON data set necessitated creation of hourly precipitation by extrapolating daily precipitation data across a 24-hour period, using SAMSON as an hourly precipitation template. Issues and uncertainties associated with this methodology are mainly associated with the lack of knowledge of when precipitation events did actually occur, given that the instruments used to measure the hourly rainfall totals in SAMSON could not measure small amounts of rainfall. This limited knowledge of hourly precipitation amounts could bias the record in two ways: (1) by stretching rain events over more hours than they occurred, or (2) by attributing more rain to a short amount of time than actually occurred—biases that would have opposing effects. One small source of consistent bias is that the cooperative summary data can be missing hours at the beginning and end of the 24-hour period, which would underestimate precipitation totals by not including any rainfall or snowfall during those periods.

***Meteorological Data Completeness / Length of Record*.** During data collection, it was found that most meteorological stations were missing significant portions of data. For instance, for most stations only about one-third of the observations are complete for the late 1960s and early 1970s. Data for these years were usually discarded and, as a result, complete data for 30 years were never available. Other stations were completely discarded because too much of the data were missing. In these cases, alternative stations nearby were used instead. It is not known whether missing data for the late 1960s or early 1970s represents a consistent bias, but it is not thought so.

Similarly, there is uncertainty as to whether the 10 to 30 years of records at each station adequately represent future weather and climatic conditions. If, for example, there is a significant climatic change (warming or cooling), the available data may not represent future conditions. Given that these are future events, and all past data records are being used in the current data, this uncertainty would be difficult to assess except through hypothetical scenarios.

Meteorological Station Representativeness. Meteorological station assignments were based on both proximity and climatic regimes as reflected by ecological habitats and physiography. Although these are likely to be accurate for many sites, it is possible that microclimates at some site station assignments could be different than those for the closest station. Although this could lead to some inaccuracy with respect to site-specific results, it is probably not a significant source of error for a national, site-based analysis.

Replacing Missing Data. When replacing missing surface or mixing height data, Atkinson and Lee (1992) reported that each occurrence of missing data can be filled in more accurately if it is examined individually and filled in using objectivity, where possible, and subjective meteorological experience as necessary. The objective methodology recommended by Atkinson and Lee (1992) was primarily used in this effort, with simple, automated linear interpolation being used for the mixing height data. Although for a particular site and time this method may not necessarily be representative of the nature of the atmosphere, this automation was necessary to complete the extensive data collection effort in a timely, consistent, and precise manner. In addition, the automated interpolation method should not introduce a consistent bias, while professional judgment can. However, independent examination of interpolation results and, perhaps, comparison of replaced data with actual data for sites with data, may be necessary to confirm this lack of consistent bias.

4.2.5 Watershed and Waterbody Data

The HWIR 3MRA model uses site-based data collected around 201 nonhazardous industrial waste disposal facilities randomly selected from EPA's *Screening Survey of Industrial Subtitle D Establishments* (Westat, 1987). As primary spatial data layers for this site-based analysis, watershed and waterbody data were collected for all 201 sites. Within the HWIR 3MRA model, watersheds and waterbody reaches define the spatial units for which soil and surface water concentrations were calculated to estimate human and ecological exposure through these media. These data include the lengths, areas, slopes, and interconnectivity of the watersheds, streams, lakes, and wetlands at a site.

Where site-based data were not available, regional and national data were used for the HWIR 3MRA. Regional and national data collected to supplement site-based watershed and waterbody data include regional water quality and flow data extracted from EPA's Storage and Retrieval System (STORET) database and national ranges and distributions for surface water inputs derived from literature and professional judgment.

4.2.5.1 <u>Methods and Data Sources</u>. To develop the HWIR 3MRA watershed and waterbody layout, geographic information system (GIS) programs were used to compile available hydrologic, digital elevation model (DEM), land use, and wetlands data sets; extract site-specific data from these data sets; delineate the watershed subbasins, waterbodies, and local watersheds; and export the resulting spatial parameters in data tables for further processing. Database programs were then used to perform the necessary calculations and relationships to create the input data sets for the 3MRA model. Most of the data processing was automated, but the GIS delineations required manual interaction.

Data collection of the regional and local watersheds and waterbodies was conducted concurrently and based on the 2-km radius AOI that corresponded to the largest WMU at each HWIR site. Figure 4-2 provides an overview of the data collection methodology for watersheds and waterbodies. The watershed and waterbody layout data required GIS processing to delineate the watersheds and waterbodies and to obtain spatially related parameters (Steps 1 through 8). Further database processing of the GIS data was required to provide the exact data and format for the HWIR 3MRA model (Step 9).

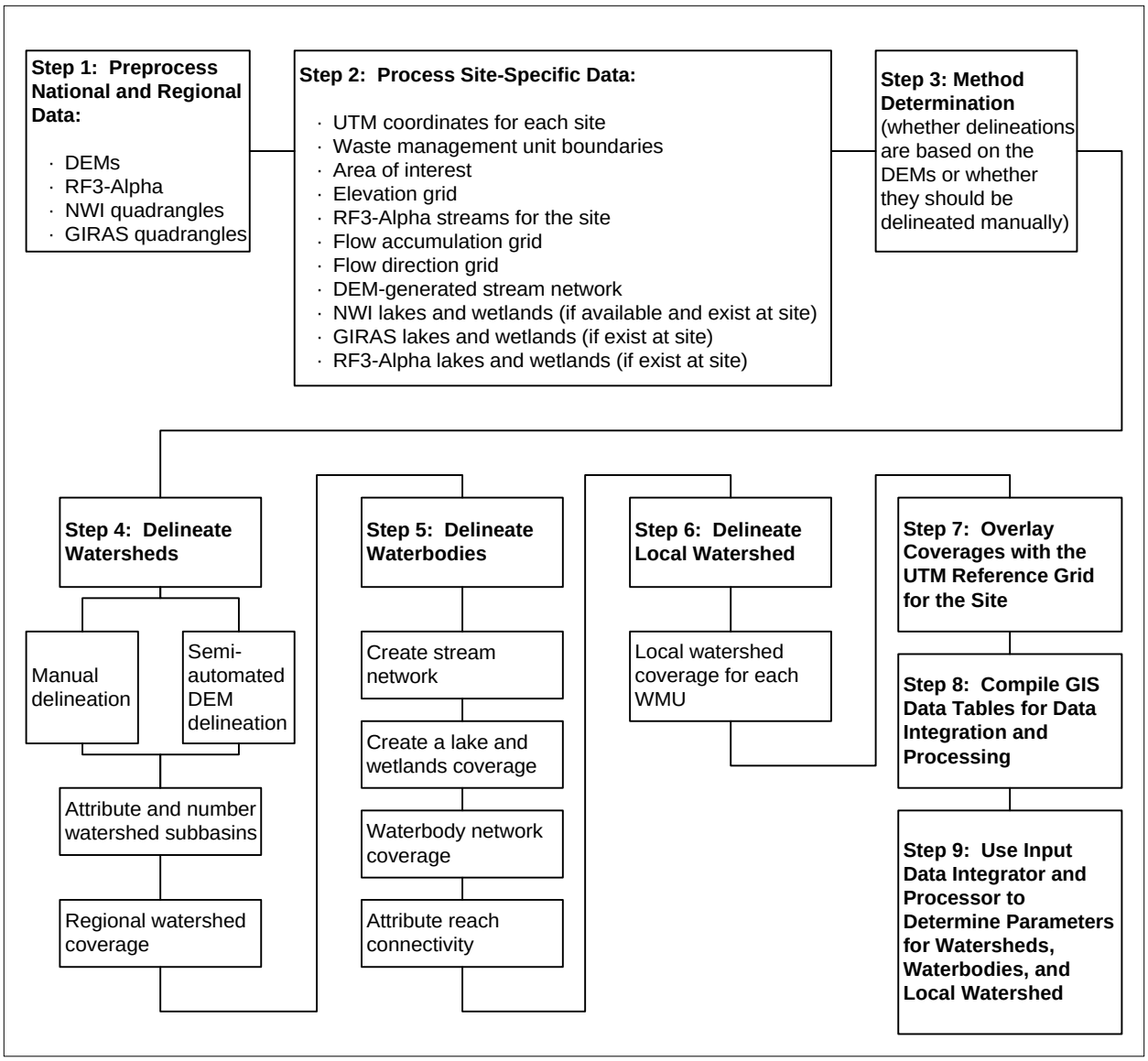

Figure 4-2. Overview of data collection methodology for watersheds and waterbodies.

To obtain the available data coverages for each site, coverages were first gathered on a national scale from the data sources shown in Table 4-5 (Step 1), and then the coverages around each site were extracted from them (Step 2). Using these site-based data coverages, watersheds and waterbodies were delineated by manual and automated processes (Steps 3 through 7). The GIS data were passed as a set of data tables for further processing to create the model input data set (Step 8). The Input Data Integrator and Processor (IDIP), comprising multiple databases and SQL (structured query language) and VBS (Visual Basic Script) script files, was designed to integrate collected data from multiple sources, perform any necessary calculations, and format the the 3MRA model input variable database (Step 9).

A consistent source of site-based data was not readily available for several categories of waterbody data, including water quality parameters, flows, depths, and sediment characteristics. Water quality and flow-related variables were collected regionally based on data from EPA's STORET database (U.S. EPA, 1990; http://www.epa.gov/OWOW/STORET). National parameters were obtained from the literature or estimated using the professional expertise of surface water modelers and other senior scientists. Section 6 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999h) describes how these data were collected.

4.2.5.1 Assumptions. The waterbody layout includes freshwater lakes, stream, rivers, and wetlands within the area of interest around each Industrial D facility. The watershed layout includes the watersheds that contribute flow to waterbodies within the AOI. Although the waterbody/watershed layout is important for several modules in the 3MRA model, the layout primarily supports the surface water, regional watershed, and overland transport (local watershed) component of the land application unit and wastepile source models. Within the context of these models and the 3MRA system, certain assumptions and definitions were

Table 4-5. Source Data for Watershed and Waterbody Delineations

Data Source	Brief Description	Scale	Data type	Internet Address / Reference
Reach Files, Version 3 (RF3)	Surface water hydrography	1:100,000	ARC/INFO coverages	http://www.epa.gov/OST/BASINS/download.htm[a] / U.S. EPA (1994b)
One degree digital elevation models (DEMs)	Elevations for creating DEM streams and watersheds	1:250,000	ARC/INFO coverages	http://edcwww.cr.usgs.gov/doc/edchome/ndcdb/ndcdb.html / USGS (1990a)
National Wetlands Inventory (NWI)	Lakes and wetlands	1:24,000	ASCII text file	http://www.nwi.fws.gov/download.htm / U.S. FWS (1995)
Geographic Information Retrieval and Analysis System (GIRAS)	Lakes, wetlands and rivers without NWI	1:250,000	ARC/INFO coverages	ftp://ftp.epa.gov/pub/spdata/EPAGIRAS / U.S. EPA (1994a)

[a] Dataset is also available as part of the EPA Office of Water's Better Assessment Science Integrating Point and Nonpoint Sources (BASINS), Version 2, software. More information about BASINS software is available online at http://www.epa.gov/OST/BASINS/download.htm.

required to develop the procedures for delineating the watersheds, watershed subbasins, and waterbodies to be modeled within the HWIR modeling framework:

- The area of interest for the HWIR risk assessment is the WMU area plus the area encompassed by a 2-km radius extending from the corner of the square WMU.

- Although the HWIR 3MRA models each of 419 site/WMU settings, data for waterbody and watershed layout were collected only once per site, within an AOI corresponding to the largest WMU present at each site.

- All order 1 through 5 waterbodies were passed to the model, except for headwater (order 1) stream reaches exiting the AOI.

- Waterbody reaches to be modeled by the waterbody model are order 2 and higher stream reaches, as defined by the Strahler ordering system (Strahler, 1957). All lakes, ponds, and wetlands connected to a waterbody network are modeled regardless of order. Waterbody reaches to be modeled were limited to those reaches lying completely or partially within the AOI.

- Order 3 and higher stream reaches, along with lakes, ponds, and certain wetlands, are assumed to support fish populations suitable for recreational anglers (i.e., are fishable).

- Each reach ordered 2 through 5 within the AOI will be modeled in its entirety or until it exits the AOI. This required extending the reaches intersecting the AOI boundary upstream to the waterbody network headwaters.

- Upstream watershed subbasins were delineated and passed to the HWIR model for all order 1 through 5 waterbody reaches within the AOI.

- Every unit of land within the AOI will be modeled for soil concentrations and runoff/erosion loads as part of a regional watershed subbasin except when no aerial deposition of chemical occurs (e.g., in the case of a metal or other nonvolatile chemical undergoing treatment in a tank or surface impoundment [for which there are no particulate emissions]).

Headwater Criterion

- The headwater criterion specifies where, in a headwater area, a stream (i.e., a defined drainage channel) starts. The final criterion used was 700,000 m^2, which provided a good match between DEM-delineated streams and RF3 stream reaches.

Additional, parameter-by-parameter assumptions may be found in U.S. EPA (1999g, 1999h).

4.2.5.3 <u>Uncertainties and Limitations</u>. Significant uncertainties and limitations for the site-based watershed and waterbody data include inconsistencies in scale for several data sources,

the resolution of the DEM data with respect to the AOI, data gaps from incomplete or inconsistent coverages in NWI and RF3 data, and site-to-site methodology differences necessitated by variability in both data and site characteristics. These uncertainties are discussed in the following sections, followed by uncertainties and limitations for the regional and national data.

Scale of Waterbody Data Layers. One of the uncertainties in the waterbody data collection effort is the use of different scale data to delineate both watersheds and waterbodies, as follows:

- DEM-generated streams were used to model streams at most sites (132). These were created from 1:250,000 scale DEMs. In another 69 sites, however, RF3-Alpha and Pacific Northwest Reach Files were used. These datasets have a nominal scale of 1:100,000.

- The three different lakes and wetlands coverages (NWI, RF3, GIRAS) used in the study had different scales (1:24,000, 1:100,000, 1:250,000) but were used interchangeably.

Ramifications of these differences and remedial measures taken include the following:

- The insertion of NWI, GIRAS, and RF3-Alpha lakes and wetlands features into the DEM stream network often caused spatial mismatches because of scale and/or registration differences. Often, for instance, lakes connected to RF3-Alpha streams did not connect to corresponding DEM-generated streams. Manual manipulation (e.g., moving a lake to connect to the DEM stream) was necessary to rectify such mismatches.

- NWI, at a larger scale (1:24,000) than either GIRAS or RF3-Alpha, often had many more wetlands features. Thus, sites without NWI coverage were biased low in terms of number of wetlands.

- Numerous, detailed NWI features posed challenges with respect to the overall scale and detail of the 3MRA (e.g., as reflected by the 100-m x 100-m grid resolution) and required manual manipulation. For example, small, adjacent features were combined and small (less than 20,000-m^2) isolated features were not included in the waterbody coverage.

- GIRAS and RF3 wetlands do not have the detailed identifiers that NWI features have. Thus, it is possible that some GIRAS and RF3 wetlands were included in the waterbody network that would not have been included if NWI coverages and descriptors had been available.

Scale of the DEM. One to 250,000 scale DEMs were used because 1:24,000-scale coverages are not yet available for the entire United States. This created a number of

scale-related problems for the analysis that can be rectified once higher resolution data are available.

The DEM-generated stream model sometimes did not match well with the RF3-Alpha data set. Generally the more relief that existed at the site, the better the match between the DEM-generated and RF3 waterbody networks. For flatter sites, the delineation software often did a poor job of following RF3-Alpha streams, creating cases of “bowling alley” or parallel streams and of streams that crossed perpendicular to RF3-Alpha streams. As a result, manual delineation was performed at many sites. The matching problem could be partially attributed to scale because 1:250,000 scale DEMs were used with a 1:100,000 RF3 stream network, which limits the detail that can be attained with a DEM, even when the DEM is conditioned to mimic the vector RF3-Alpha coverage.

All elevations represent an average elevation value in the 100-m by-100-m grid cell of the DEM. Because the DEM elevations were so coarse and the features of certain local watersheds (i.e., those for small WMUs) were so small in comparison, an intermediate local watershed coverage (L2) was created using the DEMs. From this L2 coverage, the actual local watershed was created geometrically using data out of the L2 coverage. Because the DEMs defined the L2 coverage and not the actual local watershed, the elevations for the L2 coverage were used for the local watershed.

Different Watershed Delineation Methods. Two different methods of delineating watershed subbasins were used in the HWIR study; a manual and a semi-automated DEM delineation. This probably created some site-to-site variability in delineation criteria, although the number and size of watersheds and waterbodies was generally similar from DEM- and hand-delineated sites.

Manual Processing of Data. Both the DEM and manual delineation methods involved some manual interaction, and manual data processing can be inconsistent. A watershed subbasin delineated by one staff member would not necessarily look exactly like another staff member's version at the same site. Choices such as pour point locations or what lake and wetlands features to include can result in slightly different outcomes. To minimize this variability, an informal guide and QC measures were developed and implemented to limit inconsistencies; however, any duplication of the process would encounter slightly different results based on choices made in the manual process. Some of the choices that were addressed in the guide and checked during QC include, for DEM-delineated watershed subbasins:

- Deleting overlapping areas of different watershed subbasins.
- Locating basin pour points.
- Altering DEM-generated subbasin boundaries to accommodate lakes and wetlands.

- Moving, deleting, or extending subbasin boundaries to solve problems such as fitting a lake in one watershed, or eliminating very small or "bowling alley" watersheds.

For waterbodies at both manual and DEM-delineated sites, choices included:

- Deciding which waterbodies to keep in the lakes and wetlands coverage, for instance, when wetlands features along the AOI boundary could be deleted to simplify the site delineation.

- Merging wetlands features to simplify delineation at a site. This involved moving the arcs from one feature to overlap another feature, deleting overlapping arcs, and deleting duplicate label points. Wetlands could differ in size, shape, or feature codes based on how the delineator performed this step.

- Moving waterbodies to fit within the DEM-delineated subbasins.

Although each of these actions was necessary to accommodate differences in scale and resolution for the source data sets and was appropriate in the context of a national analysis, professional judgment was involved that could lead to some inconsistencies. However, the QC measures were designed to minimize inconsistencies.

Incomplete National Wetlands Inventory. The NWI data layer currently used in the HWIR study is only partially complete for the United States. When available, NWI was used as the default data layer for lakes and wetlands. However, only half (100) of the 201 sites had NWI coverage. Sites without NWI coverage were covered (in much less detail) by GIRAS and RF3. As NWI data become available for additional sites, this could change results of the analysis, most significantly for ecological receptors.

Anthropogenically Altered Drainage. Anthropogenically altered stream networks were difficult to model with DEM-generated subbasins, the surface water model, and the 3MRA site layout. These networks include urban drainage that has been diverted into underground culverts and agricultural drainage by irrigation canals and ditches. For example, canals would split then rejoin as well as terminate on the GIS coverage as they enter underground drainage systems, and neither case can be modeled in the current 3MRA system. Most of these sites were manually delineated, with drainage networks modified as necessary to accommodate the 3MRA model requirements.

Missing Reach Type in Pacific Northwest Reach Files. The Pacific Northwest Reach Files were processed differently than the RF3-Alpha Reach Files. Notably, there was no REACHTYPE attribute in the Pacific Northwest files and a REACHTYPE of L was assigned to all lake and wetlands features, making all nonstream reaches lakes. Only seven of the HWIR sites are in the PNW region, so only lakes and wetlands at those sites were impacted.

Regional and National Data. The primary issues and uncertainties for both the regional and national surface water data arise from the lack of readily available site-specific data and are,

therefore, associated with the representativeness of the regional and national data for a particular site. However, the HWIR 3MRA is a site-based national analysis rather than a site-specific analysis; site-specific accuracy is not critical as long as the site-to-site variability is sufficient to characterize nationwide variability in model results (in this case, water and sediment concentrations from the surface water module). For regional data, site-to-site variability and accuracy was preserved to the extent practicable by keeping the region over which data were collected and statistics compiled as small as possible.

National data were collected only when site-based and regional data were not available. In these cases, national distributions, applied on a site-to-site or waterbody-to-waterbody basis, were used to represent national variability. It is not apparent whether this approach biased the model results in a consistent direction; EPA may investigate more site-specific data sources in the future to determine what, if any, bias might be associated with these data.

4.2.6 Soil Data

The HWIR 3MRA uses a site-based data collection strategy centered around 201 sites randomly selected from the 1985 Industrial D Screening Survey. Site-based soil data were collected for all 201 sites, largely using automated methods. Soil data are primarily used by the source, watershed, and vadose zone models, but fraction organic carbon is used by the farm food chain, terrestrial food web, human exposure, and ecorisk models. Additionally, the farm food chain model uses soil pH. This section addresses uncertainties associated with HWIR soil properties along with the land-use-based water erosion inputs, the universal soil loss equation (USLE) cover factor (C) and erosion control factor (P).

4.2.6.1 <u>Methods and Data Sources</u>. Figure 4-3 summarizes the HWIR data collection methodology for soils, which employs both GIS and conventional database processing to collect soil property data by map unit for watershed subbasin or WMU. GIS programs were used to identify and extract, for each Industrial D site modeled by the HWIR 3MRA, soil map units and map unit areas by watershed subbasin and by WMU. Database programs were used to extract soil data from the underlying databases by these map units and process them to generate soil properties by watershed for surface soil and by the WMU for the entire soil column (vadose zone or subsoil).

Surface soil is defined as the top 20 cm within the HWIR 3MRA modeling system. Vadose zone soil extends from the ground surface to the water table. Depending on the property, average (area- and depth-weighted) or predominant soil properties were derived for the soil depth zone of interest across each watershed subbasin or WMU. In addition, certain variables were derived, using national relationships, from site-specific soil texture or hydrologic group along with site-specific land use.

The primary sources for site-specific soil properties were the STATSGO database maintained by the U.S. Department of Agriculture (USDA, 1994) and two GIS-based compilations of STATSGO data, USSOILS (Schwarz and Alexander, 1995) and the Conterminous United States Multi-Layer Soil Characteristics (CONUS) data set (Miller and White, 1998). USSOILS, maintained by USGS, averages STATSGO data by map unit, with

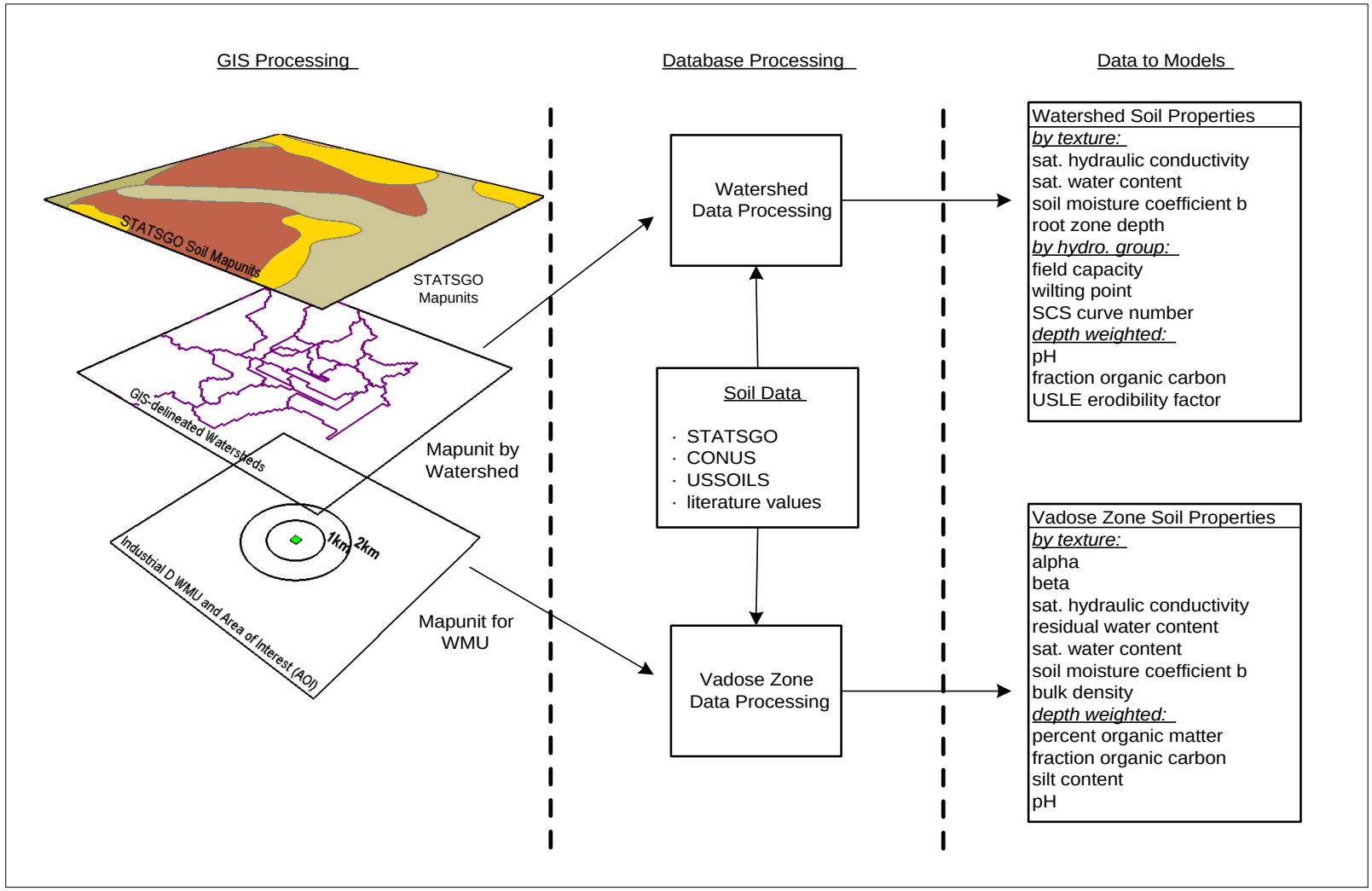

Figure 4-3. Overview of HWIR soil data collection methodology.

depth-weighted averages to the water table. CONUS, created by Pennsylvania State University, averages by STATSGO map unit and converts STATSGO soil layers into a set of 11 standardized soil layers extending to a depth of 2.5 m (60 in). The land-use-based universal soil loss equation (USLE) factors were obtained using lookup tables by Anderson land use codes obtained for each site from the GIRAS land use database (U.S. EPA, 1994a). National relationships for soil hydrological properties were obtained from Carsel and Parrish (1988), Carsel et al. (1988), and Clapp and Hornberger (1978).

Data from each of these sources were stored in a Microsoft Access database, including lookup tables for the nationwide relationships. Database query programs processed the soil data by watershed or WMU to generate the HWIR 3MRA model inputs. Parameters collected by the watershed map units included soil properties for the watershed model as well as those collected for the local watersheds in the LAU and wastepile models. Soil parameters collected by the WMU map units were used to provide subsoil data for the vadose zone model and subsoil and cover soil data for the landfill model. Additional detail on the soil data collection methodology is provided in Section 7.0 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999i).

4.2.6.2 Assumptions. In general, the site-specific soil data readily available from STATSGO and its associated databases are more than adequate for a national screening analysis like HWIR. One of the more significant assumptions used during the HWIR soil data collection is that soil properties are uniform and homogeneous areally across a watershed subbasin and with depth across the entire thickness of the vadose zone; the predominant or average values were used to characterize the soil for this national HWIR risk assessment. Soil properties do vary significantly on a much smaller spatial scale than the nationwide soil data in STATSGO, and even the STATSGO data provide resolution of soil properties with depth that was not used on this analysis.

Similarly, although the LAU and WP soil parameters were indexed by local watershed and subarea, they are the values for the watershed subbasin in which the local watershed resides. The STATSGO soil data do not have significant resolution to justify a separate collection of soil properties for the local watershed subareas.

4.2.6.3 Uncertainties and Limitations. Given the demonstrated quality of the soil data in STATSGO, the QA/QC measures designed to ensure effective data processing and transfer, and the national scale of the HWIR analysis, EPA believes that soil data are not a significant source of overall uncertainty in the 3MRA. The uncertainties associated with the soil data, however, are summarized here.

First, within the Generalized Soil Column Model in the nonwastewater source models, adsorption isotherms were approximated by treating the input adsorption isotherms for metals as random variables in the Monte Carlo sampling scheme. This ignores the possible dynamic effects of aqueous phase contaminant concentration, precipitation, dissolution, adsorption/desorption, and the geochemistry of the media (e.g., oxidation-reduction conditions) on the value of the adsorption isotherms and the fate and transport of metals in general.

Second, in a few cases, missing data were identified during QC of the original STATSGO data. In these instances, an infill value, representing the most common value nationally, was substituted. Infill values were used for sites with missing hydrologic soil groups and for four sites where the predominant soil type was organic materials. In total, infill values for soil properties were required for 25 sites. In most cases, infill values were necessary only for a couple of the watersheds at each site and there were no sites completely missing soil data.

A third uncertainty is the scale at which data were available for the local watershed. Because the scale of the STATSGO data (1:250,000) was much larger than that required to delineate the local watersheds, the soil properties for the local watershed and subareas were assumed to be the same as the regional watershed subbasin containing the local watershed.

A fourth uncertainty has to do with the correlations between hydrologic soil properties obtained from Carsel and Parish (1988) and Carsel et al. (1988). Although these were correctly sent to the 3MRA system in the input data set, the 3MRA Monte Carlo subroutine had problems using the correlations and early investigations suggest that the original data may not be correct. As a result, correlations between soil variables are not considered when variables are selected randomly by the 3MRA model. This could lead to unrealistic combinations of input parameter values.

Finally, two uncertainties associated with the land use data are lack of data currency and the scale of the land use data. The GIRAS land use coverages are from the mid-1970s to the early 1980s and may not accurately reflect current conditions at all sites; however, they are roughly contemporaneous with the 1985 Industrial D Screening Survey used to define the sites used in this analysis. The 1:250,000-scale GIRAS land use coverages also limit the spatial resolution of the land use data. However, GIRAS data are the same scale as the DEM and STATSGO coverages. Thus, although some resolution may be lost, the level of detail is fairly consistent across these data sources.

4.2.7 Farm Food Chain / Terrestrial Food Web Data

Data were collected to quantify parameters required by the Farm Food Chain (FFC) and Terrestrial Food Web (TerFW) modules to develop exposure profiles for receptors in these food webs. These parameters include bioaccumulation factors, partitioning coefficients, and ingestion rates that the modules use to estimate movement of constituents through food webs. The FFC module is used to predict concentrations in food crops, beef, and milk that are consumed by human receptors. The TerFW module is used to predict concentrations in plants, vertebrates, earthworms, and other soil invertebrates that are consumed by wildlife.

4.2.7.1 <u>Methods and Data</u>. Although the FFC and the TerFW modules are discussed separately, it should be noted that many of the parameters required by the FFC to estimate plant concentrations are also used by the TerFW.

Farm Food Chain. The FFC database was created using two sources:

- *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways* (U.S. EPA, 1999 [in press])

- *Parameter Guidance Document* (U.S. EPA, 1997g).

Data were compiled from these two sources and the remaining data gaps were addressed using other current references.

Terrestrial Food Web. In addition to the parameters on plant uptake used in the FFC module, the TerFW module requires empirical data on bioaccumulation in terrestrial prey items, including earthworms, other soil invertebrates, and vertebrates. A two-tiered search strategy was implemented to identify suitable bioaccumulation factors (BAFs): (1) a survey of compendia and databases on bioaccumulation in terrestrial systems, and (2) focused on-line literature searches to address data gaps specific to certain constituent/prey item combinations. Given the general paucity of data on bioaccumulation in terrestrial organisms, both field studies and laboratory data were used in developing suitable BAFs. Although field studies may be confounded by the conditions under which exposures occur (e.g., no controls), field data from well-designed studies generally offer a more realistic account of the potential for bioaccumulation in terrestrial biota.

4.2.7.2 Assumptions

Farm Food Chain. The following key assumptions were made in quantifying parameters for the FFC module:

- Contaminant loadings for plants occur through the following mechanisms: (1) wet and dry deposition of particulate-bound contaminants, (2) wet and dry deposition of contaminant vapor, and (3) uptake from soil and subsequent translocation into edible plant parts.

- All plants within a plant category (e.g., exposed fruits, protected vegetables) accumulate contaminants to the same degree.

- The default value of 1 used for the fraction of food items grown in contaminated soil assumes that the entire cattle diet is contaminated.

- Beef and dairy cattle accumulate constituents only through food, soil, and water ingestion.

Terrestrial Food Web. The following key assumptions were made in identifying BAFs for terrestrial prey items:

- It was assumed that empirical data relating tissue concentrations to soil concentrations reflected relevant exposure pathways for a given prey item. For

example, the BAF for small mammals incorporates ingestion of soil as well as ingestion of contaminated prey items (e.g., plants, soil invertebrates).

- The primary prey items consumed by terrestrial predators were terrestrial plants, small mammals, small birds, small herpetofauna, omnivorous vertebrates, earthworms, and invertebrates.

- Empirical data on bioaccumulation could be applied to species in the same prey category. For instance, a BAF for small mammals was applicable to shrew, mice, and small weasels.

- Uptake factors reflect central tendency values (i.e., geometric means) of the data collected. Data used to generate the geometric mean were assumed to represent a log-normal distribution.

- For some constituents, the bioaccumulation potential for terrestrial invertebrates was presumed to be similar to that of aquatic invertebrates. The data suggested a correlation between BAFs for emergent insects and BAFs for some flying terrestrial invertebrates.

4.2.7.3 <u>Uncertainty and Limitations</u>. This section reviews the key overall uncertainties as well as the parameter-specific uncertainties associated with the chemical-specific database and the exposure-related database developed to support the FFC and TerFW modules.

Chemical-Specific Database: Farm Food Chain. Values for a number of FFC parameters were the default values recommended by EPA for assessments lacking site-specific data. Although the application of default values is appropriate for national analyses, this practice introduces some uncertainty into the calculations for plant concentrations and decreases the overall resolution of the plant exposures. For example, the recommended default values used for milk and beef biotransfer factors were derived from regression analyses of pesticides data. Evidence suggests that these data may not be appropriate for organics with log K_{ow}s outside the prescribed range for milk (log K_{ow} range 2.8 to 6.5) and beef (log K_{ow} range 1.3 to 6.9). Experimentally derived values are uncertain to the extent that conditions of the study differ from conditions expected in the field. Within the constraints of these uncertainties, these values are the standard for estimating biotransfer relationships in the farm food chain and are accepted by EPA as reasonable estimates for risk assessment applications.

Soil-to-plant bioconcentration factors are strongly influenced by the chemical and physical properties of the soil as well as the plant species. Experimental uptake factors for metals derived from the plant uptake database were, in some cases, higher than previously applied uptake factors that were calculated using correlations developed by Baes et al. (1984). This was probably an artifact of including both field and greenhouse studies. In spite of the added conservatism generated by including greenhouse studies, the plant uptake database was preferred because it contains measured uptake factors that more likely reflect typical exposure scenarios for terrestrial plants.

Plant surface loss coefficients for particle-bound contaminants estimated values were taken from U.S. EPA (1997g) even though locally measured values are preferable due to the strong influences of local conditions. The U.S. EPA (1997g) noted that there was no speciation provided for mercury in the data used to derive this coefficient; therefore, the assumption was made that the values for each mercury species (total, elemental, and methyl) were equal. In addition, the plant surface loss coefficient for vapor-phase contaminants is uncertain because it is based on limited data.

Chemical-Specific Database: Terrestrial Food Web. The large number of data gaps in the TerFW database contribute to the uncertainty in estimating exposures to upper trophic level consumers in the terrestrial ecosystem. A default uptake factor of 1 was applied in cases where uptake factors were not available. The default value may be high for some constituents that do not significantly bioaccumulate. Data gaps introduce uncertainty into the analysis by limiting the ability to capture the true nature of ingestion exposure for some receptors.

The quality and quantity of studies that investigate bioaccumulation in terrestrial systems indicates that these values are associated with considerable uncertainty. As explained in detail in Section 10 (U.S. EPA, 1999l) of the data collection documentation, values for terrestrial BAFs reflect a broad range of data quality with some studies reporting only percentiles off of the raw study data (e.g., a 50th percentile from a rank order). Moreover, the bioaccumulation factors for a particular prey item vary across species, dietary preferences, seasonal resource requirements, and climatic conditions. For instance, Sample et al. (1998) indicated that vertebrates of varying dietary preferences (i.e., herbivores, omnivores, insectivores) accumulate contaminants to different degrees. Although bioaccumulation has been demonstrated for very few constituents in terrestrial systems, there is great uncertainty associated with the BAFs derived for prey items in this analysis.

For invertebrates, BAFs based on sediment exposure were adopted in the absence of data quantifying exposure via soil. This approach introduces uncertainty because these exposure pathways are not equivalent. The primary literature reports bioaccumulation data on various terrestrial insects such as beetles, especially for metals; however, these data are difficult (i.e., costly) to locate through traditional search methods because they are generally found as secondary assessments conducted within larger site-specific risk analyses. The BAFs adopted from Oak Ridge work (Bechtel Jacobs, 1998) represent the best alternative in the midst of current data limitations.

The BAFs for herpetofauna (i.e., reptiles and amphibians) were identified from primary literature searches based on sediment rather than soil exposures (Canton and Sloof, 1982). The fact that many species of herpetofauna are semi-aquatic, explains the abundance of data associated with sediment exposures. Because only sediment data were identified for herpetofauna, these factors BAFs were adopted until soil-derived uptake factors are identified; however, there is significant uncertainty in applying sediment exposures to estimate soil uptake values. In addition, the bioaccumulation data on herpetofauna were almost exclusively gleaned from studies on amphibians. Application of amphibian-based BAFs to reptiles is associated with great uncertainty given the physiological differences between these classes.

Exposure-Related Parameters. Uncertainties associated with the development of the exposure-related database are reviewed on a parameter-specific basis. Only those parameters associated with key uncertainties are reviewed.

Beef and dairy cattle consumption rates of forage, silage plants, and soil and water are EPA default values used to represent national averages for these parameters. The consumption rate for drinking water was derived from several studies covering a limited range of cattle and conditions. There are limitations associated with the use of representative default values since they which do not reflect the variability across the broader range of agricultural practices and climates. For example, the consumption rate of varies with region, physical conditions, climate, and breeds. Therefore, there is uncertainty associated with adopting average point estimates for these parameters.

The primary uncertainty associated with the fraction of wet deposition that adheres to plants is the paucity of data available to derive the value. The default parameter was derived using one study. The scarcity of relevant data generates low confidence in the default value generated. In spite of evidence suggesting that water acts to wash contaminants from the surface of the plant, the fraction of constituents that are actually removed has not been well quantified.

Available data indicate that interception fractions vary across vegetation types. Methods presented in U.S. EPA (1997g) suggest that site-specific interception fractions are preferred. However, this analysis is not site-specific. Rather, data are preferred that reflect national distributions. No suitable data were identified that represented national distributions for specific plant categories; therefore, default values were applied. Using a single value for all vegetation types introduces some uncertainty into the analysis.

The default values adopted to estimate the length of vegetation exposure to deposition have considerable uncertainty due to: (1) substantial regional variability in the duration of the growing season, and (2) insufficient data available to derive regional values (U.S. EPA, 1997g). This parameter is highly dependent on regional conditions, growing practices, and crop types. Because these factors were not accounted for in the derivation of this parameter, there is uncertainty in the values used.

In deriving default values for the empirical correction factors for plants, the assumption is made that all organic chemicals are equally lipophilic ,even though organic chemicals differ with regard to their lipophilicity. This simplification contributes to the uncertainty associated with the default values particularly for the less lipophilic organics that may have limited distribution into the interior plant tissues. This leads to conservative correction factors for less lipophilic constituents (U.S. EPA 1997g).

4.2.8 Aquatic Food Web Data

Data were collected to quantify parameters required by the aquatic food web module to develop exposure profiles for freshwater receptors. There were two databases developed in support of this module: the chemical properties database and the fish attribute database. The chemical properties database contains two types of parameters: (1) equation variables used in estimating uptake and accumulation of nonionic organic compounds into fish tissues using

chemical-specific properties (e.g., log K_{ow}, octanol/water partition coefficient) and (2) experimentally derived BAFs for ionic compounds, such as metals, that are not calculated using chemical properties. The fish attribute database was developed to characterize dietary preferences of aquatic biota and the type of aquatic habitats where they are likely to be found (e.g., warmwater streams). This database also includes the life history attributes that influence the exposures of fish and other aquatic biota. Life history parameters such as fish body weight, tissue lipid fraction, tissue water fraction, and common prey items are identified. This database supports the human health exposure analysis by identifying the types of fish that are likely to be consumed by humans.

4.2.8.1 Methods and Data Sources. Various sources were investigated to compile the databases for the aquatic food web. Each database was constructed to reflect different degrees of variability. The chemical properties database was compiled to reflect average constituent uptake factors for individual biota in the aquatic food web. In the case of the fish attribute database, the aim of the data collection effort was to represent the variability of parameters at the national level across different waterbody types (e.g., lakes and streams) and surface water temperatures (e.g., warm and cold). To accomplish these tasks, various government agencies, EPA offices, research laboratories, and primary literature sources were consulted to review publications, databases, and guidance documents to support the development of the aquatic food web databases. Section 11 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999m) describes these data sources and the data collection methodologies in detail.

Chemical Properties Database. The chemical properties database contains parameters used by the module that are related to chemical uptake in aquatic biota. Chemical uptake in fish and other prey items in the aquatic food web is a critical link in estimating exposures of terrestrial organisms foraging primarily in aquatic food webs. The uptake and accumulation of chemicals into aquatic biota can be predicted using models or measured through laboratory exposure experiments. In this analysis, measured BAFs were identified for metals and other constituents that were not amenable to the mechanistic models used for organic chemicals (e.g., PAHs, and mercury compounds). For organics, two different models were applied to estimate uptake into fish tissues, depending on the hydrophobicity of the chemical (i.e., Bertelsen et al., 1998, and Gobas, 1993). In short, the bioaccumulation factors for organic chemicals were predicted by the module and reflect the aquatic habitats (e.g., pond, stream) and water quality conditions (e.g., DOC) modeled at each site. Empirical data on the bioaccumulation potential for other constituents were reviewed, the BAFs were evaluated for data quality, and appropriate values were entered into the database.

Data collection efforts were focused on BAFs measured through laboratory or field exposures for the following categories of aquatic organisms: aquatic macrophytes, benthic filter feeders, and trophic level 3 (T3) and 4 (T4) fish. Where possible, the BAFs for T3 and T4 fish include both whole-body and filet values. For metals, uptake factors identified for whole body were also applied as fillet uptake factors.

The criteria for data selection presented in U.S. EPA (1999m) were used to ensure consistency in the interpretation and selection of appropriate BAFs. For each category of aquatic organisms, the geometric mean of BAFs identified in the literature was calculated and entered

into the database as the BAF for that constituent. Bioaccumulation data were assumed to be lognormally distributed; therefore, the geometric mean was the most appropriate central tendency estimate.

Fish Attribute Database. This database was used to reflect the variability in the parameters of fish body weight, lipid fraction (whole body and muscle), and prey preferences; the means for water fractions in whole body fish tissues and filet fractions; and the fish categories (e.g., T3 large benthivores, T4 piscivores) in the aquatic food web likely to be consumed by humans. Because certain parameters assessed in the fish attribute database can vary significantly across different waterbodies, water temperatures, and habitat structures, these parameters were characterized across various aquatic ecosystems.

The raw data on fish attributes were reviewed and processed prior to entry in the fish attribute database in an effort to capture the variability introduced into the body weights, lipid content (total body and muscle), and dietary preference parameters resulting from fish that are distributed across different environmental conditions (see U.S. EPA [1999m] for an expanded discussion). In summary, species-specific data for these parameters were collated into groups based on trophic level (i.e., 3 or 4), relative fish size (i.e., small, medium, or large), water temperature preference (i.e., warm or cold), habitat preference (i.e., stream, pond, lake, wetland), and dietary classification (i.e., benthivore, zooplanktivore, omnivore, and piscivore). Descriptive statistics (e.g., mean, minimum, and maximum) were derived for body weight, fraction lipid, and dietary preferences for adult fish of specific trophic levels and habitat classifications. For example, a mean body weight was derived for small T3 benthivores in warmwater lakes. Values for the parameters of fish body weight, fish lipid content (whole body), minimum prey preference, and maximum prey preference were derived (i.e., means, minima, or maxima).

Other parameters were adopted from various EPA documents and standards for evaluating risk in aquatic ecosystems. Values for fish lipid content (muscle) and filet fraction were taken from data presented in the *Great Lakes Water Quality Initiative* (U.S. EPA, 1995e). The fish water fraction, based on the whole fish, was adopted from the *Wildlife Exposure Factors Handbook* (U.S. EPA, 1993b). The analogous variable that estimates water fraction in muscle tissue was adopted from Bertelsen et al. (1998). A variety of sources were surveyed to identify the kinds of T3 fish (i.e., T3 edible fish) that people are likely to catch and eat (U.S. DOI, 1996; U.S. EPA, 1996e). Based on these sources and professional judgment, edible fish categories for T3 fish were designated for each aquatic habitat.

4.2.8.2 <u>Assumptions</u>. The following key assumptions were implicit in identifying data to quantify parameters for the aquatic food webs. These assumptions and relevant uncertainties are explained in greater detail in U.S. EPA (1999m). The assumptions are presented separately for each database.

Chemical Properties Database.

- Bioaccumulation data were assumed to be lognormally distributed for a given constituent/prey item combination.

- Although it is recognized that aquatic macrophytes and benthic filter feeders may be exposed through the sediment, the database does not include sediment-to-biota accumulation factors (BSAFs). In effect, the implicit assumption is that these aquatic biota are not exposed through the sediment. As implemented, the Aquatic Food Web module does not require BSAFs; however, this capability could be added now that suitable sediment data have been identified.

- For metals, it is implicitly assumed that water quality characteristics do not significantly influence the bioaccumulation of constituents in fish. The empirical BAFs are generally based on total concentrations in surface water and are not adjusted for differences in water quality (e.g., humic acids). Ideally, the BAFs would be derived based on freely dissolved metal to adjust for differences in surface waters.

- The BAFs for metals presume that the uptake and accumulation into fish is a linear phenomenon. Studies have shown that this process is likely to be nonlinear, particularly for metals that are essential nutrients.

- It was assumed that BAFs for metals in fish, based on whole body, provide a reasonable approximation of the BAF for fish filet. Metals tend not to partition into a specific tissue type (lipid, muscle, gills); rather, they are evenly distributed in fish tissues. The exception is methylmercury, which partitions almost 100 percent of the muscle tissue in fish.

Fish Attribute Database.

- Prey items in the aquatic food web were limited to the following categories: periphyton, aquatic macrophytes, phytoplankton, zooplankton, benthic filter feeders, benthic detritivores, and fish.

- Data compiled from multiple regions were presumed to be reasonable estimates of national distributions. Sufficient data across numerous regions were identified only for fish body weight, fish lipid content (whole body), minimum prey preference, and maximum prey preference.

- Data collected for fish were assumed to represent the adult life stage of fish species.

- Parameters included in this database were presumed to vary across trophic level, relative fish size, habitat preference, water temperature, and dietary preferences.

4.2.8.3 <u>Uncertainties and Limitations</u>. There are a number of uncertainties inherent in the development and application of data in the chemical properties database and the fish attribute database. These uncertainties are addressed separately for each database.

Chemical Properties Database. Developing the chemical properties database was limited by data availability; appropriate data could not be identified for all categories of aquatic prey for constituents that required measured bioaccumulation factors. In many cases, uptake and accumulation data were measured for aquatic macrophytes and benthic filter feeders based on sediment exposures rather than surface water exposures. Because uptake factors were not identified for many combinations of prey items and constituents, uncertainty is introduced into the assessment of food web exposures.

When bioaccumulation data were not available, a default value of 1 was entered into the database. Default values should not be construed as estimates of bioacccumulation potential; rather, they should be viewed only as place holders until data are identified. The risk results need to be considered within the limitations of the default values.

In generating BAFs for the chemical properties database, the central tendency estimates were calculated. There is significant uncertainty in using a point estimate for bioaccumulation potential given the wide variability in uptake and accumulation across various species (including life stages) and water quality conditions. In addition, because many distributions were based on only a few values, the central tendency estimates may be greatly influenced by extreme values (e.g., outliers) or clustered data.

Fish Attribute Database. The key uncertainties associated with the development of the fish attribute database as a whole can be grouped into the following general categories: default values and data gaps, fish body weight uncertainty, and decision criteria adopted to sort and manipulate the data.

There is uncertainty regarding how to predict fish concentrations of dioxins, and methods are available to make such predictions other than the food web approach used in this modeling. For example, another approach uses modeled sediment concentrations and then applies a biosediment factor to predict fish tissue concentrations.

The values derived for lipid fraction (whole body and muscle) and dietary prey preference were developed from a relatively diverse data set. To derive a lipid value, the lipid fractions across T3 and T4 fish, respectively, were independently averaged. This included all the lipid values derived from the GLWQI (U.S. EPA, 1995e) document in addition to primary literature studies on a variety of species and aquatic habitats. When lipid data were not available for a fish category in an aquatic food web, this average value for the lipid was used. In the case of dietary fractions, the description of what food items typically comprise fish diet varied substantially across different sources. In some instances, no information was provided on the actual dietary fractions—only the general components were known (e.g., aquatic plants, zooplankton, and benthos). In these cases, the feeding guild to which the species was assigned (e.g., benthivore) was used to inform professional judgment in selecting appropriate prey preferences. In this example, we may have selected 0.5 for benthos and 0.25 for aquatic plants and zooplankton, respectively. If no data were identified to suggest the inclusion of a particular aquatic prey item, it was assumed that the prey item was not eaten by the species in question and the null value (-999) was used.

Because a large portion of the fish body weight data were taken from the National Ambient Water Quality Assessment (NAWQA) studies, any uncertainties associated with this database influence the uncertainty in the fish attribute database. Body weight data identified through the NAWQA appear to be low for some species. The lower body weights reported could be due to any number of factors. The fish population sampled may have included juvenile fish. A review of the NAWQA biological sampling method did not explicitly define fish collection by age. Further, many of the NAWQA sample sites were located in streams, ponds, and swamps, which larger fish may not inhabit as frequently. These discrepancies may have resulted in a body weight distribution skewed toward lower values. Applying these body weight values introduces some uncertainty into how well the variability in body weights is represented.

The decision criteria adopted to aggregate and process raw data were based on professional judgment as to the most appropriate use and interpretation of existing data. For instance, fish species were often assigned to cold or warm water habitats based on a weight-of-evidence approach when definitive study data on temperature preferences could not be identified. In these cases, STORET temperature measurements taken at multiple sampling locations across different regions and drainage basins were assessed for maximum temperatures, defined as the 95th percentile value for the region of interest. If a fish species was found predominantly in regions where the maximum temperature exceeded 25 ^{o}C, then the fish was presumed to be warm water. There is some uncertainty associated with these assignments because a fish assigned as warm water could possibly tolerate both cold and warm surface water temperatures. Because this database was developed to delineate preferences (not absolutes), the weight-of-evidence approach offered a reasonable and transparent method to sort and manipulate data. Other assumptions and professional judgments used in the preprocessing effort were outlined within the preprocessing parameter discussions concerning trophic level, relative size, and habitat preferences in U.S. EPA (1999m).

4.2.9 Human Exposure Factors

Exposure factors are used in the human exposure module and the human risk module of the HWIR 3MRA Model to calculate the dose of a chemical (in mg/kg of body weight/d) based on contact with contaminated media or food, the duration of that contact, and the body weight of the exposed individuals. The human exposure module calculates exposures to two basic human receptor types (residents and farmers) from media and food concentrations calculated by other modules. Residential receptors may be recreational fishers in addition to being a resident or home gardener. Farmers may be beef farmers or dairy farmers, and either type of farmer also may be a recreational fisher. Within each of the two basic receptor types, the human exposure module calculates exposures for five age cohorts: infants (ages 0 to 1 year), children ages 1 to 5 years, children ages 6 to 11 years, children ages 12 to 19 years, and adults (ages 20 years or older). We then combine the cohorts into 3 groups: less than 1 year old; 1 to 12 years old; and 13 years old and older.

The media inputs needed for the human exposure module include ambient air concentration (both vapor and particulate), soil concentration, groundwater concentration, exposed vegetable concentration, protected vegetable concentration, exposed fruit concentration, protected fruit concentration, root vegetable concentration, beef concentration, milk

concentration, and fish filet concentration for trophic level 3 and trophic level 4 fish. For vegetables and fruits, the terms "exposed" and "protected" refer to whether the edible portion of the plant is exposed to the atmosphere.

Exposure to humans other than infants may occur through eight pathways:

- Inhalation of ambient air
- Inhalation of shower air
- Ingestion of groundwater
- Ingestion of soil
- Ingestion of fruits and vegetables
- Ingestion of beef
- Ingestion of milk
- Ingestion of fish.

However, not all receptors are exposed through all of these pathways. Residents are exposed through inhalation of ambient air, inhalation of shower air, ingestion of groundwater, and ingestion of soil. Home gardeners have the same exposures as residents, plus exposure through ingestion of fruits and vegetables. All farmers are exposed through inhalation of ambient air, inhalation of shower air, ingestion of groundwater, ingestion of soil, and ingestion of fruits and vegetables. In addition, beef farmers are exposed through ingestion of beef, and dairy farmers are exposed through ingestion of milk. Recreational fishers have the same exposures as one of the other receptor types, plus fish ingestion. Not all age cohorts are exposed through all pathways–shower exposures are calculated only for adults and children ages 12 to 19 years.

The human exposure factors used in the HWIR 3MRA model were selected and developed to characterize the consumption rates, exposure durations, and contaminated fractions necessary to calculate the receptor-specific exposures described above. In all cases they represent estimates of nationwide variability, being based on national data provided in the *Exposure Factors Handbook* (EFH) (U.S. EPA, 1997d, 1997h, and 1997i). Where appropriate, distributions are used in the HWIR analysis to capture this variability.

4.2.9.1 Methods and Data Sources. Table 4-6 lists the exposure factors used in the HWIR risk analysis, the data sources, and whether distributions (stochastic variables) or fixed values (constants) were used. The age cohorts used for HWIR were selected because a majority of the data in the EFH were provided in a similar manner, and the use of these age groups in this analysis (for all receptor types) reduced the need to manipulate the data sets. In addition, other human risk analyses performed by EPA's Office of Solid Waste (OSW) have used these same age cohort definitions.

The primary data source for HWIR human exposure factors was the EFH. In general, one of three methods was used to develop these factors:

1. When EFH percentile data were adequate, maximum likelihood estimation (MLE; Burmaster and Thompson, 1998) was used to fit selected parametric models (gamma, lognormal, Weibull, and generalized gamma) to the EFH data. The

Table 4-6. Input Parameters and Data Sources: HWIR Human Exposure Factors

Parameter	Data Source
Distributed (stochastic) variables	
Body weight (adult, child1-4)	U.S. EPA (1997d)
Inhalation rate (adult, child1-4)	U.S. EPA (1997d)
Ingestion rate: soil (adult, child2-4)	U.S. EPA (1997d)
Ingestion rate: drinking water (adult, child1-4)	U.S. EPA (1997d)
Breast milk consumption (child1)	U.S. EPA (1997h)
Consumption rate for gardener: exposed vegetables (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for farmer: exposed vegetables (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for gardener: root vegetables (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for farmer: root vegetables (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for gardener: protected vegetables (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for farmer: protected vegetables (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for gardener: exposed fruit (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for farmer: exposed fruit (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for gardener: protected fruit (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for farmer: protected fruit (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for recreational fisher: fish (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for farmer: beef (adult, child2-4)	U.S. EPA (1997h)
Consumption rate for farmer: milk (adult, child2-4)	U.S. EPA (1997h)
Shower contact time	U.S. EPA (1997i)
Total time in shower and bathroom	U.S. EPA (1997i)
Fixed variables (constants)	
Exposure frequency (adult, child1-4)	EPA policy
Exposure duration (adult, child1-4)	U.S. EPA (1997i)
Fraction contaminated: soil	EPA policy
Fraction contaminated: drinking water	EPA policy
Fraction contaminated for recreational fisher (fish)	U.S. EPA (1997h)
Fraction homegrown for gardener (exposed vegetables, root vegetables, protected vegetables, exposed fruit, protected fruit)	U.S. EPA (1997h)
Fraction homegrown for farmer (exposed vegetables, root vegetables, protected vegetables, exposed fruit, protected fruit)	U.S. EPA (1997h)
Fraction contaminated (home-raised) for farmer (beef, dairy)	U.S. EPA (1997h)
Event frequency–showering	U.S. EPA (1997i)
Fraction of fat in maternal breast milk	U.S. EPA (1997h)
Fraction of T3 fish consumed	U.S. EPA (1997h)
Fraction of T4 fish consumed	U.S. EPA (1997h)

chi-square measure of goodness of fit (Bickel and Doksum, 1977) was then used to choose the best distribution to assume for HWIR. Parameter uncertainty information (e.g., for averages, standard deviations) also was derived using the

asymptotic normality of the maximum likelihood estimate or a regression approach.

2. When percentile data were not adequate for model fitting, models were selected on the basis of results for other age cohorts or, if no comparable information was available, by assuming lognormal as a default distribution and reasonable coefficients of variation (CVs).

3. Variables for which data were not adequate for either (1) or (2) above were fixed at EFH-recommended central tendency values or according to established EPA policy. In general, variables were fixed because limited variability was expected or because available data were not adequate to generate national distributions.

For most variables for which national distributions were developed, exposure factor data from the EFH were analyzed to fit selected parametric models. Steps in this process included preparing data, fitting models, assessing fit, and preparing parameters to characterize distributional uncertainty in the HWIR model inputs. Section 8.0 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999j) describes each of these steps in detail.

4.2.9.2 <u>Assumptions</u>. Key assumptions used for development of HWIR exposure factor data are as follows:

- All human exposure model inputs were collected and processed on a national basis using EFH data. Site-specific data were not available, and, although food consumption rates and exposure duration data are grouped by four regions (Northeast, Midwest, South, and West) in the EFH, similar regional distributions were not available for other human exposure factors. In addition the regional data did not include age-specific data necessary to develop distributions by age cohorts.

- Distributions were developed independently for each variability. The variables were assumed not to be correlated.

- Because the EFH data are always positive and almost always skewed to the right (i.e., have a long right tail), three two-parameter probability models commonly used to characterize such data (gamma, lognormal, and Weibull) were fit to the data. In addition, a three-parameter model (generalized gamma) was used to allow for a likelihood ratio test of the fit of the two-parameter models.

- In the cases (soil ingestion, breast milk consumption, and inhalation rate) where EFH data were not adequate to fit a distribution, the lognormal model was assumed as a default.

- Several fractional variables (fraction beef, milk, or fish contaminated; consumption fractions) were fixed at EFH-recommended values, assuming that variability was not great enough to warrant a distribution.

- Standard EPA default values were assumed for exposure frequency and duration, in accordance with EPA policy. This included a 1-year exposure duration for noncarcinogens and the assumption that, for residents and farmers, exposure duration for carcinogens was equal to the average population mobility (9 years) reported in the EFH.

4.2.9.3 Uncertainties and Limitations. Significant uncertainties and issues in the HWIR human exposure data collection effort can be organized by the following headings: national versus regional distributions or site-based data, variable independence, source data, models (distribution selection and fixed variables), estimation methods, goodness-of-fit tests (parametric versus nonparametric approaches), and uncertainty issues. Uncertainties associated with the EFH data are not discussed for individual variables but can be found in the EFH (U.S. EPA, 1997d, 1997h, 1997i).

National versus Regional Distributions or Site-Specific Data. Available data did not allow stratification of data by regions, and collection of site-specific exposure factor data is clearly beyond project resources of this effort. However, it is likely that there is regional and local variability in exposure factors due to differences in farming or gardening practices, dietary preferences, climate, income, employment, and other socioeconomic variables. However, while recognizing that not considering site-specific conditions could impact the accuracy of site-specific results, EPA believes that, in the context of the national scope and framework for this analysis, the national distributions used are appropriate and adequate.

Variable Independence. The exposure factor distributions are developed and used by the model assuming they are independent (i.e., no correlations exist between variables). EPA recognizes that this is likely not the case for certain exposure factors, and that independent specification and sampling of distributions could result in unrealistic combinations of variables but does not have the information currently to develop or evaluate the significance of cross-correlation statistics. EPA may investigate possible correlations between variables and impacts on model results as a science support activity.

Source Data Uncertainties. For most exposure factors addressed, data analyses involved fitting distributions to selected percentiles. EPA believes that little information is lost by fitting to percentiles versus fitting to raw data but recognizes opinions that such analyses should always be based on raw data, synthesizing all credible sources. EPA may pursue settling this issue as a scientific support activity for selected parameters and also may consider including additional percentiles in future editions of the EFH to provide better data for determining the best fit.

Similarly, the percentiles for fitting the standardized age cohorts could have been obtained by fitting distributions to the original groups, generating simulated data from the fitted distributions, mixing the simulated data in proportion to the subgroup sizes, and then fitting the distributions again. Mixing proportions for this exercise could be determined by the demographics of the population (e.g., using U.S. Bureau of the Census data) of interest for the risk assessment rather than the original study sample sizes. EPA may consider this as a science support activity to test the uncertainty of the method used.

The EFH data sets for time spent in shower [showerT] and cumulative time spent in the bathroom [cumTroom] clearly are affected by rounding and grouping of data. The fitting methods do not account for these sources of inaccuracy but could be developed and explored depending upon the significance of these input variables.

Statistical Model (Distribution) Uncertainty. Three standard two-parameter probability models (gamma, lognormal, and Weibull) were used for this analysis. These distributions are special cases of a three-parameter model (generalized gamma) that contains them and allows for a likelihood ratio test of the fit of the two-parameter models. Other models are possible (e.g., Myers et al., 1998), but EPA believes that this simple setup offers a considerable improvement over using a lognormal model in all cases and is appropriate for this analysis. In support of this conclusion, the three-parameter generalized gamma module did not significantly improve on goodness of fit over the two-parameter models in 58 of 59 cases at the 5 percent level of significance.

In the cases where fixed values (constants) were assumed because of lack of percentile data, nondegenerate probability distributions could be assigned by assuming a reasonable minimum CV (based on available data for similar variables) and using a default distribution type such as lognormal or gamma. For variables that can have significant variability and impact on the analysis results (i.e., significantly affect risk estimates), specifying a minimum positive standard deviation would improve uncertainty estimation for the analysis. However, EPA does not believe that the variables that were fixed (mostly fractions) are, on the whole, subject to significant variability in terms of model results. Exposure duration may be a possible exception, and EPA will explore the sensitivity of the 3MRA model to likely ranges of variability in the duration variables.

Estimation Methods. The MLE method of estimating uncertainty parameters is generally considered the best approach currently available for most situations. There may be room for improvement in certain cases, however. Data provided in U.S. EPA (1999j) shows that the maximum likelihood estimates for the means and standard deviations agree with the data means and standard deviations much better for the gamma and Weibull models than for the lognormal model. For example, even in cases where the lognormal model fits best, the gamma estimate of the mean is often closer to the data mean than is the lognormal mean (i.e., this suggests that the lognormal maximum likelihood estimates of the mean and standard deviation can be biased). Truncation might reduce this problem, but if applied, truncated models should be fit to the data rather than fitting a model and then truncating the distribution.

Testing Goodness of Fit. Although they offer significant improvement in objectivity over visual estimation, goodness-of-fit tests are subject to some uncertainty that should to be considered in their application. One area of concern is uncertainty about how the survey statistics in the EFH were calculated. All of the statistics used to assess goodness of fit for HWIR assume a random sample, which may or may not be a valid assumption for EFH data. Many of the EFH data sources are surveys that, in many cases, are likely not to involve purely random samples. Rather, they use clustering and stratification, primarily for economic reasons. In such cases, the calculation of estimates and their standard errors, as well as test statistics, should use the survey weights and should take the study design (e.g., clustering and stratification) into

account. The EFH mentions that the SAS system was used for calculation of statistics. If the SAS UNIVARIATE procedure was used to calculate percentiles, then the percentiles are unweighted.

If the random sample assumption is not valid, the likelihood ratio test may be more valid than the chi-square test used in this analysis. Valid chi-square statistics can be devised for clustered or stratified sampling designs, but they require raw data and information on the survey design and weights. One way to avoid some of the difficult goodness-of-fit issues is to use empirical distributions (bootstrapping) when the raw data are available. This nonparametric approach, however, is less convenient for risk assessment simulations than using simple parametric probability models.

Treatment of Uncertainty. Regarding statistical treatment of uncertainty, the situation is less clear than for estimation, where there is a fairly clear consensus in favor of the maximum likelihood estimate. Relying either on the asymptotic normality of the maximum likelihood estimate or on likelihood-based contours to get parameter uncertainty distributions can be problematic when data do not exactly fit the model, which is, unfortunately, always the case (Box, 1976). A partial remedy for this problem was pointed out by Huber (1967); White (1982) offers a more recent and more readable account.

The pursuit of this theme leads to variance estimates that absorb (are inflated by) model lack of fit and that, therefore, automatically take some account of model uncertainty as well as parameter uncertainty. This finding has led to a substantial body of recent practical statistical methodology under the name of robust sandwich estimators of variance. As far as this analysis has seen, this area of study has received little attention in the risk assessment community. In one sense, it is a treatment for an affliction mentioned by Hattis and Burmaster (1994): "The application of standard statistical methodology to a single data set will nearly always reveal only a trivial proportion of the overall uncertainty." Regression estimates of uncertainty may be investigated as a means of addressing this problem.

4.2.10 Human Receptor Data

Human receptor points, which include residences and farms, are one of the primary spatial data coverages in the HWIR 3MRA data. Receptor data were collected within an area extending 2 km from the edge of each WMU modeled at the 201 sites, resulting in 419 unique setting coverages. For each of these settings, U.S. Census and other data were used to locate exposure points of concern for individual human risk and to attribute these points in terms of the population by receptor type and age. The 3MRA model uses these points and associated data to calculate human risk where people are likely to be located around a WMU and weight these estimates by population.

The human receptor variables collected for residences and farms surrounding each site are used primarily by the human risk module, but location and site layout information is also used by the air, farm food chain, aquifer, and human exposure modules to determine the points and areas for calculating exposure concentrations. For each HWIR setting, human resident and farmer population data are dimensioned by the number of receptor points or farms (index 1), eight

receptor types (index 2), and five age cohorts (index 3). Four receptor types are used to characterize residential exposure and risk:

- Residents
- Resident home gardeners
- Resident recreational fishers
- Resident home gardener/recreational fishers

and four receptor types for exposure and risk to farmers:

- Beef farmers
- Dairy farmers
- Beef farmer/recreational fishers
- Dairy farmer/recreational fishers.

Each of these eight receptor types is further dimensioned by five age cohorts:

- Child1 (infant): younger than 1 year
- Child2: 1 to 5 years
- Child3: 6 to 11 years
- Child4: 12 to 19 years
- Adult: 20 years or older

resulting in 40 receptor type/age cohort combinations.

The HWIR 3MRA estimates exposure and risk for three concentric rings defined by their distance from the WMU boundary: 500 m, 1,000 m, and 2,000 m. Human receptor and farm layout data include ring indices that indicate the ring that each receptor point and is in.

4.2.10.1 <u>Methods and Data Sources</u>. Human receptor data were assembled from the primary data sets shown in Table 4-7 using GIS technology (U.S. EPA, 1999k). The data were collected using a site-based approach, by applying national and regional receptor type information (i.e., county agricultural survey data on beef and dairy farmers, state data on recreational fishers, and a national gardener percentage) to site-specific population (U.S. Census) and land use (GIRAS) data to estimate receptor type/age cohort percentages for each human receptor point. The GIS was used to locate human receptor points and collect and analyze data on the population numbers and characteristics for the 201 study sites. The GIS allowed the use of different data layers with different scales (e.g., state, county, census block group, census block) in the collection effort. Figure 4-4 illustrates the automated GIS data collection process.

County-level agricultural census data, census block group data, census block data, and land use data are the primary spatial data layers used in the collection of human receptor data. For HWIR, resident human receptor points were located at the centroid of each census block / ring polygon; that is, a block crossing a ring boundary establishes two receptor points. In all

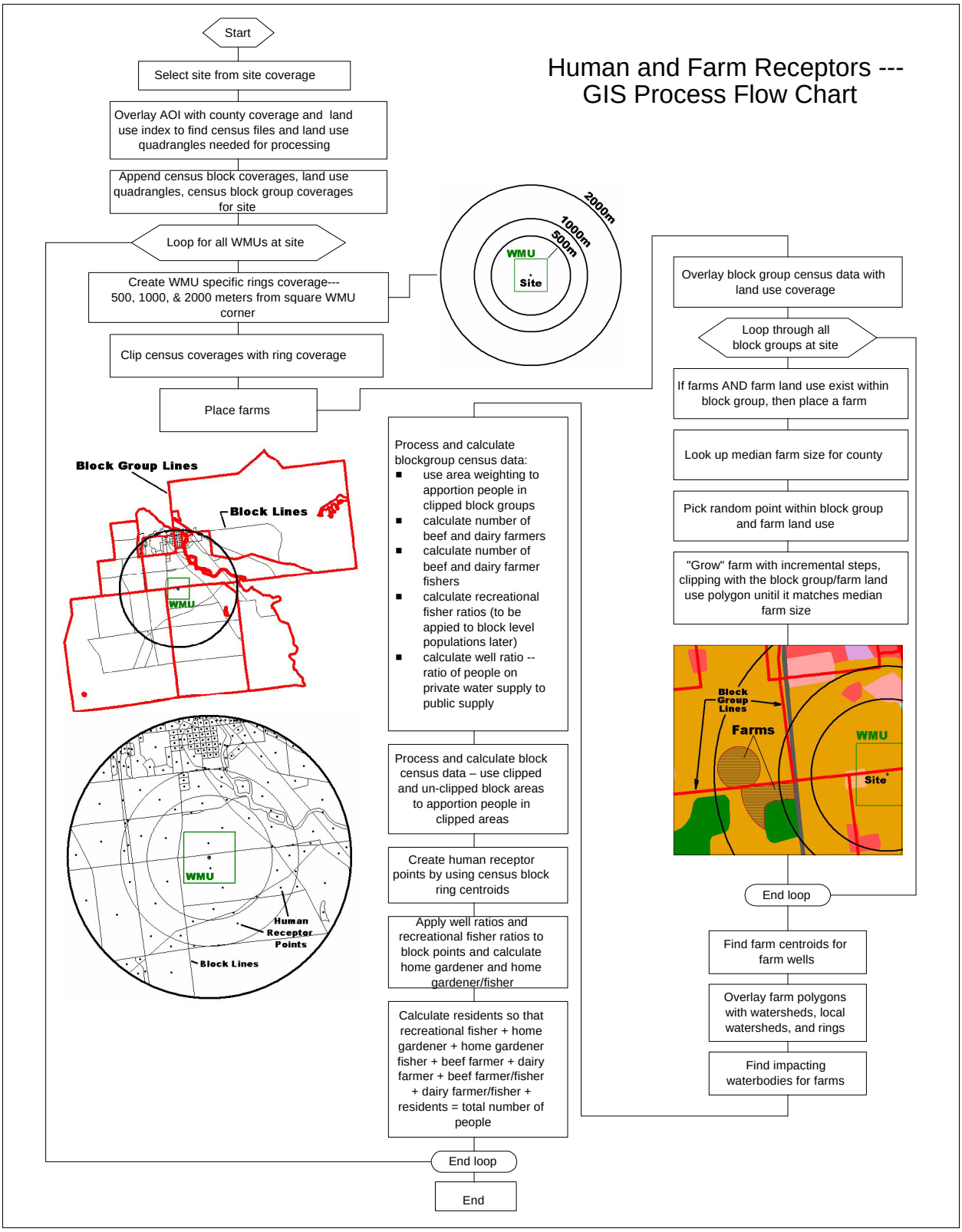

Figure 4-4. Human and farm receptor GIS process flow chart.

Table 4-7. Primary Human Receptor Data Sets, Date, and Scale

Data/Receptor Type	Data Set	Date	Scale
Location	Industrial D Screening Survey (facilities); Envirofacts (locations)	1985 (Ind. D) 1996 (Envirofacts)	1:100,000 scale mapping
Residents	Census block data STF 1B attribute data Census TIGER/line block coverages	1990	1:100,000 scale mapping
Beef and dairy farmers, farm size	Census block group data STF 3A attribute data Census TIGER/line block group coverages	1990	1:100,000 scale mapping
	GIRAS land use data (USGS, 1990b)	late 1970s to early 1980s	1:250,000 scale mapping
	Census of Agriculture	1987 and 1992	County level
Recreational fishers	National Survey of Fishing, Hunting, and Wildlife	1992	State level

cases, human receptor points were excluded from the WMU itself. In the few cases where block centroids fell within the WMU area, the corresponding receptor point was moved just outside of the WMU. Farms were located and populated using census block group boundaries, subdivided by farm land use, along with county-level agricultural census data.

Human receptor point and farm data were placed and processed within one GIS Arc Macro Language (AML) batch process program for each site/WMU setting. This program uses area-weighting to calculate population attributes for a particular study area for a site. For example, if the AOI boundary clips a block group so that only 10 percent falls within the AOI, then the total for all census items for that block group are multiplied by 0.1. Or, if a census block falls across a ring boundary, the population is split between the two human exposure points that result using the fractional area of the block falling within each ring. This area-weighting method was used to calculate census numbers within the study area for both block and block group data applied to each human receptor point. County beef and dairy farmer percentages, state recreational fisher fractions, and the national gardener fraction were applied equally across all receptor points at a site.

The methodology did not permit double-counting of receptors across type or age cohorts. Thus, the beef farmer fishers are not a subset of beef farmers. In other words, if the population of farmers and residents for a site is totaled up across all eight receptor type and five age cohorts, it will equal the total U.S. Census population, area-weighted as described above, across the entire 2-km-radius AOI.

4.2.10.2 <u>Assumptions</u>. The following assumptions were used in collecting and processing human receptor data:

- Population attributes are uniformly distributed across a given block or block group area.

- Older data sets (1990 Census, 1980s land use) adequately represent future conditions at a site.
- County-level, state-level, and national data apply equally across all receptor points at a site.
- Receptor location is represented by census block centroid.
- Average Census of Agriculture data from 1987 and 1992 best match 1990 U.S. Census population data.
- Farm size is the median acreage in the county agricultural census.
- Farms can be located anywhere within crop or pasture land use in block groups with beef or dairy farmer populations.
- Drinking water wells are present at every farm centroid and at every receptor location for which census block group data indicate there are households with private wells.

4.2.10.3 Uncertainties and Limitations. In general, there are few data gap issues associated with human receptor data; consistent data of an appropriate scale for a national analysis were available for all 201 sites addressed in the analysis. The primary issues and uncertainties are associated with the age of the data and the mismatches based on date and geographic scale. Not all data sources used for the estimation of human receptors were created at the same time or at the same scale, and most of the data are 9 years old or older (see Table 4-8). More up-to-date national data sets were not available, however.

To the extent possible, scale and data discrepancies were addressed using spatial averaging and interpolation to minimize the errors incurred from using different data layers. For example, to better combine the 1990 U.S. Census data with the 1992/1987 Census of Agriculture data, the 1987 and 1992 data were averaged to create a hybrid 1990 data set. Larger-scale (i.e., block group, county, and state) data were combined with the block data by assuming uniform characteristics across block groups, counties, and states. This step was necessary to allow automated processing of human receptor type and population data. Although this assumption does create some inaccuracies at individual sites, it is valid and appropriate for a national analysis.

From a time-frame perspective, the data are roughly contemporaneous with the 1980s, with data ranging from pre-1980 (certain GIRAS land use data) to 1985 (Industrial D Screening Survey data) to 1990 (U.S. Census, Census of Agriculture 1987-1992 averages). Although a comprehensive review of these data for temporal consistency was not conducted, the visual observation of the land use and census coverages conducted during farm placement QC uncovered no apparent inconsistencies. The general applicability of mid-1980s data to the 1999

Table 4-8. Matrix of Biota Types in Food Webs of Representative Aquatic Habitats

	Coldwater habitats				Warmwater habitats			
Biota type	**Stream**	**Wetland**	**Pond**	**Lake**	**Stream**	**Wetland**	**Pond**	**Lake**
Periphyton	/	/	/	/	/	/	/	/
Phytoplankton		/	/	/		/	/	/
Aquatic macrophytes	/	/	/	/	/	/	/	/
Zooplankton		/	/	/		/	/	/
Benthic detrivores	/	/	/	/	/	/	/	/
Benthic filter feeders	/			/	/			/
TL3benth_sm	/				/			/
TL3benth_med	/	/	/	/	/	/	/	
TL3benth_lg								/
TL3zoop_sm			/					
TL3zoop_med				/			/	
TL3zoop_lg								/
TL3omni_sm		/	/	/		/	/	
TL3omni_med	/	/	/		/	/		/
TL3omni_lg				/			/	/
TL4pisc	/	/	/	/	/	/		/

HWIR analysis has been addressed elsewhere by EPA and also should be considered a significant uncertainty in the overall analysis.

The accuracy of location data of the WMU site was initially an issue, but steps were taken to find better WMU locations for the 201 sites, including address matching for 20 sites and visual inspection of all sites and their surrounding land use in order to move them to a more plausible location (i.e., moving a WMU out of a waterbody and into an industrial land use type). However, there were a few cases where human receptor points (census block centroids) fell within the WMU and were moved just outside of the WMU boundary. Although this may add a degree of conservatism to the analysis by placing receptors very close to the unit edge, EPA does not have adequate information to assume a standard or site-specific setback for locating receptor points. A related concern is that because, for a particular site with different WMU types, WMU sizes vary with site/WMU type setting and receptors near one WMU may be within the WMU for a different setting at the same site. Although this would be a concern if HWIR was a site-specific analysis, EPA believes that, in a national analysis, where the conditions at 201 facilities are intended to represent almost 30,000 facilities across the country, this is a reasonable inconsistency required by limitations in the data and in the ability to model multiple units at one site simultaneously.

Placing a drinking water well at every at every receptor point within a block group with at least one household with a well is conservative in that more densely populated blocks within a block group may be on public water supplies but will be assumed to have wells.[2] However, in sparsely populated areas with large census blocks, placing wells only at the block centroid may miss actual well locations. Depending on the width of the WMU and contaminant plume, a broad well spacing may miss a plume entirely and there is a possibility that actual downgradient wells are not considered in the analysis. EPA does not have the information to determine the relative impacts of these assumptions, or the actual placement of wells around a site, but believes that, across the entire analysis, the conservative assumption about whether a particular receptor point has a well will balance the chance that receptor point placement has missed a potential well/plume juxtaposition.

4.2.11 Ecological Exposure Factors

Ecological exposure factors are used in the ecological exposure module of the Hazardous Waste Identification Rule modeling system to calculate the total exposure dose (in mg/kg-d) to a suite of receptors that consume contaminated media and food items within their respective habitats. The ecological exposure module calculates exposure doses based on media and food concentration inputs generated by other modules.

4.2.11.1 <u>Methods and Data Sources</u>. Methods for developing exposure factors from wildlife data extracted from various sources are detailed in the *Wildlife Exposure Factors Handbook* (U.S. EPA, 1993b). These methods are the primary methods used in the HWIR analysis. Based on information in the *Handbook* and in the other sources listed above, a database

[2] In essence, the HWIR analysis assumes that, whether block group data show that there are 1 in 100 households with wells or that all households have wells, all receptors in that block group drink ground water.

was developed containing all available data relevant to ecological exposures for the selected 57 receptor species. The preponderance of data were extracted from the *Handbook* and from Sample et al. (1997). Both the *Handbook* and Sample et al. are compilations of data from many different references and therefore include multiple values for each species for a particular data category (e.g., multiple body weights for the otter). In some cases, the reported values are the mean of data collected for a single study; in other cases, the reported values reflect a single measurement. In all cases, all reported values were entered in the database, including means, minima, and maxima. Single reported values were entered as mean values. Data collected from additional sources were treated in the same manner. These data were then processed to generate the appropriate inputs for the module.

In general, the ecological exposure factors are for the mean adult body weight for the species; male and female weights are combined, and juvenile body weights are not considered. Although some data are reported by age or sex, not enough of these data exist to develop exposure factors for juveniles or for males versus females for an adequate number of species. Moreover, age- or sex-specific uptake factors and toxicological benchmarks are not available for ecological receptors. Therefore, consumption rates and prey preferences were estimated for average adults only. Additional information on methodology and data sources can be found in Section 12 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999n).

4.2.11.2 <u>Assumptions</u>. Key assumptions include the following:

- The pathways through which each receptor species is exposed were determined by the species' dietary preferences. For example, strict herbivores are exposed through ingestion of vegetation but not through ingestion of animal prey items; species whose diet includes animal prey are exposed through prey ingestion, and so on. Exposures through ingestion of soil, sediment, and surface water were determined by the documented foraging and feeding behavior of each species.

- The body weight and ingestion rate parameters are characterized by fixed values; dietary composition, or the amount of each food item eaten, is characterized by a uniform distribution between reported minimum and maximum values.

- Exposure factors are for the mean adult body weight for the species; male and female weights are combined, and juvenile body weights are not considered. Consumption rates and prey preferences were estimated for average adults only.

- No allometric equation is available for relating body weight to drinking water ingestion for reptiles and amphibians. Therefore, a default value of 0.0001 was used for water ingestion in all herpetofauna.

- Sediment and soil ingestion rate data were reported as sediment/soil ingestion rates, without distinguishing what proportion was sediment versus soil. The ecological exposure module, however, uses discrete constituent concentrations in soil and sediment. Therefore, receptor species were assigned soil or sediment ingestion as a function of their feeding patterns. Those species feeding entirely on

terrestrial prey and food items were assumed to consume soil, and the reported sediment/soil ingestion rate was applied entirely to soil. Those receptors feeding on aquatic prey and food items, or on a combination of aquatic and terrestrial prey and food items, were assumed to ingest only sediment.

- For those species for which sediment/soil ingestion rates were not available, the reported ingestion rate of the most similar species was used. Similarity was based first on faunal class (i.e., mammal, bird, reptile, amphibian), second on size, and third on feeding behavior. Data were available for only two species of herpetofauna, the eastern box turtle and the painted turtle. Therefore, small herpetofauna with mixed diets (e.g., frogs, salamanders) were given the same rate as the eastern box turtle, and larger herpetofauna that eat a significant proportion of fish (e.g., snapping turtles and aquatic snakes) were given the same rate as the painted turtle.

- The HWIR exposure analysis required data on the full range of potential diet items across all habitats where each species could occur. In addition, the HWIR analysis required quantified data – maxima and minima. Therefore, the two general types of data for each receptor were combined, and dietary profiles were developed for each receptor species. The profiles include each documented diet item for each species, whether the item was reported in a quantitative or qualitative form.

- Maximum and minimum prey preference values were generated for each item through a subjective process implemented by project ecologists. Qualitative data were needed to complement the available dietary fraction data. In this process, qualitative information was evaluated, and an estimated dietary fraction was assigned to each item. Then, all reported and estimated dietary fractions were compared, and the minimum and maximum values were identified for each diet item.

- When fish were a reported dietary item, the fish's trophic level was included for use in the aquatic food web module. When the fish species name was reported, its trophic level was extracted from trophic level analyses in EPA's Great Lakes Study (U.S. EPA, 1995e). In all other cases, the trophic level for fish prey was assumed to be T3 or T4 based on the size of the predator (receptor species). Small receptors were assumed to eat T3 fish. For large receptors, it was assumed that 50 percent of the fish they eat are T3 and 50 percent are T4. None of the primary exposure data sources included information on the trophic level of prey items.

- When a linear distance was reported for home range or territory size, it was assumed to represent a radius. When a radius was reported, it was assumed to describe a circular area around a nest or some essential physical feature. All home range size data were entered into the exposure factors database, and the midpoint

between the minimum and maximum values was calculated for each species. The home ranges were assumed to be circular.

- Calculated averages were assumed to reflect the home range size for a species in all of the habitats where it occurs, while available data may come from a single habitat type. In fact, habitat type and quality affect foraging distance. For example, a white-tailed deer may cover a larger area when foraging in a forest than when feeding in a crop field.

- Average home range sizes also were assumed to reflect all regions of the United States and year-round food availability. Some reported home range data were for particular regions and seasons; no attempt was made to distinguish home range size by region or season.

Additional information can be found in U.S. EPA (1999n).

4.2.11.3 <u>Uncertainties and Limitations</u>. The estimation of ingestion rates was based on average, gender-neutral body weights and does not account for differences in size, season, habitat, or activity level. However, prey preferences are represented by distributions that are intended to reflect some of the natural variation in wildlife feeding behaviors.

The regression statistics used to calculate food ingestion rates for herpetofauna were derived from data on iguanid lizards (family Iguanidae) only and are not recommended for use with other reptiles or amphibians (U.S. EPA, 1993b). However, lacking any other data at all for food ingestion rates in amphibians and reptiles, this estimation methodology was used for all herpetofauna in the HWIR analysis. It is not known how the ingestion rates of iguanid lizards compare to the ingestion rates of herpetofauna in general.

Sources of water intake other than drinking water, including water derived from ingested food, were not accounted for. Depending on dietary composition, some species derive more water from food items than do others. Exposure through drinking water ingestion may, therefore, be over- or underestimated.

Soil and sediment ingestion were treated as mutually exclusive because the data do not specify which medium was reported. Many receptors probably ingest a combination of soil and sediment (e.g., raccoon). Since soil and sediment constituent concentrations are likely to differ at a given sight, exposure estimates for these two ingestion pathways would be affected.

Individual body weights were derived to represent receptors nationally. This derivation was generally based on local scale data from studies distributed throughout the United States. For some species, however, only one or a few reported body weights were available. Therefore, some species' body weights, as well as other exposure factors based on body weight, may not be representative of the species' natural variability.

4.2.12 Ecological Receptors and Habitats

Ecological receptor and habitat data include the data necessary to delineate and characterize the terrestrial and aquatic habitats at a site and populate these habitats with receptors and their home ranges. Ecological habitats and receptors were defined and identified for each of the 201 sites in the HWIR analysis. Receptor home ranges were randomly placed at each of the 419 site/WMU settings.

For the HWIR 3MRA, the representative habitats are defined by site layout inputs to the ecological exposure module; the variables representing the habitats are HabGroup, which indicates whether a habitat type is terrestrial, aquatic, or wetlands; HabType, a string variable that gives the descriptive name of the habitat; and HabIndex, which assigns an index number to each habitat type.

The ecological exposure module uses the representative habitats to determine the receptor species likely to be present at a site and then calculates receptor doses based on the diet items (plants, prey, and water) expected to be present in the habitat. Receptor variables in the ecological exposure models are ReceptorType, a string variable that gives the descriptive species name, and ReceptorIndex, which assigns an index number to each receptor species.

The database processing for HabRangeNumLWSSubA, HabRangeLWSSubAIndex, and HabRangeLWSSubAFrac differentiated between WMU types, assigning multiple subareas to the home ranges at land application units and wastepiles sites only.[3] Landfills, surface impoundments, and aerated tanks were not modeled using multiple subareas, so they had only one subarea consisting of the WMU itself. In addition, if the home range intersected the surface impoundment subarea, HabRangeNumSISrc was set to 1 for that home range.

The waterbody network and waterbody network reach connectivity for the habitats and home ranges, HabWBNIndex, HabNumWBNRch, HabWBNRchIndex, HabWBNRchFrac, HabRangeWBNIndex, HabRangeNumWBNRch, HabRangeWBNRchIndex, and HabRangeWBNRchFrac, needed to be determined on the database end, because the waterbody networks were created during database processing (see U.S. EPA, 1999g). The GIS-assigned reach connectivity was based on each reach at the site having a unique number. The database created waterbody networks, assigning each reach to a network and giving it a new identifier that included the waterbody network identifier and a new reach number. GIS-determined reach habitat and reach range connectivity were then converted to the new waterbody network identifiers using a lookup table of network identifier by GIS identifier.

The determination of fishable reaches was done by database processing as described in U.S. EPA (1999g). Home range connectivity to fishable reaches, HabRangeFishWBNIndex, HRangeNumFishWBNRch, and HRangeFishWBNRchInde, was determined in the database by using the reach number connected with each home range and a lookup table of fishable reaches (streams of order 3 or greater, lakes, and wetlands) by reach number.

[3] GIS processing always splits local watersheds into multiple subareas independent of the WMU type.

4.2.12.1 Methods and Data Sources. Methods and data sources for ecological habitat and receptor home range development and delineation are described in detail in Section 13 of *Data Collection for the Hazardous Waste Identification Rule* (U.S. EPA, 1999o). A summary of these methods follows.

Habitats. The primary goal in developing representative ecological habitats is to provide a framework for incorporating characteristics of site location into the assessment methodology. The level of refinement or degree of detail delimiting the habitats was designed to be consistent with data availability and selected endpoints. For the HWIR analysis, the selected endpoint is survival of receptor species populations. Accordingly, habitats are distinguished at a level that affects receptor species' exposure and, in particular, at a level that affects the species included in the analysis. The HWIR receptor species are those for which sufficient exposure factor, uptake, and benchmark data are available; therefore, habitat characteristics that determine these species' presence or absence at a site form the basis of the habitat classification. In general, habitat affects the presence of a species at a site by providing essential resources such as food, shelter, nesting sites or materials, and appropriate sites for behaviors such as courtship, mating, roosting, or hibernation. Therefore, habitats were developed based on the physical setting in terms of these resources. Development of the representative habitats also included a survey of existing ecological classifications.

Receptors. The ecological exposure module uses the representative habitats to determine the receptor species likely to be present at a site and then calculates receptor doses based on the diet items (plants, prey, and water) expected to be present in the habitat. Receptor groups were developed for each representative habitat. The basis of the methodology for developing habitat receptor groups is the construction of food webs. The food webs describe the trophic levels, eating strategies, and faunal classes included in each habitat. Receptors are then selected to represent these components of the food web. Receptor groups were drawn from the species contained in the wildlife exposure factors database; these are species for which at least the minimum required toxicological and exposure data are available. (See Section 12.0 for a description of the wildlife exposure factors database.) As such, the species composition of the receptor groups is based on available data as well as on species distribution and habitat information.

Habitat Delineation. The habitats occurring at each HWIR site were identified and delineated based on mapped land use and wetlands and other waterbody data. Terrestrial habitats were delineated based on digitized Anderson land use categories (Anderson et al., 1976) available from EPA's GIRAS (U.S. EPA, 1994a); a combination of National Wetland Inventory (U.S. FWS), GIRAS, and Reach File Version 3.0 Alpha Release (RF3-Alpha) (U.S. EPA, 1994b) data were used to delineate wetlands; and RF3-Alpha and DEM (USGS, 1990a) data were used to delineate waterbody margin habitats. Digitized land use features, wetlands, and waterbodies were generated in the HWIR GIS as described in U.S. EPA (1999g); these data then provided the basis for the delineation of representative habitats at each site.

A delineation tool was developed in ArcView to allow hand delineation of habitats at each site. The delineators used the digitized spatial features at each site as the initial indicators for the placement and boundaries of habitats. Additional protocols were developed to combine patches

of like habitat at a single site into one contiguous polygon and to eliminate very small isolated patches that would not provide useful habitat. A habitat delineation tool was developed to eliminate the need for habitats to be digitized by GIS staff and to thus save many hours of labor. The basic concept of the tool was to allow the users to easily pull together different layers of GIS data for a given site and delineate habitat areas with a minimal amount of training.

Database Processing. Following GIS processing, the resulting data tables were imported into Access databases, where Structured Query Language (SQL) and Visual Basic programs were used to prepare the final HWIR model-ready data set, according to HWIR 3MRA modeling system specifications.

4.2.12.2 Assumptions. The following assumptions were used during habitat delineation and placement of home ranges and receptors.

- The term habitat implies a level of detail and specificity that is meaningful for the exposure scenario at a particular facility site, but does not require extensive biological inventory or field investigation for identification or delineation.

- The primary criteria for defining the terrestrial and wetlands habitats are soil moisture and vegetation structure. It is assumed that all terrestrial and wetlands sites have a soil substrate. Terrestrial sites without a soil substrate are assumed to be industrial sites that do not support ecological communities and, therefore, do not warrant ecological risk assessment. Natural areas lacking a soil substrate (e.g., rock outcrops) are assumed to be relatively small inclusions within a habitat and not the focus of this risk assessment. Vegetation type is directly affected by these abiotic parameters and, therefore, often used as a general indicator of many abiotic characteristics.

- Wetlands ecosystems are assumed to have a food web similar to either an aquatic or terrestrial ecosystem, depending on its water regime. The three intermittently flooded wetlands habitats are characterized by infrequent inundation and do not support aquatic communities; therefore, their food web is assumed to be described by the general terrestrial food web. The three permanently flooded wetlands habitats are characterized by long-term inundation and support aquatic communities including fish, benthic organisms, and aquatic plants. Thus, their food web is assumed to be similar to the aquatic habitat food web.

- Trophic level 1 (T1) consists of species that consume only plants (i.e., the herbivores) and that are potential prey for higher trophic level species. T1 species include small or large mammals, herpetofauna, invertebrates, and birds. The soil community is a subset within T1 and includes invertebrate soil organisms that live in direct contact with soil, thus reflecting a unique exposure pathway. Within the HWIR analysis, the soil community is assumed to be T1, although the dynamics within soil communities are, in fact, very complex and include herbivores, omnivores, and carnivores at several trophic levels. This complexity is beyond the reach of the HWIR models and available supporting data.

- It could be argued that many of the species designated as T3 are not, in fact, true apex species because their eggs or young may be consumed by other species. For the HWIR analysis, however, species for which the adults are not a regularly consumed prey item are considered apex or T3 species.

- The terminology used to describe wildlife habitats in the source references varies considerably, as would be expected. Therefore, professional judgment was used to correlate the habitat terminology found in the literature with the representative habitat types. Although some interpolation was employed in certain cases, a species was not assigned to a habitat unless such assignment was supported by a reference or firsthand knowledge. Anomalous sitings or casual occurrences (sometimes reported for birds) were not included.

- Wetlands habitats, in particular, raised questions about assignment of receptor species to habitats. Sources of habitat information for wetlands species rarely refer to the degree or frequency of flooding when describing wetlands habitats. Therefore, it was difficult to differentiate between species using intermittently flooded as opposed to permanently flooded wetlands habitats. In many cases, the food or prey items attributed to a species were used as an indicator; for example, if a source reported that a species fishes in wetlands habitats, the species was assigned to the appropriate permanently flooded wetlands habitat that, by definition, supports fish. Due to the predominant use in the source literature of terms such as swamps and marshes, which imply relatively long flood duration, fewer species were assigned to the intermittently flooded wetlands habitats. Furthermore, intermittently flooded wetlands are generally less discernible from surrounding uplands than are permanently flooded wetlands and, thus, are reported less frequently in general wildlife habitat sources. These differences are reflected in the smaller receptor groups for the intermittent wetlands.

- Data interpretation methods were used to qualitatively characterize each species' diet, to establish its trophic levels, and to populate the terrestrial food web compartments for each habitat. All dietary items reported in the literature as commonly eaten were included in the characterizations. Information from species monographs and zoological profiles was emphasized over field guides. In general, dietary information from the various sources was quite consistent.

- It was assumed that receptor species occur in their assigned representative habitats regardless of a site’s position in the landscape. Thus, if a forest habitat is delineated at a site, all the species included in the forest receptor group and occurring in that particular region are assumed to be present. In fact, it is probably unlikely that all of the receptor species, particularly those less adapted to human impacts and development, would be present. Moreover, when the habitat patches at a site are small, it is questionable whether the entire receptor group would use the habitat. For example, the black bear is included in the forest habitat receptor group; but, even within its normal distributional range, it is unlikely to occur in

developed or industrialized areas or areas that consist of small habitat patches. Exposure doses were adjusted to account for the proportion of a species' required home range provided at a site; however, no adjustment was made to account for differences in species diversity in small versus large habitat patches or in disturbed versus undisturbed areas.

- Habitat delineation protocols were developed to combine patches of like habitat at a single site into one contiguous polygon and to eliminate very small isolated patches that would not provide useful habitat. The basis for these protocols is the assumption that wildlife receptors in disturbed landscapes, such as those in which HWIR sites are most likely to be located, will travel from patch to patch to forage and feed. Most developed areas of CONUS consist of a mosaic of different vegetation communities. In many cases, the individual habitat patches that make up this mosaic are not large enough to support a significant food web. When combined with similar patches in the landscape, however, these patches can provide the necessary habitat to support the food webs and receptors developed for the HWIR analysis. Thus, in order to indicate receptor movement throughout each habitat type, patches of like habitat were connected, or bridged, in the delineation process. Consideration was given to barriers to movement, such as waterbodies or industrial areas.

- When a single small patch of a particular habitat type occurred at a site, the delineators checked the surrounding area outside the area of interest for additional habitat of the same type. If none existed within a reasonable distance, the single isolated patch was subsumed in its adjacent habitat type. In these cases, the assumption was made that a small isolated habitat patch would not effectively support receptor species.

- The Anderson land use codes correlate reasonably well with the representative terrestrial habitats, and a simple crosswalk between the two classifications is possible for most categories (see Table 13-17, U.S. EPA 1999o). However, because the land use classification and the habitats were developed for different purposes and use different terminology, certain assumptions and subjective decisions were necessary to correlate the two. Many of the inconsistencies were resolved based on decision rules using the largest adjacent or proximate similar land use polygon as the default. These decision rules are based on the assumption that the predominant land use in an area is likely to exert the greatest influence on receptors occurring in the vicinity. Therefore, when the habitat designation for a polygon could not be determined with certainty, data from the largest adjacent or proximate polygon were used.

- For waterbody margin habitat delineation, elevation contours were assumed to be the best indicators of stream corridors and pond and lake margins. Using DEM contour data, an attempt was made to determine a visual natural limit for the corridor or margin. Because waterbodies occur in the landscape along elevation contours, natural boundaries were frequently evident. If no contour-based

boundaries were apparent, surrounding land use was used instead. For example, if GIRAS data indicated a forest buffer running parallel to a stream and a commercial or industrial area adjacent to the forest, the stream corridor would consist of the forest buffer. When neither contours nor land use indicated corridor or margin boundaries, a default minimal margin was delineated.

- Wetlands types with flood regimes that indicate the presence of sufficient flooding to support fish populations were delineated as permanently flooded wetlands. Those wetlands in which flooding is infrequent or of short duration were delineated as intermittently flooded wetlands.

- For sites where NHI data were not available, GIRAS data were used to delineate wetlands. GIRAS data, however, do not include information on the wetlands flood regime. All GIRAS-identified wetlands were delineated as permanently flooded, because GIRAS generally does not recognize wetlands ecosystems at the drier end of the wetlands flood regime continuum.

- GIRAS data classify wetlands as forested or nonforested and do not include any information on the flood regime. Because most national data sets generally apply the term wetlands to tidal and other aquatic habitats and do not recognize noninundated areas as wetlands, the HWIR analysis assumes that wetlands identified in GIRAS data fall within the permanently flooded wetlands habitats.

- The forested GIRAS wetlands were delineated as permanently flooded forested wetlands; the nonforested GIRAS wetlands were delineated as permanently flooded grasslands. Although some of the wetlands included in the GIRAS data are undoubtedly dominated by shrub/scrub vegetation, the data do not allow this distinction to be made. In the absence of better data, the intermittently flooded grassland habitat was considered the most appropriate alternative.

- Permanently flooded wetlands frequently occur in association with streams, rivers, lakes, and ponds. Thus, the potential arises for areas adjacent to waterbodies to include both wetlands and waterbody margin habitats. The most effective and straightforward approach to handle this situation appeared to be to default to the wetlands habitat when wetlands and waterbody margin habitats overlapped. In fact, many wildlife receptors probably forage across both waterbody margin and wetlands habitats, while other species show a preference for or tend to avoid the wetlands habitat. Because this degree of variability in habitat usage is beyond the reach of the HWIR model, wetlands habitats were delineated whenever they were indicated, including within a waterbody margin. Thus, wetlands occurring near waterbodies were not subsumed in the stream corridor or lake and pond margin habitat. In general, waterbody margin habitats occur on HWIR sites more frequently than do wetlands habitats, and a significant number of wetlands habitats would have been eliminated from the assessment if waterbody margin habitats had been delineated to subsume co-occurring wetlands.

- Using the binning approach adopted for home range placement, each receptor's average home range size was assigned to one of four bins. Each bin comprises a range of home range sizes. Thus, all receptors with home ranges in Bin 1 are assumed to have a home range size of 100,000 m^2, receptors in Bin 2 have a home range size of 1 million m^2; receptors in Bin 3 have a home range size of 10 million m^2, and receptors in Bin 4 have a home range size of 1 billion m^2. Home ranges are assumed to be circular and are placed within delineated habitats such that each bin is entirely contained within the next largest bin. This placement arrangement ensures that predator ranges overlap with that of all potential prey.

4.2.12.3 <u>Uncertainties and Limitations</u>. Generally speaking, in the continental United States, the major faunal classes (mammals, birds, reptiles, and amphibians) occur throughout all trophic levels (with the exception of amphibians in T3). However, exposure and toxicological data are not necessarily available for representative species in all classes at all trophic levels. Therefore, not all potential compartments are represented in the food web. In particular, herbivorous (T1) herpetofauna and birds and T3 reptiles are not represented in the terrestrial habitat food web. Thus, risk for these particular receptor types is not assessed. However, T1 birds and T3 reptiles are assessed in the aquatic food web.

Exposure doses were adjusted to account for the proportion of a species' required home range provided at a site; however, no adjustment was made to account for differences in species diversity in small versus large habitat patches or in disturbed versus undisturbed areas. The HWIR analysis assumes that receptor species occur in their assigned representative habitats regardless of a site's position in the landscape. In fact, it is probably unlikely that all of the receptor species, particularly those less adapted to human impacts and development, would be present. Moreover, when the habitat patches at a site are small, it is questionable whether the entire receptor group would use the habitat. Therefore, the total national risk estimates for particular habitat types or receptors may be high.

GIRAS land use/land cover data are 15 to 25 years old and, therefore, do not reflect current conditions in some locations. However, the GIRAS data set is the most complete and current national data set available.

The GIRAS data for residential land use include both high- and low-density residential land use in a single category (land use code 11). In the HWIR analysis, the residential habitat is intended to cover areas where lawns, gardens, and landscaped areas provide habitat for species such as passerine and ground birds and small mammals. However, the high-density residential areas included in the GIRAS data probably do not provide useful habitat Lacking any distinguishing data, all areas mapped as land use code 11 were delineated as residential habitats. Therefore, the occurrence of the true residential habitat may be overestimated.

The GIRAS Cropland and Pasture land use classification does not distinguish between crop fields and pastures. Pasture vegetation generally consists of a variety of herb and grass species and provides a relatively diverse habitat. Crop fields, on the other hand, are generally monocultures and may lack vegetation during the nongrowing season. Overlapping but different suites of receptors would be expected to use these two habitat types. Because the GIRAS data

do not distinguish between crop fields and pastures, the two habitat types were combined into one representative habitat (crop fields/pastures) to correspond with the land use classification.

The habitat delineation procedures consist largely of subjective evaluation of each site for the hand-delineation of habitat boundaries. These methods pose challenges for quality control and quality assurance. A totally automated habitat delineation procedure was considered but proved not to be feasible, given the enormous variety of numbers, sizes, and arrangement of habitat patches at the 201 sites. Delineation of waterbody margin habitats, in particular, is based entirely on hand-delineation and required the most subjective decision-making in the habitat delineation process. Exposure and risk estimates for these habitats, therefore, include additional uncertainty as compared to the terrestrial and wetlands habitats.

Because receptor exposure dose, as calculated in the ecological exposure module, is adjusted based on the proportion of the home range that falls within the contaminated area (see Section 12.0 for further explanation), the binning approach for placement of home ranges potentially underestimates the exposure dose for receptors at the low end of each range. For example, the eastern newt has an average home range size of 91 m^2; however, since it falls within Bin 1, its home range is assumed to be 100,000-m^2. Thus, the proportion of the eastern newt's diet taken from contaminated habitat will be calculated as the proportion of the 100,000- m^2

home range that falls within the contaminated area. Obviously, in certain cases, this will be a significantly smaller proportion of the diet than would be estimated if the true mean home range size (91 m^2) were used.

4.3 Model Uncertainty

This section presents a synopsis of each of the modules constituting the release, media fate and transport, uptake, exposure, and risk calculations of 3MRA. The modules are discussed in approximate order of their execution sequence within 3MRA. (See Figure 4-5.) Each module is discussed in the context of (1) spatial and temporal scales, (2) key assumptions, (3) methodologies, and (4) limitations/uncertainties. The methodologies subsections present an overview of the modules. Some of the modules are legacy code and, as such, have undergone extensive peer review and/or validation. For these modules, the methodologies sections provide only a brief overview. Others are new modules and their methodologies sections may be more extensive, as dictated by the desirability of a more in-depth description. This is particularly true for the source modules and the Aquatic Food Web Module. Much of the material presented herein is abstracted from the module-specific background documents, and the reader is referred to those documents for more comprehensive discussions.

4.3.1 Nonwastewater Sources (LF, LAU, WP) Modules

The Landfill (LF), Land Application Unit (LAU), and Wastepile (WP) source modules simulate time-varying releases of chemical fluxes to the atmosphere (Air Module), groundwater (Vadose Module), and nearby surface waters (Surface Water Module), as well as surficial soil concentration in the WMU (LAU only) and surficial soil concentrations in downslope buffer areas (LAU and WP) to the Farm Food Chain, Terrestrial Food Web, Human Exposure, and

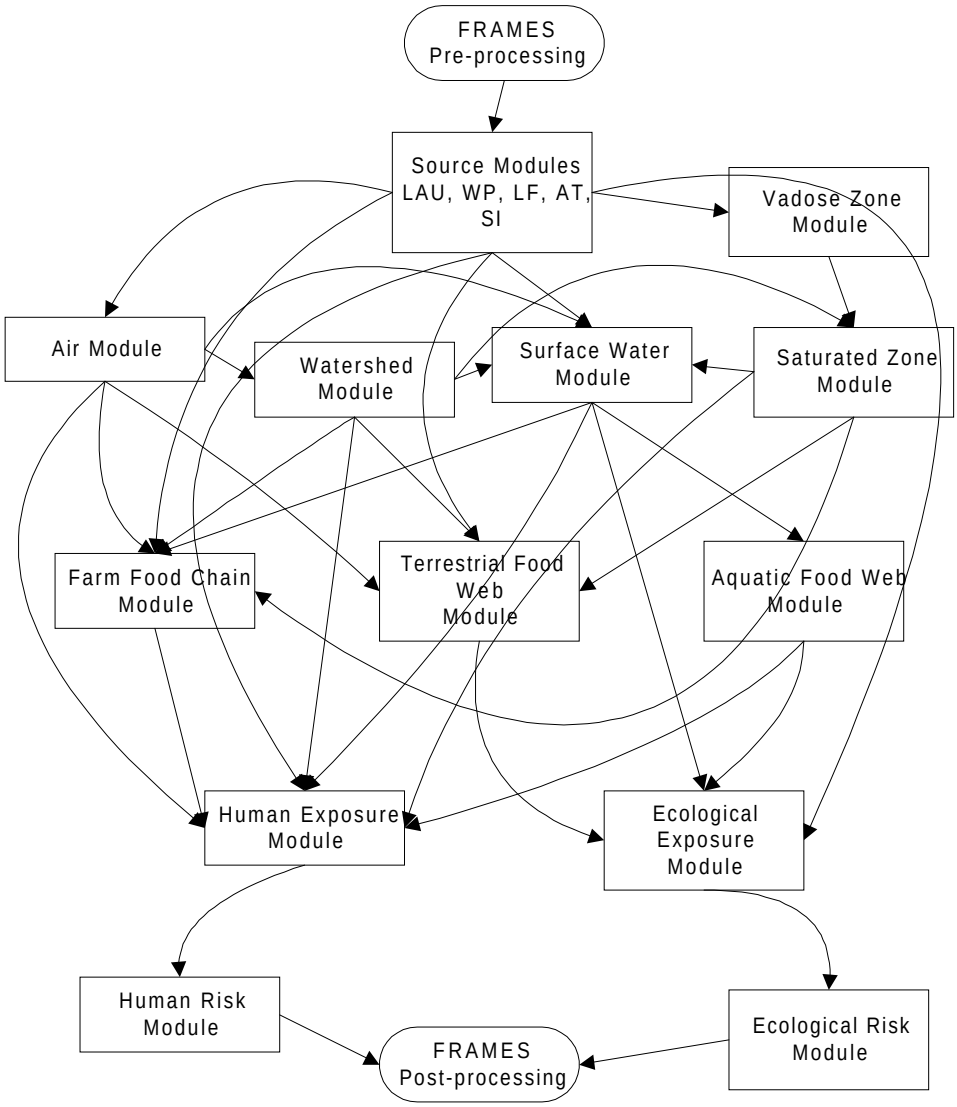

Figure 4-5. 3MRA release, media, uptake, exposure and risk modules.

Ecological Exposure modules. These three nonwastewater source modules share much functionality and, for this reason, are discussed here collectively.

4.3.1.1 <u>Spatial and Temporal Scales</u>. The LAU and WP models are two-dimensional spatially. (See Figures 4-6 and 4-7). These dimensions are vertical (depth) and longitudinal, where longitudinal is the direction along the surface flow path of stormwater runoff and erosion. The LF model is one-dimensional (vertical) because it is assumed to be unaffected by runoff and erosion. Simulated concentration gradients in either of the two dimensions are not continuous but are piece-wise constant (i.e. discretized) as a result of the solution scheme used by the semi-analytical solution algorithm. The vertical dimension is numerically represented as a series of computational cells arranged in sequential vertical layers. Concentrations can vary from cell to

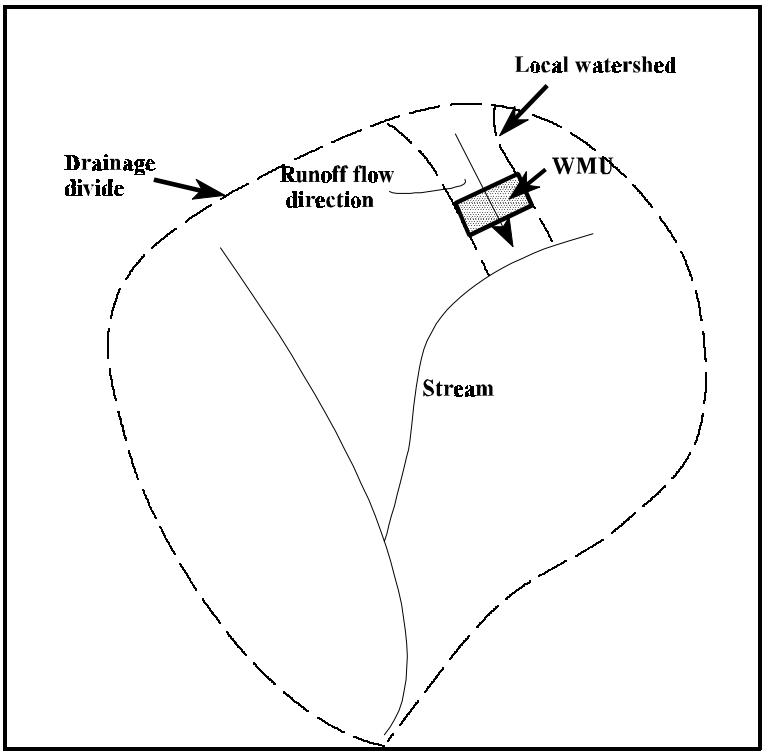

Figure 4-6. Local watershed containing WMU.

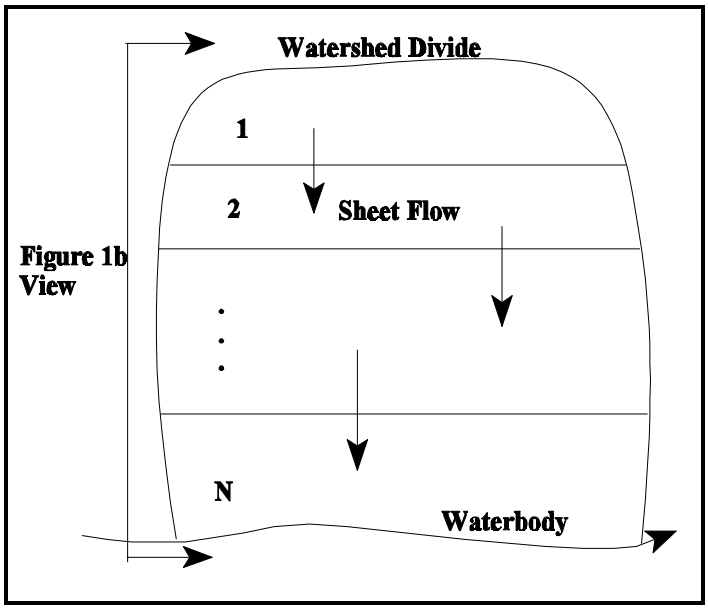

Figure 4-7a. Local watershed.

cell but are homogeneous within any given cell. The depth of a computational cell can vary somewhat as a function of run time considerations but is on the order of 1 cm, which affords a relatively high vertical resolution.

The resolution in the longitudinal dimension for the WP and LAU corresponds to a subarea within the "local watershed" (Figure 4-6). The local watershed is defined as that drainage area that just contains the WP or LAU in the lateral direction (perpendicular to the runoff flow path), and extends longitudinally downslope to the first surface waterbody. For purposes of HWIR99, within this local watershed, there are two subareas of interest. The first subarea corresponds to the surface area (footprint) of the WMU itself. The second subarea of interest is that portion of the local watershed extending from the WMU subarea downslope to the surface waterbody (i.e. the buffer subarea). Like the homogeneous concentrations within each vertical computational cell, concentrations are homogeneous within each of these two subareas, but can vary between them. (There is at some sites a third subarea, the land area upslope of the WMU subarea extending to the drainage divide, but it is of interest only for purposes of hydrology and erosion, not chemical contamination.) Longitudinal resolution in the LF model is similarly limited to the footprint of the entire LF.

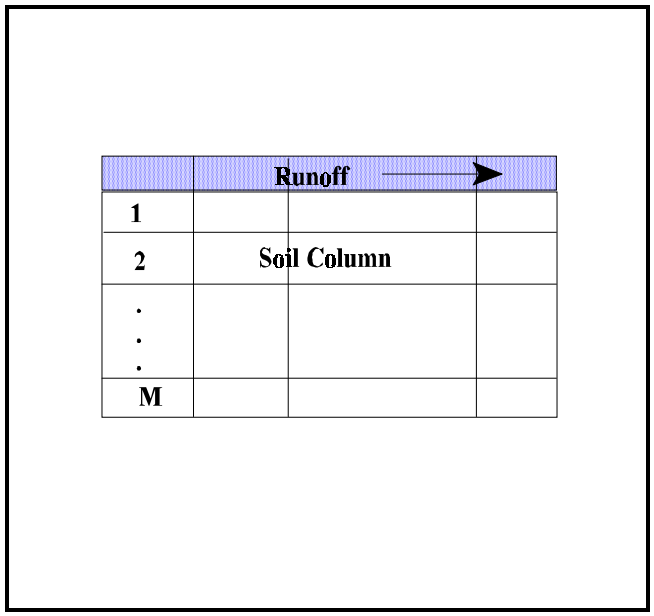

Figure 4-7b. Cross section view.

Temporally, the LAU and WP models are dynamic and were designed with the objective of simulating annual average conditions—the stated HWIR99 modeling goal. Although the desired output temporal resolution is annual average, the modules' internal time steps are in general much less than 1 year, both for purposes of numerical stability as well as for generating accurate estimates of annual average conditions. Indeed, the fundamental time step is daily, which was specified for the two following reasons:

- It was considered impractical to simulate annual average runoff without building up that annual average from daily precipitation and runoff events. (The precipitation/runoff model is nonlinear in the independent variable. One cannot simply input average annual precipitation as the independent variable and output average annual runoff.)

- An approximately daily time step is the fundamental temporal scale at which surface transport of chemical downslope in the local watershed is occurring. It was considered important to honor this time scale in simulating fate and transport from the sources.

Despite the internal daily time step, we do not consider the module's predictions to necessarily be accurate at this resolution. There is considerable uncertainty in day-to-day variation in inputs and parameter values, which would result in uncertainty in model predictions for any given day. However, when averaged over a year, much of this uncertainty is mitigated.

The preceding comments are generally applicable to the LF model as well, with one exception. For purposes of computational efficiency, the LF model uses an internally calculated, long-term average (approximately 10 to 30 years, depending on the meteorological station record) infiltration rate. Thus, the time series outputs of the LF module implicitly assume that annual average infiltration rates are constant at their long-term average.

4.3.1.2 <u>Key Assumptions</u>. The LAU, WP, and LF modules are all based on an underlying, porous media fate and transport algorithm that is termed the Generic Soil Column Model (GSCM). The assumptions and limitations of the GSCM are described completely in the background document (U.S. EPA, 1999at). Of that complete list, the key assumptions for HWIR99 are the following:

- The medium modeled, whether soil, waste, or a soil/waste mixture, can be approximated for modeling purposes as an unconsolidated, homogeneous, porous medium.

- Loss processes (e.g., decay) can be considered to proceed in accordance with first-order reaction kinetics.

- Contaminant partitioning between particulate and aqueous phases is reversible and linear. Furthermore, the partitioning coefficient is unaffected by changes in concentrations or environmental conditions (e.g., pH, temperature) during the module execution.

- Contaminant partitioning between aqueous and gaseous phases can be described by Henry's law.

- Daughter products are not considered.

Beyond these generic key assumptions for the GSCM, several module-specific additional key assumptions are applicable:

LAU

- The waste that includes the contaminant to be modeled is organic, decomposes within the LAU, and does not result in a significant buildup of the soil surface, i.e. the LAU is conceptually a compost pile. An implication of this assumption is that concentrations for persistent contaminants can, with repeated waste applications, be concentrated in the LAU soil, possibly beyond the concentration in the original waste stream in unusual cases.

LF

- The LF is assumed to be below grade; stormwater run-on, runoff, and erosion are assumed not to occur.

- For purposes of HWIR99, landfills are assumed to be unlined. (This is not a limitation of the model per se, although any "liner" simulated by the LF module is assumed not to be completely impermeable.)

- No daily soil cover is assumed. (This is an LF module limitation.) However, once an LF cell (not to be confused with the earlier "computational cell") is filled after 1 year, that LF cell is assumed to have a permanent, permeable cover, and fate and transport processes through that cover are simulated.

- For computational efficiency, the LF module assumes that the time series of output fluxes from the first LF cell is representative of time series outputs from all subsequent cells. That is, the initial cell's time series outputs are used repeatedly (although staggered in time) to represent future cells as they come "on line". Any variations in future releases due to variations in, for example, hydrological processes, are not considered. (Such hydrological variation is not available in any case, due to using only the available meteorological record.)

4.3.1.3 <u>Methodologies</u>. Methodologies for the nonwastewater source modules are discussed below in three subsections representing important individual algorithms of the overall methodology.

Generalized Soil Column Model. As previously mentioned, a local watershed containing a WMU is comprised of up to three subareas. (The LF local watershed consists of a single subarea.) Each subarea is considered as a one-dimensional (vertical) "soil column" in which vertical fate and transport are dynamically simulated using the GSCM algorithm. The GSCM is an algorithm that solves the following partial, linear, differential equation in space and time

$$\frac{\partial C_T}{\partial t} = D_E \frac{\partial^2 C_T}{\partial z^2} - V_E \frac{\partial C_T}{\partial z} - kC_T \qquad (4\text{-}1)$$

where

C_t (M/L3)	=	total (aqueous, gaseous, and particulate) contaminant concentration in the soil column at depth z and time t
D_E	=	effective diffusivity in soil (L2/T), determined as a function of gaseous diffusion, aqueous diffusion, and solids' partitioning
VE	=	effective contaminant advective velocity (L/T), is determined by the rate of groundwater infiltration modified by the solids' partitioning

k	=	aggregate first-order contaminant loss rate constant (1/T), which includes losses due to biochemical decay, hydrolysis, volatilization, and, in the surficial soil layer, contaminant losses due to stormwater runoff and erosion and wind/mechanical (e.g., vehicular activity) erosion.

Typically in environmental applications, an advection/diffusion equation such as 4-1 is solved using numerical, finite difference approximations in both the space (z) and time (t) domains. However, these numerical methods, while accurate, were found to be prohibitively slow in terms of computer execution time for HWIR99. To mitigate this run-time issue, a semi-analytical solution was developed that incorporates aspects both of numerical solutions (for the depth variable) and analytical solutions (for time-integration).

The depth domain is disaggregated first into zones that are assumed to be homogeneous with respect to all parameters in Equation 4-1. (The LAU and WP modules assume a single zone only, i.e., the tilled zone and wastepile, respectively. The LF, for HWIR99, is comprised of two zones, a permanent soil cover, and the waste zone below.) Each zone is further disaggregated into vertical cells of relatively small depth (e.g., 1 cm). For example, a 20-cm tilled zone in the LAU module would be represented as twenty 1-cm vertical cells.

For any given cell, and starting with an initial (time 0) uniform concentration throughout the cell, an analytical solution to Equation 4-1 is used to simulate the changes in contaminant concentration both within the given cell and all other cells making up the zone. The same process is then repeated for each cell, so that for each of 20 cells, for example, there are 20 concentration predictions for each time t. Each separate prediction represents the concentration at time t resulting from the initial condition for its respective cell. The principle of superposition, which applies to linear differential equations, is then invoked to add all 20 predictions to determine the overall concentration in each cell resulting from the initial condition in all cells.

The analytical solution to Equation 4-1, in fact, consists of three individual analytical solutions, one each for diffusive transport, convective transport, and aggregate first-order losses. Again, using the principle of superposition, the complete solution is attained by adding these three individual solutions at time t. Thus, the principle of superposition is used to advantage both in facilitating the complete analytical solution for a given cell and in accumulating responses from the effects of multiple cells.

Boundary conditions used are a zero concentration boundary condition at the upper boundary of the soil column and a zero concentration gradient boundary condition at the lower boundary. The upper boundary condition is consistent with the assumption that the air is a sink for volatilized contaminant mass at the surface. The zero gradient, lower boundary condition is used to approximate the assumption that the contaminant concentration immediately below the lower boundary (in the vadose) is not significantly different than it is just above the lower boundary.

Local Watershed/GSCM Algorithm. As a component of the overall HWIR multimedia exposure/risk model, the WP and LAU source emissions models are required to provide annual

average contaminant mass flux rates from the surface of the WMU and its subsurface interface with the vadose zone, total contaminant concentration in the surface material, and contaminant mass emission rate due to particulate emissions. In addition, because these WMUs are on the land surface, they are integral land areas in their respective watersheds. Consequently, they are not only affected by runoff and erosion from upslope land areas, they also affect downslope land areas through runoff and erosion. Indeed, after some period of time during which runoff and erosion has occurred from a WMU, the downslope land areas will have been contaminated and their surface concentrations could approach (or conceivably even exceed long after WMU operation ceases) the residual chemical concentrations in the WMU at that time. Thus, after extensive runoff and erosion from a WMU, the entire downslope surface area can be considered a "source," and it becomes important to consider these "extended source" areas in the risk assessment. It is for this reason that a holistic modeling approach has been taken with the WP and LAU source models to incorporate them into the watershed of which they are a part.

The Local Watershed/GSCM algorithm is based on mass balances of solids and chemicals in the runoff and the top soil column cell. The soil compartment is external to this model (see GSCM discussion above), and results from that compartment are called as needed by the software. A simplifying assumption is made that solids and chemical concentrations in the runoff are at instantaneous steady-state during each individual runoff event but can vary among runoff events; that is, a quasi-dynamic approach is used. Figure 4-8 presents the conceptual runoff quality model showing the two compartments and the fate and transport processes considered. During a runoff/erosion event, contaminant mass may be introduced into the runoff "water column" (i.e., the overland flow runoff) that occurs on the WMU subarea by two processes: (1) erosion of surficial soil solids that bear sorbed contaminant, and/or (2) diffusion of dissolved contaminant from the saturated pore space of the surficial soil into the overlying overland flow. As this overland flow makes its way downslope to the adjoining buffer subarea, the (initially

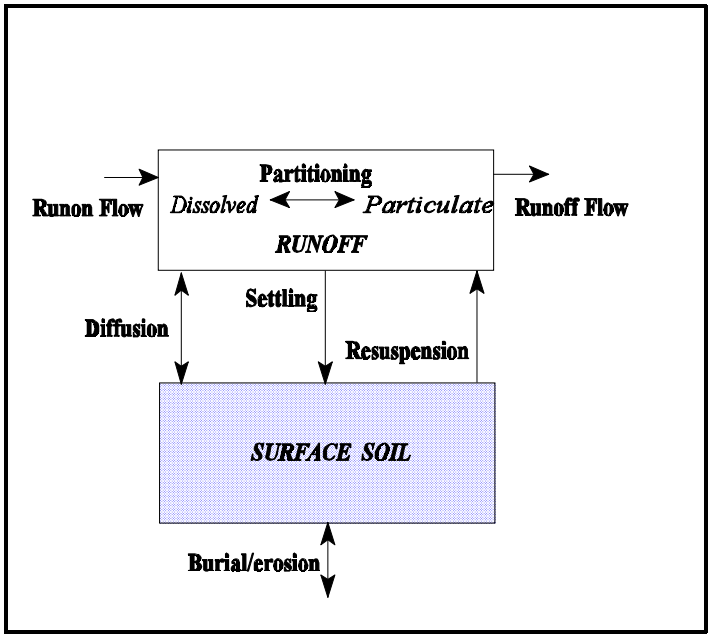

Figure 4-8. Runoff quality conceptual model.

uncontaminated) buffer can become contaminated by the same two processes: (1) settling of suspended solids bearing sorbed contaminant eroded from upslope, and (2) diffusion of dissolved chemical from the runoff water into the surficial soil pore water (assuming the concentration gradient is from the runoff to the surficial soil). The model describing this coupling between overland flow runoff and surficial soil contaminant conditions (see Figure 4-8) is in fact conceptually identical to conventional stream/sediment water quality models (substitute "stream" for "overland flow" and "sediment" for "surficial soil") in which one of the fundamental assumptions is that there is no lateral movement of the "bedload"; that is, all solids (and sorbed contaminant) transport occurs through vertical settling/resuspension/burial (erosion). This somewhat simplified conceptual representation of the complex processes of sheet-flow runoff, interflow runoff (stormwater moving laterally through pore spaces), and erosion/deposition was believed to be an appropriate simplification for HWIR99.

Hydrology Model. The hydrology model provides two overall functions: (1) estimates of daily soil moisture, runoff, evapotranspiration, and infiltration, i.e. a daily water balance, and (2) estimates of daily soil erosion on days when stormwater runoff occurs. These daily estimates, which are also subarea-specific, are then used by the Local/GSCM Watershed algorithm in its daily time step to build up the annual average output variable values previously described.

The daily soil moisture water balance is performed for the root zone of the soil column. At the end of a given day t, the soil moisture in the root zone of an arbitrary subarea i is updated as

$$SM_{i,t} = SM_{i,t-1} + P_t + RO_{i-1,t} - RO_{i,t} - ET_{i,t} - IN_{i,t} \qquad (4\text{-}2)$$

where

$SM_{i,t}$	=	soil moisture (cm) in root zone at end of day t for subarea i
$SM_{i,t-1}$	=	soil moisture (cm) in root zone at end of previous day for subarea i
P_t	=	total precipitation (cm) on day t
$RO_{i-1,t}$	=	storm runoff (cm) on day t coming onto subarea i from i-1
$RO_{i,t}$	=	storm runoff (cm) on day t leaving subarea i
$ET_{i,t}$	=	evapotranspiration (cm) from root zone on day t for subarea i
$IN_{i,t}$	=	infiltration (groundwater recharge) on day t (cm) for subarea i

given daily precipitation and estimates of the other variables. Precipitation is undifferentiated between rainfall and frozen precipitation; that is, frozen precipitation is treated as rainfall. The methodologies used to estimate runoff, evapotranspiration, and infiltration are described briefly below.

Runoff is calculated using the widely used "curve number" methodology developed by the U.S. Department of Agriculture Soil Conservation Service (SCS, 1986). In the HWIR99 imlementation of the SCS curve number methodology, 5-day antecedent soil moisture is tracked and used to adjust the curve number accordingly; that is, the curve number is increased (more runoff) relative to a default value when antecedent moisture is high and decreased from the default value when antecedent moisture is low.

Potential evapotranspiration (PET) is estimated by the Hargreaves methodology (Hargreaves, 1975), which is daily temperature-based and also includes factors to account for solar declination and month. When soil moisture is abundant, actual evapotranspiration (ET) is equal to PET. When soil moisture is limiting, ET is less than PET and is calculated according to a functional relationship that considers PET, available soil moisture, the soil moisture field capacity, and the soil moisture wilting point (Dunne and Leopold, 1978).

Daily infiltration is assumed to occur when the available soil moisture exceeds the soil moisture field capacity. The water available for infiltration is the difference between the current soil moisture and the soil moisture field capacity. However, if the infiltration rate using this available water exceeds the saturated hydraulic conductivity, the infiltration rate is set equal to the saturated hydraulic conductivity. In this event, to maintain a water balance, a feedback loop is needed to modify runoff and possibly ET so that the water available for infiltration does not exceed the saturated hydraulic conductivity.

Daily storm event-specific soil erosion loads are estimated using a modification of the universal soil loss equation (USLE) (Wischmeier and Smith, 1978). The daily modification results from disaggregating the long-term average USLE erosivity factor, R, to first an hourly R-factor (precipitation days only), under the assumption that erosivity is directly proportional to hourly rainfall intensity. The hourly R values are then aggregated to a daily basis to estimate the R-factor for each precipitation day. The USLE equation was applied in such a manner to estimate the daily soil erosion attributable to each subarea of a local watershed. This was achieved essentially using a "backwards" algorithm whereby the USLE was first applied to the entire local watershed to get the watershed total soil loss. To estimate the soil loss attributable to, for example, only the buffer, the USLE was next applied to the local watershed excluding the buffer subarea. The difference in the two soil loads represents the load attributable to the buffer. A sediment delivery ratio factor was also included in the USLE equation to account for intra-subarea deposition.

4.3.1.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- *Partitioning coefficients for metals.* The GSCM was developed originally for organic contaminants, for which K_d is calculated internally as the product of K_{oc} and *foc*. When applied to metals, K_d would ideally be calculated as a function of not only environmental conditions such as pH, but also the metal concentration as it changes over space and time. This sophistication is not included, and metals K_{ds} are constants read in from the chemical properties database. These values are randomly sampled from a distribution as part of the Monte Carlo strategy, however, which mitigates this limitation to some extent.

- *(NAPL) limitations.* With organic contaminants, the GSCM is not applicable if nonaqueous phase liquid (NAPL) is present. Similarly, with metals, the presence of a precipitate is not allowed. The presence of NAPL (precipitate) is determined by comparing C_T to the theoretical maximum contaminant concentration in soil without NAPL (precipitate), determined by the aqueous solubility, saturated soil-gas concentration of the contaminant, and the sorptive capacity of the soil. This is

not considered to be a significant limitation, however, because the waste concentrations (Cws) were carefully selected to minimize such solubility problems.

- *Daughter products not considered.* The model allows consideration of only one contaminant at a time and does not simulate fate and transport of reaction products in its current form. With further model development, it would be possible to track the production of reaction products in each soil column cell and use basically the same algorithm that is used for the parent compound to model the fate of reaction products.

- *Order of implementation of analytical solutions.* The solution technique used, sequential solutions to the three-component differential equations of the governing differential equation, allows computational efficiency. However, systematic errors could result from the choice of the order in which these solutions are applied. The size of the error would be dependent on the relative loss rates associated with the three processes. For example, if the first-order loss rate due to degradation were high and losses due to degradation were calculated first, then less contaminant mass would be available for diffusive and advective losses. The current algorithm prioritizes diffusive losses since the diffusion equation is solved first. This is followed by first-order losses and advection in that order.

- *Approximation to upper boundary condition.* A boundary condition at the soil/air interface of $C_T = 0$ was assumed. This is consistent with the assumption that the air is a sink for volatilized contaminant mass. However, because the diffusion coefficient used in the governing equation includes diffusion in both the air and aqueous phases of the soil, contaminant mass that is transported upward in the soil column via diffusion can include mass in both the air and aqueous phases. While this is appropriate within the soil where the ratio of air to water is relatively constant, the assumption breaks down at the soil/air interface itself. To account for the fact that contaminant mass in the aqueous phase should not be lost out of the surface of the soil column—which, e.g., would lead to nonzero volatilization fluxes for nonvolatile contaminants—the volatilization flux at the surface is assumed to include only the diffusive flux due to gas phase diffusion. Mass estimated to be lost from the surface due to aqueous phase diffusion is added back into the surface soil column cell, augmenting the contaminant concentration there and maintaining mass balance. This is an approximation, justified on the basis of computational efficiency; nonetheless, the approximation should be in reasonable agreement with what actually occurs in nature.

- *Approximations to inner boundary conditions.* For the LF, at the waste/subsoil inner boundary, under the assumption that the sorptive capacity of the subsoil has been exhausted and that partitioning is no longer applicable, a boundary condition where the aqueous phase contaminant concentration gradient is zero ($dC_L/dz = 0$) should be applied to ensure that dissolved concentrations cannot increase across the subsoil. Such an increase could occur if mass is allowed to diffuse into the

subsoil from the waste zone. Since diffusive flux occurs in the aqueous and gas phases only, $dC_L/dz = 0$ implies no diffusive flux across this boundary. This boundary condition is approximated in the landfill model by: (1) setting the boundary condition equal to $dC_T/dz=0$ at the bottom of the waste zone, which prevents diffusive flux from waste to subsoil, and (2) modeling only advection and decay in the subsoil zone. With use of these approximations, the existence of a subsoil zone simply shifts the leachate flux profile by the amount of time required for the contaminant to travel from the top to the bottom of the subsoil zone. Similarly, at the LF's cover soil/waste inner boundary, a boundary condition ideally should be applied where the sums of the advective and diffusive fluxes on either side of the boundary are equal. However, using the current solution technique, it is not possible to implement such a rigorous boundary condition. As an approximation, at the bottom of the cover soil zone, a no diffusive flux ($dC_T/dz = 0$) boundary condition is applied. At the top of the waste zone, a zero concentration ($C_T = 0$) boundary condition is applied. The diffusive flux from the waste into the cover soil zone is added to the bottom-most soil column cell in the cover soil zone. This set of boundary conditions allows diffusive transfer of contaminant mass from the waste to the cover soil. Conversely, advective, but not diffusive, transfer of contaminant mass from the cover soil to the waste zone is allowed. Use of these conditions would tend to overestimate contaminant mass losses from the waste zone into the cover soil zone and overestimate the volatile emissions.

- *Burial/erosion minor mass balance error.* The burial/erosion mechanism introduces a minor mass balance error into the model. The model for surface soil/runoff water fate and transport is based on a conceptual model originally developed for use in a stream/sediment application where the sediment compartment location relative to a reference point below the surface can move vertically (”float”) as burial and erosion occur. In that moving frame of reference, burial/erosion of contaminant does not introduce a mass balance error. However, in this HWIR99 application, the frame of reference is not allowed to float, but is fixed by the elevation of the lower boundary (e.g. top of the vadose zone). Thus, if sorbed chemical is eroded from the surface computational cell, that surface cell, which is vertically fixed, must have a “source” that is internal to the modeled soil column to compensate for this sink, or its internal mass balance is not maintained. The magnitude of this mass balance error is equal to the mass of eroded soil from the surface over the duration of the simulation times its average sorbed chemical concentration. In most cases, this error as a percentage of the total chemical mass in the modeled WMU will be quite small, and that has been confirmed in multiple executions of the model.

- *Spatial dilution of hot spots in the local watershed buffer subarea.* For HWIR99, the local watershed buffer is considered as a single, homogeneous subarea by the model to facilitate watershed delineation activities and decrease run-time. This will result in spatial averaging of concentrations in the buffer.

- *Sheet flow assumption across buffer subarea.* The conceptual local watershed model construct assumes that runoff and erosion occurs as sheet flow from the WMU across the buffer subarea to the waterbody. This implicitly assumes no short-circuiting of chemical loads directly from the WMU to the downslope waterbody, such as might occur if runoff/erosion created ditches or swales. (It would also occur in instances where engineered storm drainage captures runoff from the WMU.) To the extent that such short-circuiting might occur, the model will underestimate waterbody chemical loadings. There is a mass balance, so that any such underestimation would come at the expense of overestimating soil concentrations in the (bypassed) buffer zone.

- *Hydrological responses are limited by the available record.* The hydrology model uses available historical records of meteorological data. When the number of years simulated exceeds the number of years in the record, the record is repeated. Thus, unusual hydrological events (e.g., major storms) are limited to those actually observed. Events not observed, but possible nonetheless, that could result in increased source releases or media transport will not be included in the simulation.

- *The modules have not been field-validated*. Calibration and validation of the modules to field data would lend strong credibility to the underlying algorithms.

4.3.2 Wastewater Sources (SI, AT) Modules

The Surface Impoundment (SI) and Aerated Tank (AT) modules simulate time-varying releases of chemical flux to the atmosphere (Air Module) and, for the SI only, leachate flux to the Vadose Module. Both modules employ many of the same underlying equations and computer code and, because of this similarity, are described together in this section. Indeed, the only difference between the two modules is the leachate pathway for the SI; the AT is assumed to have an impermeable tank bottom.

4.3.2.1 <u>Spatial and Temporal Scales</u>. The spatial dimensionality of the SI and AT is zero-dimensional, i.e. both are modeled as completely stirred tank reactors (CSTRs) with no spatial concentration gradients. (The suspended solids algorithm, however, does acknowledge concentration gradients across the water column by estimating an average [as opposed to well-mixed, uniform] total suspended solids concentration.)

Temporally, a pseudo-steady-state monthly time step is used. Each month, the model updates certain parameters based on average monthly environmental conditions (temperature, windspeed, precipitation, and evaporation). It is then assumed that the system equilibrates instantaneously to these new conditions and a steady-state solution is obtained for that month. The resulting 12 monthly values for all outputs are then averaged and reported as annual averages.

4.3.2.2 <u>Key Assumptions</u>. The following key assumptions are noted:

- Two-compartment model: "mostly" well-mixed liquid compartment; a well-mixed sediment compartment, which includes a temporary accumulating solids zone. (“Mostly” well-mixed refers to the acknowledgment for purposes of estimating liquid compartment total suspended solids concentrations that spatial gradients will exist, and an average concentration is estimated.)
- First-order kinetics for volatilization in liquid compartment
- First-order kinetics for hydrolysis in both liquid and sediment compartment
- First-order kinetics for biodegradation with respect to both contaminant concentration and biomass concentration in liquid compartment
- First-order kinetics for biodegradation in sediment compartment
- Darcy's law for calculating the infiltration rate
- First-order biomass growth rate with respect to total biological oxygen demand (BOD) loading
- First-order biomass decay rate within the accumulating sediment compartment
- No contaminant in precipitation/rainfall
- Linear contaminant partitioning between adsorbed solids, dissolved, and vapor phases.

4.3.2.3 <u>Methodologies</u>. The AT or SI is divided into two primary compartments: a "liquid" compartment and a "sediment" compartment. Mass balances are performed on these primary compartments at time intervals small enough that the hydraulic retention time in the liquid compartment is not significantly impacted by the solids settling and accumulation. Figure 4-9 provides a general schematic of a model construct for an SI; the AT model construct is similar except there is no infiltration (leachate to groundwater loss mechanism) in the AT.

In the liquid compartment, there is flow both in and out of the waste management unit. There is also a leachate flow to the sediment compartment and out the bottom of the WMU for surface impoundments. Within the liquid compartment, there is contaminant loss through volatilization, hydrolysis, biodegradation (presumably aerobic), and particle burial (net sedimentation). The sediment compartment has contaminant losses due to (anaerobic) biodegradation and hydrolysis. Some contaminant mixing between the liquid and sediment compartment occurs due to contaminant diffusion and due to particle sedimentation and resuspension.

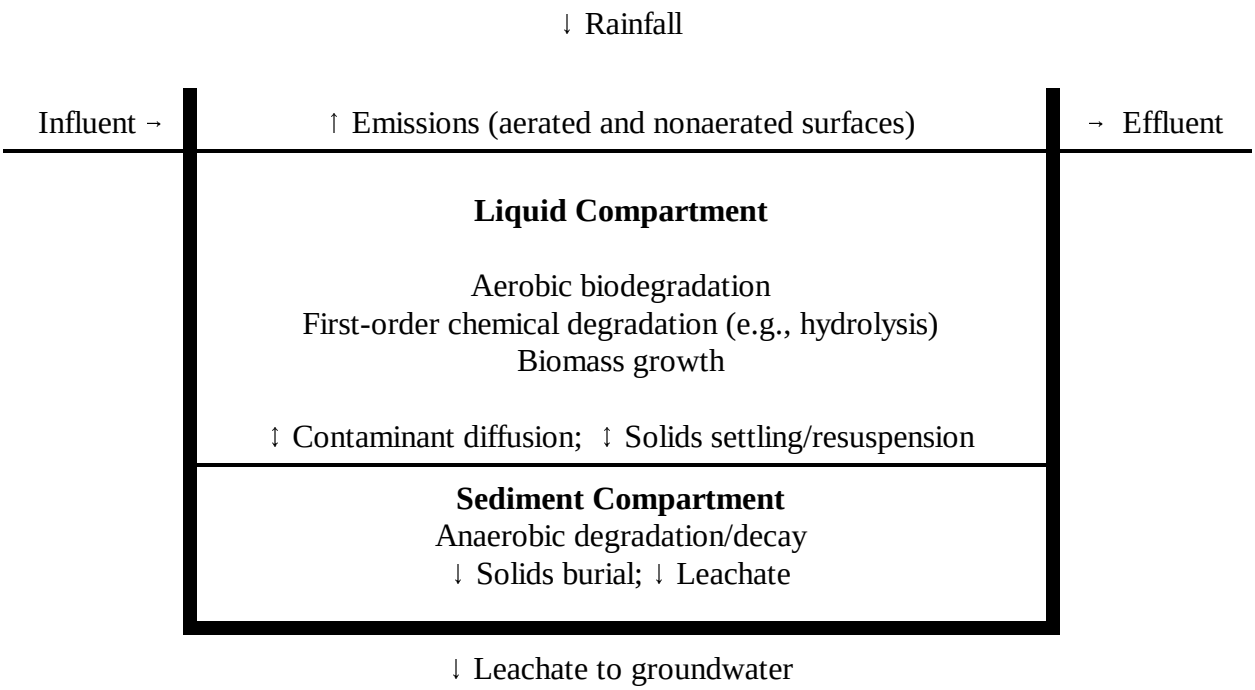

Figure 4-9. Schematic of general model construct for tanks and surface impoundments.

Solids generation occurs in the liquid compartment due to biological growth; solids destruction occurs in the sediment compartment due to sludge digestion. Using a well-mixed assumption, the suspended solids concentration within the WMU is assumed to be constant throughout the WMU. However, some stratification of sediment is expected across the length and depth of the WMU so that the effective total suspended solids (TSS) concentration within the tank is assumed to be a function of the WMU's TSS removal efficiency, rather than equal to the effluent TSS concentration. The liquid (dissolved) phase contaminant concentration within the tank, however, is assumed to be equal to the effluent dissolved phase concentration (i.e., liquid is well mixed). Consequently, the term "mostly well mixed" is used to describe the liquid compartment.

The steady-state, mass balance equations on which the module is based are summarized below.

Constitutent Mass Balance in the Liquid Compartment. In the liquid compartment, there is flow both in and out of the WMU. There is also a leachate flow to the sediment compartment and out the bottom of the WMU for surface impoundments. Within the liquid compartment, there is contaminant loss through volatilization, hydrolysis, and biodegradation. Additionally, contaminant is transported across the liquid/sediment compartment interface by solids settling and resuspension and by contaminant diffusion. At steady state, the constituent mass balance for the liquid compartment is:

$$Q_{infl}\ C_{tot,infl} = Q_{out}\ C_{tot,out} + Q_{leach}\ C_{tot,1} + (K_{OL}\ A + k_{hyd}\ V_1\ \theta_{liq,1})\ C_{liq,1} + V_1\ (k_{bm}\ k_{ba}\ [TSS]_1)\ C_{tot,1} + v_{sed}\ A\ [TSS]_1\ C_{sol,1} - v_{res}\ A\ [TSS]_2\ C_{sol,2} - v_{diff}\ A\ (C_{liq,2} - C_{liq,1}) \quad (4\text{-}3)$$

where

Q_{infl}	=	volumetric flow rate of influent (m^3/s)
$C_{tot,infl}$	=	total contaminant concentration in influent stream (mg/L = g/m^3)
	=	$C_{in} \times f_{wmu}$ (assumes density of hazardous waste and other influent wastes are equal)
Q_{out}	=	volumetric flow rate of effluent (m^3/s)
$C_{tot,out}$	=	total contaminant concentration in effluent stream (mg/L = g/m^3)
Q_{leach}	=	leachate flow rate from WMU (m^3/s)
$C_{tot,1}$	=	total contaminant concentration in liquid compartment [and effluent] (mg/L = g/m^3)
K_{OL}	=	overall volatilization mass transfer coefficient (m/s)
A	=	total surface area of WMU (m^2)
k_{hyd}	=	hydrolysis rate (1/s)
V_1	=	volume of liquid compartment in WMU= d_1 A (m^3)
d_1	=	depth of liquid compartment (m)
$\theta_{liq,1}$	=	volumetric liquid content of liquid compartment (m^3/m^3)
$C_{liq,1}$	=	liquid phase contaminant concentration in liquid compartment (mg/L = g/m^3)
k_{bm}	=	complex first-order biodegradation rate constant (m^3/Mg-s)
k_{ba}	=	ratio of biologically active solids to the total solids concentration (i.e., $k_{ba} = [MLVSS]_1/[TSS]_1$)

$[TSS]_1$ = concentration of total suspended solids (TSS) in liquid compartment and in effluent ($g/cm^3 = Mg/m^3$)

$[MLVSS]_1$ = concentration of biomass as mixed liquor volatile suspended solids (MLVSS) liquid compartment and in effluent ($g/cm^3 = Mg/m^3$) $C_{tot,1}$ = total contaminant concentration in the WMU ($mg/L = g/m^3$)

v_{sed} = solids settling or sedimentation velocity (m/s)

$C_{sol,1}$ = solid phase contaminant concentration in liquid compartment (mg/kg = g/Mg)

v_{res} = solids resuspension velocity (m/s)

$[TSS]_2$ = concentration of total suspended solids in the sediment compartment ($g/cm^3 = Mg/m^3$).

$C_{sol,2}$ = solid phase contaminant concentration in sediment compartment (mg/kg = g/Mg)

v_{diff} = mass transfer coefficient between liquid and sediment compartments (m/s)

$C_{liq,2}$ = liquid phase contaminant concentration in sediment compartment ($mg/L = g/m^3$)

C_{in} = contaminant concentration in hazardous waste ($mg/L = g/m^3$)

f_{wmu} = mass fraction influent waste that is hazardous (Mg/Mg).

Constituent Mass Balance in the Sediment Compartment. Within the sediment compartment, there is contaminant loss through hydrolysis and biodegradation. Additionally, contaminant is transported across the liquid/sediment compartment interface by solids settling and resuspension and by contaminant diffusion. For surface impoundments, there is also leachate flow from the liquid compartment (which includes entrained sediment) and "filtered" leachate out the bottom of the WMU. At steady state, the general constituent mass balance for the sediment compartment is:

$$Q_{leach}\ C_{tot,1} = Q_{leach}\ C_{liq,2} + k_{bs}\ V_2\ C_{tot,2} + (v_{res} + v_b)\ A\ [TSS]_2\ C_{sol,2} + V_2\ \theta_{liq,2}\ k_{hyd}\ C_{liq,2} - v_{sed}\ A[TSS]_1\ C_{sol,1} + v_{diff}\ A\ (C_{liq,2} - C_{liq,1}) \quad (4\text{-}4)$$

where

k_{bs} = (anaerobic) biodegradation decay rate of contaminant (1/sec)

$C_{tot,2}$ = total contaminant concentration of sediment compartment (g/m^3)
v_b = solids burial velocity (m/sec)
$\theta_{liq,2}$ = volumetric liquid content of sediment compartment (m^3/m^3).

Solids Mass Balance in Liquid Compartment. Sedimentation and resuspension provide a means of sediment transfer between the liquid and sediment compartments. As seen in Equations 4-3 and 4-4, sedimentation and resuspension are assumed to occur in the quiescent areas. For systems in which biodegradation occurs within the liquid compartment, there is also a production of biomass associated with the decomposition of organic constituents. At steady state, the sediment mass balance for the liquid compartment is:

$$Q_{infl}([TSS]_{infl}+\lambda\epsilon_{BOD}C_{BOD}) = Q_{out}[TSS]_{out} + Q_{leach}[TSS]_1 + v_{sed}A[TSS]_1 - v_{res}A[TSS]_2 \quad (4\text{-}5)$$

where

λ = biomass yield (g-biomass (dry basis)/g-BOD)

ϵ_{BOD} = biological oxygen demand (BOD) removal efficiency of WMU (Mg/m^3)

C_{BOD} = biological oxygen demand of influent (Mg/m^3)

$[TSS]_{out}$ = concentration of total suspended solids (TSS) in the effluent (g/cm^3 = Mg/m^3).

Solids Mass Balance in Sediment Compartment. In the sediment compartment, as in the liquid compartment, sedimentation and resuspension provide a means of sediment transfer between the liquid and sediment compartments. In the sediment compartment, however, there is some accumulation of sediment during the time step. This sediment accumulation is also referred to as sediment burial, and the rate of sediment accumulation is determined by the burial velocity. The sediment mass balance for the sediment compartment is:

$$(Q_{leach} + v_{sed}\ A)\ [TSS]_1 - v_{res}\ A\ [TSS]_2 = v_b\ A\ [TSS]_2 \quad (4\text{-}6)$$

The primary output of the AT and SI model is the annual average volatilization rate (K_{ol} A $C_{liq,1}$).

SI Vadose Zone Infiltration. The SI model also outputs the average annual infiltration rate and the associated average annual leachate contaminant flux rate. The leachate flux rate is calculated as the dissolved contaminant concentration in the sediment pore water times the infiltration rate of leachate flow. The infiltration algorithm was developed specifically for use in HWIR99 and estimates infiltration rates as a function of liquid depth in the WMU, the thickness and hydraulic conductivity of the sediment compartment (considered as two layers, unconsolidated and consolidated), a native soil layer immediately below the WMU that has been

clogged by infiltrating sediment, and an unclogged native soil layer below the clogged layer. A schematic diagram of the conceptual model for the infiltration rate algorithm is shown in Figure 4-10. An iterative method is used to converge on the infiltration rate satisfying the physical requirement that the hydraulic pressure head at the groundwater table must be atmospheric pressure. A flowchart depicting how that iterative method is implemented within the monthly time step is provided in Figure 4-11.

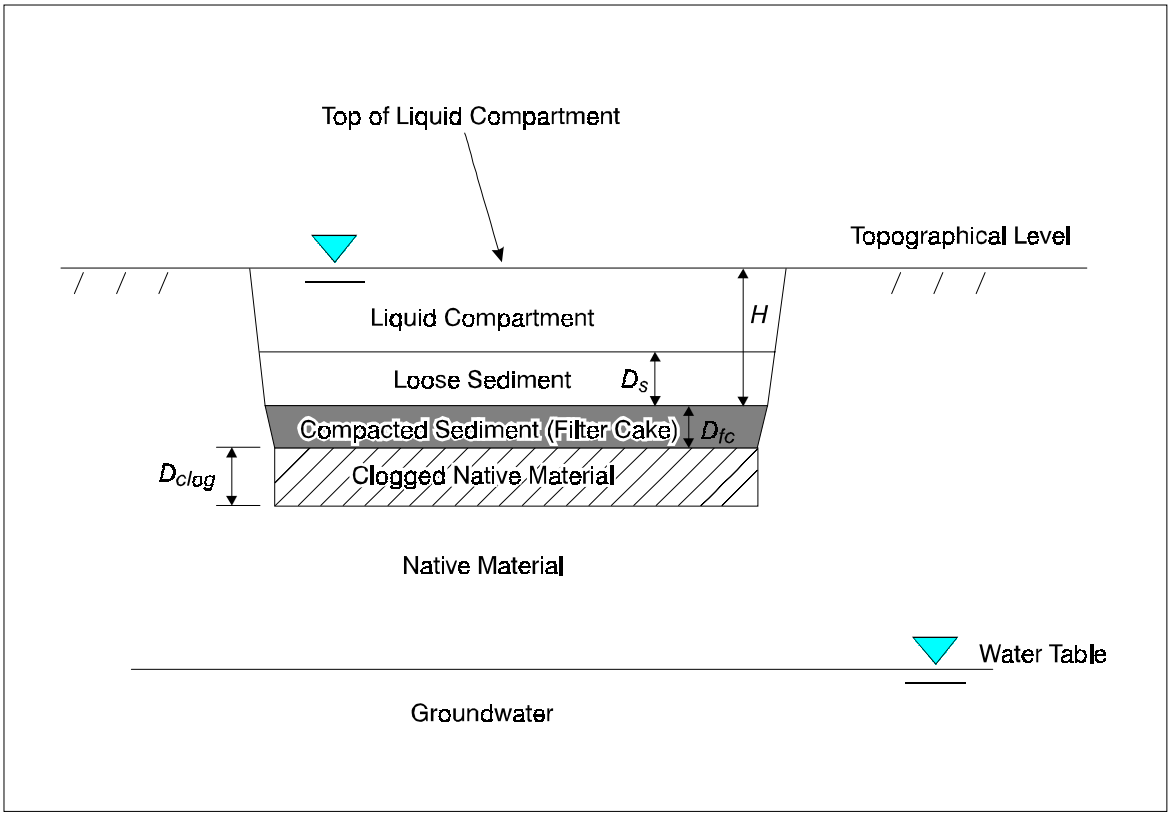

Figure 4-10. Conceptual model for SI infiltration algorithm.

4.3.2.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- *Pseudo steady-state assumption.* The pseudo-steady-state assumption is probably most limiting for the SI due to possible large volumes. It is possible (likely for large units) that a true steady-state condition is never reached. Thus, to the extent that the steady-state assumption results in treatment efficiencies that are different, on a monthly average, than would be expected under real (non-steady-state) conditions, the resulting chemical releases to air and groundwater will be in error. In general, one would expect steady-state conditions to represent more efficient treatment; therefore, some bias toward underestimation of releases may occur.

- *Strictly applicable only to dilute aqueous wastes.* Due to the simplicity of the biodegradation rate model employed and the use of Henry's law partitioning coefficients, the model is most applicable to dilute aqueous wastes. At higher

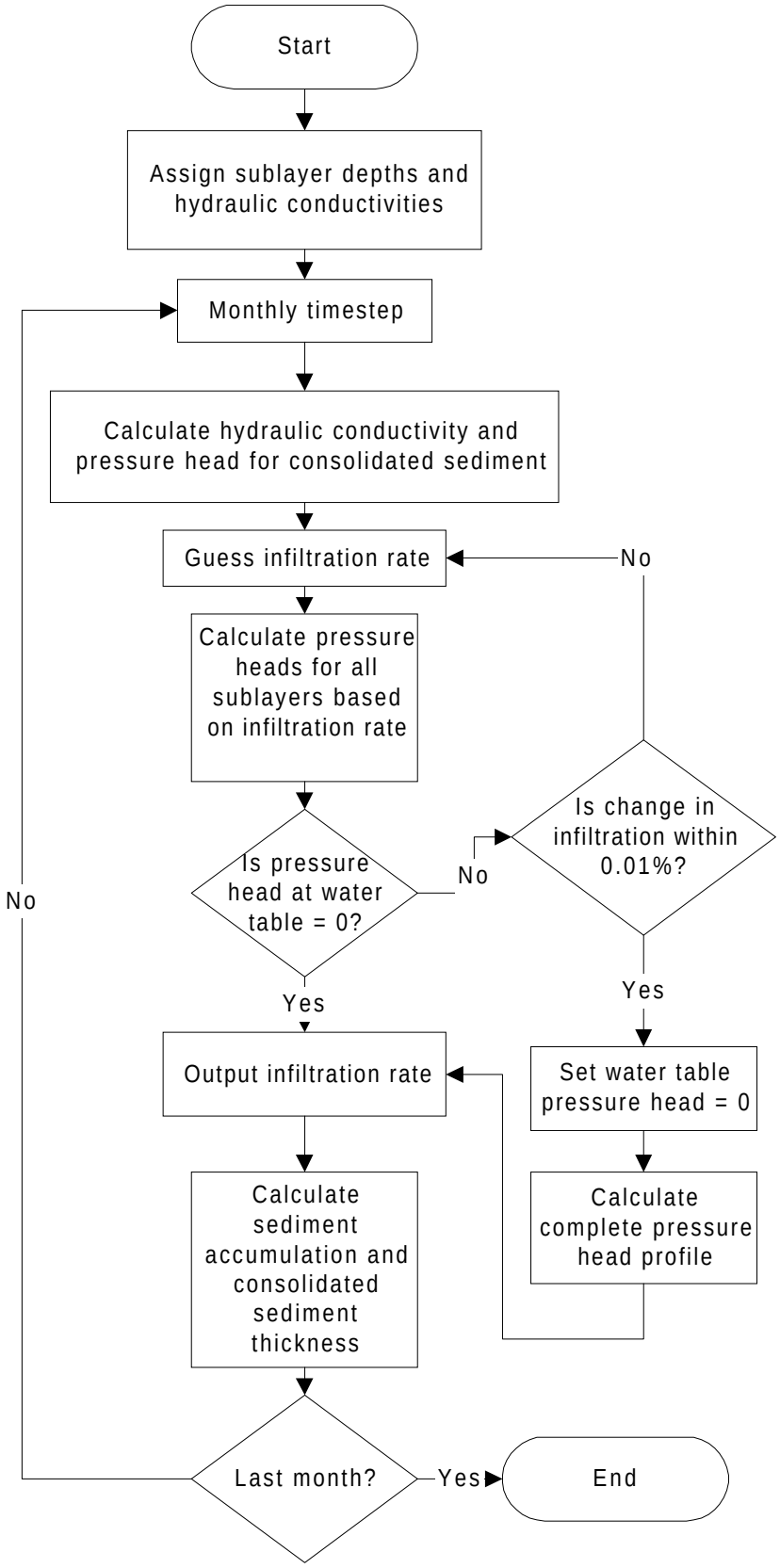

Figure 4-11. Flowchart of infiltration algorithm.

contaminant concentrations, biodegradation of toxic constituents may be expected to exhibit zero-order or even inhibitory rate kinetics. For waste streams with high contaminant or high total organic concentrations, vapor phase contaminant partitioning may be better estimated using partial pressure (Raoult's law) rather than Henry's law. Also, as daughter products are not included in the model, any contaminant emissions or leachate generated as a reaction intermediate or end product from either biodegradation or hydrolysis are not included in the model output.

- *Completely mixed (CSTR) assumption.* The CSTR assumption, especially for large SIs, will be erroneous at many of the sites; that is, the hydraulic regime is more likely to be near-plug flow than completely mixed. Such an error, at least in theory, would result in a systematic underestimation of the WMU removal efficiency and a subsequent overestimation of releases. This is because an ideal CSTR is less efficient under first-order reaction kinetics than is an ideal plug flow WMU of the same volume. It should be noted, however, that in reality CSTRs are often more efficient than plug flow WMUs, despite this theoretical handicap, because they buffer toxic effects on process microorganisms of incoming spills more effectively. This consideration would mitigate to some extent any such bias due to an erroneously assumed CSTR.

- *The module has not been field-validated*. Calibration and validation of the module to field data would lend strong credibility to the underlying algorithms.

4.3.3 Air Module

The Air Module simulates time series chemical concentrations in ambient air throughout the AOI and deposition fluxes of vapors and particulates to soils, surface waters, and plant materials. The ambient concentrations are used by the Human Exposure Module while the deposition fluxes are used by the Watershed Module, Surface Water Module, Terrestrial Food Chain, Farm Food Chain, and Ecological Exposure modules.

4.3.3.1 <u>Spatial and Temporal Scales</u>. The Air Module outputs are based on analytical solutions to the underlying mass balance and hydrodynamic equations; consequently, the spatial resolution of the outputs is continuous. However, the continuous output scale does not imply infinitely small spatial resolution. Area sources are approximated by the underlying solution technique as being disaggregated into a number of smaller area sources, each of which is then represented by a representative point source. Thus, some discretization error is introduced into the analytical solution, which compromises the spatial resolution.

Temporally, the Air Module outputs are a time series of annual average concentrations and deposition fluxes. The transport processes, however, are based on steady-state assumptions which give rise to time-invariant, normalized, intermediate transport outputs. These normalized (or unit) transport outputs are then multiplied by time-varying chemical emission releases from the source modules to determine the chemical concentrations and deposition fluxes. In summary, the

Air Module outputs are an annual average time series, but are based on steady-state transport assumptions.

4.3.3.2 Key Assumptions. The following key assumptions are noted:

Steady-state transport processes. The steady-state transport processes are based on long-term average meteorological conditions, as mentioned above.

4.3.3.3 Methodologies. The air model used in the Air Module is the Industrial Source Complex — Short Term (ISCST3) model. ISCST3 is a steady-state Gaussian plume model. The model provides estimates of pollutant concentration, dry deposition (particles only), and wet deposition (particles and gases). To address specific needs (mostly relating to computational burden) of HWIR99, several new features were added to ISCST3. These new features are summarized in the subsections below. A complete description of the technical algorithms in the HWIR99-specific version of ISCST3 (ISCST3-HWIR) can be found in U.S. EPA (1999a). The operational instructions are in U.S. EPA (1999b).

Revised Plume Depletion Scheme. The version of ISCST3 distributed by EPA's Office of Air Quality Planning and Standards contains the Horst (1983) plume depletion algorithm. This algorithm was found to be computationally intensive and gave spurious results for some cases. A new plume depletion and settling algorithm developed by Venkatram (1998) was implemented into ISCST3-HWIR, resulting in a faster, more robust approach. This approach is based on depleting material in a surface-based internal boundary layer that grows with distance from the source. In conjunction with this change, the deposition velocity algorithm was also modified by removing the inertial impaction term. The inclusion of this term appears to provide deposition velocity estimates that are too high for some particle sizes.

Sampled Chronological Input Model (SCIM). To reduce model run time, an option was added to ISCST3-HWIR to sample the long-term meteorological record at regular, user-specified intervals and scale the model results at the end of the run to produce the annual average estimates. This method is called Sampled Chronological Input Model. An advantage of this method is that hourly meteorology is used, which maintains the serial correlation between wet deposition and concentration. The user specifies two sampling intervals. Using the first interval, the meteorological data are sampled, ignoring any recorded precipitation and the concentration and dry deposition are calculated for each receptor location of interest. The second interval specifies the sampling rate for the hours of meteorological data during which precipitation was recorded. This sampling rate is used to determine the concentration, dry deposition, and wet deposition. The estimates from these separate schemes are combined at the end of the model run using a weighted average based on the number of hours sampled.

Output by Particle Size. For HWIR99, inhalation risks are determined for pollutants with particles sizes $\leq 10\ \mu m$. Therefore, a new output option was required to allow examination of concentration and deposition by particle size.

4.3.3.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

Errors Introduced by SCIM Interval. The SCIM sampling will introduce some error into the results. A rigorous analysis was performed, however, to select a SCIM interval that did not significantly compromise results based on use of all meteorological data.

Lack of Dry Deposition of Vapors Methodology. One of the largest areas of uncertainty is the deposition of gases. There are currently no air models that contain algorithms specifically designed to model the dry deposition of VOCs and SVOCs. Previous modeling exercises have used a transfer coefficient to model the dry deposition of gases. The concern with this approach is that deposition is calculated outside of the model, which precludes the consideration of the deposition in the amount of material depleted from the plume. This presents challenges in trying to satisfy conservation of mass objectives. To calculate the wet deposition of gases, chemical-specific scavenging coefficients should be used. However, these values are not readily available. An alternative approach is to select a single scavenging coefficient for all gases that is based on approximating the gases as very small particles. This approach may lead to underprediction of wet deposition for some gases and overprediction for others. An appropriate methodology for dry deposition of gases remains an issue for HWIR99.

4.3.4 Watershed Module

The Watershed Module simulates time series concentrations of chemical in watershed soils, which are used by the Terrestrial Food Chain, Farm Food Chain, and Human Exposure modules, and mass fluxes of chemical, eroded soil, surface water runoff, and baseflows to the Surface Water Module. It also simulates infiltration estimates that are used as regional aquifer recharge flows by the Saturated Zone Module.

4.3.4.1 <u>Spatial and Temporal Scales</u>. The Watershed Module is based on conceptual and mathematical models that are very similar to those already described for the LAU and WP sources, i.e. the combined “local watershed/soil column” algorithm. As previously mentioned, this algorithm is a dynamic, two-dimensional, fate and transport model that also includes hydrological functionality. There are two general differences between the way the algorithm is implemented in the LAU/WP Modules and the Watershed Module. First, in the Watershed Module, the algorithm is applied to each watershed making up the AOI with no further disaggregation – watersheds are not disaggregated into “subareas” as were the local watersheds containing either the LAU or WP, although that functionality is available in the Watershed Module software should it ever be needed. With no longitudinal disaggregation, the algorithm as applied in the Watershed Module is one-dimensional (vertical) only. Each watershed is independent of other watersheds and is simulated individually.

The other difference involves the size of the computational time step used to determine contaminant concentrations in runoff water. In the LAU/WP source modules, contaminant concentrations in runoff water and in the surface soil column layer are determined on a daily basis, even though the computational time step in the subsurface soil column layers is typically much larger than 1 day. (The time step used for any given year is a function of the annual average effective convection velocity of the contaminant in soil and the soil column layer thickness.) It

was determined that the daily time step was necessary in the LAU/WP source module's implementation of the watershed/soil column algorithm for the two following reasons:

- It was considered impractical to simulate annual average runoff without building up that annual average from daily precipitation and runoff events. (The precipitation/runoff model is nonlinear in the independent variable. One cannot simply input average annual precipitation as the independent variable and output average annual runoff.)

- An approximately daily time step is the fundamental temporal scale at which surface transport of chemical downslope in the local watershed is occurring. It was considered important to honor this time scale in simulating fate and transport from the sources.

For the Watershed Module, both of these considerations are still valid. However, indirect soil concentrations resulting from aerial deposition are likely to be significantly less than soil concentrations resulting from direct runoff/erosion from a source, and aerial deposition rates are only known on an average annual basis, not daily. For these reasons it was decided that, in the Watershed Module, to minimize run time and accommodate data limitations

- Soil erosion and runoff models would be executed on a daily time step. Daily results would then be used to determine annual average soil erosion (CSL) and runoff volume (Q).

- Annual average Q and CSL would be used to estimate the annual averages of the other runoff/erosion related parameters.

- The computational time step used by the watershed/soil column algorithm would be the same as that calculated each year for the subsurface soil column layers, i.e. it is based on numerical considerations, not physical. This time step does not exceed 1 year as a maximum.

- The annual average runoff-related parameters and the annual average aerial deposition rates would be used in applying the watershed/soil column algorithm at each computational time step.

In summary, annual average soil erosion and runoff are estimated on a daily time step, while the remainder of the model (contaminant mass fate and transport simulation) is executed on a computational time step that is typically much larger than 1 day and can vary each year of the simulation. All outputs are ultimately reported as annual averages, regardless of their individual computational time steps.

4.3.4.2 <u>Key Assumptions</u>. The following key assumptions are made by the module:

Watershed as a CSTR. Each watershed is assumed to be spatially homogeneous and no concentration gradients within a given watershed are simulated.

Annual average erosion/runoff rate constants can be used. See above discussion.

Surface water baseflow can be represented by the 30Q2 low flow statistic. See following discussion.

4.3.4.3 Methodologies. The methodology for dynamically simulating fate and transport of aerially deposited chemical to any given watershed is fundamentally identical (with the exceptions as noted above) to that previously described for the GSCM applied to an individual local watershed subarea for one of the nonwastewater source modules. (See Section 4.3.1.3.) In the Watershed Module application, an entire regional watershed is equivalent to the local watershed's subarea, so that the "soil column" for the GSCM is the watershed in its entirety.

The additional functionality added by the Watershed Module that is not included in the nonwastewater source modules is the estimation of dry weather flow emanating from each watershed as groundwater discharge into adjacent surface waterbodies. This dry weather flow constitutes the baseflow for those surface waterbodies and, when added to stormwater runoff flows, represents total surface water flow in those waterbodies. That methodology is summarized below.

For a given stream reach, baseflow can vary seasonally, or even near-continuously, as groundwater levels and/or interflow varies and can be estimated for a given time period by analysis of runoff hydrographs that include runoff as well as pre- and postrunoff flows. For HWIR99 purposes, however, it was considered unnecessary (and computationally impractical) to attempt to estimate within-year variability in baseflows. Rather, a single estimate was sought that would reasonably characterize annual average baseflow conditioned on stream reach order (or tributary drainage area), year, and hydrologic region.

The issue then became — what single flow statistic best represents annual average baseflow for a given region, reach order, and year? The widely available annual average streamflow would, in general, tend to overestimate baseflow. (Some losing streams might be exceptions.) Conversely, the common low flow statistic, 7Q10 (the minimum 7-day average flow expected to occur with a 10-year return period, i.e. at least once in 10 years), would, in general (if not always), tend to underestimate baseflow. As a compromise, it was assumed for HWIR99 that the 30Q2 low flow, i.e. the minimum 30-day average flow occurring, on average, at least once every other year, is a reasonable estimate of annual average baseflow for any given year. This flow statistic was not widely available from USGS gaging data and therefore was developed as a part of the HWIR99 effort. The procedure used was the following:

1. For each of the 18 USGS Hydrologic Units (HUCs) in the conterminous United States, retrieve from EPA's STORET database the long-term historical record of daily average streamflows for each USGS gage in that region and the gage's tributary drainage area.

2. Statistically analyze each gage's daily flow record to estimate 30Q2 values by gage.

3. Fit a regression model of the form $30Q2 = aA^b$ (a power function) to the data for all gages in a given region, where A is the gage tributary area. (In a few of the 18 regions, a linear model, i.e. 30Q2 = a + bA, provided a slightly better fit in the sense of explaining greater overall variation [R^2]. However, the improvement in R^2 was not considered to be significantly great as to outweigh the considerable advantage of the power function model of predicting zero flow for zero tributary area, which the linear model with an intercept term does not achieve.)

4.3.4.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are made by the module:

***Spatial Dilution of Hot Spots*.** Because each watershed is assumed to be uniform with respect to chemical concentrations in soil, hot spots resulting from nonuniform aerial deposition will not be detected.

***30Q2 Equivalent to Baseflow*.** There is uncertainty in the baseflow estimates, both in regard to using the correct low flow statistic (e.g., 30Q2) as well as in representing the variability in the baseflow for any given watershed. For a given watershed, the 30Q2 estimate of constant baseflow is a point estimate, generated from a regression model. The variability around that point estimate represented by the regression data could be quantified and used in a Monte Carlo sense, but it is not. Thus, the same baseflow will always be estimated for a given hydrologic region and watershed size.

***Module Has Not Been Field-Validated*.** Calibration and validation of the module to field data would lend strong credibility to the underlying algorithms.

4.3.5 Vadose Module

The Vadose Module estimates chemical fluxes to the Saturated Zone Module given leachate flows and fluxes from the source module.

4.3.5.1 <u>Spatial and Temporal Scales</u>. The Vadose Module outputs are spatially continuous in one dimension, the flow path dimension. Temporally, a quasi-steady-state approach is used. The transport processes are assumed to be at steady state, but are applied to incoming, annually varying leachate fluxes from the source modules, resulting in an annual time series of loads to the Saturated Zone Module.

4.3.5.2 <u>Key Assumptions</u>. Key assumptions of the module are:

- Steady-state transport processes.
- One-dimensional flow regime.

4.3.5.3 <u>Methodologies</u>. Flow in the vadose zone is modeled as steady-state, one-dimensional, and vertical from underneath the source and the surficial soil outside the unit toward the water table. The lower boundary of the vadose zone is the water table. The flow in the

vadose zone is predominantly gravity-driven, and therefore the vertical flow component accounts for most of the fluid flux between the source and the water table. The flow rate is determined by the long-term average infiltration rate through the waste management unit. Contaminant is transported in the vadose zone by advection and dispersion. Initially, the vadose zone is assumed to be contaminant-free and contaminants are assumed to migrate vertically downward. The technical details on the Vadose Module are provided in the background document for the vadose zone (U.S. EPA, 1999aa and ac).

The Vadose Module receives the net rate of vertical downward percolation from the waste management unit through the unsaturated zone and to the water table. Infiltration rates and contaminant mass fluxes emanating from the unit are provided as a time series of annual average rates. The Vadose Module and Saturated Zone Module require an effective steady-state infiltration rate and annual average contaminant concentrations. In calculating the effective infiltration rate, the Vadose Module conserves mass and uses the full time series of annual average rates.

The output of the Vadose Module is a time series of contaminant concentrations for each species, the times at which the concentrations are reported, the effective infiltration rate, and the duration of the source boundary condition.

4.3.5.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are made by the Module:

- Transient effects of the flow are not considered. (Year-to-year variability in infiltration is not considered.)
- Multiphase flow and transport are not permissible. NAPL flow and transport are not permissible.
- Vapor-phase diffusion is not allowed.
- “Fingering” effects in the vadose zone are excluded.
- Clay lenses or potential flow and transport barriers in the vadose zone are not considered.
- Decay is limited to first-order. Lag time for decay is not considered.
- The transport domain in the saturated zone is kept constant. Effects due to mounding caused by infiltration from waste management units are not considered. These effects would decrease the depth of the flow and transport domain in the vadose zone.

4.3.6 Saturated Zone Module

The Saturated Zone Module simulates time series of chemical concentrations in wells throughout that part of the AOI affected by a contaminated groundwater plume emanating from

the WMU. These concentrations serve as inputs to the Farm Food Chain, Terrestrial Food Web, and Human Exposure modules. The Saturated Zone Module also generates time series of chemical loadings to surface water reaches that are affected by the contaminated plume as input to the Surface Water Module.

4.3.6.1 Spatial and Temporal Scales. The spatial scale is pseudo three-dimensional. Steady-state flow is assumed with time-variable contaminant fate and transport.

4.3.6.2 Key Assumptions. The major simplifying assumptions used to simulate contaminant transport in the saturated zone are:

- The flow field is at steady state.
- Flow in karst environments is not modeled.
- The aquifer is homogeneous and initially contaminant free.
- Adsorption onto the solid phase is described by an equilibrium isotherm.
- Chemical and/or biochemical degradation of the contaminant can be described as a first-order process.
- The contaminants exist in two phases: solids and liquids. The liquid phase is considered a dilute solution of the contaminant.
- The flow field is not affected by traversing streams, nor by extraction wells.
- Mass lost to streams located between the wells and the waste management units is assumed to be small compared with the bulk of the contaminant mass in the saturated zone. All the streams are assumed to be gaining streams. Down-gradient wells beyond the streams are assumed to be unaffected by the presence of streams.

4.3.6.3 Methodologies. For HWIR99, the Saturated Zone Module simulates groundwater flow using a one-dimensional steady-state solution for predicting hydraulic head and Darcy velocities. The aquifer is assumed to be of uniform thickness, subject to recharge along the top of the aquifer with a regional hydraulic gradient. The saturated zone transport module simulated the advective-dispersive transport of dissolved one dimension with the other two dimensions added analytically (pseudo three dimensional). The technical details on the Saturated Zone Module are provided in the background document for the saturated zone (U.S. EPA, 1999aa, 1999ab).

In implementing the Saturated Zone Module, we set the initial contaminant concentration to zero. The concentration gradient along the downstream boundary is zero, and the lower aquifer boundary is taken to be impermeable. A zero concentration condition is used for the upstream aquifer boundary. Contaminants enter the saturated zone through a patch source on the

upper aquifer boundary directly beneath the source. Recharge of contaminant-free infiltration water occurs along the upper aquifer boundary outside the patch source. Transport mechanisms considered are advection, dispersion, linear or nonlinear equilibrium adsorption, and first-order decay.

The module requires as an input an effective, steady-state recharge rate. The primary outputs of the Saturated Zone Module are annual average concentrations at observation/receptor well locations for all chemical species and annual average mass fluxes to streams for all chemical species.

Although we did not implement this feature, the Saturated Zone Module can factor the effects of fractures in porous media into the modeling. Similarly, we also have the ability to incorporate effects of heterogeneity in aquifers (U.S. EPA, 1999ag), but did not implement this feature due to time constraints. Both of these capabilities are discussed further in the technical background document (U.S. EPA, 1999aa). However, due to time limitations, we did not implement these features.

The mobility of metals in the subsurface is dependent on the geochemical properties of the soil and groundwater. To account for the metal-specific interactions with various sub-surface environments, we used nationwide distributions of key geochemical parameters. In this methodology, we used the MINTEQA2 metals speciation code to generate nonlinear adsorption isotherms for each metal. We produced a set of isotherms for each metal reflecting the range of geochemical environments that is expected to be encountered at waste sites across the nation. We then used this set of isotherms to generate two subsets of isotherms for each metal: one for the vadose zone, and the other for the saturated zone.

4.3.6.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are made by the Module:

- Transient effects of the flow, recharge, and infiltration are not considered.
- Spatially varied recharge is not considered.
- Source geometry is limited to an idealized square, with two opposite sides parallel to the flow direction.
- Multiphase flow and transport are not modeled. NAPL flow and transport are not modeled.
- Contribution of contaminant to the saturated zone via vapor-phase diffusion above the water table is not modeled.
- Karst conditions are not modeled.
- Decay is limited to first-order. Lag time for decay is not considered.

- The presence of different hydrogeologic zones in the flow and transport domain is not considered.

- The transport domain in the saturated zone is kept constant. Effects due to significant mounding caused by infiltration from waste management units are not considered.

- Domain geometry is limited to the idealized rectangular shape. Other geometries are not considered.

- Only the gaining streams, with axes normal to the groundwater flow direction, are permitted. Effects of streams on the flow field are not considered.

- Only receptor wells with small extraction rates are considered. Effects of extraction on the groundwater flow field are not considered.

- There are many sources of uncertainty associated with the distribution coefficients generated by MINTEQA2. These can be categorized as: uncertainty arising from model input parameters, uncertainty in database equilibrium constants, and uncertainty due to application of the model.

4.3.7 Surface Water Module

The Surface Water Module receives chemical loadings due to direct WMU runoff and erosion from the source module, chemical loadings in runoff and erosion from aerially deposited loads from the Watershed Module, hydrological inputs (soil loads, runoff flows, and baseflows) also from the Watershed Module, and groundwater loads from the Saturated Zone Module. Outputs are used by the Farm Food Chain, Aquatic Food Web, Human Exposure, and Ecological Exposure modules.

4.3.7.1 <u>Spatial and Temporal Scales</u>. EXAMS is a compartment model that assumes no concentration gradients exist within each compartment being modeled. This zero-dimensional assumption is applied to lakes, ponds, and wetlands for HWIR99. For streams and rivers, each reach (waterbody segment between tributaries) is considered as a separate compartment. Thus, streams and rivers are represented as pseudo one-dimensional systems. Temporally, EXAMS is applied as a dynamic model.

4.3.7.2 <u>Key Assumptions</u>. Key assumptions of the Module include the following:

- No solids settling processes. A simplistic solids model has been included that treats suspended solids as a conservative substance; that is, solids are assumed to remain in the water column and not settle.

4.3.7.3 <u>Methodologies</u>. Chemical mass released from a WMU can enter the local surface waterbody network in runoff and erosion directly from the WMU, from atmospheric deposition to the water surface, in runoff and erosion from adjoining watershed subbasins, and by interception

of contaminated groundwater. The chemical is then subject to transport and transformation processes occurring within the waterbody network, resulting in variable chemical concentrations in the water column and in the underlying sediments. These chemical concentrations are the basis for direct exposure to human and ecological receptors and indirect exposure through uptake in the aquatic food web.

The Surface Water Module takes the loadings calculated by the source, atmospheric, watershed, and groundwater modules, along with data on meteorology, hydrology, environmental conditions, and chemical reactivity, and calculates the chemical concentrations throughout the waterbody network over time. The Surface Water Module consists of the core model EXAMS II (U.S. EPA, 1997a) and the interface module EXAMSIO (U.S. EPA, 1999au) . EXAMS is a general surface water fate model for organic chemicals. This compartment model has been used routinely by both EPA and industry analysts for the analysis of expected pesticide concentrations in generically defined environments, such as farm ponds. It has also been used for site-specific analysis of pesticide concentrations in various waterbodies around the world. The interface module EXAMSIO was developed specifically for this HWIR99 project. It reads data from other HWIR modules and databases and builds EXAMS input files describing the waterbody environment and chemical properties, along with the command file that specifies the chemical loading history and controls the EXAMS simulation. Control is passed to EXAMS, which conducts the simulation and produces intermediate results files. EXAMSIO then processes the intermediate files and passes the output data back to the proper HWIR databases.

4.3.7.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

The surface water module as implemented by EXAMSIO and EXAMS employs several simplifications to meet HWIR99 project requirements and constraints. The project design calls for repeated long simulations (200 to 10,000 years) executed quickly (seconds to minutes). This requirement limits the temporal resolution at which simulations can be conducted. Another important constraint is limited site-specific data. This constraint limits the accuracy with which a particular site can be described. The major model simplifications made in response to these project constraints include the use of average-yearly hydrological and loading inputs, the use of national distributions to specify some site-specific environmental conditions, and the use of a simple solids balance with no settling and burial. For sites that experience periodic drying, a small positive flow equivalent to 5 mm/yr of direct precipitation onto the waterbody surface was assumed to keep the model functioning.

These simplifications lead to a degree of model error in the calculated concentrations. Using annual-average loadings and flows rather than daily loadings and flows will lead to calculated annual-average concentrations that are biased somewhat high, depending on the correlation between flow and loading at a particular site. This bias is somewhat mitigated for reactive and volatile chemicals where the loss rate is proportional to the concentration. The simple solids balance will overestimate suspended solids concentrations slightly in streams and more significantly in ponds, wetlands, and lakes. Calculated total water column chemical concentrations will be high, while the dissolved chemical fraction will be low. The net result for dissolved water column chemical concentrations, which are used for fish exposure, is not expected to be biased significantly high or low.

The effect of assuming a small positive flow equivalent to 5 mm/yr of direct precipitation onto the waterbody to prevent drying is more difficult to evaluate. This procedure conducts chemical loads downstream within a remnant aquatic reach rather than within runoff over a dry bed. While the mass balance is maintained, the chemical and solids concentrations will tend to be elevated within the remnant reach. These elevated concentrations are probably realistic for years in which evaporation exceeds all hydrologic inflows.

4.3.8 Terrestrial Food Web Module

The Terrestrial Food Web Module estimates contaminant concentration in soil and uptake and concentration into associated prey and plant items. These concentrations are subsequently used as inputs by the Ecological Exposure Module. The Terrestrial Food Web Module addresses only concentrations in ecological receptor food webs. Human food webs are considered by the Farm Food Chain Module.

4.3.8.1 Spatial and Temporal Scales. The spatial scale of chemical concentrations in soils, plants, and prey items is an average across the home ranges of the ecological receptors. The underlying assumption is that receptors are mobile within their home range and spatially integrate soil and plant concentrations. The time scale is annual average.

4.3.8.2 Key Assumptions. Key assumptions of the module are the following:

Homogeneous Concentrations in Fruits and Vegetables Assumed. The exposure methodology makes no provision for the possible chemical concentration gradients within fruits or vegetables that might result in different concentrations in edible portions than when averaged throughout the food item.

Resuspension and Redeposition on Plants Are Not Considered. Plant concentrations are a function of the deposition of the contaminants that have been emitted from the waste management unit. Plant concentrations do not consider resuspension and redeposition. These processes can occur due to tillage, wind erosion, vehicular resuspension, and rain splash, but are not considered by this model.

Some Chemicals Rely Heavily on Empirical Uptake Data. This limitation is similar to that noted for the Farm Food Chain module. In essence, the paucity of data on uptake and accumulation of constituents in other terrestrial food items introduces significant uncertainty into this module. This limitation was explained in some detail in the proposed HWIR95; however, little progress has been made in the data and science required to model chemical uptake and accumulation in terrestrial habitats.

4.3.8.3 Methodologies.

Soil Concentrations. Soil concentrations are disaggregated from watershed-average concentrations, as calculated by the Watershed Module, to receptor home range-average concentrations in the Terrestrial Food Web Module. Home range-average soil concentrations are estimated both for surficial soils as well as depth-averaged concentrations. Surficial soil

concentrations are incidentally ingested by prey while eating food items in the soil, and those concentrations are passed on to the Ecological Exposure Module. Depth-averaged concentrations are relevant to uptake into plants through their root systems and are used internally in this module.

Prey Concentrations. Chemical concentrations in ecological receptor prey are estimated as a function of depth-averaged soil concentrations, chemical- and prey-specific bioaccumulation factors, and the fraction of each habitat that makes up the prey's home range. Prey considered in the module are earthworms, invertebrates, small mammals, small birds, omniverts, herbiverts, and small herpetafauna. Because these prey categories are typically made up of multiple species (e.g., a Cerulean Warbler, Marsh Hen, and Northern Bobwhite are all small birds), the module evaluates the prey concentration within each prey category as a maximum and minimum value among the associated species within the category. This maximum and minimum are then passed onto the Ecological Exposure Module where prey concentrations are randomly sampled from this feasible range. Concentrations in prey are estimated as

$$Creceptor_{HabRange} = CTdaAve_{HabRange} \cdot ChemBAFreceptor \cdot HomeRange_{frac} \qquad (4\text{-}7)$$

where

$Creceptor_{HabRange}$	=	concentration in prey animal
$CTdaAve_{HabRange}$	=	depth-averaged soil concentration in the home range
ChemBAFreceptor	=	chemical- and prey-specific bioaccumulation factor
$HomeRange_{frac}$	=	fraction of home range within habitat.

Plant Concentrations. The following plant categories are considered to be ecological receptor food items: exposed fruit, exposed vegetables, forage, silage, and grain. There are three mechanisms by which contaminants can bioaccumulate in vegetation: deposition of particle-bound contaminants to exposed plant tissues and subsequent uptake, deposition of vapor-phase contaminant to exposed plant tissues and subsequent uptake, and root uptake and translocation. Exposed fruit, exposed vegetables, forage, and silage are considered to be exposed vegetation; therefore, particle-bound contaminants, vapor-phase contaminants, and root uptake are all considered in those calculations. Grain is considered to be a protected vegetation; therefore, air contaminants are not included in the concentration calculation. Silage refers to any kind of plants grown for animal consumption, which could be partly protected and partly exposed. Consequently, silage concentrations are calculated as an exposed vegetation but are also assigned an empirical parameter to account for being partly exposed.

Plant concentrations are calculated as a function of the relevant media concentration (soil or air deposition) and an empirical concentration factor, which varies depending on the uptake mechanism and chemical type. Uptake mechanisms are air-to-plant biotransfer of ambient air concentrations, air-to-plant transfer from direct deposition (vapor and particulate), and root concentration. The general types of chemicals that the empirical concentration factors have been

developed for are dioxin, mercury, metal (nonmercury), organic (nondioxin), and "special." (See Section 4.3.10 for a discussion of these chemical types.)

4.3.8.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- Empirically based approach. The use of empirical uptake parameters introduces uncertainty into results, particularly if any extrapolation beyond the range of the empirical data is involved.

- Assumed BAF of 1.0 as default when data are missing. BAFs can vary significantly. Assumption of a default value of 1.0 introduces a major uncertainty.

4.3.9 Farm Food Chain Module

The Farm Food Chain Module estimates chemical concentrations in terrestrial food items that humans eat, based on uptake of soil, air, surface and/or groundwater concentrations and their subsequent concentration in the food items. The food items include plants (fruits and vegetables), beef tissue, and milk. These food item concentrations are subsequently used as inputs by the Human Exposure Module. It should be noted that these contaminated food items occur **both** on farms, for farmer receptors, as well as residential garden plots, for nonfarmer (residential) receptors; thus, "Farm" Food Chain is somewhat of a misnomer, or is at least incomplete. The Farm Food Chain Module is very similar to the Terrestrial Food Web Module in its use of plant uptake and concentration factors. Indeed, many of the plant uptake equations are identical between the two modules. Further, in lieu of the prey concentrations, this Module calculates beef tissue and milk concentrations for human ingestion, but the methodologies are analogous. One difference between the two modules is that farm-average soil concentrations are not calculated by this Module, as were home range-averages by the Terrestrial Food Web Module. Rather, they are calculated by the Human Exposure Module.

4.3.9.1 <u>Spatial and Temporal Scales</u>. The spatial scale of chemical concentrations in human food items is the average across the farm. The time scale is annual average.

4.3.9.2 <u>Key Assumptions</u>. Key assumptions of the Farm Food Chain Module are the following:

Similar Uptake and Accumulation of Contaminants Is Assumed for All Species of Fruits and Vegetables. The FFC module algorithms do not distinguish between different species of fruits in vegetables in predicting plant concentrations. It is assumed that uptake and accumulation of contaminants is similar across the categories of interest (e.g., exposed vegetables) and that the empirical data and algorithms provide a reasonable method to estimate plant concentrations.

Resuspension and Redeposition on Plants Are Not Considered. Plant concentrations are a function of the deposition of the contaminants that have been emitted from the waste management unit. Plant concentrations do not consider resuspension and redeposition. These

processes can occur due to tillage, wind erosion, vehicular resuspension, and rainsplash, but are not examined by this model.

Inhalation and Dermal Exposure Are Not Considered in Cattle. Beef and dairy cattle calculations only consider contaminant pathways of food, soil, and water ingestion. Any other pathway such as inhalation or dermal exposure are not considered in this module.

4.3.9.3 <u>Methodologies</u>. Contaminated plants that humans are assumed to ingest are considered as belonging to three categories: exposed fruits and vegetables, protected fruits and vegetables, and root vegetables. Chemical concentrations in these plants are estimated using similar methodologies (empirical concentration factors) to those previously described in the Terrestrial Food Web Module and will not be repeated here.

Beef and dairy cattle (and their products, beef tissue and milk) become contaminated through ingestion of soil, plants, and water. Soil concentrations are estimated by disaggregating the watershed-average soil concentration down to the farm level using area-weighted averaging factors. Uptake into cattle feed plants from the soil is estimated using uptake and concentration factors as previously described. Drinking water concentrations are assumed to result from untreated surface water (as estimated by the Surface Water Module), if at least one surface water reach occurs within the farm's boundaries. Otherwise, untreated groundwater well concentrations (located at the farm centroid) from the Saturated Zone Module are used.

Given the soil, plant, and drinking water concentrations, the chemical concentration in beef tissue is estimated as:

$$Abeef = 1000\ [[[(P_{forage_{farm_{DW}}} \cdot Qp_{forage_{beef}} \cdot f_{forage_{beef}}) + (P_{grain_{farm_{DW}}} \cdot Qp_{grain_{beef}} \cdot f_{grain_{beef}}) + (P_{silage_{farm_{DW}}} \cdot Qp_{silage_{beef}} \cdot f_{silage_{beef}})] + [CTssAve \cdot Qs_{beef} \cdot Bs]]\ Ba_{beef}] + [(1000\ WBNRchConcWaterDiss \cdot Qw_{beef} \cdot Ba_{water})] \quad (4\text{-}8)$$

where

Abeef	=	concentration of contaminant in beef (µg/g whole weight)
1,000	=	units conversion factor (1,000 g/kg)
$P_{forage_farm_DW}$, $P_{grain_farm_DW}$, $P_{silage_farm_DW}$	=	forage, grain, and silage contaminant concentrations (mg/kg wet weight), respectively
Qp_{forage_beef}	=	quantity of forage eaten by beef cattle (g plant DW/d)

f_{forage_beef}	=	fraction of forage grown in contaminated area (unitless)
Qp_{grain_beef}	=	quantity of grain eaten by beef cattle (g plant DW/d)
f_{grain_beef}	=	fraction of grain grown in contaminated area (unitless)
Qp_{silage_beef}	=	quantity of silage eaten by beef cattle (g plant DW/d)
f_{silage_beef}	=	fraction of silage grown in contaminated area (unitless)
CTssAve	=	average concentration in surficial soil from across farm area (µg/g soil)
Qs_{beef}	=	quantity of contaminated soil eaten by beef cattle (g soil/d)
Bs	=	bioavailability fraction of contaminant in soil relative to vegetation (unitless)
Ba_{beef}	=	biotransfer factor for beef cattle from plants and drinking water (d/g tissue whole weight)
WBNRchConcWaterDiss	=	dissolved concentration in surface water used as drinking water source by beef cattle (mg/L)
1,000	=	units conversion factor (1,000 µg/mg)
Qw_{beef}	=	quantity of drinking water consumed by beef cattle (L/d)
Ba_{water}	=	default biotransfer factor of dissolved contaminant in drinking water set at 1 (d/g).

Milk concentrations are similarly estimated by substituting a dairy biotransfer factor (Ba_{dairy}) for the beef cattle biotransfer factor (Ba_{beef}) in the above equation.

4.3.9.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- *Plant categories are based on exposure factors rather than plant physiology.* The categories of plants for which concentrations are calculated (e.g., protected fruit; exposed vegetables; root vegetables) are based on the aggregation of plants for exposure assessment purposes. Assuming that uptake and accumulation of

contaminants is similar for these categories introduces significant uncertainty in the module results. It is presumed that less uncertainty would be associated with estimating plant concentrations based on similar plant physiology and principles of botany.

- *The FFC module relies heavily on empirical data for some constituents.* Because mechanistic models are not available for some constituents (e.g., metals), the module requires empirical data to estimate concentrations in beef, milk, and plants. This introduces considerable uncertainty in the estimates since the study conditions seldom simulate the actual site conditions (e.g., soil conditions; plants of interest).

- *The FFC module uses a number of default parameters recommended by EPA.* Although EPA has done extensive research to develop reasonable default values for certain parameters, they have been clear in suggesting that site or regional data are preferred. Implementing the module with these default values introduces a degree of uncertainty that, given the state-of-the-science, can not be quantified.

- *The methodology used to predict concentrations in beef in milk has not been previously tested.* The FFC module introduces new methods to account for the contribution to beef and milk contaminant concentrations from contaminated ground and surface water (e.g., farm pond). There is some uncertainty in developing a new methodology to account for contaminant uptake in cattle from drinking water.

4.3.10 Aquatic Food Web Module

The Aquatic Food Web Module estimates chemical concentrations in aquatic organisms that are consumed by human and ecological receptors (Human and Ecological Exposure modules).

4.3.10.1 <u>Spatial and Temporal Scales</u>. The spatial resolution is identical to that of the inputs from the Surface Water Module; that is, each pond, wetlands, lake, or individual reach of a stream or river is assumed to be a CSTR with no spatial gradients. Temporally, the module can be considered as quasi-steady-state where the annually varying outputs of the Surface Water Module are updated each year under an instantaneous, steady-state assumption.

4.3.10.2 <u>Key Assumptions</u>. Key assumptions of the module are the following:

- *All waterbodies that define aquatic habitats are fishable.* The module assumes that all third-order stream reaches (and above), ponds, lakes, and certain permanently flooded wetlands support a multicompartment aquatic food web. The simple food webs developed for each of these aquatic habitats provide a useful framework for predicting tissue concentrations in aquatic organisms for a national assessment. Nevertheless, it is a certainty that not all of the waterbodies

designated as fishable in this analysis will be of sufficient quality to sustain a multicompartment food web.

- *Variability in aquatic systems is reasonably represented.* The underlying framework developed for the Aquatic Food Web Module (as applied in a national analysis) is the eight representative aquatic habitats. It is implicitly assumed that these eight habitats provide adequate resolution of the major types of freshwater systems within the constraints of available data and modeling tools.

- *Hydrophobic organics may be defined as organic chemicals with log $K_{ow} \geq 4.0$.* Although a strict definition for hydrophobic organics has not appeared in the literature, the Aquatic Food Web Module assumes that a reasonable cutoff is a log K_{ow} value of 4.0. Comparisons of predicted bioaccumulation factors (BAFs) derived with mechanistic models versus BAFs derived using regression equations suggests that, below log K_{ow} = 4.0, the difference in BAF estimates is below the level of resolution that these models are capable of.

- *The model construct is applicable to waterbodies other than coldwater lakes.* A number of journal articles and reference texts were reviewed in evaluating appropriate mechanistic models to simulate the uptake and accumulation of hydrophobic organics in aquatic organisms. From that review, it was determined that the underlying theory for these models is remarkably similar and that there is no inherent advantage in selecting one model over another. Although the Gobas (1993) model was calibrated for coldwater lakes (i.e., Lake Ontario), it was determined that this model construct was appropriate for use on other aquatic systems under the general assumption of steady-state conditions.

4.3.10.3 <u>Methodologies</u>. The methodology and equations used in the Aquatic Food Web module consist of two components: (1) an underlying framework of eight representative aquatic habitats, and (2) chemical-specific subroutines that are used to calculate tissue concentrations in fish and other aquatic organisms. A critical first step in developing the Aquatic Food Web Module was to create a framework that could capture the variability in aquatic habitats and fish species found in various hydrological regions across the United States. To represent this variability, simple freshwater food webs were constructed to depict the major functional and structural components of a "healthy" aquatic ecosystem (see Figure 4-12 for lake example). Taxa of plants, prey, and predatory fish were selected to represent components of the aquatic food webs according to major categories of aquatic biota in freshwater systems: aquatic macrophytes, phytoplankton, periphyton,

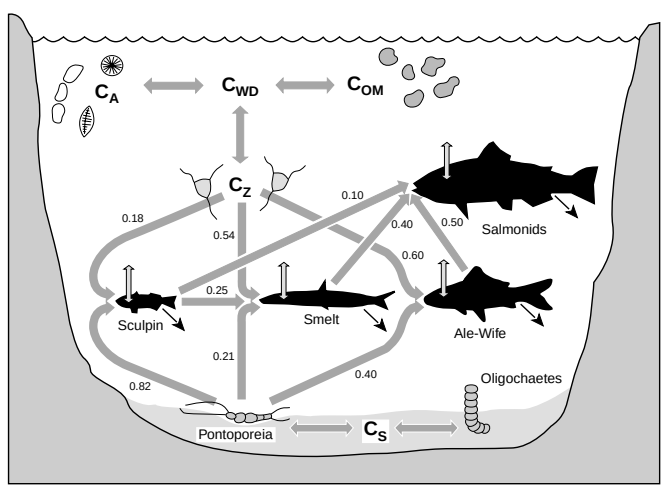

Figure 4-12. Example of simplified food web for lake habitat (Gobas, 1993).

zooplankton, benthic detritivores, benthic filter feeders, and fish in trophic levels (TL) 3 and 4. Four functional classes of trophic level (TL) 3 fish were selected based on feeding guilds and habitat niche and included zooplanktivores, benthivores, omnivores, and piscivores. The TL3 fish classes were further grouped according to size (small, medium, and large) to ensure that feeding habits were reasonable (i.e., small fish do not eat large fish). These habitat-specific food webs simulate pathways of chemical movement and biological uptake in the system and are used to estimate exposure concentrations in aquatic food items consumed by predators that inhabit the margins of the waterbody (e.g., stream corridor, lake margins). Table 4-7 summarizes the biota comprising the food webs for each habitat considered.

The methodology developed to estimate tissue concentrations is largely based on the physical and chemical properties of the constituents of concern. Mechanistic models, regression equations, and empirical data (derived using a weight-of-evidence approach) are all used to predict the tissue concentrations in aquatic biota. The Aquatic Food Web Module recognizes five chemical types and, based on the chemical properties, calls the appropriate subroutines and data. The chemical types are: dioxin-like chemicals (D), organic chemicals (O), special chemicals (S), metals (M), and mercury (Hg). The conceptual approach for each chemical type is summarized below.

Dioxin-like chemicals (D) - Dioxin-like chemicals and other relatively insoluble, nonmetabolizable organic chemicals (designated as O) are considered hydrophobic organics. The cutoff for hydrophobic organics is log $K_{ow} \geq 4.0$, and, therefore, all "D" constituents and any "O" constituent with a log K_{ow} in that range are modeled using a steady-state bioaccumulation model based on the work of F.A.P.C Gobas (Gobas, 1993). In addition to the log K_{ow} for the chemical of interest, the module requires inputs on the species assigned to the aquatic food web (e.g., lipid fraction, body weight, dietary preferences) as well as selected variables on the waterbody such as the fraction organic carbon in bed sediment. The module does not require information on water quality parameters such as dissolved and particulate organic carbon (DOC and POC); the Surface Water Module performs the partitioning and provides contaminant concentrations in surface water (dissolved and total) and in sediment (dissolved in pore water and total). The Aquatic Food Web Module contains a subroutine that calculates whole-body tissue concentrations (adjusted for lipid content) for aquatic macrophytes, benthic organisms, and TL3 and TL4 fish, and outputs these values for use by the Ecological Exposure Module. The Aquatic Food Web Module also calculates the filet concentrations for TL4 fish and for species of TL3 fish presumed to be edible for humans and outputs these values for use by the Human Exposure module.

Organic chemicals (O) - As noted above, the Aquatic Food Web Module recognizes hydrophobic organics as those constituents for which ChemType = "D" or ChemType = "O" **and** the log K_{ow} value is greater than or equal to 4.0. Other nonmetabolizable organic chemicals are considered as hydrophilic and the dominant exposure route is presumed to be via gill uptake (i.e., gill uptake is much greater than food/particle ingestion). This is a particularly important assumption since it is the concentration gradient in the fish gut that is believed to be the mechanism by which the tissue concentration in fish increases up the food chain for certain contaminants (i.e., biomagnification). If the uptake via gill exposure is the dominant pathway, a more simplistic approach is appropriate, namely, the use of empirically derived regression

equations. Following a review of numerous regressions used to predict the bioconcentration factors (BCFs) in fish, the methodology developed by Bertelsen et al. (1998) was chosen as the most appropriate approach. In addition, the Bertelsen methodology provides an algorithm to directly calculate the BCF for muscle tissue (i.e., filet) in fish. The tissue concentrations in aquatic organisms other than fish (e.g., benthos) are predicted using a partitioning approach as described in Gobas (1993).

Special chemicals (S) - This chemical type includes, primarily, constituents for which "special" subroutines or data are required for the module to excecute. For the Aquatic Food Web Module, this chemical type is generally used to indicate that the constituent is significantly metabolizable by fish. For example, it is widely accepted that aquatic organisms (particularly fish) readily metabolize polycyclic aromatic hydrocarbons (PAHs) and that BAFs predicted for PAHs with log K_{ow} values greater than ~ 5.0 overestimate the bioaccumulation potential. Consequently, empirical data on either bioaccumulation or metabolism are needed to predict tissue concentrations for PAHs. For the proposed HWIR95, BAFs were identified from the open literature and from an EPA report developed to support the Great Lakes Water Quality Initiative (*Derivation of Proposed Human Health and Wildlife Bioaccumulation Factors for the Great Lakes Initiative*, Stephan, 1993). Subsequent to HWIR95, the EPA Office of Solid Waste (OSW) performed additional analyses to investigate other data sources for PAH bioaccumulation factors as well as alternative methods for their derivation. After a comprehensive literature survey and review, it was determined that the empirical database on PAH bioaccumulation was insufficient to support defensible BAFs for most PAHs. Thus, two alternatives have been proposed to estimate tissue concentrations for PAHs. The first alternative involves the use of interval analysis (or fuzzy arithmetic) to derive a BAF from empirical data as described by Spencer and Beaulieu (1997). In brief, the interval analysis predicts a range of bioaccumulation factors associated with a given likelihood. The second alternative requires metabolic rates for use in the bioaccumulation model. The model developed by Gobas (1993) is designed to predict appropriate BAFs at steady-state conditions for hydrophobic organic chemicals, provided that the metabolic rates are available. These alternatives, of course, are not mutually exclusive and research is ongoing to determine the most appropriate approach for PAHs.

Metals (M) - Bioaccumulation factors for metals are estimated exclusively from empirical data. Few models are available that can be used in a national-scale analysis to estimate metals transport and accumulation in the food web from surface waters and sediments. Consequently, OSW has devoted considerable effort toward identifying studies and developing criteria for selecting appropriate bioaccumulation factors for metals.

Although uptake and accumulation is not of concern for all metals, the impact of surface water characteristics (particularly dissolved organic carbon) on bioavailability is significant. Several modeling approaches have been developed recently that may be used to predict bioavailability (e.g., the Windermere Humic Aqueous Model - WHAM), and water effects ratios (WER) provide empirical ratios that may be used to adjust water quality criteria to account for the mitigating effects of natural waters. Moreover, the effects and accumulation of essential metals change with concentration (i.e., bioconcentration is nonlinear); thus, a single BCF ratio may be inappropriate.

Based on current information on accumulation of metals in aquatic organisms, essential metals (e.g., Cu, Zn) were distinguished from nonessential metals (e.g., Cd, Pb) in evaluating data on uptake and accumulation. In addition, information on speciation was considered in deriving appropriate BCFs or BAFs since the environmental form of the metal may be very different than the metal salt studied in the laboratory. As discussed in the data collection documentation, a weight-of-evidence approach has been used to derive BAFs for metals that recognizes the importance of considering the essentiality of metals and the potential effects of water quality parameters. (Despite these efforts, it was not possible to account for regional variability in water quality and species characteristics in deriving the majority of bioaccumulation factors for metals.)

Mercury (Hg) - Tissue concentrations of mercury were estimated using empirical data from the *Mercury Report to Congress* (U.S. EPA, 1997h). The development of the Surface Water Module and the Aquatic Food Web Module were closely coordinated so that mercury modeling is conducted in a consistent manner by both modules. Specifically, the Aquatic Food Web Module calls for the concentration of methylmercury in surface waters provided by the SW module and applies bioaccumulation factors for methylmercury for TL3 and TL4 fish, as appropriate, to calculate the tissue concentration. Although EPA recognizes that the surface water (and sediment) characteristics exert a significant influence on the uptake and accumulation of mercury in the tissues of aquatic organisms, the modeling system was not designed to accommodate the level of complexity in site-specific models such as the Mercury Cycling Model (MCM - Hudson et al., 1994). To reduce the uncertainty in fish tissue concentrations of mercury, future research could be conducted on distributions of BAFs for methyl mercury relative to the water quality characteristics of the aquatic habitat. (Alternatively, the modeling system could be modified to incorporate a mercury-specific module into the system that would bypass both the SW module and the Aquatic Food Web Module. The ChemType variable could be used to initiate a “sub-module” in the system and provide a more mechanistic option for mercury modeling in aquatic systems.) As implemented in the current Aquatic Food Web Module, point estimates for methylmercury BAFs will be used to predict tissue concentrations in aquatic organisms.

4.3.10.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- *Steady-state conditions are generally assumed.* Because annual average concentrations are provided by the SW module, the Aquatic Food Web Module assumes steady-state conditions. As a result, the module cannot be used to evaluate the impacts from storm events nor can it be used to distinguish the impacts on tissue concentrations from peak events and subsequent averaging from long-term, low-level exposures. For example, a storm event may contaminate a given reach for relatively short periods of time, probably well below the duration required for organisms to reach steady-state for most chemicals.

- *The module relies heavily on empirical data for many chemicals.* For chemicals other than nonmetabolizable organics, mechanistic models are not used to predict tissue concentrations. Hence, the Aquatic Food Web Module estimates tissue

concentrations by multiplying empirical factors (primarily bioconcentration factors, or BCFs) by water concentrations. As discussed in the data collection documentation on the Aquatic Food Web parameters, these BCFs are measured under conditions that may not be relevant to all possible conditions (and species) included in the HWIR99 analysis.

- *The module does not allow for separate treatment of essential metals.* Bioconcentration of essential metals is not linear and modeling approaches are available to account for nonlinearity (see Bergman and Dorward-King, 1997). Bioconcentration of essential metals tends to be much greater at low concentrations than at higher concentrations since organisms actively seek to sequester necessary nutrients. Because many metals are regulated in biological systems, the apparent bioconcentration of metals at low concentrations may simply result in metal accumulation at "healthy" levels.

- *The module currently lacks the capability to use sediment concentrations directly in predicting tissue concentrations.* The Aquatic Food Web Module was developed primarily to use dissolved and total contaminant concentrations to predict tissue concentrations. Although sediment concentrations are used in predicting uptake and accumulation into benthic dwellers, the Aquatic Food Web Module lacks the necessary algorithms to use these data directly to predict concentrations in plants or fish. For certain constituents (e.g., dioxins), it may be useful to build this functionality into the module to provide greater flexibility in data use.

- *The module has not been validated in field studies.* Much of the modeling theory on which the Aquatic Food Web Module is based is widely accepted and has been used in numerous analyses. In particular, the methods used to predict concentrations of hydrophobic organics have been validated in coldwater lakes. However, the module has not been validated for other freshwater aquatic habitats, nor has it been validated *in toto* for application in a national-scale analysis.

4.3.11 Human Exposure Module

The Human Exposure Module calculates the applied dose (mg of constituent per kg of body weight), that is, the exposure, to human receptors from media and food concentrations calculated by the Air, Watershed, Surface Water, Saturated Zone, Farm Food Chain, Terrestrial Food Web, and Aquatic Food Web modules.

4.3.11.1 <u>Spatial and Temporal Scales</u>. The spatial scale of the estimated exposures is set by the scale of the inputs. For nonfarm receptors (residential receptors); the scale for air inhalation and drinking water and crop ingestion exposures is a point estimate (at the centroid of the associated census block within a distance ring). For residential fish ingestion exposures, the scale is an average over up to three randomly selected fishable reaches within the AOI. For farm receptors, the spatial scale for air inhalation and crop and beef ingestion is farm-averaged. For drinking water ingestion, it is a point estimate at the farm centroid. For fish ingestion, it is

averaged over up to three randomly selected fishable reaches, as for residential fishers. The temporal scale is annual average as dictated by the media and uptake module output scales.

4.3.11.2 <u>Key Assumptions</u>. Key assumptions include the following:

- Residential receptors are stationary and are all located at the census block/ring centroid.
- No background concentrations of chemical are present.

4.3.11.3 <u>Methodologies</u>. The Human Exposure Module calculates exposures for two basic receptor types: residential receptors (residents and home gardeners) and farmers. Residential receptors may also be recreational fishers in addition to being a resident or home gardener. Farmers may be beef farmers or dairy farmers, and either type of farmer may also be a recreational fisher. The subcategories within residential receptors and farmers differ in the particular exposures they incur. For example, a resident (only) differs from a home gardener in that home gardeners are exposed to contaminated vegetables but residents are not. Within each of the two basic receptor types, the Human Exposure Module calculates exposures for 5 age cohorts: infants (ages 0 to 1 year), children ages 1 to 5 years, children ages 6 to 11 years, children ages 12 to 19 years, and adults (ages 20 years and up).

The media inputs that are needed for the Human Exposure Module include ambient air concentration (both vapor and particulate), soil concentration, groundwater concentration, exposed vegetable concentration, protected vegetable concentration, exposed fruit concentration, protected fruit concentration, root vegetable concentration, beef concentration, milk concentration, and fish filet concentration for trophic level 3 and trophic level 4 fish. For vegetables and fruits, the terms "exposed" and "protected" refer to whether the edible portion of the plant is exposed to the atmosphere.

Exposure to humans other than infants may occur through eight pathways: inhalation of ambient air, inhalation of shower air, ingestion of groundwater, ingestion of soil, ingestion of fruits and vegetables, ingestion of beef, ingestion of milk, and ingestion of fish. However, not all receptors are exposed via all of these pathways. Residents are exposed via inhalation of ambient air, inhalation of shower air, ingestion of groundwater, and ingestion of soil. Home gardeners have the same exposures as a resident, plus exposure via ingestion of fruits and vegetables. All farmers are exposed via inhalation of ambient air, inhalation of shower air, ingestion of groundwater, ingestion of soil, and ingestion of fruits and vegetables. In addition, beef farmers are exposed via ingestion of beef, and dairy farmers are exposed via ingestion of milk. Recreational fishers have the same exposures as one of the other receptor types plus fish ingestion. Not all age cohorts are exposed via all pathways, shower exposures are calculated only for adults and children ages 12 to 19 years.

Infant exposure occurs via breast milk ingestion. The Human Exposure Module tracks noninfant exposure by the eight pathways described above. For infant exposure via breast milk, the maternal exposure via all pathways must be summed. Therefore, infant exposures are calculated for eight maternal exposure configurations: resident, home gardener, beef farmer,

dairy farmer, resident/recreational fisher, home gardener/recreational fisher, beef farmer/recreational fisher, and dairy farmer/recreational fisher. The mother is assumed to be an adult (as opposed to a teenager) for the purpose of calculating maternal dose in the infant breast milk pathway.

The Human Exposure Module performs calculations for residential receptors (residents and home gardeners) for a single x-y coordinate in each census block in the study area, and for farmers for a single, randomly located form in each census block group that supports farming land use. The x-y coordinate is located at the centroid of the census block group or farm. (Where census blocks or farms overlap the concentric, radial distance "rings," they are disaggregated into subblock groups or subfarms, each located exclusively in its respective ring. Populations are allocated to subblock groups or subfarms based on area-weighting of the subarea to the total area, i.e., populations are assumed to be uniformly distributed areally). Recreational fish exposures are calculated and averaged across up to three randomly selected reaches over the entire study area. The random selection of reaches is made once for recreational fishers who are residential receptors, and once for recreational fishers who are farmers.

An example exposure calculation for air inhalation average daily dose is presented below. It should be understood that exposures are calculated as a time series, with one value for each year in the time period under consideration.

$$Inh_{air} = \frac{C_{ambient} \times IR}{BW} \tag{4-9}$$

where

Inh_{air}	=	average daily dose for ambient air (mg/kg/d)
$C_{ambient}$	=	total ambient air concentration (mg/m^3)
IR	=	inhalation rate (m^3/d)
BW	=	receptor body weight (kg).

4.3.11.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- The spatial point estimates of average daily doses for a nonfarm receptor assume all receptors in that census block/ring are located at the centroid. Thus, to the extent that some receptors in fact reside at, or spend appreciable time at, more highly contaminated areas within the block/ring, their average daily doses will be underestimated. The converse is also true; that is, the centroid average daily dose will overestimate dose in other areas where concentrations are less. If chemical

concentrations decrease approximately linearly with distance away from the WMU, there is probably little if any bias introduced by the centroid assumption. If concentrations decrease nonlinearly (e.g., first-order air deposition), the centroid assumption may underestimate the true average daily dose across the block/ring to an unknown extent. The effect of any such bias will also be influenced by the size of the census block — relatively larger blocks have greater potential for a bias.

- The above limitation is not applicable for farm receptors for air and soil-related pathways, because it is assumed that farm receptors are mobile and spatially integrate soil and air concentrations across the farm area. That is, an estimate of the mean air and soil concentration across the farm area is determined and used for farm exposure calculations for air and soil-related pathways. A groundwater well is located within the farm area at its centroid, which introduces a source of error for the water ingestion pathway, although there should be no systematic bias in this error. If the located well happens to be in the contaminated groundwater plume when in reality it is not, then groundwater ingestion exposures are artificially high. The converse is also true.

- The number of farms assumed to exist in the study area of interest (which affects the farm-receptor population possibly at risk) is the number of counties that make up the AOI that also contain farms. For example, if the AOI includes two counties, and both counties contain farming land use, then two farms will be assumed to exist within the AOI. This could introduce a risk-conservative bias if the AOI does not in fact contain a farm.

- Estimated exposures due to fish ingestion are also subject to random sampling error for both farm and residential receptors. Residential and farming fishers are assumed to be mobile and catch fish from up to three randomly selected, fishable reaches throughout the AOI. These selected reaches may or may not reflect actual preferred fishing locations in the AOI. There is no reason to expect any systematic bias in estimated fish ingestion exposure, however.

- *Incremental exposure is modeled.* The HWIR99 model generates incremental exposures in accordance with standard practice. No provision is made for considering background exposures for the purpose of generating aggregate or total risk, HQ, or MOE estimates for modeled receptors.

- *Homogeneous concentrations in fruits and vegetables assumed.* The exposure methodology makes no provision for possible chemical concentration gradients within fruits or vegetables that might result in different concentrations in edible portions than when averaged throughout the food item.

- *Food preparation effects.* No diminution of chemical concentration in food items is assumed to occur through food preparation, e.g., washing of fruits and vegetables.

4.3.12 Human Risk Module

The Human Risk Module estimates cancer risks and noncancer hazard quotients to human receptors located throughout the AOI for each year for which exposures are incurred. Exposures are read in from the Human Exposure Module outputs. These risks and HQs are performed for individual exposure pathways (e.g., air inhalation), as well as aggregations of pathways (e.g., across all ingestion pathways). Results are ultimately expressed as cumulative frequency histograms of the numbers of receptors that incur various risk or HQ categories during the most critical year of risk or HQ. These histograms are specific to receptor type, cohort type, exposure pathway (or aggregation), and distance from the WMU.

4.3.12.1 Spatial and Temporal Scales. Spatial and temporal scales are identical to those associated with the outputs from the Human Exposure Module, with one exception. Cancer risk is calculated as a temporally average risk over an assumed exposure duration of 9 years.

4.3.12.2 Key Assumptions. Key assumptions of the module are:

- *Carcinogenic risks are linearly disaggregated from lifetime exposure to exposure duration period.* In accordance with standard practice, incurred risks are assumed to be lifetime exposure risks that are reduced in direct proportion to the fraction of a lifetime actually exposed, i.e., 350 of 365 days per year (15 days away per year) for each year of the exposure duration.

- *Cohort age-specific exposure doses are assumed to be independent among cohort age classes.* As a cohort ages through an exposure duration into subsequent age classes, each subsequent exposure dose is assumed to be independent of the preceding cohort age class dose. For example, an individual in one cohort age class with a dose corresponding to a relatively high body weight would not necessarily maintain that relative body weight, and consequently high dose, as it ages into the next age class.

- *Maximum HQ and MOE estimates are conservatively based on a single year of exposure.* Unlike carcinogenic risk calculations, which use a moving average over multiyear exposure periods, HQ and MOE calculations treat each year independently, i.e., their time series reflects 1-year average values. Thus, a single high year of maximum exposure would not be "diluted" by a multiyear averaging period. This is a conservative approach.

4.3.12.3 Methodologies.

Risk/HQ/MOE Estimates. The Human Risk Module calculates pathway-specific cancer risk, noncancer HQ, and/or noncancer MOE (for breast feeding infants only) for each receptor type and subpopulation (cohort) in each relevant geographic location across the AOI, given the average daily dose (ADD) as output from the Human Exposure Module. These calculations are performed for each year for which there is a nonzero ADD. Cancer risk estimates are calculated

as 9-year average risks, in accordance with the assumed exposure duration. HQ and MOE estimates are 1-year averages.

Frequency Histograms. Given the resulting time series of pathway-specific, receptor type/cohort-specific, and location-specific risks, HQs, or MOEs, a time series of frequency histograms is constructed. The histogram for any given year is constructed as a series of class intervals ("bins") defined by risk or HQ ranges. For example, the first bin for a risk histogram is defined by risk values between 0 and 10^{-8}. For any given year, the histogram contains the pathway-specific risk or HQ distribution of the number of people (corresponding to a given receptor type/cohort) across locations. For example, if there are 150 people cumulatively across locations incurring risks corresponding to bin 3 (10^{-7} to 10^{-6}) due to, say, the air inhalation pathway in a given year, then bin number 3 of that histogram for that year contains the value 150.

Determination of Critical Years. Given the resulting time series of pathway-specific and receptor type/cohort-specific risk and/or HQ histograms, that year is determined during which the **total** risk or HQ to all receptors is a maximum. Total risk is calculated as the number of receptors in a given bin times the average risk for that bin, summed across all bins. Thus, if 10 receptors have an average risk of $5x10^{-6}$ and 100 receptors have an average risk of $5x10^{-5}$, the total risk for that year would be estimated as $(10)(5x10^{-6}) + (100)(5x10^{-5})$ or 0.00505.

4.3.12.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- *Risk/HQ/MOE estimates are aggregated for certain receptor types.* The four receptor types considered by the Human Risk Module (resident, residential gardener, farmer, fisher) are fewer than the number of receptor types considered and output by the Human Exposure Module in order to maintain output storage requirements at reasonable levels. The Human Risk Module internally aggregates dairy farmers and beef farmers into a single farmer (dairy and beef) and aggregates all of the Human Exposure Module's receptor type-specific fishers (e.g., resident fisher) into a single fisher receptor. Thus, some resolution is lost by this aggregation; for example, the risks specific to farmers who drink contaminated milk, but do not consume contaminated beef, would not be available.

- *Synergistic or antagonistic effects among multiple chemicals, or individual chemical speciation, on risk/HQ/MOE are not considered.* The Human Risk Module is executed by the FRAMES system within a system-level chemical loop so that only one chemical is considered at any single execution of the module, and it is implicitly considered to be independent of other chemicals.

- *Cancer slope factors do not vary with cohort age.* Age-specific differences in exposure responses are not considered.

- *The resolution of the risk and HQ histograms is limited by the number of bins.* Six bins were included for risk distributions (0 to 10^{-8}, 10^{-8} to 10^{-7}, 10^{-7} to 10^{-6}, 10^{-6}

to 10^{-5}, 10^{-5} to 10^{-4}, and 10^{-4} to 1.0). For HQ histograms, four bins were included (0 to 0.1, 0.1 to1.0, 1.0 to 10.0, and >10.0.)

4.3.13 Ecological Exposure Module

The Ecological Exposure Module calculates the applied dose (in mg/kg-d) to ecological receptors that are exposed to contaminants via ingestion of contaminated plants, prey, and media (i.e., soil, sediment, and surface water). These dose estimates are then used as inputs to the Ecological Risk Module. The concentration inputs required by the Ecological Exposure Module are provided by the Terrestrial Food Web module, the Aquatic Food Web module, the Surface Water Module , and the Surface Impoundment Module.

4.3.13.1 <u>Spatial and Temporal Scales</u>. The spatial resolution is set by the resolution of the input media and food item concentrations. Soil, plant, and terrestrial prey concentrations are receptor home range-average. Surface drinking water and aquatic food item concentrations are reach-average, where "reach" denotes a stream or river reach, or a pond, lake, or wetlands. Drinking water concentrations from groundwater are point estimates. The temporal resolution is an average daily dose for each year of the total simulated period.

4.3.13.2 <u>Key Assumptions</u>. Key assumptions of the module are:

- *All areas delineated as habitat support wildlife.* It is assumed that habitats delineated at each site are capable of sustaining a variety of wildlife. Because the predator-prey interactions for each habitat are represented by a simple food web, each habitat is assumed to be of sufficient quality to support multiple trophic levels and, at least, one reproducing pair of upper trophic level predators. Hence, exposure estimates reflect essentially free access to any of the food items suggested in the database on ecological exposure factors.

- *There are no other chemical stressors in the study area.* Because this is a site-based (rather than site-specific) assessment, it was assumed that ecological receptors were not subjected to other stressors within the study area. Background concentrations of constituents were not considered in developing exposure estimates, nor were other potential stressors such as habitat fragmentation.

- *No less than 10 percent of the diet is attributed to the study area.* In many instances, the home range for a given receptor exceeds the size of the habitat. In general, it was assumed that the percent of the home range that "fits" into the habitat is a suitable surrogate with which to scale exposures. However, the purpose of this analysis is to determine acceptable waste concentrations assuming that suitable portions of the study area (e.g., forests) would be used as habitat by wildlife. Therefore, it was assumed that no less than 10 percent of the diet should originate from the study area, even if the fraction of the home range inside the habitat fell below 10 percent.

- *Spatial averaging of exposures is defined by habitat and home range.* For this site-based assessment of representative habitats, it was assumed that a reasonable approach to define the spatial extent of exposure for each receptor was to place the home range within the habitat boundaries. However, alternative approaches were considered, including the calculation of exposure point concentrations based on a random walk across various habitats.

4.3.13.3 Methodologies. The Ecological Exposure Module calculates average daily doses, or exposures, for each receptor placed within a terrestrial or freshwater aquatic habitat. Thus, exposure is a function of the receptor's home range (or portion, thereof), the spatial boundaries of the home range, the food items (plants and prey) that are available in a particular home range, the dietary preferences for food items that are available, and the media concentrations in the receptor's home range. In essence, the module attempts to estimate an applied dose for birds, mammals, and selected herpetofauna that reflects the spatial and temporal characteristics of the exposure (i.e., exposure is tracked through time and space). Terrestrial habitats considered by the analysis are grasslands, shrub/scrub, forests, crop fields, and residential areas. Aquatic habitats considered are reach order 3 or higher streams, lakes, ponds, and wetlands.

The conceptual approach in developing the ecological exposure assessment for HWIR99 (including the Ecological Exposure Module and supporting databases) was to reflect the natural variability in exposure by considering major sources of variability. In particular, the approach considers variability through (1) the development of representative habitats; (2) selection of receptors based on ecological region; (3) the recognition of opportunistic feeding and foraging behavior using probabilistic methods; (4) the creation of dietary scheme specific to region, habitat, and receptor; and (5) the application of appropriate graphical tools to capture spatial variability in exposure. The underlying framework for the Ecological Exposure Module is based on a representative habitat scheme to increase the resolution of general terrestrial and freshwater systems.

Depending on the type of habitat and chemical-specific uptake and accumulation, animals may be exposed through the ingestion of plants (both aquatic and terrestrial), soil invertebrates, aquatic invertebrates, fish, terrestrial vertebrates, media, or any combination that is reflected by the dietary preferences of the particular species. For example, an omnivorous animal that inhabits a freshwater stream habitat may ingest fish, small terrestrial vertebrates found in the stream corridor, terrestrial and aquatic plants, surface water, and soil. The dietary preferences are independent of the chemical type and, therefore, contaminant concentrations in some food items may be near zero for chemicals that do not bioaccumulate. The dietary preferences for each receptor are supported by an extensive exposure factors database containing information on, for example, dietary habits, natural history, for over 50 representative species of interest. The module includes an innovative approach to characterizing the diet: a probabilistic algorithm that cycles through the database on minimum and maximum prey preferences to simulate dietary variability.

The major calculation steps performed by the Ecological Exposure Module that are required to calculate an applied dose may be summarized as follows:

1. Select receptor of interest.

2. Get media concentrations from outputs of appropriate other modules.

3. Calculate average media concentrations to which receptor is exposed. (Average soil concentrations within the home range have previously been calculated by the Terrestrial Food Web Module. Average surface drinking water concentrations within the home range are calculated by this module.)

4. Construct diet for receptor of interest (i.e., composition and preferences). (A constrained, random dietary preference fraction sampling algorithm was developed to select dietary fractions at random between the minimum and maximum assuming a uniform distribution. The algorithm maintains overall dietary preferences and allows for the dietary composition to reflect the full range of variability inherent in many wildlife diets. Once the dietary preferences and the concentrations in various food items are determined, the effective concentration in the overall receptor diet is calculated as the sum of plant and prey concentrations in the individual items that it consumes times the fraction of the diet associated with each item.)

5. Get plant and prey concentrations for dietary items from Terrestrial Food Web Module outputs.

6. Sum intake from media and food sources.

7. Calculate potential applied dose by adjusting for body weight.

8. Calculate applied dose by prorating dose by habitat/home range ratio.

The applied daily dose to a receptor in a terrestrial habitat is calculated as follows. Applied daily doses to receptors in aquatic habitats are similarly calculated, with appropriate adjustments made for sediment ingestion in lieu of soil ingestion.

$$Dose^{i}_{rec} = \frac{(CR^{i}_{food} \cdot C^{i}_{food}) + (C^{i}_{soil} \cdot CR^{i}_{food} \cdot Soil^{i}_{frac}) + (Csw^{i}_{ave} \cdot CR^{i}_{water})}{BodyWt^{i}_{Rec}} \cdot HomeRange_{frac} \qquad (4\text{-}10)$$

where

$Dose^{i}_{rec}$ = time-dependent applied dose of constituent to receptor i (mg/kg-d)

CR^{i}_{food} = consumption rate of food for receptor i (kg/d)

C^i_{food} = effective concentration in food consumed by receptor i, weighted by prey preferences (mg/kg)

C^i_{soil} = average concentration in soil to which receptor i is exposed (mg/kg)

$Soil^i_{frac}$ = soil dietary fraction for receptor i

$C^i_{sw_ave}$ = total average concentration in water to which receptor i is exposed (mg/L)

CR^i_{water} = consumption rate of water for receptor i (L/d)

$BodyWt_{Rec}$ = body weight of receptor i (kg)

$HomeRange_{frac}$ = fraction of the receptor's home range within habitat.

4.3.13.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- *Plant categories were defined by analogy*. Vegetation categories relevant to wildlife were extrapolated from the plant categories defined for use in the Farm Food Chain Module. The cross reference for vegetative categories consumed by wildlife is presented in the Terrestrial Food Web Module documentation.

- *Annual average concentrations define exposure.* The exposure profiles generated with the Ecological Exposure Module are based on the average annual concentrations in food items and media. Consequently, concentration spikes due to episodic events (e.g., rain storms) or elevated source releases following waste additions are not evaluated.

- *Exposures are predicted only for adult animals.* Because concentrations are annualized, the module predicts exposures only for adult animals; intrayear contaminant exposures to juveniles, often with very different dietary preferences, are not predicted.

- *Dietary preferences remain constant over the year.* The Ecological Exposure Module constructs the dietary preferences for each receptor based on dietary data covering one or more seasons. Some of the seasonal variability in the diet is captured indirectly by the hierarchical algorithm used to determine the dietary preferences. However, the algorithm is implemented on data across multiple seasons and, therefore, does not necessarily reflect seasonal differences.

- *Exposure estimates reflect a single home range setting.* The Ecological Exposure Module calculates the applied doses to receptors for a single random placement of

four home range sizes. (Each receptor is assigned to one of four discrete home range sizes, depending on the receptor-specific home range. The four home ranges are spatially linked in that the ranges overlap in a manner that reflects the dietary preferences of the predator species.) As a result, the four home ranges in the site layout may not reflect the spatial variability in exposure patterns, particularly for large habitats (i.e., habitats that cover substantially greater areas than most of the home ranges).

4.3.14 Ecological Risk Module

The Ecological Risk (EcoRisk) module calculates the hazard quotients to ecological receptors assigned to various habitats at a site. Hazard quotients (HQs) are defined as: (1) the ratio between an ecological benchmark (EB in units of dose) and the applied dose received during ingestion of contaminated media and food items and (2) the ratio between a chemical stressor concentration limit (CSCL in units of concentration) and the medium of interest (soil, sediment, or surface water). The HQs are assigned to one of five risk bins and presented according to following roll ups for the maximum years of exposure:

- Distance from the source (i.e., 1 km, 2 km, or across the entire site)
- Habitat type (e.g., grassland, pond, permanently flooded forest)
- Habitat group (i.e., terrestrial, aquatic, and wetland)
- Receptor group (e.g., mammals, amphibians, soil community)
- Trophic level (i.e., producers, TL1, TL2, TL3 top predators).

In addition, the maximum HQ is calculated and all of the attributes of the maximum are reported, such as the receptor, the habitat, and distance. The inputs required by the EcoRisk Module are provided by the Ecological Exposure (EcoEx) module, the Terrestrial Food Web module (TerFW), and the Surface Water (SW) module.

4.3.14.1 <u>Spatial and Temporal Scales</u>. For most ecological receptors, the spatial resolution for the EcoRisk module is, to a large degree, a function of the resolution established by the Ecological Exposure module. The applied dose for each ecological receptor is based on the average media and food concentrations for the home range area to which the receptor is assigned. In turn, the home range area is defined in terms of the habitat and predator-prey interactions, that is, the home ranges are constrained by the habitat boundaries[4] and overlap, as appropriate, to represent predator-prey interactions. However, for two ecological receptors - the aquatic and benthic communities - the spatial resolution is at the level of the aquatic habitat rather than at the level of the stream reach. The temporal resolution is based on annual average applied doses (for comparison with EBs) and media concentrations (for comparison with CSCLs).

[4] If the home range area is larger than the area of the habitat, the home range is presumed to extend beyond the 2 km radius that defines the area of interest rather than cross the habitat boundaries within the area of interest. If the home range is smaller than the habitat, the entire home range is presumed to fall within the habitat boundaries.

4.3.14.2 <u>Key Assumptions</u>. Many of the key assumptions of the EcoRisk module were described in Section 4.3.13 on the EcoEx module. For example, the assumption that all areas delineated as habitat support wildlife also applies to the EcoRisk module in that HQs calculated within each habitat are presumed to reflect potential risks to ecological receptors. However, there are several assumptions that are unique to the EcoRisk module:

- *It is implicitly assumed that one and only one population of each wildlife species is carried by a given habitat.* Although there may be a number of receptors assigned to a habitat, multiple populations of shrews or robins are not evaluated. Each receptor population has the same spatial characteristics, as defined by the home range. Hence, there is one HQ calculated for each receptor in each habitat.

- *Receptors are not assumed to be subjected to any other stressors at the same time that they are exposed to a single constituent.* The implicit assumptions are that: (1) there are no background concentrations and (2) the receptor physiology is not compromised in any way.

- *Maximum HQ estimates are based on a single year of exposure.* This assumption is also discussed under the section on scenario uncertainty. However, it is important to point out the ecological HQ estimates are based on the smallest increment of time that the 3MRA system is designed to work with, annual averages. This time step represents much longer than lifetime exposures for some receptors, and substantially less than lifetime for other receptors.

4.3.14.3 <u>Methodologies</u>. The EcoRisk module calculates HQs for a suite of ecological receptors assigned by habitat and ecological region. These receptors include eight major groups: (1) mammals, (2) birds, (3) herpetofauna, (4) terrestrial plants, (5) soil fauna, (6) aquatic plants and algae, (7) aquatic organisms, and (8) benthic organisms. In calculating receptor-specific HQs, the EcoRisk module does the necessary accounting to develop distributions based on the specific receptor and habitat groupings of interest, as described above. The EcoRisk module reads information relating to habitats and receptors within the area of interest around a waste management unit, reads in information about the chemical concentrations each receptor is exposed to, calculates hazard quotients (HQs) based on the EB or CSCL and the chemical exposure information, and provides summaries of ecological risk information for the simulation for critical years where maximum HQs are experienced.

For a given year, the set of HQ data is stored as a series of distributions. Each distribution contains data for a specific receptor or habitat group of interest. The distributions are composed of a series of bins for different ranges of HQ values. Bins are populated based on the number of receptors with HQ values in the range defined for the given bin.

Each site is constructed as a set of habitats, each located within one or more ecorings at the site, and a set of receptors inhabiting ranges within each of those habitats. Habitats have a variety of characteristics, including a unique index identifier, a habitat type and group, a number of Reaches, a number of ranges containing receptors, and the receptors associated with each

range. Reaches, habitats, and ranges also have chemical concentrations associated with them. Each receptor has an index, type, name, group, trophic level.

Outputs are generated for three areas of the site relative to the distance from the edge of the waste management unit. These distances are termed EcoRings and depict the following: (1) habitats that fall within 1 km of the WMU, (2) habitats that fall between 1 and 2 km from the WMU, and (3) habitats within 2 km of the WMU (i.e. across the entire site). It is important to note that the HQ results for habitats that intersect both EcoRings are attributed to the risk results for both of those distances. In other words, the habitat risks are not apportioned by distance, they are reported as though they are positioned entirely within each distance ring. Because the fundamental unit of this analysis is the representative habitat (not distance to the waste management unit), it was considered inappropriate to truncate risks by distance.

The primary functions of the Ecorisk module are, for each distance ring, to:

- Read in all data required to calculate HQs for all receptors (e.g., EBs, CSCLs, site layout characteristics such as water hardness)
- Calculate HQs for all receptors within the area of interest for each year of the simulation
- Calculate cumulative distribution functions for each year of the simulation (this is performed in much the same manner as with the Human Risk module)
- Identify and output the cumulative distribution function for various receptor and habitat groups for the year in which the maximum total HQ was experienced
- Identify and output information about the receptor experiencing the maximum HQ across all years of the simulation and the year in which the maximum occurred.

4.3.14.4 <u>Limitations/Uncertainties</u>. The following limitations or uncertainties are noted:

- *The HQ estimates are generated based on one, and only one, home range area.* For the purposes of creating the site layout file, four home range areas are placed in each habitat. Once these areas are delineated and appropriate receptors are assigned, the spatial characteristics of the risk for each home range is established. Variability associated with exposures in different areas of the habitat is not reflected in this scheme. This limitation may result in significant differences for receptors with small home ranges, and can influence the risk estimates for predators with large home ranges (i.e., home range ≈ habitat) since tissue concentrations in prey items are constrained by the same spatial characteristics. As a result, the HQs are associated with some uncertainty with regard to the spatial character of the exposure in any given area of the habitat.

- *The effects of multiple stressors (chemical and non-chemical) are not considered in developing estimates of potential ecological risk.* This is a source of considerable uncertainty in the HQ estimates. The EcoRisk module is executed within the FRAMES system within a system-level chemical loop such that only a single chemical can be evaluated per iteration of the model. Naturally, there are data limitations regarding the quality of ecotoxicological data needed to develop multi-chemical benchmarks. Nevertheless, the inability to consider other sources of risk is a limitation of the risk module.

- *The HQ estimates for the aquatic and benthic communities, respectively, are resolved at the habitat, rather than reach level.* There is some uncertainty associated with calculating risks to aquatic life across an entire aquatic habitat (as defined within the study area). It has been shown that many species of fish utilize only segments of stream habitats and, therefore, HQs at the reach level may be more appropriate. Conversely, establishing artificial boundaries between stream reaches is contrary to the goals of the assessment strategy, namely, to evaluate ecological risks using the habitat as the fundamental unit.

- *The HQ estimates reflect different endpoints at varying levels of effect.* The HQ methodology - the ratio of an exposure to a benchmark - is applied uniformly across all ecological receptors. However, the data supporting the HQ calculation varies in that they include endpoints from lethality to reproductive fitness and address both population- and community-level effects by inference. To some degree, the HQ estimates for different receptor groups represent different risk metrics. The interpretation of these HQ estimates is, therefore, accompanied by some uncertainty in understanding the potential ecological significance of the measures of effect.

4.4 System/Technology Uncertainty

The software system has been designed and implemented with a strong focus on quality assurance and quality control (QA/QC). The software system is made up of three primary components: the site-based databases, the system software, and the modules for performing the required exposure and risk assessments. The system software organizes the waste site information and prepares individual datasets that are used to simulate contaminant release, multimedia fate and transport, and human and ecological exposure and risk. The system software also manages the execution of the numerous modules that simulate specific steps in the risk assessment process (e.g., source release, surface water fate and transport, ecological risk). The software development steps that we followed (and address QA/QC) include:

1. Software system design based on detailed and peer reviewed HWIR assessment methodology.

2. Software system designed using object-oriented design principles and using existing EPA models (ISCST, EXAMS, EPACMTP).

3. Detailed system specifications are documented and reviewed before software coding is initiated.

4. Data dictionaries are developed to fully define (and constrain) each data item that is shared within the system.

5. Database development is designed and executed in close coordination with software system development.

6. Individual developers design and conduct first-level testing of all code before assimilation into the larger software system.

7. System software and component modules are assimilated into a unified system with extensive testing of information flow and related data integrity.

8. Execution of an initial "technical" verification (i.e., tracking the actual numbers through the system) of the software system using a single combination of waste site, chemical, and waste unit type.

9. Execution of limited "production" runs using a subset of the total number of waste site/chemical/waste unit type combinations. Production runs are oriented toward producing exemption levels.

10. Execution of initial full-scale production runs (i.e., using all site/chemical/waste unit type) combinations.

11. *Execution and documentation of final tests for individual components of the software system. (This step has been delayed due to the extended nature of the development process and overall project schedule.)*

12. Execution of second full-scale production runs (i.e., the runs that produced the exemptions levels included in this proposal).

5.0 References

Abt Associates Inc. 1995. *Review and Analysis of Toxicity Data to Support the Development of Uncertainty Factors for Use in Estimating Risks of Contaminant Stressors to Wildlife*. Review Draft. Prepared for U.S. Environmental Protection Agency, Office of Water. June.

Ahlborg, U.G., G.C. Becking, L.S. Birnbaum, et al. 1994. Toxic equivalency factors for dioxin-like PCBs. Report on a WHO-ECEH and IPCS consultation, December 1993. *Chemosphere* 28:1049-1067.

Anderson, J.R., E.E. Hardy, J.T. Roach, and R.E. Witmer. 1976. *A Land Use and Land Cover Classification System for Use with Remote Sensor Data*. U.S. Geological Survey Professional Paper 964. Reston, VA. Available online at ftp://www-nmd.usgs.gov/pub/ti/LULC/lulcpp964.

Atkinson, D., and R.F. Lee. 1992. *Procedures for Substituting Values for Missing NWS Meteorological Data for Use in Regulatory Air Quality Models.* U.S. Environmental Protection Agency. Research Triangle Park, North Carolina.

ATSDR (Agency for Toxic Substances and Disease Registry). various years. Toxicological Profiles. U.S. Department of Health and Human Services, Public Health Service, Centers for Disease Control and Prevention, Atlanta, GA.

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides Through Agriculture.* Oak Ridge National Laboratory, Oak Ridge Tennessee.

Bechtel Jacobs Company LLC. 1998. *Biota Sediment Accumulation Factors for Invertebrates: Review and Recommendations for the Oak Ridge Reservation*. BJC/OR-112. Prepared for the U.S. Department of Energy, Office of Environmental Management.

Bergman, H.L. and J. Dorward-King (Eds). 1997. *SETAC Pellston Workshop on Reassessment of Metals Criteria for Aquatic Life Protection*; 1996 Feb 10-14. SETAC Press, Pensacola, FL.

Bertelsen, S.L., A.D. Hoffman, C.A. Gallinat, C.M. Elonen, and J.W. Nichols. 1998. Evaluation of Log Kow and Tissue Lipid Content as Predictors of Chemical Partitioning to Fish Tissues. *Environmental Toxicology and Chemistry*, Vol. 17, No. 8, pp. 1447-1455.

Bickel, P.J., and K.A. Doksum. 1977. *Mathematical Statistics*. San Francisco, CA: Holden-Bay.

Box, G.E.P. 1976. Science and statistics. *J. Amer. Stat. Assoc*. 71(356):791-799.

Burmaster, D.E, and K.M. Thompson. 1998. Fitting second-order parametric distributions to data using maximum likelihood estimation. *Human and Ecological Risk Assessment* 4(2):319-339.

CalEPA (California Environmental Protection Agency). 1994. *Benzo(a)pyrene as a Toxic Air Contaminant. Part B Health Assessment.* Office of Environmental Health Hazard Assessment, California Air Resources Board, Berkeley, CA.

CalEPA (California Environmental Protection Agency). 1995. List of California cancer potency factors (dated 4/4/95). Available online at http://www.calepa.cahwnet.gov/oehha/docs/covrltrb.htm.

CalEPA (California Environmental Protection Agency). 1997. *Technical Support Document for the Determination of Noncancer Chronic Reference Exposure Levels.* Draft for Public Review. Office of Environmental Health Hazard Assessment, Air Toxicology and Epidemiology Section, Berkeley, CA. Available online at http://www.calepa.cahwnet.gov/oehha/docs/rels/getrels.htm.

CalEPA (California Environmental Protection Agency). 1999a. *Air Toxics Hot Spots Program Risk Assessment Guidelines: Part II. Technical Support Document for Describing Available Cancer Potency Factors.* Office of Environmental Health Hazard Assessment, Berkeley, CA. Available online at http://www.oehha.org/scientific/hsca2.htm.

CalEPA (California Environmental Protection Agency). 1999b. *Air Toxics Hot Spots Program Risk Assessment Guidelines: Part III. Technical Support Document for the Determination of Noncancer Chronic Reference Exposure Levels.* SRP Draft. Office of Environmental Health Hazard Assessment, Berkeley, CA. Available online at http://www.oehha.org/hotspots/RAGSII.html.

Canton, J.H. and W. Slooff. 1982. Toxicity and accumulation studies of cadmium (Cd^{2+}) with freshwater organisms of different trophic levels. *Ecotoxicology and Environmental Safety*, pp. 113-128.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. Characterizing the uncertainty of pesticide leaching in agricultural soils. *Journal of Contaminant Hydrology* 2(2):111-124.

Carsel, R.F., and R.S. Parrish. 1988. Developing joint probability distributions of soil water retention characteristics. *Water Resources Research* 24(5):755-769.

Clapp, R.B., and G.M. Hornberger. 1978. Empirical equations for some hydraulic properties. *Water Resources Research* 14:601-604.

Clickner, R P., and J. Craig. 1987. Using business establishment size in an environmental survey. *Proceedings of the American Statistical Association: Survey Research Section.* American Statistical Association, 1429 Duke St., Alexandria, VA 22314-3402. 5pp.

Clickner, R.P. 1988. Sampling weights for the industrial Subtitle D screening survey. July 28 , 1988, Westat memorandum to Zubair Saleem, U.S. EPA, Office of Solid Waste.

Dunne, T. and L.B. Leopold, *Water in Environmental Planning*, W.H. Freeman and Co., New York, 1978.

Eijsackers, H. 1994. Ecotoxicology of Soil Organisms: Seeking the Way in a Pitch Dark Labyrinth. *Ecotoxicology of Soil Organisms*. CRC Press, Inc., p. 3.

Gobas, F.A.P.C. 1993. *A model for predicting the bioaccumulation of hydrophobic organic chemicals in aquatic food-webs: Application to Lake Ontario.* Ecological Modeling, 69:1-17.

Hargreaves, G.H., "Moisture Availability and Crop Production, " Trans. Am. Soc. Agric. Eng., vol. 18, no. 5, pp 980-984, 1975.

Hattis, D., and D. Burmaster. 1994. Assessment of variability and uncertainty distributions for practical risk analyses. *Risk Analysis* 14(5):713-730.

Horst, T.W. 1983. A correction to the Gaussian source-depletion model. In *Precipitation Scavenging, Dry Deposition and Resuspension*, H.R. Pruppacher, R.G. Semonin, W.G.N. Slinn, eds., Elsevier, NY.

Huber, P.J. 1967. The behavior of maximum likelihood estimates under nonstandard conditions. In: *Proceedings of the Fifth Berkeley Symposium in Mathematical Statistics*, University of California Press.

Hudson, R.J. M., S.A. Gherini, C.J. Watras, and D.B. Porcella. 1994. Modeling the Biogeochemical Cycle of Mercury in Lakes: The Mercury Cycling Model (MCM) and its Application to the MTL Study Lakes. In: *Mercury Pollution: Integration and Synthesis.* C.J. Watras and J.W. Huckabee (Eds), Lewis Publishers, Boca Raton, FL, pp. 473-523.

McKone, T.E. Human Exposure to Volatile Organic Compounds in Household Tap Water; the Indoor Inhalation Pathway. *Environ. Sci. Technol*. 21(12):1194 - 1201, 1987.

Miller, D.A., and R.A. White. 1998. A Conterminous United States Multi-Layer Soil Characteristics Data Set for Regional Climate and Hydrology Modeling. Earth Interactions, 2. Available online: http://EarthInteractions.org.

Mineau, P., B.T. Collins, and A. Baril. 1996. On the use of scaling factors to improve interspecies extrapolation of acute toxicity in birds. *Regulatory Toxicology and Pharmacology*. 24:24-29.

Myers, L., J. Lashley, and R. Whitmore. 1998. *Development of Statistical Distribution for Exposure Factors*. Research Triangle Institute, Research Triangle Park, NC, for U.S. EPA's Office of Research and Development.

NCDC (National Climatic Data Center), ERL (Environmental Research Laboratories), and NWS (National Weather Service). 1995. Cooperative Summary of the Day TD3200–Period of record through 1993 CD-ROM. National Climatic Data Center. Asheville, NC.

Neuhauser, E.F., P.R. Durkin, M.R. Malecki, and M. Anatra. 1986. Comparative toxicity of ten organic chemicals to four earthworm species. *Comp. Biochem. Physiol*. 83C(1):197-200.

Nisbet, I.C.T., and P.K. LaGoy. 1992. Toxic equivalency factors (TEFs) for polycyclic aromatic hydrocarbons. *Regulatory Toxicol. Pharmacol*. 16:290-300.

Sample, B.E., M.S. Alpin, R.A. Efroymson, G.W. Suter, and C.J.E. Welsh. 1997. *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants*. Prepared for the U.S. Department of Energy, Office of Environmental Policy and Assistance, Air, Water, and Radiation Division. Prepared by Oak Ridge National Laboratory, Oak Ridge, TN.

Sample, B.E., J.J. Beauchamp, R.E., Efroymson, G.W. Suter II, and T.L. Ashwood. 1998. *Development and Validation of Bioaccumulation Models for Earthworms.* ES/ER/TM-220. Oak Ridge National Laboratory.

Schwarz, G.E., and R.B. Alexander. 1995. *State Soil Geographic (STATSGO) Data Base for the Conterminous United States. Edition: 1.1.* Open-File Report 95-449. U.S. Geological Survey, Reston, VA. http://water.usgs.gov/lookup/getspatial?ussoils.

Science Advisory Board, An SAB Report: Review of EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP). Prepared by the OSWER Exposure Model Subcommittee of the Environmental Engineering Committee. EPA-SAB-EEC-95, 1995.

Science Advisory Board, An SAB Report: Review of a Methodology for Establishing Human Health and Ecological Based Exit Criteria for the Hazardous Waste Identification Rule (HWIR), Prepared by the HWIR Subcommittee of the Executive Committee. EPA-SAB-EC-96-002, May 1996.

Seegert, G., J.A. Fava, and P.M. Crumbie. 1985. How representative are the data sets used to derive national water quality criteria? Pp.527-537. In: Cardwell, R.D., R. Purdy, and R.C. Bahner. *Aquatic Toxicology and Hazard Assessment: Seventh Symposium*, American Society for Testing and Material, Philadelphia, PA.

Small, Mitchell J., Yoram Cohen and Paul F. Deisler, Jr. 1998. *Review of ORD/OSW Integrated Research and Development Plan for the Hazardous Waste Identification Rule (HWIR).*

Smith, E.P. and J. Cairns, Jr. 1993. Extrapolation methods for setting ecological standards for water quality: Statistical and ecological concerns. *Ecotoxicology* 2:203-219.

Smith, S.L., D.D. MacDonald, K.A. Keenleyside, C.G. Ingersoll, and L.J. Field. 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. *J. Great Lakes Res*. 22(3):624-638.

Spencer, M., and S.B. Beaulieu. 1997. Bioaccumulation Factors for Polycyclic Aromatic Hydrodcarbons in Fish: Methods for Dealing with Uncertainty. Independent Consultant, Sheffield, England.

Stephan, C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman, and W.A. Brungs. 1985. *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses.* PB85-227049. National Technical Information Service, Springfield, VA.

Stephan, C.E. 1993. *Derivation of Proposed Human Health and Wildlife Bioaccumulation Factors for the Great Lakes Initiative*. Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Duluth, MN.

Strahler, A.N. 1957. Quantitative analysis of watershed geomorphology. *Transactions of the American Geophysical Union* 8(6):913-920.

Suter II, G.W., D.S. Vaughan, and R.H. Gardner. 1983. Risk assessment by analysis of extrapolation error, a demonstration for effects of pollutants on fish. *Environ. Toxicol. Chem.* 2:369-378.

Suter II, G.W. and A.E. Rosen. 1987. Endpoints for responses of fish to chronic toxic exposures. *Environ. Toxicol. Chem*. 6:793-809.

U.S. Bureau of the Census. *1987 and 1992 Census of Agriculture. Geographic Area Series State and County Data. 1987 and 1992*. (http://www.census.gov/econ/www/ag0100.html)

U.S. Bureau of the Census. 1990. *TIGER: The Coast-to-Coast Digital Map*. 1990 (GIS coverage - block and block groups, roads, etc.) (http:/www.census.gov/geo/www.tiger/)

U.S. Bureau of the Census. 1992. *Census of Population and Housing*. Summary Tape File on CD-ROM Technical Documentation. STF 1-B and STF 3A. Washington, DC. (http:/www.census.gov/mp/www/rom/msrom6ac.html and http:/www.census.gov/mp/www/rom/msrom6ae.html)

USDA (U.S. Department of Agriculture). 1994. *State Soil Geographic (STATSGO) Data Base*. Natural Resources Conservation Service, Fort Worth, TX. Available online: http://www.ftw.nrcs.usda.gov/stat_data.html.

U.S. DOC (Department of Commerce) and U.S. DOE (Department Of Energy). 1993. *Solar and Meteorological Surface Observation Network 1961-1990*. CD-ROM, Version 1.0. National Climatic Data Center, Asheville, NC.

U.S. DOI (Department of the Interior). 1996. *National Survey of Fishing, Hunting, and Wildlife-Associated Recreation.* Fish and Wildlife Service and U.S. Department of Commerce, Bureau of the Census.

U.S. EPA (U.S. Environmental Protection Agency). 1982. *Exposure Analysis System (EXAMS): User's Manual and System Documentation*. EPA-600/3-82-023. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1987. *1986 National Survey of Hazardous Waste Treatment, Storage, Disposal, and Recycling Facilities (TSDR) Database*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1990. *Environmental and Information Systems Compendium*. EPA 500-9-90-002. Office of Water, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1991a. *MINTEQA2/PRODEFA2, A Geochemical Assessment Model for Environmental Systems: Version 3.0, User's Manual*. EPA/600/3-91/021. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1991b. *Hazardous Waste TSDF - Background Information for Proposed RCRA Air Emission Standards*.

U.S. EPA (Environmental Protection Agency). 1993a. *Provisional Guidance for Quantitative Risk Assessment of Polycyclic Aromatic Hydrocarbons*. EPA/600/R-93-089. Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 1993b. *Wildlife Exposure Factors Handbook*. EPA/60/P-92-003C. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1994a. *Quadrangles of Landuse/Landcover GIRAS Spatial Data in the United States (1:250,000 scale)*. Office of Information Resources Management, Washington, DC. (http://www.epa.gov/ngispgm3/nsdi/projects/giras.htm).

U.S. EPA (Environmental Protection Agency). 1994b. *The U.S. EPA Reach File Version 3.0 Alpha Release (RF3-Alpha) Technical Reference, First Edition*. Office of Wetlands,

Oceans, and Watersheds, Office of Water, Washington, DC. Available online at http://www.epa.gov/owowwtr1/NPS/rf/techref.html

U.S. EPA (Environmental Protection Agency). 1994c. *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry*. EPA/600/8-90/066F. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995a. *Policy for Risk Characterization at the U.S. Environmental Protection Agency*. Science Policy Council, Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995b. *Guidance for Risk Characterization*. Science Policy Council, Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995c. *1990 Census Block Boundaries in the Conterminous U.S.* Office of Information Resources Management, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995d. *1990 Census Block Group Boundaries in the Conterminous U.S.* Office of Information Resources Management, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995e. *Great Lakes Water Quality Initiative Criteria Documents for the Protection of Aquatic Life in Ambient Water*. EPA-820-B-95-004. Office of Water, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995f. *Trophic Level and Exposure Analyses for Selected Piscivorous Birds and Mammals*. Volumes I, II, and III. Office of Science and Technology and Office of Water, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996a. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP): Background Document*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996b. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP): User's Guide*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996c. *Background Document for EPACMTP, Metals Transport in the Subsurface*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996d. *Proposed Guidelines for Carcinogenic Risk Assessment.* EPA/600/P-92-003C. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996e. *1995 Update: Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water*. EPA-820-B-96-001. Office of Water, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997a. *Exposure Analysis System (EXAMS II): User's Guide for Version 2.97.5*. EPA-600/R-97/047. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997b. *System Design Development Guidance*. Directive No. 2182. Office of Solid Waste, Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997c. *Test and Verification of EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP)*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997d. *Exposure Factors Handbook (EFH)*. EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997e. *Health Effects Assessment Summary Tables (HEAST), FY 1997 Update*. EPA-540-R-97-036. Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997f. *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions*. EPA/600/6-90/003. National Center for Environmental Assessment, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 1997g. *The Parameter Guidance Document. A Companion Document to the Methodology for Assessing Health Risks Associated with Multiple Pathways Exposure to Combustor Emissions*. National Center for Environmental Assessment, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 1997h. *Exposure Factors Handbook, Volume II, Food Ingestion Factors*. EPA/600/P-95/002Fb. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997i. *Exposure Factors Handbook, Volume III, Activity Factors*. EPA/600/P-95/002Fc. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997j. *Mercury Study Report to Congress* (Volumes I - VIII). EPA 452/R-97/003. Office of Air Quality Planning and Standards and Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1998a. *Risk Assessment Forum. Guidelines for Ecological Risk Assessment - Final*. EPA/630/R-95/002F.

U.S. EPA (Environmental Protection Agency). 1998b. *Anaerobic Biodegradation Rates of Organic Chemicals in Groundwater: A Summary of Field and Laboratory Studies*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1998c. *Testing of the Sampled Chronological Input Model (SCIM) option in the enhanced ISCST3 Model for Use in the Hazardous Waste Identification Rule (HWIR99).*

U.S. EPA (Environmental Protection Agency). 1998d. *MINTEQA2/PRODEFA2, A Geochemical Assessment Model for Environmental Systems: User Manual Supplement for Version 4.0.*

U.S. EPA (Environmental Protection Agency). 1998e. *Diffuse-Layer Sorption Reactions for use in MINTEQA2 for HWIR Metals and Metalloids.*

U.S. EPA (Environmental Protection Agency). 1998f. *ORD/OSW Integrated Research and Development Plan for the Hazardous Waste Identification Rule (HWIR).* Office of Research and Development/Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1998g. *The U.S. EPA TEF Values.* Office of Research and Development, National Center for Environmental Assessment. Available online at http://www.epa.gov/nceawww1/dchem.htm.

U.S. EPA (Environmental Protection Agency). 1999a. *Correlation between Liquid, Sludge, and Solid Waste Forms and Surface Impoundment, Land Application Unit, and Landfill Disposal Options*. Office of Solid Waste, Washington, DC. February.

U.S. EPA (Environmental Protection Agency). 1999b. *A Framework for Finite-Source Multimedia, Multipathway and Multireceptor Risk Assessment: 3MRA.* Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999c. *FRAMES-HWIR Technology Software System for 1999: System Overview*. Office of Research and Development, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999d. *Data Collection for the Hazardous Waste Identification Rule. Section 1. Introduction and Overview Section 2: Spatial Layout.* Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999e. *Data Collection for the Hazardous Waste Identification Rule. Section 3: Waste Management Unit Data*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999f. Data Collection for the Hazardous Waste Identification Rule. Section 4: Meteorological Data. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999g. *Data Collection for the Hazardous Waste Identification Rule. Section 5: Watershed and Waterbody Layout*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999h. *Data Collection for the Hazardous Waste Identification Rule. Section 6: Surface Water Data*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999i. *Data Collection for the Hazardous Waste Identification Rule. Section 7: Soil Data*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999j. *Data Collection for the Hazardous Waste Identification Rule. Section 8: Human Exposure Factors*. Office of Solid Waste, Washington, DC. June.

U.S. EPA (Environmental Protection Agency). 1999k. *Data Collection for the Hazardous Waste Identification Rule. Section 9: Human Receptor Data*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999l. *Data Collection for the Hazardous Waste Identification Rule. Section 10: Farm Food Chain and Terrestrial Foodweb Data*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999m. *Data Collection for the Hazardous Waste Identification Rule. Section 11.0 Aquatic Food Web Data.* Draft Report. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999n. *Data Collection for the Hazardous Waste Identification Rule. Section 12: Ecological Exposure Factors*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999o. *Data Collection for the Hazardous Waste Identification Rule. Section 13: Ecological Receptors and Habitats*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999p. *Data Collection for the Hazardous Waste Identification Rule. Section 14: Ecological Benchmarks*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999q. *Data Collection for the Hazardous Waste Identification Rule. Section 15: Human Health Benchmarks*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999r. *Data Collection for the Hazardous Waste Identification Rule. Section 16: Miscellaneous Data*. Office of Solid Waste, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1999s. *Site Selection Methodology for HWIR99 Sampling*. 29 Office of Research and Development, Washington, DC. June.

U.S. EPA (Environmental Protection Agency). 1999t. *Source Modules for Non-Wastewater Waste Management Units (Land Application Units, Waste Piles, and Landfills): Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999u. *Source Modules for Tanks and Surface Impoundments: Background and Implementation for the for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999v. *Documentation for the Air Module for the FRAMES-HWIR System*. Office of Solid Waste, Washington, DC. June.

U.S. EPA (Environmental Protection Agency). 1999w. *Air Module Pre- and Post-Processor: Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999x. *User's Guide for the Industrial Source Complex (ISC3) Dispersion Models for Use in the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) for HWIR99: Description of Model Algorithms*. Office of Solid Waste, Washington, DC. June.

U.S. EPA (Environmental Protection Agency). 1999y. 1999. *Watershed Module: Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999z. *Surface-Water Module for the Hazardous Waste Identification Rule (HWIR99)*. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1999aa. *Vadose and Saturated Zone Modules Extracted from EPACMTP for HWIR99*. Draft. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ab. *Pseudo-Three Dimensional Aquifer Module for HWIR99: Module Verification Document*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ac. *Vadose Zone Module for HWIR99: Module Verification Document*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ad. *Landfill Module for HWIR99: Module Verification Document*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ae. *Surface Impoundment Module for HWIR99: Module Verification Document*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999af. *A Study to Assess the Impacts of Fractured Media in Monte-Carlo Simulations*. With appendices. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ag. *Incorporation of Heterogeneity into Monte-Carlo Fate and Transport Simulations*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ah. *Changes in the MINTEQA2 Modeling Procedure for Estimating Metal Partitioning Coefficients in Groundwater for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ai. 1999. *Chemical Database for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999aj. *Background Document for the Human Exposure Module for HWIR99 Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) Model*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ak. *Background Document for the Human Risk Module for HWIR99 Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) Model*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999al. *Farm Food Chain Module: Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) Model for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999am. *Aquatic Food Web Module: Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) Model for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999an. *Ecological Exposure Module: Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) Model for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ao. *Ecological Risk Module: Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ap. 1999. *Terrestrial Food Chain Module: Background and Implementation for the Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA) for HWIR99*. Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999aq. *Partitioning Coefficients for Metals in Surface Water, Soil, and Waste for HWIR99*. Office of Research and Development, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999ar. *Analysis of NAPL Formation Potential and Cosolvency Effect*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1999as. *Risk Characterization Report for the HWIR99 Multimedia, Multipathway and Multireceptor Risk Assessment (3MRA).* Office of Solid Waste, Washington, DC. July.

U.S. EPA (Environmental Protection Agency). 1999at. *Integrated Risk Information System (IRIS).* National Center for Environmental Assessment, Office of Research and Development, Washington, DC. Available online at http://www.epa.gov/iris/subst/index.html.

U.S. EPA (Environmental Protection Agency). 1999au. EXAMSID (to be completed)

U.S. EPA (Environmental Protection Agency). various years. Risk assessment issue papers. Superfund Technical Support Center, National Center for Environmental Assessment (NCEA), Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). In press. *Methodology for Assessing Health Risks Associated With Multiple Exposure Pathways to Combustor Emissions.* EPA, National Center for Environmental Assessment, NCEA-0238.

U.S. FWS (Fish and Wildlife Service). 1991. *The National Survey of Fishing, Hunting, and Wildlife-Associated Recreation.* U.S. Department of the Interior, Washington, DC.

USGS (U.S. Geological Survey). 1990a. *Digital Elevation Models, National Mapping Program Technical Instructions, Data Users Guide 5*. 2nd Printing (Revised). Reston, VA. Available online at: http://edcwww.cr.usgs.gov/glis/hyper/guide/1_dgr_dem.

USGS (U.S. Geological Survey). 1990b. *USGeoData 1:250,000 and 1:100,000 Scale Land Use and Land Cover and Associated Maps Digital Data*. U.S. Geological Survey Reston. VA. Available online at ftp://www-nmd.usgs.gov/pub/ti/LULC/lulcguide.

U.S. FWS (Fish and Wildlife Service). 1995. *National Wetlands Inventory (NWI) Metadata.* U.S. Fish & Wildlife Service. St. Petersburg, FL. Available online at ftp://www.nwi.fws.gov/metadata/nwi_meta.txt.

VanStraalen, N.M. and C.A.J. Denneman. 1989. Ecotoxicological evaluation of soil quality criteria. *Ecotoxicology Environmental Safety* 18:241-251.

Venkatram, A. 1988. *A Simple Model for Dry Deposition and Particle Settling*. Subcontractor Progress Report 2 (including addendum). EPA Contract No. 68D70002, Work Assignment No. 1-001.

Westat. 1987. *Screening Survey of Industrial Subtitle D Establishments. [Draft Final Report]*. Westat, Inc. EPA Contract No. 69-01-7359. Prepared for U.S. EPA, Office of Solid Waste, Washington, DC.

White, H. 1982. Maximum likelihood estimation of misspecified models. *Econometrica* 50(1):1-25.

Wischmeier, W. H., and D. D. Smith. 1978. Predicting rainfall erosion losses. A guide to consevation planning. In: *Agricultural Handbook*. 537 Edition. U.S. Department of Agriculture, Washington, DC.

Appendix A

Table of Results

Landfills - in mg/kg HWIR Results - All Cohorts by Scenarios 2000 meters

Landfills - 2000m - by Cohorts		Scenario 1						Scenario 2					
		10-6 99% Pop 95% Prob HH (0.1) Eco (1)						10-6 99% Pop 90% Prob HH (1) Eco (1)					
		Infants		1-12 yrs old		13 yrs old & older		Infants		1-12 yrs old		13 yrs old & older	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	NA	0E+00	0.2	9E-07	0.06	NA	NA	0E+00	3	9E-07	0.7
		Scenario 3						Scenario 4					
		10-5 99% Pop 90% Prob HH (1) Eco (1)						10-5 95% Pop 90% Prob HH (1) Eco (10)					
		Infants		1-12 yrs old		13 yrs old & older		Infants		1-12 yrs old		13 yrs old & older	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	NA	0E+00	3	9E-06	0.7	NA	NA	NA	10	NA	10

note 1: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.

note 2: The values in the highlighted cells are the same as the highest waste concentration evaluated.

note 3: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).

NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Exposure Pathways 2000 meters

Landfills - 2000m - by Exposure		Scenario 1													
		10-6		99% Pop		95% Prob		HH (0.1)		Eco (1)					
		Air Inhalation		Soil Ingestion		Water Ingestion		Crop Ingestion		Beef Ingestion		Milk Ingestion		Fish Ingestion	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	; 0E+00	; 0E+00	9E-10	0E+00	9E-07	0.09	; 0E+00	2	7E-06	0.4	7E-06	0.4	9E-10	; 0E+00

		Scenario 1											
		10-6		99% Pop		95% Prob		HH (0.1)		Eco (1)			
		Shower Inhalation		Breast Milk		All Inhalation		All Ingestion		All Ingest & Inhal		Groundwater Total	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	1E-07	; 0E+00	NA	NA	1E-07	; 0E+00	1E-06	0.1	NA	NA	1E-06	0.1

note 1: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 2: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 3: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Exposure Pathways 2000 meters

Landfills - 2000m - by Exposure		Scenario 2													
		10-6		99% Pop		90% Prob		HH (1)		Eco (1)					
		Air Inhalation		Soil Ingestion		Water Ingestion		Crop Ingestion		Beef Ingestion		Milk Ingestion		Fish Ingestion	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	0E+00	0E+00	1E-09	0E+00	9E-07	0.8	2E-05	6	6E-07	5	6E-06	5	1E-09	0E+00

		Scenario 2											
		10-6		99% Pop		90% Prob		HH (1)		Eco (1)			
		Shower Inhalation		Breast Milk		All Inhalation		All Ingestion		All Ingest & Inhal		Groundwater Total	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	9E-09	0.6	NA	NA	9E-09	0.6	1E-06	1	NA	NA	1E-06	1

note 1: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 2: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 3: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Exposure Pathways 2000 meters

Landfills - 2000m - by Exposure		Scenario 3													
		10-5		99% Pop		90% Prob		HH (1)		Eco (1)					
		Air Inhalation		Soil Ingestion		Water Ingestion		Crop Ingestion		Beef Ingestion		Milk Ingestion		Fish Ingestion	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	0E+00	0E+00	2E-09	0E+00	7E-06	0.8	2E-05	6	7E-06	5	2E-05	5	2E-09	0E+00

		Scenario 3											
		10-5		99% Pop		90% Prob		HH (1)		Eco (1)			
		Shower Inhalation		Breast Milk		All Inhalation		All Ingestion		All Ingest & Inhal		Groundwater Total	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	5E-06	0.6	NA	NA	5E-06	0.6	1E-05	1	NA	NA	9E-06	1

note 1: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 2: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 3: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Exposure Pathways 2000 meters

Landfills - 2000m - by Exposure		Scenario 4													
		10-5		95% Pop		90% Prob		HH (1)		Eco (10)					
		Air Inhalation		Soil Ingestion		Water Ingestion		Crop Ingestion		Beef Ingestion		Milk Ingestion		Fish Ingestion	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	10	NA	10	NA	10	NA	10	NA	10	NA	10	NA	10

		Scenario 4											
		10-5		95% Pop		90% Prob		HH (1)		Eco (10)			
		Shower Inhalation		Breast Milk		All Inhalation		All Ingestion		All Ingest & Inhal		Groundwater Total	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	10	NA	NA	NA	10	NA	10	NA	NA	NA	10

note 1: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 2: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 3: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Scenarios 2000 meters

Landfills - 2000m - by Scenarios		Scenario 1			Scenario 2			Scenario 3			Scenario 4		
		10-6	99% Pop	95% Prob	10-6	99% Pop	90% Prob	10-5	99% Pop	90% Prob	10-5	95% Pop	90% Prob
Chemical Name	**CASRN**	**HH (0.1)**	**Eco (1)**	**Lowest**	**HH (1)**	**Eco (1)**	**Lowest**	**HH (1)**	**Eco (1)**	**Lowest**	**HH (1)**	**Eco (10)**	**Lowest**
Acrylonitrile	107-13-1	0	note 1	0	0	note 1	0	1	note 1	1	10	note 1	10

note 1: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.

note 2: The values in the highlighted cells are the same as the highest waste concentration evaluated.

Landfills - in mg/kg HWIR Results - All Sub-Populations 2000 meters

Landfills - 2000m - by Sub-Populations		Scenario 1							
		10-6			99% Pop 95% Prob			HH (0.1) Eco (1)	
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	CASRN	Risk	HQ	Risk	HQ	Risk	HQ	Risk	HQ
Acrylonitrile	107-3-1	7E-06	1	2E-06	0.1	8E-07	0.06	9E-07	0.06

		Scenario 2							
		10-6			99% Pop 90% Prob			HH (1) Eco (1)	
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	CASRN	Risk	HQ	Risk	HQ	Risk	HQ	Risk	HQ
Acrylonitrile	107-3-1	7E-06	6	1E-06	0.9	6E-07	0.7	8E-07	0.6

Landfills - 2000m - by Sub-Populations		Scenario 3							
		10-5			99% Pop 90% Prob			HH (1) Eco (1)	
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	CASRN	Risk	HQ	Risk	HQ	Risk	HQ	Risk	HQ
Acrylonitrile	107-3-1	2E-05	6	9E-06	0.9	6E-06	0.7	7E-06	0.6

		Scenario 4							
		10-5			95% Pop 90% Prob			HH (1) Eco (10)	
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	CASRN	Risk	HQ	Risk	HQ	Risk	HQ	Risk	HQ
Acrylonitrile	107-3-1	NA	10	NA	10	NA	10	NA	10

note 1: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 2: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 3: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Cohorts by Scenarios 2000 meters

Solids - 2000m - by Cohorts		Scenario 1						Scenario 2					
		10-6 99% Pop 95% Prob HH (0.1) Eco (1)						10-6 99% Pop 90% Prob HH (1) Eco (1)					
		Infants		1-12 yrs old		13 yrs old & older		Infants		1-12 yrs old		13 yrs old & older	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	NA	0E+00	0.2	9E-07	0.06	NA	NA	0E+00	3	9E-07	0.7
		Scenario 3						Scenario 4					
		10-5 99% Pop 90% Prob HH (1) Eco (1)						10-5 95% Pop 90% Prob HH (1) Eco (10)					
		Infants		1-12 yrs old		13 yrs old & older		Infants		1-12 yrs old		13 yrs old & older	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	NA	0E+00	3	9E-06	0.7	NA	NA	NA	10	NA	10

note 1: The levels for this chemical reflect only results for landfills. Exemption levels for solids would be selected from the landfill and wastepile results.

note 2: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.

note 3: The values in the highlighted cells are the same as the highest waste concentration evaluated.

note 4: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).

NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Exposure Pathways 2000 meters

Solids - 2000m - by Exposure		Scenario 2													
		10-6		99% Pop		90% Prob		HH (1)		Eco (1)					
		Air Inhalation		Soil Ingestion		Water Ingestion		Crop Ingestion		Beef Ingestion		Milk Ingestion		Fish Ingestion	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	0E+00	0E+00	1E-09	0E+00	9E-07	0.8	2E-05	6	6E-07	5	6E-06	5	1E-09	0E+00

		Scenario 2											
		10-6		99% Pop		90% Prob		HH (1)		Eco (1)			
		Shower Inhalation		Breast Milk		All Inhalation		All Ingestion		All Ingest & Inhal		Groundwater Total	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	9E-09	0.6	NA	NA	9E-09	0.6	1E-06	1	NA	NA	1E-06	1

note 1: The levels for this chemical reflect only results for landfills. Exemption levels for solids would be selected from the landfill and wastepile results.
note 2: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 3: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 4: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Scenarios 2000 meters

Solids - 2000m - by Scenarios		Scenario 1			Scenario 2			Scenario 3			Scenario 4		
		10-6	99% Pop	95% Prob	10-6	99% Pop	90% Prob	10-5	99% Pop	90% Prob	10-5	95% Pop	90% Prob
Chemical Name	**CASRN**	**HH (0.1)**	**Eco (1)**	**Lowest**	**HH (1)**	**Eco (1)**	**Lowest**	**HH (1)**	**Eco (1)**	**Lowest**	**HH (1)**	**Eco (10)**	**Lowest**
Acrylonitrile	107-13-1	0	note 2	0	0	note 2	0	1	note 2	1	10	note 2	10

note 1: The levels for this chemical reflect only results for landfills. Exemption levels for solids would be selected from the landfill and wastepile results.

note 2: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.

note 3: The values in the highlighted cells are the same as the highest waste concentration evaluated.

Landfills - in mg/kg HWIR Results - All Exposure Pathways 2000 meters

Solids - 2000m - by Exposure		Scenario 3													
		10-5		99% Pop		90% Prob		HH (1)		Eco (1)					
		Air Inhalation		Soil Ingestion		Water Ingestion		Crop Ingestion		Beef Ingestion		Milk Ingestion		Fish Ingestion	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	0E+00	0E+00	2E-09	0E+00	7E-06	0.8	2E-05	6	7E-06	5	2E-05	5	2E-09	0E+00

		Scenario 3											
		10-5		99% Pop		90% Prob		HH (1)		Eco (1)			
		Shower Inhalation		Breast Milk		All Inhalation		All Ingestion		All Ingest & Inhal		Groundwater Total	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	5E-06	0.6	NA	NA	5E-06	0.6	1E-05	1	NA	NA	9E-06	1

note 1: The levels for this chemical reflect only results for landfills. Exemption levels for solids would be selected from the landfill and wastepile results.
note 2: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 3: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 4: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - All Exposure Pathways 2000 meters

Solids - 2000m - by Exposure		Scenario 4													
		10-5		95% Pop		90% Prob		HH (1)		Eco (10)					
		Air Inhalation		Soil Ingestion		Water Ingestion		Crop Ingestion		Beef Ingestion		Milk Ingestion		Fish Ingestion	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	10	NA	10	NA	10	NA	10	NA	10	NA	10	NA	10

		Scenario 4											
		10-5		95% Pop		90% Prob		HH (1)		Eco (10)			
		Shower Inhalation		Breast Milk		All Inhalation		All Ingestion		All Ingest & Inhal		Groundwater Total	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	10	NA	NA	NA	10	NA	10	NA	NA	NA	10

note 1: The levels for this chemical reflect only results for landfills. Exemption levels for solids would be selected from the landfill and wastepile results.
note 2: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.
note 3: The values in the highlighted cells are the same as the highest waste concentration evaluated.
note 4: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).
NA: The pathway and chemical combination is not available or was not evaluated.

Landfills - in mg/kg HWIR Results - Sub Populations 2000 meters

Solids - 2000m - by Sub-Populations

		Scenario 1							
		10-6		99% Pop		95% Prob		HH (0.1)	Eco (1)
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	7E-06	1	2E-06	0.1	8E-07	0.06	9E-07	0.06

		Scenario 2							
		10-6		99% Pop		90% Prob		HH (1)	Eco (1)
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	7E-06	6	1E-06	0.9	6E-07	0.7	8E-07	0.6

Solids - 2000m - by Sub-Populations

		Scenario 3							
		10-5		99% Pop		90% Prob		HH (1)	Eco (1)
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	2E-05	6	9E-06	0.9	6E-06	0.7	7E-06	0.6

		Scenario 4							
		10-5		95% Pop		90% Prob		HH (1)	Eco (10)
		Beef/Dairy Farmer		Gardener		Fisher		Resident	
Chemical Name	**CASRN**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**	**Risk**	**HQ**
Acrylonitrile	107-13-1	NA	10	NA	10	NA	10	NA	10

note 1: The levels for this chemical reflect only results for landfills. Exemption levels for solids would be selected from the landfill and wastepile results.

note 2: Ecological impacts were not evaluated due to the lack of chronic ecological toxicity values.

note 3: The values in the highlighted cells are the same as the highest waste concentration evaluated.

note 4: A value of 0E+00 indicates that the risk is negligible (i.e., less than 1E-08).

NA: The pathway and chemical combination is not available or was not evaluated.