

WEST VIRGINIA VOLUNTARY REMEDIATION AND REDEVELOPMENT ACT



GUIDANCE MANUAL

VERSION 2.1

WVDEP Mission Statement

“Use all available resources to protect and restore West Virginia’s environment in concert with the needs of present and future generations.”

West Virginia Voluntary Remediation and Redevelopment Act Guidance Manual

DISCLAIMER

This document is intended as guidance only. The procedures and information contained herein are intended to assist in implementing the *Voluntary Remediation and Redevelopment Act* (“VRRRA”), W. Va. Code §22-22-1 *et seq.*, and the rules promulgated pursuant thereto. This guidance document is not intended to and does not create any rights, claims, causes of action, or remedies in addition to those afforded by VRRRA, the rules promulgated pursuant thereto, or other statutes, rules, or laws of West Virginia. This guidance document is not intended to and does not replace or change any part or provision of VRRRA or the rules promulgated pursuant thereto.

This guidance document has been issued by the West Virginia Division of Environmental Protection (“DEP”), Office of Environmental Remediation (“OER”), for use by its staff, by the regulated community, and by the citizens of this State. There is no intent on the part of the DEP or OER to give this guidance the same weight or deference as a statute, rule, adjudication or rule of law. This document provides a framework within which the DEP can exercise its administrative discretion. The DEP specifically reserves the right to deviate from this guidance document where circumstances may warrant such action.

West Virginia Voluntary Remediation and Redevelopment Act Guidance Manual

ACKNOWLEDGEMENTS

The Guidance Manual was written by the Technical Subcommittee of the Voluntary Remediation and Redevelopment Act Steering Committee. The Division gratefully acknowledges the work of the many persons who gave tirelessly of their time, energy and expertise.

Technical Subcommittee Members:

Mary Anderson, League of Women Voters
Jim Bodamer, WV Manufacturers Association, Chair
Ann Bradley, WV Chamber of Commerce
Marguerite Carpenter, WV Manufacturers Association
Ken Ellison, Division of Environmental Protection
Jim Kotcon, West Virginia University
Ron Potesta, WV Association of Consulting Engineers
Rudy Schuller, WV Oil & Natural Gas Association

The Division also acknowledges the assistance and contributions of individuals supporting the Technical Subcommittee:

Rachael Bell, WVU Graduate School
Jerome Cibrik, Union Carbide
David Hight, Division of Environmental Protection
William Jones, WVU Graduate School
Gale Lea, Jackson & Kelly
Mark Mummert, Environmental Consulting Inc.
Jan Taylor, National Institute for Chemical Studies
Mindy Yeager, Potesta & Associates

The Division wishes to acknowledge the National Institute for Chemical Studies for coordinating the expert peer review. The following expert peer reviewers have provided recommendations on the technical accuracy and completeness of the Guidance Manual, many of which were incorporated into the final version.

Waste and Chemical Management Division, EPA Region III
Hazardous Site Cleanup Division, EPA Region III
Terrie Baranek, ECT.CON Inc.
William Brattin, ISSI, Incorporated
Nancy Doerrer, American Industrial Health Council
Bruce Fishman, RBR Consulting, Inc.
Glenn Suter, Oak Ridge National Laboratory

Additional thanks go to key DEP staff support.

Rhonda McGlothlin
Wilma Pomeroy

**WEST VIRGINIA
VOLUNTARY REMEDIATION AND REDEVELOPMENT ACT
GUIDANCE MANUAL**

| | | |
|--------------------|--|--------|
| Section 1.0 | OVERVIEW OF THE VOLUNTARY REMEDIATION AND REDEVELOPMENT PROGRAM | 1 - 1 |
| 1.1 | General Structure of the Program | 1 - 3 |
| 1.1.1 | Responsibilities of the Licensed Remediation Specialist | 1 - 3 |
| 1.1.2 | Remediation Standards Concept | 1 - 4 |
| 1.1.3 | Flexible Use of Remediation Standards Options | 1 - 4 |
| 1.1.4 | Applicability of the VRRRA to a Site | 1 - 7 |
| 1.2 | Sequence of Actions for Implementation of VRRRA | 1 - 7 |
| 1.2.1 | Site Assessment | 1 - 8 |
| 1.2.2 | Submission of an Application for Voluntary Remediation and Redevelopment of a Site | 1 - 8 |
| 1.2.3 | Public Notification and Involvement | 1 - 10 |
| 1.2.4 | Remediation Standard Selection | 1 - 14 |
| 1.2.5 | Development of Risk-Based Concentrations | 1 - 14 |
| 1.2.6 | Submittal of the Remedial Action Workplan | 1 - 14 |
| 1.2.7 | Remedy Implementation | 1 - 15 |
| 1.2.8 | Closure / Remediation Verification | 1 - 15 |
| 1.2.9 | Final Report Submitted for WVDEP Approval | 1 - 15 |
| 1.2.10 | WVDEP Issuance of Certificate of Completion | 1 - 16 |
| 1.3 | Interaction of the Voluntary Remediation Program With Other Environmental Programs | 1 - 18 |
| 1.4 | References | 1 - 19 |
| Section 2.0 | SITE ASSESSMENT | 2 - 1 |
| 2.1 | Site Characterization Objectives | 2 - 1 |
| 2.2 | Preliminary Characterization | 2 - 3 |
| 2.2.1 | Evaluation of Historical and Current Land Uses to Identify COPCs | 2 - 3 |
| 2.2.2 | Preliminary Evaluation of Site Physical Characteristics | 2 - 4 |
| 2.2.3 | Preliminary Identification of Potential Human and Ecological Receptors | 2 - 5 |
| 2.2.4 | Develop a Conceptual Site Model | 2 - 9 |
| 2.2.5 | Risk Evaluation | 2 - 11 |

| | | |
|--------|---|--------|
| 2.3 | Develop Data Requirements for Sampling and Analysis Plans | 2 – 11 |
| 2.3.1 | Risk Assessment Data Requirements | 2 – 14 |
| 2.3.2 | Data Requirements for Remedial Action Design (if applicable) | 2 – 15 |
| 2.3.3 | Data Requirements for Modeling (if applicable) | 2 – 16 |
| 2.4 | Developing specific Investigation Techniques for SAPs | 2 – 20 |
| 2.4.1 | Data Quality Considerations | 2 – 21 |
| 2.4.2 | Selection of Analytical Methods | 2 – 22 |
| 2.4.3 | Health and Safety Considerations | 2 – 23 |
| 2.4.4 | Surface and Subsurface Soils | 2 – 23 |
| 2.4.5 | Storm Water Runoff | 2 – 28 |
| 2.4.6 | Site Infiltration and Vadose Zone Characteristics | 2 – 30 |
| 2.4.7 | Groundwater | 2 – 30 |
| 2.4.8 | Surface Water and Sediment Sampling | 2 – 36 |
| 2.4.9 | Indoor Air Quality (IAQ) | 2 – 39 |
| 2.4.10 | Tanks, Drums and Asbestos Containing Materials (ACM) | 2 – 41 |
| 2.4.11 | Decontamination | 2 – 44 |
| 2.4.12 | Investigation Derived Waste | 2 – 45 |
| 2.4.13 | Modeling | 2 – 52 |
| 2.5 | Background Concentrations | 2 – 58 |
| 2.5.1 | Definition of Background | 2 – 58 |
| 2.5.2 | Comparison of Site Contaminant Concentrations to Background | 2 – 62 |
| 2.6 | Contaminants of Concern | 2 – 62 |
| 2.6.1 | Field or Laboratory Contaminants | 2 – 63 |
| 2.6.2 | Low Concentrations and Low Frequency of Detection | 2 – 63 |
| 2.6.3 | Unusually High Sample Quantitation Limits | 2 – 64 |
| 2.6.4 | Comparison to Background | 2 – 65 |
| 2.6.5 | Evaluate Essential Nutrients | 2 – 65 |
| 2.6.6 | Screen Against De Minimis or Benchmark Levels to Identify COCs | 2 – 65 |
| 2.6.7 | Additional Issues for Consideration | 2 – 65 |
| 2.7 | Presentation of COCs in Tabular Format | 2 – 67 |
| 2.8 | References | 2 – 67 |
| 2.8.1 | Preliminary Characterization | 2 – 67 |
| 2.8.2 | Risk Assessment Data Requirements | 2 – 67 |
| 2.8.3 | Data Requirements for Remedial Action Design | 2 – 68 |
| 2.8.4 | Data Requirements for Modeling | 2 – 68 |
| 2.8.5 | Investigation Techniques for Sampling and Analysis Plans | 2 – 68 |
| 2.8.6 | Background | 2 – 72 |
| 2.8.7 | Contaminants of Concern | 2 – 73 |

| | | |
|--------------------|---|--------|
| Section 3.0 | HUMAN HEALTH STANDARDS | 3 – 1 |
| 3.1 | Introduction | 3 – 1 |
| 3.2 | Human Health De Minimis Risk-Based Standard | 3 – 3 |
| 3.2.1 | De Minimis Standards for Soil | 3 – 4 |
| 3.2.2 | De Minimis Standards for Ground Water | 3 – 4 |
| 3.2.3 | Implementing the De Minimis Standards | 3 – 5 |
| 3.3 | Uniform Risk-Based Standard | 3 – 5 |
| 3.3.1 | Uniform Standards for Groundwater | 3 – 6 |
| 3.3.2 | Uniform Standards for Soil | 3 – 7 |
| 3.3.3 | Establishing the Uniform Standards | 3 – 7 |
| 3.3.4 | Uncertainty Analysis | 3 – 8 |
| 3.3.5 | Attaining Compliance with the Uniform Standard | 3 – 8 |
| 3.4 | Site-Specific Risk-Based Standard | 3 – 10 |
| 3.4.1 | Baseline Risk Assessment | 3 – 10 |
| 3.4.2 | Implementing Site-Specific Risk-Based Standards | 3 – 23 |
| 3.5 | References | 3 – 25 |
| | | |
| Section 4.0 | ECOLOGICAL RISK-BASED STANDARDS | 4 – 1 |
| 4.1 | De Minimis Ecological Screening Evaluation | 4 – 6 |
| 4.1.1 | Identifying Potential Receptors of Concern | 4 – 7 |
| 4.1.2 | Determination of Exposure Pathway | 4 – 8 |
| 4.1.3 | Exposure Characterization | 4 – 9 |
| 4.1.4 | Reporting Requirements | 4 – 9 |
| 4.2 | Uniform Ecological Evaluation | 4 – 12 |
| 4.2.1 | Benchmarks and Generic Exposure Models for Uniform Ecological Evaluation | 4 – 12 |
| 4.2.2 | Applicant-Derived Benchmarks for Uniform Uniform Ecological Evaluation | 4 – 12 |
| 4.2.3 | Risk Characterization Based on the Uniform Ecological Evaluation | 4 – 14 |
| 4.2.4 | Reporting Requirements for the Uniform Ecological Evaluation | 4 – 18 |

| | | |
|--------------------|---|--------|
| 4.3 | Ecological Site-Specific Risk-Based Standards | 4 – 18 |
| 4.3.1 | Problem Formulation | 4 – 19 |
| 4.3.2 | Quantitative Exposure Analysis | 4 – 20 |
| 4.3.3 | Ecological Response Analysis | 4 – 24 |
| 4.3.4 | Risk Characterization | 4 – 24 |
| 4.3.5 | Remediation Standards Based on Ecological Risk | 4 – 26 |
| 4.3.6 | Uncertainty Analysis | 4 – 26 |
| 4.3.7 | Reporting Requirements | 4 – 27 |
| 4.4 | References | 4 – 29 |
| Section 5.0 | RESIDUAL RISK ASSESSMENT | 5 – 1 |
| Section 6.0 | PROBABILISTIC RISK ASSESSMENT | 6 – 1 |
| Section 7.0 | REMEDY SELECTION AND EVALUATION | 7 – 1 |
| 7.1 | General | 7 – 1 |
| 7.2 | Identification of Candidate Remedies | 7 – 2 |
| 7.3 | Initial Screening of Candidate Remedies | 7 – 2 |
| 7.3.1 | Screening Criteria | 7 – 4 |
| 7.3.2 | Screening Method | 7 – 4 |
| 7.4 | Evaluation of Short-List Remedies | 7 – 5 |
| 7.4.1 | Evaluation Criteria | 7 – 5 |
| 7.4.2 | Evaluation Method | 7 – 8 |
| 7.5 | Inclusion of Natural Attenuation in Remedy Evaluation | 7 – 9 |
| 7.5.1 | Developing Evidence in Support of Natural Attenuation | 7 – 10 |
| 7.5.2 | Simulation of Natural Attenuation | 7 – 13 |
| 7.5.3 | Conduct an Exposure-Pathway Analysis | 7 – 14 |
| 7.5.4 | Develop a Long-Term Monitoring Plan | 7 – 15 |
| 7.6 | References | 7 – 15 |
| 7.6.1 | Remedy Selection, Contaminant-Specific | 7 – 15 |
| 7.6.2 | Remedy Selection, Technology-Specific | 7 – 16 |
| 7.6.3 | Remedy Evaluation | 7 – 17 |
| 7.6.4 | Electronic Data Bases | 7 – 18 |
| 7.6.5 | Cost Analysis / Economics | 7 – 18 |
| 7.6.6 | Natural Attenuation | 7 – 18 |

| | | |
|----------------------|--|-------|
| Section 8.0 | REMEDIAL ACTION WORKPLAN | 8 – 1 |
| 8.1 | Purpose | 8 – 1 |
| 8.2 | Information Required | 8 – 1 |
| 8.3 | Remediation Standards | 8 – 1 |
| 8.4 | Remediation Measures | 8 – 2 |
| | 8.4.1 Selection of Alternatives | 8 – 2 |
| | 8.4.2 Natural Attenuation | 8 – 2 |
| | 8.4.3 Uncertainty or Risks | 8 – 2 |
| 8.5 | Remediation | 8 – 3 |
| 8.6 | Submittal | 8 – 3 |
| | | |
| Section 9.0 | FINAL REPORT | 9 – 1 |
| 9.1 | Contents | 9 – 1 |
| 9.2 | Appendices | 9 – 1 |
| 9.3 | Additional Documentation | 9 – 1 |
| 9.4 | Certification | 9 – 2 |
| | | |
| Appendix A: | Checklist For Conceptual Site Model Development | |
| Appendix B: | Determining Background Concentrations | |
| Appendix C-1: | Determination Of The Applicable Human Health Standard | |
| Appendix C-2: | Checklist To Determine The Applicable Ecological Standard | |
| Appendix D: | Equations For The Uniform Human Health Standards For Soil And Drinking Water | |
| Appendix E: | Relative Absorption Factors And Bioavailability | |
| Appendix F: | Risk Assessment For Lead | |
| Appendix G: | References To Benchmark Screening Levels | |
| Appendix H: | Site Specific Risk Assessment | |

Appendix I: Probabilistic Methodologies In Risk Assessment

Appendix J: Office Of Water Resources In-Stream Monitoring Procedures To Determine Impact On The Surface Water From Non-Point Source Site Remediation Projects

ACRONYM LIST

| | |
|---------|---|
| ACM | Asbestos Containing Materials |
| ASTM | American Society for Testing Materials |
| API | American Petroleum Institute |
| ATSDR | Agency for Toxic Substances and Disease Registry |
| BOD | Biological Oxygen Demand |
| BTEX | Benzene, Toluene, Ethylene, Xylene |
| CEPPO | Chemical Emergency Preparedness and Prevention |
| CERCLA | Comprehensive Environmental Response, Compensation, and Liability Act |
| CERCLIS | CERCLA List |
| CFR | Code of Federal Regulations |
| CLP | Contract Laboratory Program |
| COCs | Chemicals of Concern |
| COD | Chemical Oxygen Demand |
| COPCs | Chemicals of Potential Concern |
| CSF | Cancer Slope Factor |
| CSR | Code of State Regulations |
| CWA | Clean Water Act |
| DQOs | Data Quality Objectives |
| Eh | Redox Potential |
| EPCRA | Emergency Planning and Community Right-to-Know Act |
| ERNS | Emergency Response Notification System |
| FEMA | Federal Emergency Management Agency |
| FISs | Flood Insurance Studies |

| | |
|----------|--|
| FSP | Field Sampling Plan |
| GC-MS | Gas Chromatography-Mass Spectrometry |
| GPA | Groundwater Protection Act |
| HASP | Health and Safety Plan |
| HAZWOPER | Hazardous Waste Operations and Emergency Response |
| IAQ | Indoor Air Quality |
| IDW | Investigation Derived Waste |
| IDWMP | IDW Management Plan |
| K_{oc} | Organic-Carbon Partition Coefficient |
| K_{ow} | Octanol-Water Partition Coefficient |
| LDRs | Land Disposal Restrictions |
| LRS | Licensed Remediation Specialist |
| MDL | Method Detection Limit |
| MTBE | Methyl Tertiary Butyl Ether |
| NAD | North American Datum |
| NAPLs | Non-Aqueous Phase Liquids |
| NESHAP | National Emission Standards for Hazardous Air Pollutants |
| NIOSH | National Institute for Occupational Safety and Health |
| NIST | National Institute of Standards |
| NJDEP | New Jersey Department of Environmental Protection |
| NPDES | National Pollutant Discharge Elimination system |
| NPL | National Priority List |
| NVLAP | National Voluntary Laboratory Accreditation Program |
| NWWA | National Water Works Association |
| OER | Office of Environmental Remediation |

| | |
|----------|---|
| OPPE | Office of Policy, Planning and Evaluation |
| OSHA | Occupational Safety and Health Administration |
| OSWER | Office of Solid Waste and Emergency Response |
| OWM | Office of Waste Management |
| PAHs | Polyaromatic Hydrocarbons |
| PCBs | Polychlorinated Biphenyls |
| PCE | Tetrachloroethene |
| PCP | Pentachlorophenol |
| PELs | Permissible Exposure Limits |
| PIP | Public Involvement Plan |
| POTN | Publicly Owned Treatment Works |
| PPE | Personal Protective Equipment |
| PQL | Practical Quantitation Limit |
| QAPP | Quality Assurance Project Plan |
| QA / QC | Quality Assurance / Quality Control |
| RCRA | Resource Conservation and Recovery Act |
| RfC | Reference Concentration |
| Rfd | Reference Dose |
| the Rule | Voluntary Remediation and Redevelopment Rule (Title 60 Code of State Rules, Series 3) |
| SAP | Sampling and Analysis Plan |
| SCS | Soil Conservation Service |
| SIC | Standard Industrial Classification |
| SQL | Sample Quantitation Limit |
| TCE | Trichloroethene |

| | |
|-------|--|
| TCLP | Toxic Compound Leaching Procedure |
| TEGD | Technical Enforcement Guidance Document |
| TICs | Tentatively Identified Compounds |
| TNT | Trinitrotoluene |
| TOC | Total Organic Carbon |
| TPH | Total Petroleum Hydrocarbons |
| TSCA | Toxic Substances Control Act |
| TSD | Treatment, Storage and Disposal |
| TSS | Total Suspended Solids |
| TVOCs | Total Volatile Organic Compounds |
| USDA | United States Department of Agriculture |
| USEPA | United States Environmental Protection Agency |
| USGS | United States Geological Survey |
| UST | Underground Storage Tank |
| UTM | Universal Transverse Mercator |
| VRA | Voluntary Remediation Agreement |
| VRRRA | Voluntary Remediation and Redevelopment Act |
| VOCs | Volatile Organic Compounds |
| WHO | World Health Organization |
| WV | West Virginia |
| WVDEP | West Virginia Division of Environmental Protection |
| WVDNR | West Virginia Division of Natural Resources |
| XRF | X-ray Fluorescence |

West Virginia Voluntary Remediation and Redevelopment Act Guidance Manual

1.0 OVERVIEW OF THE VOLUNTARY REMEDIATION AND REDEVELOPMENT PROGRAM

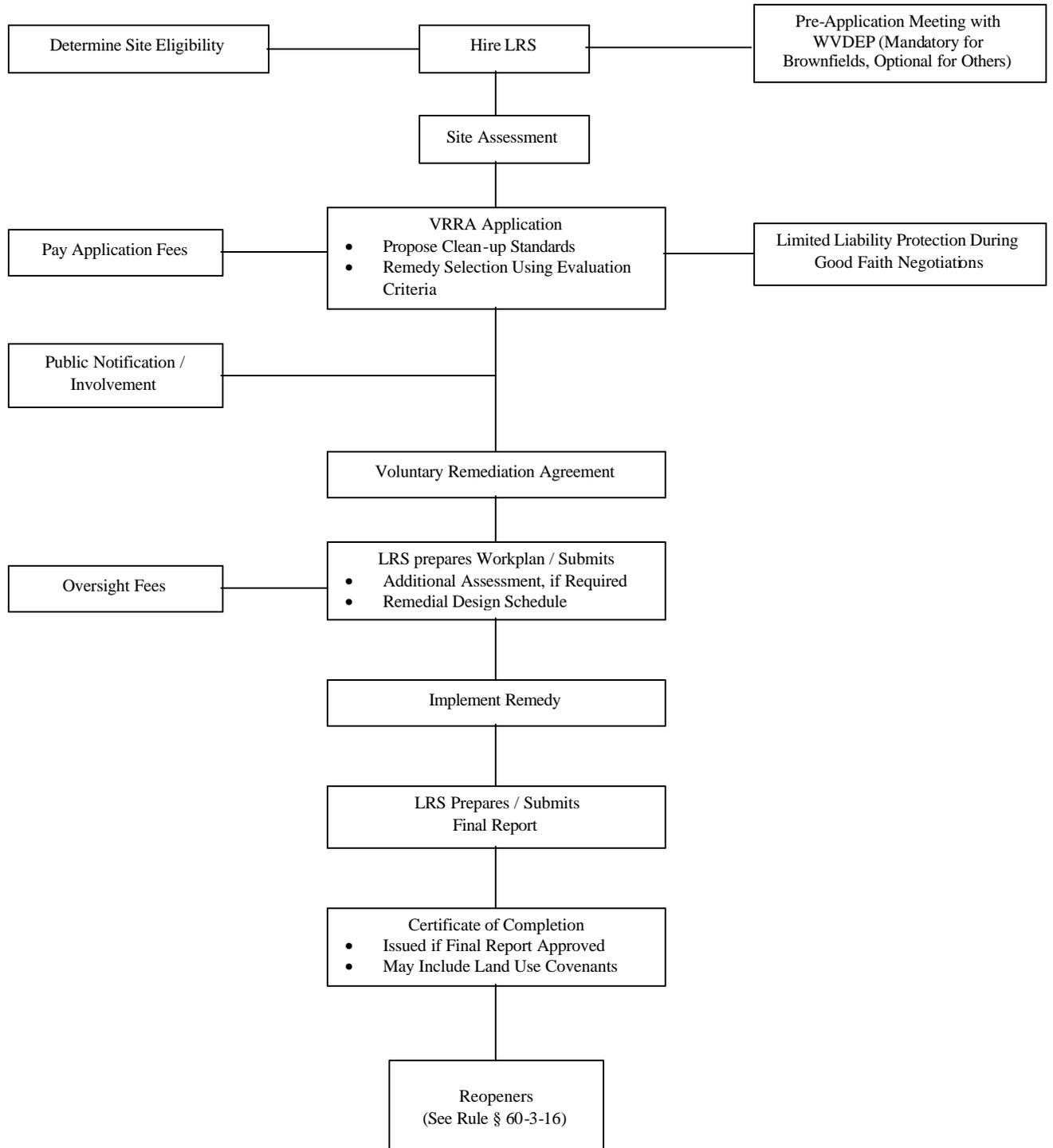
The Voluntary Remediation and Redevelopment Act (VRRRA) was enacted by the West Virginia (WV) Legislature for the purpose of encouraging the voluntary clean-up of contaminated sites and redevelopment of abandoned and under-utilized properties. Properties in the state are not being productively used because of contamination or the perception of contamination. Because many of these properties are located in areas with existing industrial infrastructure, redevelopment of these sites can be less costly to society than developing pristine sites.

The VRRRA encourages voluntary remediation and redevelopment through an administrative program set out in the WV Code of State Regulations, Title 60, Series 3 entitled the Voluntary Remediation and Redevelopment Rule (the Rule), which became effective on July 1, 1997. The VRRRA limits enforcement actions by the WV Division of Environmental Protection (WVDEP), provides financial incentives to entice investment in brownfield sites, and limits liability under environmental laws and rules for those who remediate sites under the standards provided in the Rule.

Both the VRRRA and the Rule were cooperatively developed by a diverse group of stakeholders. This process has led to a strong program that is protective of communities and the environment while promoting economic development in West Virginia. The VRRRA provides for flexibility in the voluntary clean-up of under-utilized properties and marks a turning point in state environmental policy. Figure 1-1 depicts the process to be followed in the Voluntary Remediation Program. Details of the process are provided in this section.

This Guidance Manual is provided to assist those who would like to participate in West Virginia's Voluntary Remediation Program. The information provided in this document is, indeed, guidance. Although based on the VRRRA and the Rule, this document does not carry the weight of law or regulation. The following information is intended to provide guidelines and to help lead an applicant through the Voluntary Remediation Program. Technical and scientific methods included with this guidance are acceptable to the WVDEP but are, by no means, the only acceptable alternatives. WVDEP recognizes that every site is unique and that no one guidance manual will be able to contain all of the scientifically valid methods of assessing and remediating contaminated properties. VRRRA encourages flexibility in remediating these under-utilized and contaminated sites, both for the VRRRA participant and for WVDEP. This section is merely an overview of the requirements for the VRRRA program. Details of the program are provided in the following sections.

Figure 1-1
WEST VIRGINIA VOLUNTARY REMEDIATION PROGRAM
-Overview-



1.1 General Structure of the Program

This section provides a summary of the requirements of the Voluntary Remediation Program. Details concerning these requirements are provided in the subsequent sections and appendices. WV's voluntary remediation program has as its foundation the VRRRA and the associated Rule. Participation in the program may be initiated by the owner or operator of a site, a developer, prospective purchaser, or other interested party. The purpose of participation is to identify and address potential contamination at the site. Voluntary remediation also establishes that the property complies with applicable remediation standards. Brownfield remediation is a special case of voluntary remediation. Brownfield sites also are industrial or commercial properties that are abandoned or inactive, however, voluntary remediation of brownfields involves the use of public funds for the site assessment or remediation. Because of the use of public funds, a much higher degree of public involvement is required for brownfield clean-ups.

Any person who wants to participate in the Voluntary Remediation Program must execute a Voluntary Remediation Agreement with the Director of the WVDEP. The Voluntary Remediation Agreement must provide for: 1) the services of a Licensed Remediation Specialist; 2) recovery of costs incurred by the WVDEP in excess of fees submitted by the applicant; 3) a schedule for payment of recoverable costs; 4) descriptions of the work plan and other reports that will be submitted; 5) a listing of applicable environmental requirements for the site; 6) technical standards for work at the site; 7) any engineering or institutional controls or land use covenants applicable for the site; and 8) criteria for reopening and modification of the agreement.

Once the Voluntary Remediation Agreement is executed, the Director is barred from beginning any enforcement actions against the applicant for the site and contamination under the agreement, unless there is an imminent threat to the public. The WVDEP is charged with the overall administration of the Voluntary Remediation Program and its responsibilities are carried out through the Office of Environmental Remediation (OER). The OER performs an oversight function with respect to work that is performed through the Program. The oversight function extends to review and approval of work plans and reports; periodic inspection of sites accepted into the Program; access to and review of all records relating to activities under the Program and performing sampling at sites in the Program. It is anticipated that the degree of OER oversight will increase with the size and complexity of the site.

1.1.1 Responsibilities of the Licensed Remediation Specialist

Within the Voluntary Remediation Program, all activities must be supervised by a Licensed Remediation Specialist. A **Licensed Remediation Specialist (LRS)** is a person certified by the Director of the WVDEP as qualified to perform professional remediation services and to supervise the remediation of contaminated sites.

The overriding duty of the LRS is to protect the safety, health and welfare of the public in the performance of his/her professional duties. The LRS is responsible for any release of contaminants from the site that occurs during approved remediation activities. If a release not contemplated by the Voluntary Remediation Agreement occurs during remediation activities, the LRS must immediately notify the WVDEP.

It is expected that a single LRS will supervise all site remediation activities. The LRS must be highly qualified, but it is unlikely that a single individual will have all of the skills and knowledge to perform all activities associated with the remediation. The LRS must only perform assignments for which he or she is qualified by training and/or experience in those specific technical fields. He/she should seek assistance from other qualified professionals as needed in performing work at the site.

The LRS is employed by the owner or developer of the contaminated site at usual and customary professional rates. However, the LRS must be completely objective in developing and reviewing work plans, reports, and opinions. The LRS represents the interests of the public as well as providing technical supervision of all remediation activities.

1.1.2 Remediation Standards Concept

The Voluntary Remediation and Redevelopment Rules provides for a range of risk-based soil, sediment and groundwater remedial objectives for site remediation. Risk-based remediation standards allow for current and future land and water uses to be considered in the clean-up process, while providing adequate protection of human health and the environment. The incorporation of site-related information may allow for more cost-effective remediation based on identified site risks.

1.1.3 Flexible Use of Remediation Standards Options

For any voluntary remediation, one or more remediation standards may be utilized. At some sites, the property may have areas ranging from severely contaminated to nearly pristine. For different sections of the property, different remediation standards may be appropriate. Under the Rule, these risk-based standards are available: 1) De Minimis risk-based; 2) Uniform risk-based; 3) site-specific risk-based; and 4) a combination of these remediation standards. The flexibility of the Rule allows for standards to be utilized as appropriate for any particular site. The VRRRA recognizes that each site is unique and may require the use of valid scientific procedures not specifically detailed in this guidance. In this case, the applicant and OER should work together to develop mutually agreeable methodology.

Regardless of the remedial standard selected, selection of the remedial action is based on achieving cost-effective protection of human health and the environment. In determining the remedial action, applicants must consider and document the following criteria:

- Effectiveness in protecting human health and the environment;
- Long-term reliability of remedial actions in achieving the standards;
- Short-term risks to the public, workers and the environment;
- Acceptability of the remedial action to the affected community;
- Engineering practicality of implementation;

- Achievement of protectiveness goals at lowest cost; and
- Consideration of net environmental benefits.

Risk-based standards are derived considering the current use of the property or a reasonably anticipated future use. Under the VRRRA, the two categories of land uses are residential and industrial. Industrial land use encompasses all kinds of commercial operations including manufacturing, distribution, warehousing, and waste management. Residential land use includes use of the land for homes, schools, nursing homes, recreation areas and the like. Recreational exposure is similar to residential in areas such as parks and playgrounds due to direct exposure to soils. Other recreational land use, such as gymnasiums and swimming pools would not have these types of exposure and may be considered under the commercial/industrial land use scenario. Please confer with the OER to determine if the exposure assumptions under consideration are appropriate for a proposed land use. Current and future use assessments may also include waters that may be part of or adjacent to the site. Different exposures and risks may be present for land uses such as agricultural and forestry and may warrant site-specific analysis. If land uses are being considered which involve exposure pathways not described in this document, please confer with the OER for future guidance. It is recognized that different exposures and risks may be present for these land uses and may warrant site-specific analysis.

1.1.3.1 Human Health Options

There are three options available for determining human health remediation standards. A **De Minimis Risk-Based Standard** may be used where the contaminant concentrations detected are below the risk-based concentrations provided in Table 60-3-B of the Rule and do not present significant risk to human health. If De Minimis levels are below the **natural** background concentrations, the natural background concentrations will be used as the De Minimis standard. The De Minimis Standard assumes the applicant has completed the Applicable Standard Checklist (Appendix C-1) to aid in determining whether the De Minimis Standard can be used for their site.

A **Uniform Risk-Based Standard** uses pre-approved equations and exposure factors identified in this Guidance Manual and other site-specific variable to calculate compound-specific remediation levels. The uniform standard also assumes that the applicant has completed the Applicable Standard Checklist (Appendix C-1), whether or not the site meets the De Minimis standard. OER reviews the checklist in evaluating the remediation plan. Default assumptions and equations for determining clean-up levels under the Uniform standard are located in Appendix D-2. Exposure factors which may be replaced by site-specific information are also listed in Table D-2.

A **Site-Specific Risk-Based Standard** uses a site-specific analysis of contamination and develops a remedial approach that considers the criteria in Subsection 1.1.3. If the standards developed in this way are below anthropogenic background levels of the contaminants at the site, the background levels will be considered the site-specific risk-based level.

A combination of these remediation standards may be used to implement a site remediation plan. The rules allows for any of the above standards to be used. An applicant may use site-specific risk-based standards whether attainment of other standards has or has not been attempted. A more detailed discussion of these standards appears in Section 3 of this manual.

1.1.3.2 Ecological Options

Unlike human health standards, the ecological standards are a series of evaluations to determine whether ecological receptors of concern are present and are affected. A De Minimis Ecological Screening is the minimum evaluation to be completed on any site. Increasing levels of assessment may need to be accomplished to assure ecological protection.

Where complete pathways to significant ecological receptors exist, the applicant may use uniform “benchmark” levels as remedial goals, or may engage in a more elaborate, site-specific ecological risk assessment to set site clean-up goals.

A **De Minimis Ecological Screening Evaluation** is an evaluation of the nature and extent of contaminants on the site. The evaluation should determine if potential exposure pathways are completed. If complete exposure pathways are not present between contaminants and ecological receptors of concern, no significant risk to ecological receptors is assumed. If complete pathways are present and the conditions outlined in Subsections 9.5.a through 9.5.b of the Rule have been considered, additional evaluation of ecological risk may be required.

A **Uniform Ecological Evaluation** compares site contaminant levels to benchmark values that pose no significant risks to ecological receptors of concern. If benchmark values are below natural or anthropogenic background, background concentrations will be considered the Uniform Ecological Standard. If an applicant proposes a remediation standard based on other existing standards which exceed the benchmark values, the Director may require a site-specific ecological risk assessment if he or she feels the other standards are not protective of the ecological receptors of concern.

A **Site-Specific Ecological Risk-Based Standard** uses site-specific analysis of present contamination and develops a remedial approach that considers the remedy criteria above. A site-specific analysis may be conducted using guidance from Subsection 4.3 of this document.

1.1.3.3 Combining Human Health and Ecological Processes

A combination of human health and ecological standards may be used to implement a complete site remediation plan. Parts of the site may be cleaned up to different standards. For some sites or areas of a site, human health standards may be protective of ecological receptors as well. At other areas, ecological standards may be the more stringent clean-up goal.

1.1.4 Applicability of the VRRRA to a Site

The Voluntary Remediation and Redevelopment Act can apply to almost any site in West Virginia. For both currently operated and abandoned sites, the voluntary remediation program is available. Sites eligible to participate in the program are those sites which are **not**:

- subject to United States Environmental Protection Agency (USEPA) unilateral enforcement orders under CERCLA (Comprehensive Environmental Response, Compensation and Liability Act) or RCRA (Resource Conservation and Recovery Act);
- listed or proposed to be listed on the National Priority List (NPL) under CERCLA;
- subject to WVDEP unilateral enforcement orders under Chapter 22 of the West Virginia Code; or
- created through gross negligence or willful misconduct.

Note: Sites that are covered by a federal or state consent order are not precluded from participating in the Voluntary Remediation Program. In such circumstances, it would be the applicant's burden to assure that the requirements of **both** the consent order and the Voluntary Program are satisfied.

A site may participate in the voluntary remediation program as a **brownfield site** if it meets the general eligibility criteria for the program **and**:

- the applicant did not cause or contribute to the site contamination;
- the site was an industrial or commercial property either abandoned or not in active use by the owner as of July 1, 1996; and
- the applicant qualifies as a brownfield applicant by applying for or obtaining a site assessment loan from the Brownfields Revolving Fund or by using other public funds from the state, county, or municipality.

1.2 Sequence of Actions for Implementation of VRRRA

To implement a remediation under the VRRRA, a series of steps must be taken. After eligibility for the program has been determined, the potential applicant should hire an LRS to supervise site assessment and remediation activities and, if required (for brownfield sites) or desired, request a pre-application meeting with the WVDEP. There is some flexibility in the order in which some of these steps may be taken (see Figure 1-1).

1.2.1 Site Assessment

Every application to participate in the Voluntary Remediation program must include a site assessment. The VRRRA allows considerable flexibility in the extent of the site assessment that is necessary to support an application to participate. At a minimum, the site assessment must include a legal description of the site, a description of the site's physical characteristics, the general operational history of the site, and any available information concerning the nature, location, and extent of any known contamination on or near the site.

At some point, however, whether with the application or after the Voluntary Remediation Agreement has been negotiated, sufficient information must be provided to assure that the site has been adequately characterized both horizontally and vertically. Characterization under the rule means an evaluation of the site's physical and environmental features to determine: 1) if a release has occurred; 2) levels of chemicals of concern in environmental media; and 3) likely physical distribution of the chemicals of concern. Evaluation of the physical and environmental features may include a review of subsurface geology, soil properties and structures, hydrology and surface characteristics. Data are to be collected on groundwater and surface water quality, land and resource use and potential receptors, both human and ecological. These data are generated as appropriate to support remedial action decisions under the Program. Further guidance on site characterization is found in Section 2.0.

1.2.2 Submission of an Application for Voluntary Remediation and Redevelopment of a Site

To participate in the Voluntary Remediation and Redevelopment Program, an application must be filed with the WVDEP. If the application is for a brownfield site, applicants must have a pre-application conference with the Director and comply with loan procedures (Section 15 of the VRRRA Rule) before submitting an application. Other applicants may request a pre-application conference with the Director, if desired.

An application form is available from the WVDEP Office of Environmental Remediation. Applicants should submit the completed application to the Chief of the Office of Environmental Remediation, 1356 Hansford Street, Charleston, WV 25301-1401. Each application must be accompanied by an application fee made payable to the Voluntary Remediation Administrative Fund.

The application fee is based on (1) the size of the property, (2) the years of operation for non-residential purposes, and (3) the SIC (Standard Industrial Classification) Code for the operations that occurred on the property. The fee will be determined using the point system described below:

Size of Property

- surface area less than one acre 10 points
- surface area less than 5 acres and greater than 1 acre 20 points
- surface area equal to or greater than 5 acres 30 points

Years of Operation (for a non-residential activity)

- 10 years or fewer 10 points
- more than 10 years but fewer than 20 years 20 points
- 20 years or more 30 points

SIC Codes (activities that may fall in more than one SIC Code will be assigned the code with the highest points)

- SIC 26 (Paper & Allied Products) 30 points
SIC 28 (Chemicals & Allied Products)
SIC 29 (Petroleum Refining)
SIC 30 (Rubber and Misc. Products)
SIC 31 (Leather & Leather Products)
SIC 33 (Primary Metals)
SIC 34 (Fabricated Metal Products)
SIC 35 (Industrial & Commercial Machinery)
SIC 36 (Electronic & Other Electrical Equipment)
SIC 37 (Transportation Equipment)
SIC 38 (Measuring, Analyzing & Controlling Equipment), and
SIC 39 (Miscellaneous Manufacturing)
- SIC 10 thru 14 (Mining) 20 points
SIC 20 (Food)
SIC 21 (Tobacco Products)
SIC 22 (Textiles)
SIC 24 (Lumber & Wood Products, except Furniture)
SIC 27 (Printing & Publishing)
SIC 32 (Stone, Clay, Glass and Concrete)
SIC 46 (Pipelines) and
SIC 49 (Electric, Gas & Sanitary Service)
- Any other SIC Code 10 points

Total points accumulated from the above criteria determine the application fee. The application fees are:

| | |
|-------------------------------|---------|
| Total points of 30 or 40 | \$1,000 |
| Total points of 50 or 60 | \$3,000 |
| Total points of 70, 80, or 90 | \$5,000 |

If the application covers two or more non-contiguous locations, the locations are similar in terms of contaminants and surface/subsurface characteristics, and no one location is larger than 2 acres, the application fee is \$5,000. Any individual location with surface area greater than 2 acres requires a separate application and fee.

1.2.3 Public Notification and Involvement

The WVDEP encourages participants in the voluntary clean-up program to communicate with local government officials and interested members of the public regarding information gathered and remediation plans for the site. Early, frequent and meaningful involvement of the interested public with voluntary remediation activities can create a strong and cooperative project that meets the needs of both the developer and the public.

While the discussion below lays out the minimum requirements for public notification and involvement, focused and creative programs for public interaction with voluntary clean-up activities are recommended. For simple and non-controversial sites (such as an abandoned gas station), little public interest and, therefore, involvement might be expected. For a large, complicated site, especially in highly populated areas, public interest and involvement is expected to be much greater.

In cases where many stakeholders have strong interest in a site clean-up, a multi-pronged approach to public participation may be warranted. For identifying public concerns, methods such as public meetings, presentations to community organizations, advisory committees, site visits and the like are often useful. If public input is to be used to develop ideas and solve problems, small group meetings, field offices, requests for comments and ombudsmen are among many public participation techniques that can be utilized. Stakeholder evaluation of remediation proposals can be aided if the applicant provides such assistance as workshops, model demonstration projects and environmental impact statements. Public participation activities can also be helpful in resolving conflicts. More information about meaningful public participation can be found in a variety of published documents including *Ecology, Impact Assessment, and Environmental Planning* (Westman, 1985) from which some of the above information was taken. Other resources for risk communication and public involvement guidance include:

- USEPA Office of Chemical Emergency Preparedness and Prevention (CEPPO)
USEPA Superfund Office
Emergency Planning and Community Right-to-Know Act
(EPCRA) Hotline (800) 424-9436
- USEPA Office of Policy, Planning and Evaluation (OPPE) (202) 260-4538
- USEPA Risk Communication Hotline (202) 260-5606
- USEPA Office of Pollution Prevention and Toxic Substances (202) 260-3944

Specific requirements of the VRRRA relating to public notification and involvement differentiate between brownfields and other voluntary remediation sites.

1.2.3.1 Brownfield Sites

The potential developers of brownfield sites, whether private or public entities, must notify the public about remediation activities and provide for public involvement in planning for remediation and redevelopment. The requirements are discussed below.

Notice of Intent to Remediate

The brownfield applicant must file a Notice of Intent to Remediate with the Director of the WVDEP. The WVDEP will publish a summary of the Notice of Intent to Remediate in a WVDEP publication of general circulation and will issue a news release summarizing the Notice. The summary will include information on the public's right to know to become involved with the development of remediation and reuse plans for the site. The news release will be sent to media outlets covering the area of the remediation activity.

The WVDEP may also post the Notice on the Division's Internet web site. (<http://charon.osmre.gov>.)

In addition to these general methods of notifying the public, WVDEP will also directly notify other government bodies of the Intent to Remediate. In the area where the brownfield site is located, the municipality and county commission, any county or municipal land use agency, and the Regional Planning and Development Council will be notified.

Depending on the site and its interest to state and federal agencies, WVDEP may also notify the USEPA, the United States Army Corps of Engineers, the state Bureau of Public Health and other state or federal agencies. All of these parties will be notified of the Director's decision on a Certificate of completion at the end of the remediation.

The Notice of Intent will contain the following information:

- name and business address of the brownfield applicant; and
- the geographic location of the site.

In addition, the Notice will include, as far as is known:

- the current, former and proposed future uses of the site;
- known and suspected contaminants on-site;
- proposed methods to remediate the site; and
- proposed methods for controlling possible health exposure.

For persons interested in the potential remediation, the Notice will also provide the location where the Notice may be reviewed, and the names, addresses and telephone numbers of the applicant's public contact and the WVDEP contact for comments and questions.

The WVDEP Office of Environmental Remediation will make the Notice available for public inspection and copying upon request. The Notice will also be available in the municipal and/or county commission offices where the remediation is proposed and in the county library.

A 30-day comment period and an informational meeting for interested parties are required for all brownfield sites. The applicant must provide notice to the public of the start of the comment period and of the informational meeting. This notice must be given in the following ways.

- by a 3' x 4' sign at the site stating "This site is under consideration for environmental cleanup and participation in the state's brownfield program under the Voluntary Remediation and Redevelopment Act." The sign will include the WVDEP brownfield office address and phone number.
- by a weekly 4" x 4" box ad in a local newspaper for 4 weeks which includes the information from the Notice of Intent to Remediate plus the time, date and location of the informational meeting.
- by sending a copy of the box ad to local authorities and the county/municipal land use agency, or the area's Regional Planning and Development Council.

The WVDEP will draft the box advertisement and send it to the brownfield applicant for publication. A certified and notarized proof of publication must be submitted to the WVDEP within 4 weeks of the final publication date.

Anyone may file a request to participate in remediation and reuse planning. Interested parties should write the Director during the 30-day comment period. By writing during the comment period, interested parties may automatically participate, in person or by a representative.

Brownfield Public Involvement Plan (PIP)

Each brownfield applicant must establish a PIP if requested by the public, by the county or municipality where the site is located, or by the Director. The PIP is intended to make it easy for the public to be involved with the planning for activities that may affect their communities. The PIP will be developed by the applicant in consultation with those persons who have requested to be involved in the remediation and reuse planning. The plan must be developed within 30 days of receiving notice from the Director that a request to participate has been received. The PIP will then be submitted to the Director for his/her review and approval.

The minimum requirements for a PIP include the following:

- provisions for more community meetings;
- opportunities for review and comment by participants on the work plan and voluntary agreement; and
- a means of communicating with the public which may include, but is not limited to point of contact, mailing and telephone lists, newsletter, doorstep notice, media advertisements, news releases, presentations, and a citizen advisory panel for the remediation and reuse plan.

Participants other than the applicant may petition the Director for technical assistance to make their participation in remediation and reuse planning more meaningful. This technical assistance may include review of site-related documents; explanations of technical information and translation of technical jargon to layperson's language; and assistance in communicating concerns to the applicant, WVDEP or others. If the Director receives such a petition, the Director and the brownfield applicant will, by mutual agreement, develop a technical assistance component to the PIP, paid for by the applicant.

1.2.3.2 Voluntary Remediation Projects (non-brownfields)

Public Notice of Applications for Voluntary Remediation Projects

For voluntary remediation projects not involving a brownfield site, public notice must also be given. Just as required for brownfields, WVDEP must publish a summary of the application for voluntary clean-up in a WVDEP publication of general circulation and in a news release to media outlets in the remediation area. The summary will include the following information:

- Name and address of the applicant;
- Location of the site;
- Current use of the site;
- Suspected contaminants on site;
- Proposed cleanup methods and proposed methods to control possible health effects;
- Location where the application can be reviewed;
- Name, address, phone number of applicant's public contact person;
- Name, address, phone number of WVDEP contact for comments and questions.

The Voluntary Remediation Application will be available for inspection and copying at the WVDEP Office of Environmental Remediation and in municipal/county commission offices where the project is located. Copies may also be placed in the county library. As additional information is subsequently developed concerning the site, the DEP may elect to add such information to the public repository.

Public Involvement and Notification in Development of Remediation Goals

Where a residual cancer risk level of greater than 1×10^{-6} is proposed for a residential land use or greater than 1×10^{-5} for commercial or industrial use, an informational meeting and 30-day public comment period are required. Notification of the public will follow the steps detailed in the previous sections for brownfield public notification.

1.2.4 Remediation Standard Selection

Once it is determined following the site assessment that remediation of a site is necessary, remediation standards are to be selected based on potential health effects, ecological effects and current and reasonably anticipated future use.

1.2.5 Development of Risk-Based Concentrations

Risk-based concentrations under the Voluntary Remediation and Redevelopment Rule must be based on scientifically valid toxicity information. This information may come from recognized USEPA sources, from other government agency reports, or from peer-reviewed scientific literature. Site-specific risk-based standards must consider the potential for exposure to site contaminants under current and reasonably anticipated future land and water use.

For human health risk, carcinogen standards must be established at levels which represent an excess upper-bound lifetime risk of between one in ten thousand (1×10^{-4}) to one in one million (1×10^{-6}). That is, the additional risk of cancer to an individual exposed to the site over a lifetime of 70 years must not be greater than one in ten thousand to one in one million. All risk estimates are to be expressed to only one significant figure.

Remediation standards for systemic toxicants must be contaminant concentration levels to which the human population could be exposed without appreciable risk of harmful effects, that is, where the hazard quotient does not exceed 1.0. Where multiple systemic toxicants affect the same organ or act by the same method of toxicity, the Hazard Index (sum of the hazard quotient) shall not exceed 1.0. Where multiple systemic toxicants do not affect the same organ, the Hazard Index shall not exceed 10.0 (§60-3-9.4.b of the Rule). If the Hazard Quotient exceeds 1.0, further evaluations may be necessary as discussed in Section 3.4.1.3, Approach for Calculating Noncancer Risks. All risk estimates are to be expressed to only one significant figure.

1.2.6 Submittal of the Remedial Action Workplan

The applicant or the applicant's LRS must submit any workplan required by the Voluntary Remediation Agreement to the Director. The Director may approve or disapprove the workplan, within 30 days of receipt, based on its quality and completeness. The Director may require the following in a thorough remedial action workplan:

- Health and Safety Plan;
- Documentation of the investigation that led to preparation of the workplan;
- Description of the assessments to be performed to further determine the nature and extent of actual or threatened releases;
- Description of the risk assessment conducted to show the appropriateness of remedy selection;

- Statement of work to accomplish the remediation in accordance with the risk protocol and remediation standards in the Rule;
- Implementation schedule for the Remediation Workplan; and
- Verification sampling plan to determine the adequacy of the remediation.

Within 5 days of a determination to disapprove a workplan, the Director must let the applicant know of the disapproval and provide a list of the reasons for disapproval. The Director will also indicate any additional information needed to gain approval. The applicant must either resubmit the workplan or formally terminate the Voluntary Remediation Agreement. If a final workplan is not approved or disapproved with 30 days of its receipt, the workplan will be deemed approved.

1.2.7 Remedy Implementation

Following approval of the workplan, remediation activities may begin. Remediation standards may be attained through one or more remediation activities that may include treatment, removal, engineering or institutional controls, natural attenuation and/or innovative measures. The West Virginia program is intended to be flexible and to allow innovation and good science, but does not require unusual or novel procedures.

1.2.8 Closure/Remediation Verification

When the remedy implemented to meet the applicable standards is in place, the LRS may prepare the Final Report. The Final Report should include all data and information needed to document and verify that all applicable standards have or will be met. Any ongoing work and monitoring must be described including planned activities and schedules. Supporting documentation should be included. The LRS will submit the final report to the person or organization undertaking the voluntary remediation.

1.2.9 Final Report Submitted for WVDEP Approval

The Final Report, if submitted by the applicant to WVDEP for approval, must include:

- Names, addresses, telephone and fax numbers for the current owners and/or operators of the site;
- Names, addresses, telephone and fax numbers of the owners and/or operators conducting the remediation and the LRS. Individual names and titles must be provided;
- Site location, its legal description and a site location map;

- Any ongoing work and monitoring must be described including planned activities and schedules; and
- Any institutional controls, such as deed restrictions or land use covenants, must be confirmed by including copies of properly recorded documents. The institutional controls must be shown on a site map.

The completeness and accuracy of the Final Report must be certified by an authorized agent of the applicant and by the LRS. The LRS and the applicant's agent may be the same person. The certified Final Report may be submitted to the Director for approval with a request for a Certificate of Completion. Upon review of the final report, the Director determines whether it was properly issued.

1.2.10 WVDEP Issuance of Certificate of Completion

When the Director receives a request for a Certificate of Completion, he/she must determine if the site meets the proper requirements. These requirements are that the applicable standards for the areas and contaminants covered by the Voluntary Remediation Agreement have been met, and that the applicant has complied with the Agreement and approved work plans for the site. The Director must determine that the Final Report was properly issued within 60 days of the submission of the request for a Certificate of Completion.

If the Director determines that the Final Report was properly issued, a Certificate of Completion must be issued within 30 days of his/her decision. If the report was not properly issued, the Director must promptly notify the applicant in writing. The notification will include the reasons why the report was not approved and will indicate any further actions necessary for issuance of the Certificate.

An applicant who receives a notice of disapproval may take one of the following steps. The applicant may: 1) instruct his or her LRS to take the further actions as indicated in the notification; 2) appeal the Director's decision to the Environmental Quality Board; or 3) formally terminate the Voluntary Remediation Agreement.

The successful applicant will receive a Certificate of Completion for the remedial activities covered under the Voluntary Remediation Agreement. A Certificate of Completion will incorporate the following information:

- Description of the site;
- Description of contaminants for which the standards have been met;
- The Voluntary Remediation Agreement;
- The Final Report prepared by the LRS; and
- Any land use covenant or deed restriction including, where applicable, a description of any institutional or engineering controls required for the site.

The Certificate of Completion will certify that:

- the site meets the applicable standards; and
- the applicant, current and future owners and occupiers and their successors, public utilities, remediation contractors, the LRS, and lenders:
 - 1) are relieved from liability to the state for the release that caused the contamination, and the state shall not bring civil, criminal or administrative action as long as the site meets the standards in effect at the time the certificate was issued, and
 - 2) shall not be subject to citizen suits with regard to the contamination that was the subject of the Voluntary Remediation Agreement.

The Certificate of Completion becomes effective upon signature by the Director or, if applicable, when any land use covenant is filed, whichever occurs last.

1.2.10.1 Certificates of Completion Issued by Licensed Remediation Specialist

Under certain circumstances, a Certificate of Completion for a completed remediation project may be issued by the LRS in charge of the site. The LRS, after assuring the site is eligible for the Voluntary Remediation Program, must issue a final report to notify the Director of his or her intent to issue a Certificate of Completion when the remediation is complete.

When the LRS is sure that the site meets the De Minimis Risk Based standards for Human Health and passes the De Minimis Ecological Screening Evaluation, he or she may issue the Certificate to the owner of the site. The Certificate of Completion will be similar to the forms used by the WVDEP (sample form available in Appendix 60-3 of the rule).

The Director may object to the issuance of a Certificate of Completion by a Licensed Remediation Specialist. Following notification by the LRS that he/she intends to issue the Certificate, the Director has 30 days to object. If the Director fails to object within this time period, the Certificate of Completion may be properly issued by the LRS.

If the Director objects to the issuance of the Certificate of Completion by the LRS, the applicant may: 1) appeal the decision to the Environmental Quality Board; 2) undertake further actions identified by the Director as necessary to cause the Certificate to be issued; or 3) terminate the Voluntary Remediation Agreement.

A Certificate of Completion issued by a LRS becomes effective when signed by the LRS (after notice to the Director as described above), or if applicable, when any land use covenant is filed, whichever occurs last.

1.2.10.2 Reopeners

The protections of the Certificate of Completion are transferable beyond the current owners. It is possible that the property, at sometime in the future, may not satisfy the obligations of the Certificate and the Voluntary Remediation Agreement. For instance, if a site received its Certificate of Completion based on achievement of risk-based standards for an industrial use, and in the future, the then current owner converts the site to residential use, the approved standards for the site are no longer being met. Under conditions such as these, the Director must begin actions to rescind the covenant as it applies to the then current owner and/or operator and must insure that the site is brought into compliance.

The Certificate of Completion also may be revoked or further remediation required if the Director determines that a reopener has been triggered. Reopeners include:

- Fraud – There is evidence that fraud was committed in regard to attainment of standards set forth in the Voluntary Remediation Agreement.
- New Information – New information confirms the existence of previously unknown contamination within the site and that contamination exceeds the standards in the Voluntary Remediation Agreement.
- Increased Level of Risk – If the level of risk at a site is significantly increased beyond the level of protection established through the Voluntary Remediation Agreement. This could occur as a result of substantial changes in exposure conditions, including changes in land use or new information about contaminants that revises exposure assumptions. Significantly increased risk means the level of risk has increased by a factor of five or the hazard index exceeds the criteria listed in Section 1.2.5 of this Guidance Manual.
- Release after July 1, 1996 – The release addressed by the Voluntary Remediation Agreement occurred after July 1, 1996, on a site that was **not** used for industrial activity before that date, and 1) the remedy selected for the remediation relied, in some aspects, on institutional or engineering controls, and 2) treatment, removal or destruction of the contaminant has now become technically and economically practical.
- Remediation methods failed to meet remediation standards – The remediation methods employed on the site failed to meet the remediation standard(s) in the Voluntary Remediation Agreement.

1.3 Interaction of the Voluntary Remediation Program With Other Environmental Programs

The VRRRA contemplates that remediation performed under it will satisfy the requirements of all environmental statutes administered by the WVDEP. It is the responsibility of the Office of Environmental Remediation, through the provisions of each VRRRA and the work plans submitted for its approval, to assure that this goal is achieved.

1.4 References

United States Environmental Protection Agency (USEPA). 1994. *Risk Assessment, Management, Communication: A Guide to Selected Sources*. EPA/749B-94-001. Office of Pollution Prevention and Toxic Substances.

Westman, W.E. 1985. *Ecology, Impact Assessment, and Environmental Planning*. John Wiley & Sons, N.Y. 532 pp.

2.0 SITE ASSESSMENT

The applicant must perform a site assessment to identify actual or potential contaminants at the site. The site assessment should provide the Director with sufficient information about the site to allow for evaluation of whether it is eligible for the program, but may not be complete enough to describe the full extent of contamination. Some applicants may prefer to perform a more extensive site assessment before filing an application while others may want to perform a more complete characterization after a voluntary agreement with the Director has been executed. The advantages to completing the site assessment after the agreement is executed are: protection against enforcement actions are in place and the applicant can seek input from the agency on the next phases of the site assessment.

At a minimum the site assessment should include the following information:

- **Site History** - A brief description of historical operations and regulatory status to enable the Director to evaluate eligibility. The site history should also describe land and water resource uses on site and in proximity to the site.
- **Conceptual Site Model** - The model is based on historical site usage and analytical data from sampling of soils, media of concern (e.g., groundwater or surface water). The conceptual model identifies actual and expected chemicals of potential concern (COPCs), the nature and extent of contamination, the pathways for migration of contamination, and the potential receptors.
- **Sampling and Analysis Plan** - The sampling and analysis methodologies and quality control procedures should be described or in the case where no sampling is planned, justification for not sampling should be provided.
- **Analytical Data** - Samples of environmental media should be collected in order to describe the physical and environmental setting and support the conceptual model. The sampling should be conducted to determine: if a release has occurred, the concentrations of COPCs in the environmental media, and the physical descriptions of the COPCs.

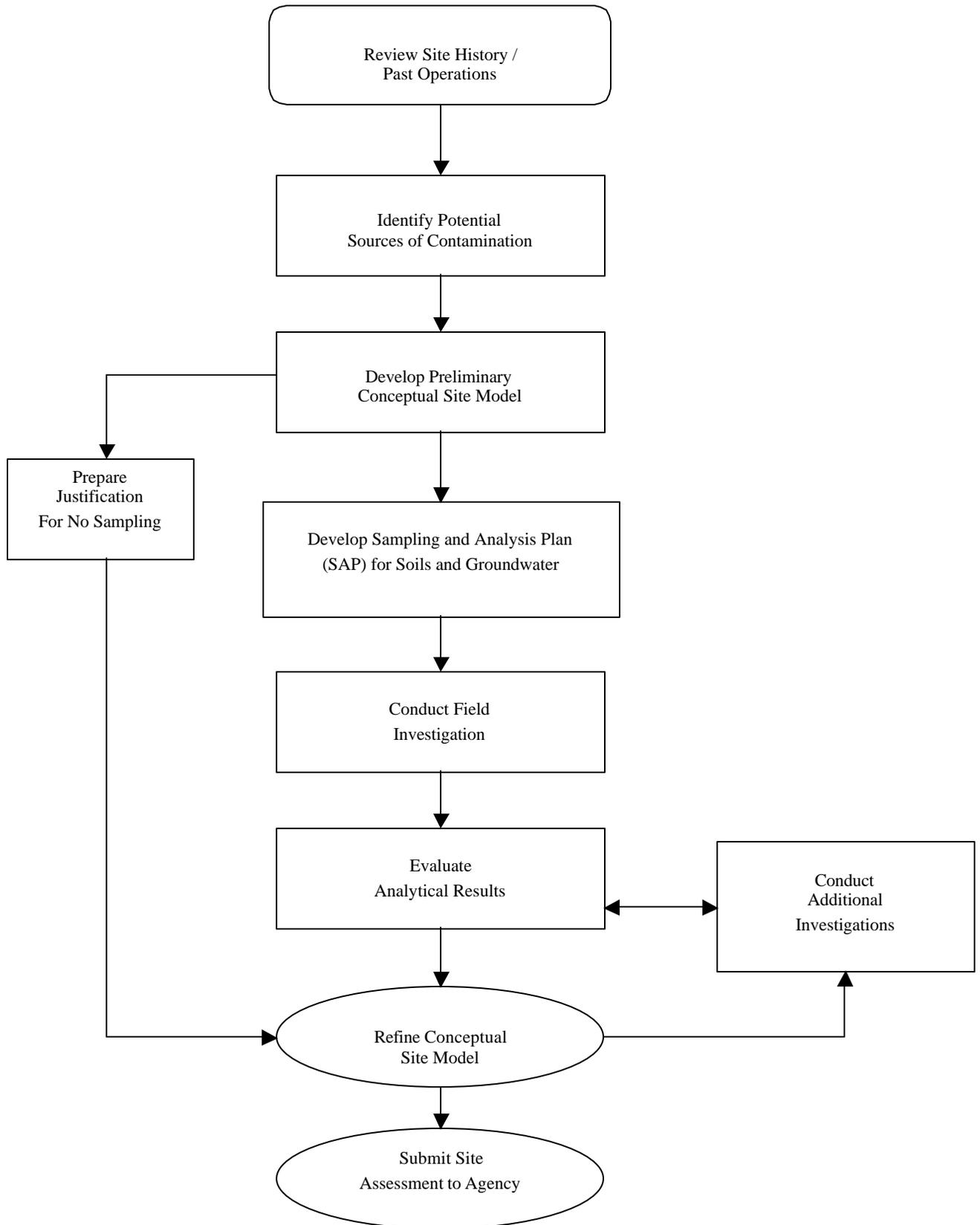
The following sections describe in more detail the components of the site assessment described above.

2.1 Site Characterization Objectives

The objectives of site characterization are as follows:

- (1) Identification of potential site-related contaminants reasonably expected to be at or near the site.
- (2) Determination of the presence or absence of those contaminants in the media of concern.
- (3) Identification of the nature and extent of contamination.

Figure 2-1: Site Assessment Process



- (4) Identification of potential pathways for contaminant migration.
- (5) Identification of the potential receptors of the contamination.

The process through which site characterization will proceed is illustrated on Figure 2-1. In many cases site characterization will be an iterative process that will continue until adequate data is developed to allow for evaluation of potential risks posed by the site and/or appropriate remedial alternatives for the site.

This section of the Guidance Manual was prepared to assist the LRS in meeting the above listed objectives. This Guidance Manual was not intended to restrict the methods by which site characterizations are performed. Rather, it was intended to provide a framework to assist the LRS in development of a work plan for a site characterization and to provide a listing of other guidance documents that may be helpful in the design of an effective site characterization program.

2.2 Preliminary Characterization

Site characterizations should generally be initiated with a literature review and brief site visit by the LRS (i.e., a preliminary characterization). The three primary areas of research during the preliminary investigation should include a review of the following: (1) information about the site history to identify the COPCs and anticipated areas where those chemicals have been handled, (2) information about the physical characteristics of the site that may influence the distribution and migration pathways of the COPCs, and (3) a listing of the potential environmental receptors and associated exposure pathways. This information will be used to develop the conceptual site model.

2.2.1 Evaluation of Historical and Current Land Uses to Identify COPCs

The scope of work for the historical investigation will depend on the nature of the property (e.g., gasoline station versus chemical plant) and the requirements of the potential property buyer/developer. The American Society for Testing and Materials (ASTM) has developed a generally accepted standard for historical research of properties titled "*ASTM E 1527-974 Standard Practice for Environmental Site Assessment: Phase I Environmental Site Assessment Process*". The ASTM procedure includes: (1) review of government databases, (2) review of historical use information, (3) interviews with people who are knowledgeable about the site, (4) interviews with government officials, (5) review of physical setting references (e.g., a United States Geological Survey [USGS] topographic map), (6) review of existing site-specific environmental data and (7) a site reconnaissance.

Government data reports can be obtained directly from the agencies or through private firms who specialize in database tracking. Examples of the federal databases that are available include, but are not limited to: the NPL, CERCLA list (CERCLIS), RCRA Treatment, Storage and Disposal (TSD) Facilities list, RCRA Generators list, and the Emergency Response Notification System (ERNS) list.

Other valuable sources of information include, but are not limited to: existing environmental investigation reports, operational maps, references in local libraries, Sanborn fire

insurance maps, chain of title information, aerial photographs, 100-year flood plain maps, and other such records.

If possible, people who are knowledgeable about the site (e.g., plant managers, former employees) should be interviewed. The interviews should focus on the identification of the COPCs and where they had been handled. Topics of discussion could include: availability of historical and current site maps, locations of former underground and/or above ground tanks, locations of underground utilities or septic systems, transformers, waste piles, drum storage areas, historical releases, compliance history, knowledge of adjacent properties, and so forth.

In addition to interviews with site personnel, it may be appropriate to contact government officials (e.g., WVDEP, fire department, local city engineer, USEPA). The government officials may have knowledge or records of accidents and/or the compliance history of the site. For review of the files maintained by the WVDEP, you should contact the **Public Information Office at (304) 759-0515**.

One or more site visits should be performed by the LRS or authorized representative to: confirm the accuracy of available mapping, to confirm information obtained during the historical reviews and interviews, and to look for visual evidence of potential contamination sources (e.g., stained soils, fill/vent pipes from underground storage tanks (USTs), stressed vegetation, drums, waste piles, etc.). It is beneficial to have a knowledgeable former site employee participate in the site visit to identify potential areas of concern to the LRS.

2.2.2 Preliminary Evaluation of Site Physical Characteristics

Requirements for the written description of the site are specified in Section 4.2.f of the Rule. The written description should include (1) the site address, (2) significant landmarks (e.g., nearby roads, surface water bodies, wetlands, buildings), (3) tax map references, (4) deed book and property number, and (5) site survey coordinates. The survey coordinates may be provided using the Universal Transverse Mercator (UTM) zone 17 NAD datum (preferred), geographic latitude and longitude, or the State Plane Coordinate System (NAD 27 datum). The site map must be accurate to within 12.2 meters (40 feet).

The site location should be shown on a large scale map (e.g., USGS 7.5 minute quadrangle) as well as a smaller scale map that shows major site features (e.g., buildings, streets, tanks, water wells, gas wells). For large and complex sites, it may be appropriate to have a surveyed topographic base map prepared for the site. For smaller sites, a simple scaled sketch may suffice.

In addition to documentation of the surficial site features, information about the subsurface conditions may need to be developed. Topics to be considered may include, but are not limited to:

- Characteristics of the site soils (grain size, permeability, ability to drain)
- Depth to bedrock

- Site stratigraphy
- Bedrock characteristics/lithology/fractures
- Significant structural features (e.g., faults, folds, sink holes)
- Depth to groundwater
- Predicted direction of groundwater flow
- Relative permeability of the site formations, aquifer, confining layers
- How water is removed from the site (e.g., evaporation, runoff, infiltration)
- Aquifer thickness
- Presence of water bodies (e.g., lakes, ponds, streams, springs, wetlands)
- Prediction of probable migration pathways for the COPCs
- Relationship between groundwater hydraulics and surface water features
- Aquifer properties (e.g., hydraulic conductivity, transmissivity)

Initially this information could be developed by review of existing investigation reports, published literature, as well as information gathered during the site reconnaissance by a geologist or qualified LRS. Published references concerning local and regional soils, geology and hydrogeology can be obtained from local libraries, the USGS, the State Geological Survey, and/or the United States Department of Agriculture (USDA).

If information about the physical characteristics of the site is not developed, the LRS should present the reasons why such an assessment was not necessary (e.g., the only COPCs are inside a building - for example asbestos insulation - and do not realistically represent a threat to the other site media).

2.2.3 Preliminary Identification of Potential Human and Ecological Receptors

A preliminary identification of potential human and ecological receptors will need to be established prior to evaluation of the risks that may be posed by the site under existing and/or future land use scenarios. The checklist in Appendix A guides the applicant in identifying receptors. This initial evaluation should consist of a literature review and site visit by the LRS or his or her representative. During the site visit, the following general items should be described:

- Current and likely potential future land uses
- Site structures, including access restrictions (e.g., fencing, locked gates, natural barriers)

- Any visible signs of trespassers
- Approximate percentage of grass/exposed soil areas, paved areas, and buildings
- Location, distance to, and description of on-site and adjacent water bodies (e.g., streams, rivers, lakes, swamps)
- Surface soil type (e.g., sandy, silty, exposed or vegetated)
- Visible signs of contamination (e.g., stained soils, stressed vegetation)
- Potential migration pathways off-site and/or to sensitive environments (e.g., drainage patterns, topography)
- Observed flora and fauna

More thorough investigations (e.g., wetland delineations) may be required if identified receptors potentially may be impacted by site activities.

The following two sections present items that are specific to the identification of potential human and ecological receptors. After this information is gathered, it is important to combine this data with information collected following the procedures described in Sections 2.2.1 and 2.2.2 to determine if contaminant migration pathways to the receptors or sensitive environments are possible. If migration to a receptor or sensitive environment is possible, then a study (e.g., sampling and/or risk assessment) may need to be conducted to further evaluate potential impacts to the receptor of potential concern.

2.2.3.1 Human Receptors

In addition to the general items presented above, specific items related to human receptors should be researched. Some of the specific items to be evaluated are as follows:

- Describe the current and reasonably foreseeable future use of the site (e.g., residential, commercial, industrial). This will be used to estimate the amount of time human populations (including adults and children) are currently present on site and potentially may be present on site. In addition, describe the current use of the area surrounding the site and closest off-site human receptor(s), and sensitive populations (e.g., schools, hospitals, retirement homes).
- Identify the source for the local drinking water supply (e.g., groundwater, springs, and surface water). Also identify if the drinking water supply is public, private, or both (i.e., some houses that are connected to public water supplies also may have a private well). This information may be obtained from the local water authority, health department, state and/or USGS. A door-

to-door well survey may be necessary in some instances (e.g., if contaminated groundwater is migrating off site towards potential human receptors).

- Identify any known or anticipated recreational activities (e.g., recreational fields, playgrounds, fishing, swimming, boating) that may result in an increased potential for human exposure.
- Identify the location of septic tanks and leach fields and note proximity to water wells.

The above data can be investigated during the site visit, through a review of the zoning records, conversations with residents, and correspondence with the appropriate state or local government offices. In addition, this information may be available from previous investigations for the site or surrounding areas, USGS topographic maps, and other literature/maps.

2.2.3.2 Ecological Receptors of Concern

Ecological receptors of concern are defined as specific ecological communities, populations, or individual organisms protected by federal, state or local laws and regulations or those local populations which provide important natural or economic resources, functions, and values. The ecological assessment portion of this preliminary evaluation consists primarily of a literature review and a site visit to determine the presence or absence of ecological receptors of concern that potentially may be impacted by contaminants originating from the site. Significant ecological investigations (e.g., fish surveys, endangered species surveys, jurisdictional wetland delineations) are not anticipated to be conducted as part of this preliminary evaluation.

More detailed ecological investigations may be required if the preliminary evaluation concludes that there may be impacts to potential ecological receptors of concern. The presence or absence of the valued environments listed below should be considered when identifying potential ecological receptors of concern since they may represent habitat for populations which provide important natural or economic resources, functions, and values. More detail is provided in Subsection 4.1.4 of this Guidance for identifying ecological receptors of concern.

- Surface water bodies or wetlands that function as feeding, breeding, nesting, resting, or wintering habitat for migratory waterfowl or other aquatic birds
- Surface water bodies or wetlands that function as spawning or nursery areas critical for the maintenance of fish/shellfish species
- Critical habitat for federal or state designated threatened, endangered, or otherwise protected species as defined in 50 Code of Federal Regulations (CFR) 424.02
- Habitat known to be used or potentially used by federal or state designated threatened, endangered, or otherwise protected species

- Area designated as a National Preserve
- Federal land designated for protection of natural ecosystems
- Designated or administratively proposed Federal Wilderness Area
- National or State Parks or Forests
- National or State Wildlife Refuges or other wildlife management areas
- State-designated natural area
- Climax community (the final successional stage of constant species composition such as an old growth forest)
- Areas important to the maintenance of unique biotic communities
- Critical areas identified under the Clean Lakes Program
- Federal or State scenic or wild river
- Trout-stocked streams or wild trout streams with verified trout production
- Federal or State fish hatcheries
- National river reach designated as recreational
- Breeding areas for dense aggregations of birds, mammals, amphibians, or reptiles
- Other Federal, State, or local-Designated Critical Biological Resource Areas or Conservation Areas
- Other valued environments not listed above that may be noted during the site visit or literature search.

During this preliminary evaluation, most of this information may be obtained by contacting the appropriate federal or state agency. The following agencies may be contacted to obtain some of the above information. It should be noted that this list is not inclusive of all agencies that may be able to provide information about valued environments.

- WV Division of Natural Resources (WVDNR), Charleston, WV (<http://www.state.wv.us>)
- WVDEP, Nitro, WV (<http://www.state.wv.us>)
- WVDNR, Natural Heritage Program, Elkins, WV (<http://www.abi.org/nhp/us/wv/index.html>)
- U.S. Fish and Wildlife Service, Elkins, WV (<http://www.fws.gov>)
- National Park Service, Washington DC (<http://www.nps.gov>)
- Nature Conservancy, Washington DC (<http://www.tnc.org>)

A site visit is necessary to initially evaluate the presence of items that may not be identified in the literature (e.g., small water bodies, or wetlands). The site visit also is important to better identify potential contaminant migration pathways from the site to ecological receptors of concern and valued environments. Consideration should be given to development restrictions and/or permits that may need to be obtained if site activities potentially could impact valued environments (e.g., development in wetland areas, alteration of endangered species habitat). Refer to Subsection 4.1.4 for details on ecological site characterization and management goals.

2.2.4 Develop a Conceptual Site Model

The conceptual site model will be used for development of the sampling program, risk evaluation, and remedial design. Because of the model's importance to all aspects of the project, it should be developed early in the project. The purpose of the model is to provide a visual representation of and to identify the following:

- Anticipated contaminants (e.g., volatile organics, metals, pesticides, explosives, petroleum)
- Primary and secondary source areas (e.g., residual chemicals in abandoned tanks, lagoons, sumps, contaminated soils)
- The release mechanism (e.g., leaking tanks, infiltration of precipitation through contaminated soils)
- Potential migration pathway(s) (e.g., groundwater, wind blown, river transport, utility conduits)
- Anticipated media of concern (e.g., soil, groundwater, surface water, sediments, air, building materials)

- Potential exposure pathways (e.g., leaching, percolation, groundwater transport)
- Potential receptor(s) of concern (e.g., humans, fish, birds)
- Ecological interactions

The conceptual site model is developed based on:

- Historical information about former site activities (particularly with respect to the handling and management of chemicals)
- Information about the physical and chemical characteristics of the media of potential concern, that will influence the distribution and migration pathways of the contaminants of concern
- A listing of the environmental receptors of potential concern

Note: It is important that the conceptual site model initially include all sources, media and exposure pathways that are of reasonable or at least plausible concern, now or in the future.

As the investigation proceeds and additional data are generated, the conceptual site model will be refined. At that point certain source areas and pathways can be excluded from the model, as appropriate. It is essential that when such exclusions occur that the rationale is documented in the text of the conceptual site model. Also the model text should clearly indicate which pathways of exposure were quantified in the risk assessment. In an adequately developed site conceptual model, a reviewer can easily determine which pathways have been addressed in the quantitative portions of the risk assessment and which have been addressed qualitatively.

It may be helpful to illustrate the site conceptual model through various figures (e.g., hydrogeologic cross sections and/or pathway analysis diagrams). Figure 2-2 is an example of a pathway analysis diagram for a hypothetical site that has completed some initial phases of investigation. Figure 2-3 illustrates a more complex site, which has more data available for consideration in the conceptual site model. These illustrations can help the LRS and project team evaluate the best way to eliminate the existing potential exposure pathways and identify data gaps which will require further evaluation. The model can also be used as a communication tool for public interaction.

The LRS generally can develop a reasonable conceptual model after completion of historical and geological research about the property and after conducting a site reconnaissance. However, the conceptual model will need to be modified/updated, as more data becomes available. Site investigations may require multiple phases of sampling and testing before the model is complete enough to select design and implement an appropriate remedial strategy and/or before the risks posed by the site can be fully evaluated. However, a well-defined conceptual model will aid in acceleration of remediation and redevelopment of the site.

2.2.5 Risk Evaluation

The conceptual site model will provide the framework for evaluation of risks that may be posed by the site. The risk evaluation is an integral part of the site characterization process, and will guide what, if any, remedial actions will be taken at the site; and ultimately how and when the site can be put back into productive use. The format and complexity of the risk evaluations will depend on the site characteristics as defined in the conceptual site model. Guidance on how the risk evaluations are to be performed is provided in Chapters 3, 4, 5, 6, 7, and 8.

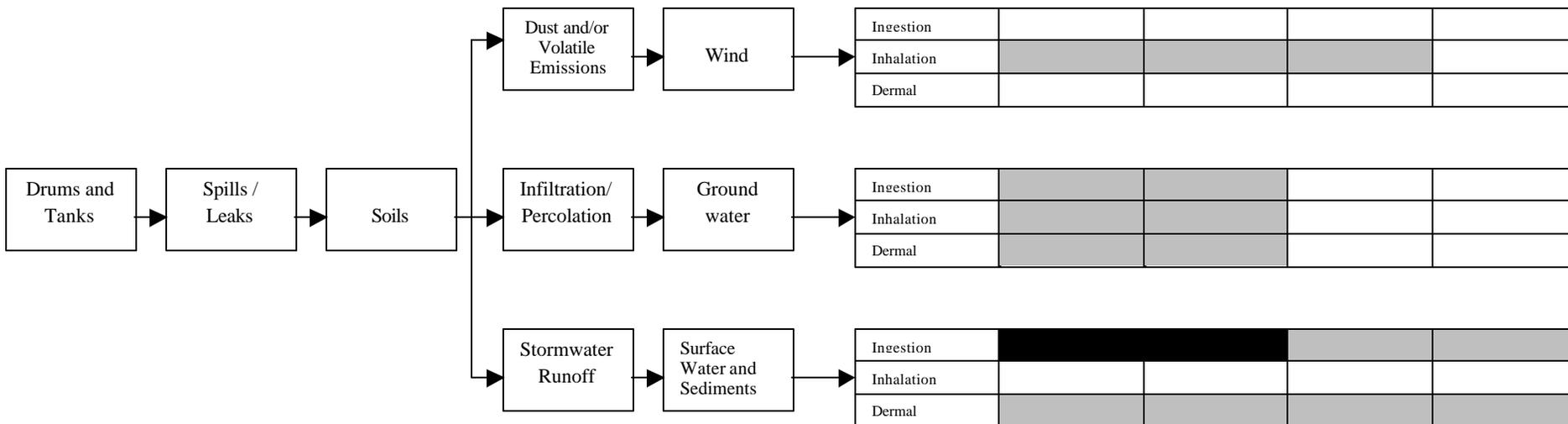
2.3 Develop Data Requirements for Sampling and Analysis Plans

The exact nature of the sampling and analysis plans (SAP) will be highly dependent on the quality and amount of analytical data available at the time a site enters the VRRRA program. In some cases, site investigations may have been completed prior to entering the program. If the investigations are complete at the time of application into the program, the Director will need adequate documentation of the work performed. Although not required, it is generally beneficial for the LRS to discuss the SAP with the WVDEP prior to implementation to avoid rework that may later be requested by the Director.

Prior to development of the SAP, the site specific data requirements will need to be determined. The primary driver for determination of the analytical data requirements will be the risk evaluation requirements. In particular, the site investigation will need to quantify the concentrations of the COPCs in the media of concern at detection levels low enough to allow for evaluation of the data for the reasonably anticipated exposure scenarios. The data may also be needed for preparation of a remedial action plan or to support contaminant transport modeling to determine if the data indicates a potential for human health, or ecological risks, and/or exceed regulatory criteria.

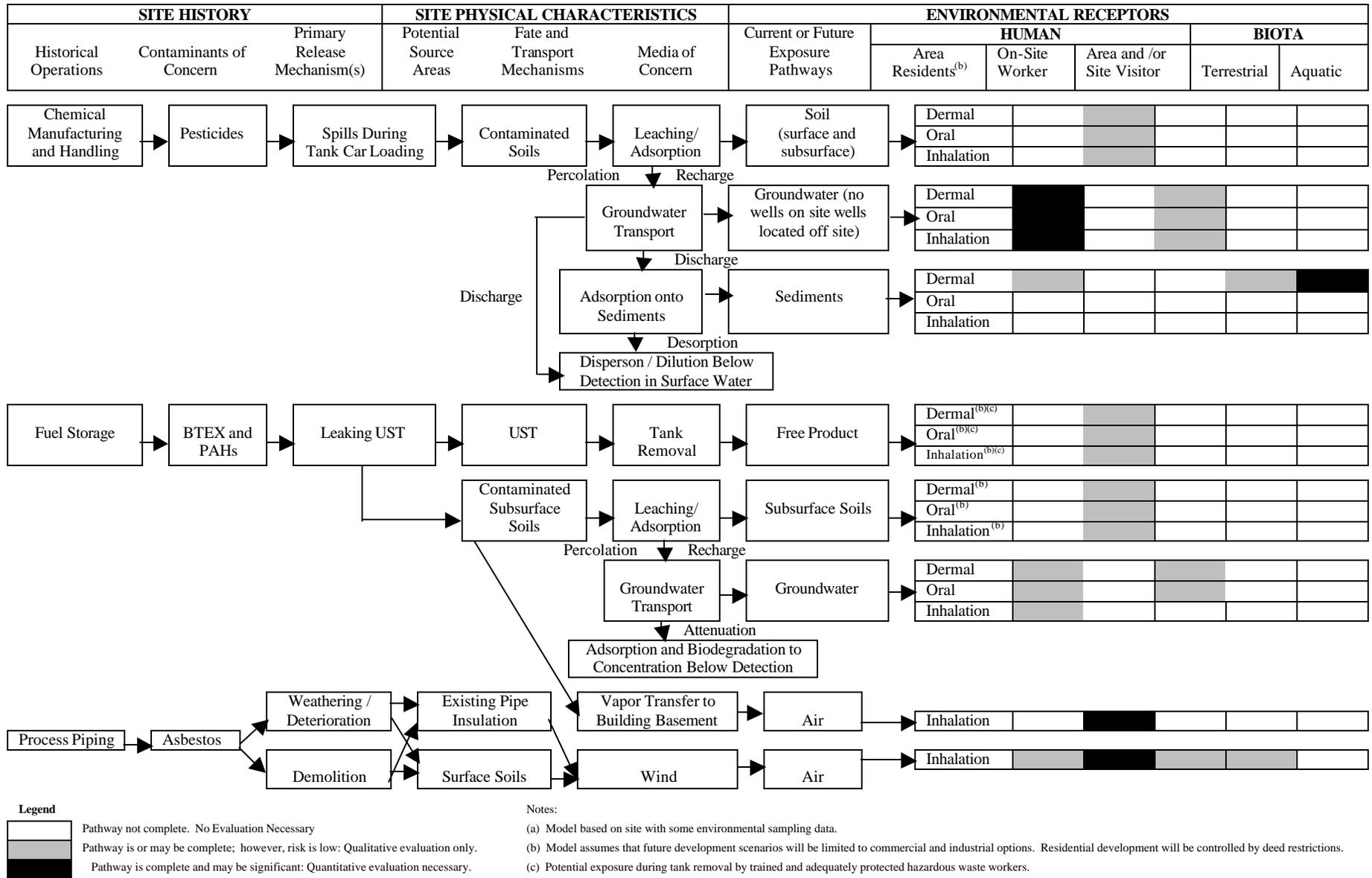
FIGURE 2-2: EXAMPLE CONCEPTUAL SITE MODEL FOR HYPOTHETICAL ABANDONED SERVICE STATION

| SITE HISTORY | | SITE PHYSICAL CHARACTERISTICS | | | | ENVIRONMENTAL RECEPTORS | | | | |
|----------------|---------------------------|-------------------------------|-------------------|---------|---------|-------------------------|----------------|--------------|-------------|---------|
| Primary Source | Primary Release Mechanism | Secondary Mechanism | Secondary Release | Pathway | Sources | HUMAN | | | BIOTA | |
| | | | | | | Exposure Route | Area Residents | Site Workers | Terrestrial | Aquatic |



- Pathway not complete.
- Pathway is or may be complete; however, risk is low.
- Pathway is complete and may be significant.

FIGURE 2-3: EXAMPLE CONCEPTUAL SITE MODEL FOR HYPOTHETICAL ABANDONED INDUSTRIAL RIVER FRONT PROPERTY^(a)



The following sections describe some of the data considerations for risk assessment, remedial design and modeling.

2.3.1 Risk Assessment Data Requirements

Risk assessments should be performed in general accordance with Chapters 3 to 8 of this guidance. A comparison to De Minimis standards, uniform risk-based standards or site-specific risk-based standards may be conducted. Typical data required to perform the risk evaluation are as follows:

- Field investigation data (e.g., source testing, media sampling), especially with respect to:
 - Background constituent concentrations by media (either naturally occurring or anthropogenic concentrations)
 - Site-specific constituent concentrations by media
 - Quantification of present and future exposures (e.g., exposure pathways; present and potential future land use; media that are or may be contaminated; locations of actual and potential exposure; and present concentrations at appropriate exposure points)
 - Potential receptor information
 - Type and duration of possible exposures (e.g., chronic, intermittent)
 - Environmental setting (e.g., climate, geology, hydrogeology, topography, nearby surface water)
- Appropriate data for statistical analysis (e.g., sufficient data to satisfy concerns about distributions of sampling data and statistics)
- Exposure assumptions (e.g., exposure duration, typical body weight, ingestion rate)
- Data needs for fate and transport models (see Section 2.3.3)
- Data evaluation/usability, especially with respect to
 - COPCs
 - Analytical quantification levels
- Data validation of analytical data

- Toxicity information (e.g., reference dose [RfD], reference concentration [RfC], cancer slope factor [CSF])

Further details on the data requirements for risk assessment may be found in the references listed below:

- USEPA. 1989. Risk Assessment Guidance for Superfund Volume I. Human Health Evaluation Manual (Part A) Interim Final. Office of Solid Waste and Emergency Response (OSWER). Washington, D.C. EPA/540/1-89-002. March 1989.
- USEPA. 1992. Guidance for Data Usability in Risk Assessment (Part A) Final. Office of Emergency and Remedial Response. Washington, D.C. Publication 9285.7-09A. April 1992.
- USEPA. 1989. Risk Assessment Guidance for Superfund, Volume II: Environmental Evaluation. OSWER. Washington, D.C. EPA/540/1-89-001. March 1989.
- USEPA. 1995. Land Use in the CERCLA Remedy Selection Process. Elliott P. Laws, Asst. Administrator. OSWER Directive 9355.7-04. May 25, 1995.
- USEPA. 1997. Integrated Risk Information System On-line Database. Environmental Criteria and Assessment Office. Cincinnati, Ohio, 1997.

2.3.2 Data Requirements for Remedial Action Design (if applicable)

The objective of data collection for remedial action design is to provide the LRS with the necessary information to complete the following tasks:

- Screening of potentially applicable technologies
- Evaluation of the cost, effectiveness, and implementability of applicable technologies
- Development of the detailed design parameters associated with the scope, duration and operation of the remedial action and remediation system.

Physical and chemical characteristics of the media of concern, that require remedial action, should be compiled as early as possible. Consideration of the data requirements for technology selection and design during preparation of sampling and analysis plans can reduce sampling costs by avoiding remobilization and inefficient data collection, while expediting the evaluation of appropriate remedial technologies. Evaluation of remedial alternatives early in the site characterization process will aid in identifying data gaps that may delay or prevent remediation and site closure.

Data requirements for soils typically include the traditional engineering properties of soils, data on soil chemistry, vertical and horizontal contaminant profiles, and the overall range and diversity of contamination across the site. Analytical data requirements for water (usually groundwater) may include chemistry, oxygen demand, pH, flow volume, flow direction, and/or other parameters. Because of the turbid nature of some samples from temporary wells, questionable results for metals analysis may not be useable data for risk assessment.

Remedial technologies can be grouped into five general categories, which are: thermal, physical, chemical, biological, and stabilization. These categories and some of the specific treatment alternatives associated with these categories are briefly introduced in Section 7 of this guidance. The tables that accompany this section present in matrix form some of the characteristics of the media to be treated that can impact the selection of a particular treatment category or treatment alternative.

Table 2-1 lists soil characteristics which can be investigated during site characterization to support technology selection, with a general interpretation of the meaning of high and low values for each characteristic. Table 2-2 provides similar information for water-related treatment categories. Engineering input should be obtained on a site-specific basis prior to the preparation of sampling and analysis plans or selection of treatment alternatives.

The USEPA's Engineering Bulletin "Technology Pre-selection Data Requirements," EPA/540/S-92/009 (USEPA 1992) can be reviewed for additional information related to data requirement for remedial design. The Engineering Bulletins are series of documents that summarize the latest information available on selected treatment and site remediation technologies and related issues. Information is also available from the USEPA's "Guide for Conducting Treatability Studies under CERCLA."

Cleanup professionals who need current information on remediation technologies can access the USEPA's Cleanup Information Bulletin Board System CLU-IN online at www.clu-in.org and www.clu-in.org/remdtext.html.

2.3.3 Data Requirements for Modeling (if applicable)

The objective of a model is to predict whether or not site contaminants (pre- and/or post-remediation) will create a condition that will impact a receptor of potential concern above risk-based acceptable criteria. A conceptual site model as discussed in Section 2.2.4 is required for all sites. Other types of models can also be used to assist the LRS in selection of the most appropriate remedial design. Mathematical models are not applicable to all sites. Generally, they can best be performed after significant site-specific data have been collected.

Soil models may be used to demonstrate that residual soil contamination will not impact the quality of groundwater beneath the site above the risk-based concentrations. Groundwater models may be used for many types of demonstrations including:

- Groundwater flow modeling to illustrate that receptors will not be in the path of the existing groundwater flow or that the remedial technology will intercept the contaminant plume.
- Contaminant fate and transport modeling to illustrate that the chemicals of concern will not reach the receptors above the risk-based concentrations.
- Natural attenuation modeling to evaluate whether the chemicals of concern will be attenuated by one or more mechanisms before reaching the receptor(s).

| Table 2-1: Soil Characteristics That Assist In Treatment Technology Preselection | | | | | |
|---|-----------------------------------|-----------------|-------------------|----------------|------------|
| Characteristic | Treatment Technology Group | | | | |
| | Physical | Chemical | Biological | Thermal | S/S |
| Particle Size | H | V | V | H | H |
| Bulk Density | V | | | H | |
| Particle Density | H | | | | |
| Permeability | H | | H | | |
| Moisture Content | V | | H | L | L |
| pH and Eh | | V | V | V | |
| Humic Content | L | L | L | V | L |
| Total Organic Carbon (TOC) | | V | H | H | V |
| Biochemical Oxygen Demand (BOD) | | | H | | |
| Chemical Oxygen Demand (COD) | | H | H | | |
| Oil and Grease | V | L | | | L |
| Organic Contaminants | | | | | |
| Halogenated | V | V | L | H | L |
| Non-Halogenated | V | V | V | H | L |
| Inorganic Contaminants | | | | | |
| Volatile Metals | | V | | L | |
| Nonvolatile Metals | H | V | L | L | H |

H = higher values support preselection of technology group

L = lower values support preselection of technology group

V = effect is variable among options within a technology group

S/S = Soil Stabilization

| Table 2-2: Water Characteristics That Assist In Treatment Technology Preselection | | | | |
|--|-----------------------------------|-----------------|-------------------|----------------|
| Characteristic | Treatment Technology Group | | | |
| | Physical | Chemical | Biological | Thermal |
| pH and Eh | | V | V | V |
| Total Organic Carbon (TOC) | | V | H | H |
| Biochemical Oxygen Demand (BOD) | | | H | |
| Chemical Oxygen Demand (COD) | | H | H | |
| Oil and Grease | V | L | | |
| Suspended Solids | H | L | V | |
| Dissolved Solids | V | H | V | |
| Nitrogen and Phosphorus | | | V | |
| Acidity and Alkalinity | V | V | L | |
| Dissolved Oxygen | | | H | |
| Organic Contaminants | | | | |
| Halogenated | V | V | L | H |
| Non-Halogenated | V | V | V | H |
| Metals | V | H | L | L |

H = higher values support preselection of technology group
L = lower values support preselection of technology group
V = effect is variable among options within a technology group

There are many types of models varying in complexity from analytical models (which are essentially equations) with minimal data input requirements to comprehensive numerical models that require as input considerable data about the spatial distributions of aquifer characteristics in one or more layers.

Before the data collection (investigation) phase is performed, the LRS should determine model objectives and model type, to justify the data requirement. The data input requirements and the planned uses of the output from the anticipated model(s) should be listed and discussed in the SAP, as applicable.

When selecting a fate and transport model, it is critical that site-specific information be reviewed along with model specifications to ensure that the model is capable of simulating site conditions and contaminant properties that may have significant impact on site-specific contaminant transport.

2.3.3.1 Physical Characteristics of Each Water-Bearing Zone

Certain site-specific information should be collected during the investigation phase(s) for subsequent input into the model(s), which may include, but is not limited to:

- Depth to groundwater

- Recharge
- Saturated hydraulic conductivity (permeability) or transmissivity
- Unsaturated hydraulic conductivity
- Aquifer thickness
- Groundwater flow direction (including any off-site pumping effects)
- Hydraulic gradient (maximum and average)
- Groundwater seepage velocity
- Dispersivity
- Bulk density (saturated zone)
- Organic carbon fraction (saturated and unsaturated zones)
- Total porosity (saturated and unsaturated zones)
- Effective porosity (or specific yield)
- Cation exchange capacity
- Clay mineral content

2.3.3.2 Chemical Characteristics of the COPCs

In addition to the physical conditions of the site, the chosen model must be able to handle all contaminant-specific properties that may significantly affect fate and transport. One critical factor will be whether COPC's include organic contaminants (such as benzene or trichloroethene) or inorganic contaminants (such as lead or chromium). The most important properties affecting organic contaminant transport are compound partition coefficients [such as the Henry's Law constant and the organic-carbon partition coefficient (K_{oc})] and the amount of organic carbon in the soil. Transport of inorganic contaminants, however, is heavily influenced by soil properties such as pH, redox potential and clay content (Luckner, 1991; Tyler, 1982; Korte, 1976). Properties to consider may include:

- Horizontal and vertical extent of contamination
- Volume of release (or initial concentration near source at time of release)
- Solubility

- Acid and base hydrolysis
- Oxidation-reduction potential
- Valence state of the contaminant
- Vapor pressure
- Henry's Law constant
- K_{oc} or octanol-water partition coefficient (K_{ow})
- Degradation (daughter) products
- Degradation rates of parent and daughter products

Note that it is not necessary for the chosen model to simulate all of the above listed properties, but only those properties that are relevant to the specific site.

2.4 Developing Specific Investigation Techniques for SAPs

The data requirements, as defined in Section 2.3, will be used to identify appropriate investigation techniques to be implemented at the site. The data requirements and proposed investigation techniques will be documented in the SAP. The investigative techniques should be designed to collect sufficient data for the LRS to refine the conceptual site model until:

- Adequate data is available for remedial design and/or
- The site risks are determined to be minimal

The following subsections provide guidance for the investigation of various environmental media. The sampling program should only investigate the media of potential concern. Therefore, several of the following sections may not be applicable for a given site, depending on the nature and extent of the contaminants of concern. For example, a former gasoline station property may not need to be investigated for groundwater contamination if it can be documented that the residual soil contamination did not extend to the water table during tank removal operations, and the residual soil contamination was removed.

The use of composite samples must be consistent with the Data Quality Objective and analytical methods selected. The LRS should balance (1) performing the site investigations in a phased manner to avoid unnecessary investigations of certain media (e.g., groundwater), and (2) minimizing the number of phases of investigation by anticipation of applicable data needs for risk assessment, remedial design and modeling (as applicable) early in the site investigation process.

2.4.1 Data Quality Considerations

The data quality objectives (DQOs) for the project should be established prior to preparation of a SAP. The DQOs are based on the project objectives, key performance requirements for the data operations and conceptual sampling design. The sampling design and DQOs are used to develop the SAP, which generally consists of the Quality Assurance Project Plan (QAPP) and Field Sampling Plan (FSP). The SAP provides detailed site-specific objectives, specifications, and procedures needed to conduct a successful field investigation. The SAP specifies the sampling strategies; number, type and location of samples; and the level of quality control.

The quality of data collected during a field investigation is based on the project DQOs. Under the West Virginia VRRRA, environmental monitoring and measurement efforts must be validated. WVDEP prefers level III validation although other levels may be discussed with them on a site by site basis as well as the percentage of samples that will be validated at each level. The WVDEP must be able to verify that investigative work, risk assessment, confirmatory sampling and other remediation tasks will be conducted in a manner that will provide reliable analytical results and an accurate conceptual site model. Examples of quality requirements are as follows:

- analytical reporting limits must be at or below the cleanup criteria
- field screening techniques must include proper instrument calibration
- sample collection procedures must not impair the sample integrity

Further guidance on quality requirements may be found in the references listed below:

- USEPA 1993. EPA Quality System Requirements for Environmental Programs (Draft) EPA/QA/R-1
- USEPA 1993. EPA Requirements for Quality Assurance Project Plans for Environmental Data Operations (Draft Final) EPA/QA/R-5
- USEPA 1993. Guidance for Planning Data Collection in Support of Environmental Decision-Making Using the Data Quality Process. EPA/QA/G-4
- USEPA 1993. Guidance for Conducting Environmental Data Assessments (Draft) EPA/QA/G-9
- USEPA 1993. Data Quality Objective Process for Superfund-Interim Final Guidance. EPA 540-R-93-071
- USEPA 1998. EPA Guidance for Quality Assurance Project Plans EPAQA/G-5

The LRS generating data under the VRRRA program has the responsibility to implement procedures to assure that the precision, accuracy, completeness, and representativeness of the data are known and documented. To ensure that this responsibility is met, each LRS must prepare a SAP for each project.

Quality assurance/quality control (QA/QC) procedures will be performed in accordance with applicable professional technical standards, West Virginia requirements, government regulations and guidelines, and specific project goals. QA/QC procedures are required for both on site analyses (e.g., field screening, pH, specific conductance) and off site analyses. The level of the QA/QC shall be based on the project DQOs. Samples collected during VRRRA activities are to be logged on a chain-of-custody form. The following QC samples are generally applicable to VRRRA fieldwork:

- Field duplicate samples
- Equipment and trip blank samples

Split samples may also be appropriate at the discretion of WVDEP.

At least ten percent of the analytical data or some other percentage agreed to by the WVDEP must be validated. Standard USEPA protocols for validation (e.g., Contact Laboratory Protocol or SW-846) should be used. However, these protocols may be modified with the director's approval, depending on the type of analyses performed and DQOs. In some cases, data from previous non-validated investigations may be utilized in the site assessment. However, new validated data must be generated to substantiate the findings of the earlier studies.

2.4.2 Selection of Analytical Methods

Routine analytical services used for projects under the VRRRA should use USEPA or other approved methods such as those listed in:

- USEPA, 1983. Methods of Chemical Analysis for Water and Waste, EPA 600/4-79-020, 1983 rev.
- USEPA, 1986. Test Methods for Evaluating Solid Waste, Office of Solid Waste and Emergency Response, Washington, DC, November 1986 revised January 1995, SW-846 Third Edition.

Non-standard methods must be approved by the Director.

At a minimum, a description of the analytical method, QA/QC requirements, and detection limits should be provided for review in all work plans and the Final Report. It would also be prudent to summarize other analytical requirements such as sample containers, and preservation techniques for the benefit of the field sampling team. A table showing this information by media and sample location will facilitate review. All required QA/QC, as specified in the analytical method, should be implemented during the analysis unless the

laboratory can demonstrate that modifications to the method provide better results. The QA/QC information to be reported is based on the DQOs for the parameter.

2.4.3 Health and Safety Considerations

The Occupational Safety and Health Administration's (OSHA) Hazardous Waste Operations and Emergency Response (HAZWOPER) standard is applicable to VRRRA site investigations. OSHA has adopted the HAZWOPER standard under its General Industry Standards (29 CFR 1910.120) and Construction Industry Standards (29 CFR 1926.65). A major component of this standard that will effect VRRRA site investigations is the requirement for development of a site or project-specific health and safety plan (HASP). The HASP must consist of the following elements:

- Safety and health hazard analysis by task
- Employee training requirements (e.g., 40-hour initial training and 8-hour refreshers)
- Personal protective equipment (PPE) requirements by task
- Medical surveillance
- Air and personnel monitoring
- Site control program
- Decontamination
- Emergency response plan
- Confined space entry procedures (if applicable)
- Spill containment

The specifics of the elements listed above will vary for each site investigation based upon the site conditions and the planned activities. Other OSHA regulations, in addition to the HAZWOPER standard, will need to be addressed, such as: the permissible exposure limits (PEL) of air contaminants, chemical-specific standards, respiratory protection program, lockout/tagout procedures, and proper excavation procedures.

2.4.4 Surface and Subsurface Soils

Surface soil and subsurface soil characterizations are primarily performed to obtain information for determining whether the soil has been impacted (contaminated) due to past chemical handling activities at the site. Various investigative techniques can be used for characterizing soil. These techniques include both intrusive techniques such as conventional

drilling, and non-intrusive techniques such as geophysical methods. Some techniques provide qualitative data that can be used to first identify an area(s) of concern. Once the areas of concern are identified, other investigation techniques (e.g., soil borings or test pits) may be necessary to collect quantitative sample data for characterizing the area.

The chemical analysis of the soil samples provides quantitative data that can be used to determine what contaminants are in the soil [e.g., metals, polychlorinated biphenyls (PCBs), solvent constituents, petroleum constituents] and the horizontal and vertical extent of contamination. The soil sample data is used to assess risk to human health under current as well as future exposure scenarios. Potential impacts to groundwater and ecological receptors can also be assessed using this data.

Physical testing of the soil (e.g., grain size analysis, compaction properties) and identification of soil types (e.g., clays, sands, loams, fill) can be performed to obtain properties that may be useful in evaluating various treatment or containment alternatives. The physical properties of the soil can also be used for determining the fate and transport potential for various contaminant types.

Surface and subsurface soil characterizations can be conducted in a variety of ways depending on site conditions, the end use of the data, cost factors, level of quality, and the level of accuracy required. In general, surface soil is defined as the top 2 feet of soil. The applicant should be careful not to dilute contaminants confined to the uppermost surface layers by sampling across too great a depth. This “layer” of soil is sampled primarily to assess human health impacts via direct exposure to the soil. Surface soil chemical data can also be used to “locate” the area of contamination and to assess the overall horizontal extent of contamination once a “hot” area is located. Subsurface soil is defined in the Rule as soil below two-feet in depth. The soil above the water table is referred to as the vadose zone. Data collected from subsurface soils can be used to assess the horizontal and vertical extent of contamination, to evaluate human health risks due to exposure via construction, or to identify “hot” zones that may serve as a source of groundwater contamination.

The remainder of this section focuses on the specific techniques and their potential applicability to performing site characterizations. The following references provide general guidance for characterizing soils:

- ASTM. 1987, Standard Guide for Investigating Soil and Rock (D-420-97) (Vol. 4.08)
- ASTM, Site Characterization - Environmental Purposes With Emphasis on Soil/Rock/Vadose Zone/Groundwater (D-5730)
- USEPA. 1991. Subsurface Characterization for Subsurface Remediation. EPA/625/4-91/026.
- USEPA. 1991. Description and Sampling of Contaminated Soils: A Field Pocket Guide. EPA/625/12-91/002

- USEPA. 1992. Preparation of Soil Sampling Protocol: Techniques and Strategies NTIS PB-92220532

2.4.4.1 Non-Intrusive Characterization Techniques

There are several remote sensing methods that can be employed for site characterizations. These techniques include visible photography, infrared photography, and thermal infrared scanning. In most cases, remote sensing techniques are used to identify changes in land use, determine groundwater preferential flow pathways, and detecting near surface leachate/contamination.

Surface geophysical techniques are usually employed in the initial stages of the field program for locating subsurface anomalies (e.g., drums, debris, and pipelines) or characterizing the geology or contaminant plumes. The most routinely used techniques include ground penetrating radar, electromagnetic induction, electrical resistivity, seismic refraction, metal detection, and magnetometry.

The following guidance documents provide information on remote sensing and surface geophysical methods, and focus on the usability/limitations of each technique:

- USEPA. 1993. Subsurface Characterization and Monitoring Techniques. EPA/625/R-93/003a.
- USEPA. 1984. Geophysical Techniques for Sensing Buried Wastes and Waste Migration. EPA/600/7-84/064.
- USEPA. 1993. Use of Airborne, Surface, and Borehole Geophysical Techniques at Contaminated Sites: A Reference Guide. EPA/625/R-92/007.

2.4.4.2 Field Screening and Field Analytical Characterization Techniques

Field screening methods provide an indication of the presence or absence of a particular type of contamination in site media (primarily soil and groundwater) based on a threshold level for a given technique. Screening methods provide relative concentrations for chemical classes, but not usually accurate chemical specific concentrations. In most cases, field-screening techniques are performed during the initial phase of the site characterization to confirm suspected areas of concern, help locate an area of concern, or identify soil samples that may be contaminated.

In most cases, field-screening techniques are limited to volatile contamination, although field screening can also be performed for other suites of compound [e.g., PCBs, polynuclear aromatic hydrocarbons (PAHs), metals, pesticides]. Field analytical methods include all chemical analysis methods capable of providing chemical-specific quantitative data in the field. Field analytical techniques are usually more rapid and less expensive than full-scale laboratory analyses. The most commonly used field screening and analytical techniques are: headspace

screening of soil samples, soil gas surveys, field immunoassay test kits and X-Ray Fluorescence (XRF).

A soil gas survey is designed to characterize subsurface soil (and groundwater) contamination. Because the technique involves the testing of vapors from the soil, the technique is primarily suited for characterizing volatile organic compounds such as solvents and some components of petroleum products. The sampling operation is relatively quick and produces a small diameter boring (usually only a few feet in depth). The samples may be collected quickly by vacuum/suction, or through the use of passive absorbent media, that is left in the boring for a few days. The soil gas samples may be analyzed in the field using a gas chromatograph or submitted to a qualified laboratory to assess the presence of specific contaminants (e.g., benzene, toluene, ethylene, xylene, (BTEX), trichloroethene (TCE), tetrachloroethene (PCE)). The results are usually plotted on isoconcentration maps. By producing the data in a rapid format, field decisions can be made with respect to delineation of contaminants during the initial phase of investigation.

Field test kits are used for on-site detection of contaminants. The test kits offer reasonably accurate results within a relatively short period of time. The tests are analyte-specific, and sensitive to levels necessary for regulatory compliance. Test systems can be purchased for characterizing PCBs, total petroleum hydrocarbons (TPH), PAHs, pentachlorophenol (PCP), trinitrotoluene (TNT) and other chemicals in soil.

The following references provide additional information with respect to field screening and analytical techniques:

- USEPA. 1987. A Compendium of Superfund Field Operations Methods, Part 2. EPA/540/P-87/001 (OSWER Directive 9355.0-14).
- USEPA. 1988. Field Screening Methods Catalog: User's Guide. EPA/540/2-88/005.
- USEPA. 1991. Second International Symposium, Field Screening Methods for Hazardous Waste and Toxic Chemicals, EPA/600/9-91/028.

2.4.4.3 Intrusive Characterization Techniques

Intrusive characterization techniques are required to obtain surface or subsurface soil samples. Intrusive characterization techniques primarily include test pitting, drilling, direct push, and hand held methods.

Test pitting offers the advantage of visually inspecting subsurface features and debris which may be contained under the ground surface. However, test pitting is limited to a depth of approximately 15 to 20 feet or until the water table is encountered. Test pitting is performed using a conventional backhoe.

Subsurface drilling is required to characterize subsurface soil and bedrock conditions, and to install piezometers and monitoring wells. Subsurface drilling provides precise detail with

respect to sampling depths and soil/geologic characterization. Drilling methods are selected based on the following: availability and cost; suitability for the type of geologic conditions at a site; and potential effects on sample integrity. A wide variety of drilling methods have been developed based on various needs and site conditions. The hollow-stem auger method is one of the most commonly used methods for drilling in unconsolidated deposits, whereas air rotary is widely employed when drilling in consolidated deposits. The following references provide additional information pertaining to drilling and soil sampling methods:

- Aller, Linda, et al. 1989. Handbook of Suggested Practices for the Design and Installation of Ground-Water Monitoring Wells. National Water Well Association.
- USEPA. 1993. Subsurface Characterization and Monitoring Techniques - A Desk Reference Guide - Volumes 1 and 2. EPA/625/R-93/003a&b.
- ASTM. 1993. Standard Guide for Investigating and Sampling Soil and Rock. D420-93, (Vol. 4.08).
- ASTM. 1991. Guide for Soil Sampling From the Vadose Zone. D4700-91. (Vol. 4.08).
- ASTM. 1993. Draft Standard Guide for the Use of Hollow-Stem Augers for Geoenvironmental Exploration and Installation of Subsurface Water-Quality Monitoring Devices. D18.21 Ballot 93-03, April 28, 1993.
- ASTM. 1993. Draft Standard Guide for the Use of Direct Rotary Drilling for Geoenvironmental Exploration and Installation of Subsurface Water-Quality Monitoring Devices. D18.21 Ballot 93-03, April 28, 1993.
- ASTM. 1993. Draft Standard Guide for the Use of Air-Rotary Drilling for Geoenvironmental Exploration and Installation of Subsurface Water-Quality Monitoring Devices. D18.21 Ballot 93-03, April 28, 1993.
- ASTM. 1995. Standard Practice for Soil Investigation and Sampling by Auger Borings. D1452-80, (Vol. 4.08) - Reapproved 1995.
- ASTM. 1993 Practice for Diamond Core Drilling for Site Investigation D-2113-83 (Vol. 4.08)- Reapproved 1993.
- ASTM. 1983. Standard Practice for Thin-Walled Tube Sampling of Soils. D1587-94, (Vol. 4.08).
- ASTM. 1992. Method for Penetration Test and Split-barrel Sampling of Soils D-1586 -84 (Vol. 4.08) - Reapproved 1992.

Direct push technology (such as cone penetrometers, Geoprobe^R, and other trade names) is used to collect lithologic data and/or soil samples for chemical analyses. The advantage of

direct push technology is that it takes less time than conventional drilling and is less expensive. In addition, this technique results in less investigation-derived waste (IDW). The disadvantage of the direct push technology is that it cannot penetrate rock or difficult geologic conditions and is limited in depth relative to other drilling methods. The following references provide additional information on the use of this sampling technique:

- ASTM. 1986. Standard Test Method for Deep, Quasi-Static, Cone and Friction-Cone Penetration Tests of Soil. D3441-86. (Vol. 4.08).
- Christy, T.M. and S.C. Spradlin. 1992. The Use of Small Diameter Probing Equipment for Contaminated Site Investigations. Groundwater Management 11:87-101 (6th NOAC).
- Chiang, C.Y. et al., Characterization of Groundwater and Soil Conditions by Cone Penetrometry. In: Proceedings (6th) National Water Works Association (NWWA)/American Petroleum Institute (API) Conference, Dublin, Ohio. pp. 175-189.

Hand-held sampling techniques include the use of scoops, shovels, and augers. Scoops and shovels are used in cases where the purpose of the sampling is to obtain surface soil samples (top 6 to 12 inches only). Hand or power augering is quick and less expensive than the other methods, but the technique is limited to the depth in which samples can be collected and geologic conditions. Normally, sampling to shallow depths such as one to two feet can be accomplished via hand augering. With the use of a power auger, the depth will be greater. Surface and shallow subsurface soil sampling is usually performed via hand or power augering. Additional information can be obtained from the following references:

- USEPA. 1987. A Compendium of Superfund Field Operations Methods, Part 2. EPA/540/P-87/001 (OSWER Directive 9355.0-14).
- USEPA. 1991. Description and Sampling of Contaminated Soils: A Field Pocket Guide. EPA/625/12-91/002.

2.4.5 Storm Water Runoff

Storm water runoff may be a potential contaminant transport mechanism on some VRRRA sites. Contaminated soils, contaminated trenches or drainage swales, storm sewers or other conveyance structures may leach contaminants into storm water that may cause an expansion of the contaminated area of concern and/or contribute contaminants to a surface water body. Structures, depressions or ditches may have been used to convey process water when the property was in active use. Although process waters may no longer be conveyed through drainage avenues, remnant contamination from those historical operations may be present and may be leaching back into the storm water. Therefore, it may be appropriate to sample storm water that passes through potentially contaminated areas.

As a result of the National Pollutant Discharge Elimination System (NPDES) which was authorized by the Clean Water Act (CWA), many industrial facilities were required to prepare

and implement a storm water pollution prevention plan by October 1, 1993. West Virginia is an NPDES-delegated state and therefore, administers this federal program within the state. A review of existing information on file at the WVDEP under this program could eliminate the need for sampling or for identifying potential contaminants of concern with respect to storm water or process water drainage systems.

Typical storm water sampling under NPDES generally includes eight pollutant parameters: oil and grease, biological oxygen demand (BOD), chemical oxygen demand (COD), total suspended solids (TSS), nitrate, nitrite nitrogen, kjeldahl nitrogen, total phosphorus, pH, and specific pollutants of concern at the site. These parameters may be appropriate to include for sites where the future land use would likely require an NPDES permit.

Sampling protocol for storm water generally requires that:

- sampling begins at a predetermined 0.1-inch of rainfall, with 72 hours of dry time having elapsed from the time of the last 0.1-inch storm event.
- a grab sample be taken within 30 minutes of the onset of a storm event.
- composite sampling be conducted for 3 hours or the duration of the storm event.

The logistics of meeting these protocols may be problematic on abandoned sites where no personnel are available for timely mobilization. Therefore, it may be necessary to modify these protocols to match the site data requirements versus the logistical realities of the investigation.

Composite samples can be either flow-weighted or time-weighted. If the applicant is conducting flow-weighted composite sampling, then, the storm water discharge flow should be estimated each time a sample aliquot is collected. Common flow measurement techniques include weirs and flumes, velocity methods, volumetric methods, slope and depth methods, and runoff coefficient methods.

Sampling can be conducted either manually or with an automated monitoring system. There are many benefits to using an automated monitoring system including: enhanced safety, more accurate documentation of the storm event, enhanced data quality and reduced field man-hours. However, this approach may not be appropriate for preliminary evaluation of storm water runoff. It may be more appropriate to coordinate the storm water sampling with other site investigation activities.

The following documents are available to assist the LRS with design of a storm water-sampling program:

- USEPA. 1992. Storm Water Management for Industrial Activities (EPA 832-R-92-006).
- USEPA. 1992. NPDES Storm Water Sampling Guidance Document (EPA 833-B-92-001).

2.4.6 Site Infiltration and Vadose Zone Characteristics

Contaminants released on the land surface might travel (infiltrate) to the shallow subsurface above the water table (the vadose zone) and descend (percolate) to the water table. The relative rates of infiltration and percolation can provide an indication of the likelihood whether or not contaminants could descend to the groundwater. Additionally, contaminated soils themselves could become secondary sources of groundwater contamination.

Where soils are found to be contaminated, the information derived from the site investigations of infiltration and permeability rates would contribute to the feasibility evaluation or remedial design of soil washing, vapor extraction systems, and other remedial technologies.

The initial qualitative evaluation of vadose conditions can occur during a site reconnaissance (Section 2.2.2), although the reconnaissance should be supplemented by review of soil surveys (USDA - Soil Conservation Service [SCS]), the relevant County Report by the State Geological Survey; other similar sources of information on infiltration, percolation and recharge potentials; or site-specific information.

Should further site investigations become advisable, field tests of infiltration rate should be scheduled during other site activities, such as well sampling or drilling activities. Other tests, such as soil-moisture tension (lysimetry) or isotherm quantification (calculating the rates of adsorption and remobilization of a particular chemical by the site soils), should only be scheduled if warranted by the requirements of the risk assessment or remedial design.

The appropriate field methods for permeability testing of the vadose zone, either at land-surface or in a borehole, are found in the ASTM and USDA-SCS references. Methods for laboratory testing of consolidated and unconsolidated materials should follow the appropriate ASTM method. The following provides references for some of the field methods that may be selected for this investigation:

- ASTM. 1990. Test Method for Measurement of Hydraulic Conductivity of Saturated Porous Materials Using a Flexible Wall Permeometer. D5084-90 (Vol. 04.09).
- ASTM. 1991. Guide for Soil Sampling from the Vadose Zone. D4700-91 (Vol. 4.08).
- ASTM. 1994. Practice for Thin-Walled Tube Sampling of Soils. D1587-94, (Vol. 4.08).
- ASTM. 1994. Test Method for Infiltration Rate of Soils (in Field) Using Double Ring Infiltrometer. D-3385-94 (Vol. 04.08).

2.4.7 Groundwater

Groundwater characterizations are performed when there is a potential for leaching/percolation of contaminants through the site soils into the uppermost water-bearing

zone (and potentially into deeper zones). The primary objective of a groundwater investigation is to determine whether the concentrations of COPCs exceed regulatory limits as specified under West Virginia's Requirements Governing Groundwater Standards (Legislative Rule 46CSR12) or other risk-based standards. A second objective of the groundwater investigation is to determine the vertical and horizontal extent and magnitude of COPCs and the potential threat to human and ecological receptors. A third objective of the groundwater characterization is to evaluate and quantify site hydrogeologic conditions that will govern the COPC fate and transport. Groundwater investigations are particularly important if the water-bearing zone is an aquifer or is hydraulically connected to an aquifer that is used as a source of drinking water.

Groundwater characterization may not need to be performed during the initial phases of investigation if it is considered an unlikely media of concern. For example, groundwater may not need to be investigated if only surface soils are contaminated and the uppermost aquifer is known to be deep (e.g., 100 feet) and subsurface soils consisting of clays and silts. However, a groundwater investigation (that was initially considered unnecessary) will need to be conducted during later phases of investigation if soil sampling indicates that the initial conceptual model was in error, and that significant contaminant leaching/percolation has occurred.

Data generated for groundwater characterizations may be collected during a single or over several phases. An example-phased investigation could be as follows:

1. Installation of temporary or permanent groundwater sampling points (e.g., temporary direct push borings, well points, monitoring wells, extraction wells)
2. Collection of groundwater samples to determine the presence/absence of the COPCs.
3. Collection of data necessary for estimation of groundwater flow rate and transport mechanisms (e.g., groundwater elevation data, porosity estimates, hydraulic conductivity estimates, definition of aquifer boundaries)
4. Installation of additional wells and collection of additional samples, to determine the extent of a plume, to better evaluate remedial alternatives, or for groundwater model calibration.

In addition to the standard investigation techniques, other investigation techniques may be employed to collect groundwater data (e.g., surface geophysics, borehole geophysics, soil gas surveys, remote sensing, tracers).

Factors that would impact the level of effort for completion of the groundwater characterization may include the following:

1. Concentrations of the identified COPCs relative to the risk based standards
2. Presence of non-aqueous phase liquids (NAPLs)
3. Complexity of site hydrogeologic conditions (e.g., non-stratified vs. stratified (alluvium, colluvium, fractured bedrock, fill material))

4. Point source vs. non-point source release mechanism for the COPC
5. Chemical properties of the COPC (e.g., solubility, K_{OC} , density, vapor pressure)
6. Attenuation processes
7. Location of human and ecological receptors
8. On-site and off-site wells
9. Facility structures and utilities (e.g., preferential migration pathways)
10. Other site-specific factors

2.4.7.1 Well Installation and Groundwater Quality Investigation

Groundwater sampling points can be established using a variety of temporary or permanent wells (such as, temporary wells installed via direct push technologies, well points, monitoring wells, extraction wells). Additionally, springs and seeps may be used as sampling points, since they typically represent zones of preferred groundwater migration. The number of sampling points necessary to adequately characterize a site is to be based on the site-specific characteristics.

Monitoring wells must be designed and installed in accordance with West Virginia's regulatory requirements as defined in the Monitoring Well Regulations (47CSR59) and the Monitoring Well Design Standards (47CSR60). The regulatory framework allows the LRS to make a selection of the specific drilling technologies and well materials to be utilized.

Note: Monitoring wells with turbid samples may not provide reliable analytical results for metals.

Selection of appropriate drilling techniques, well installation techniques, well materials, well diameter, and sampling techniques are dependent on a wide variety of site specific geologic and hydrogeologic factors as well as the characteristics of the COPC. Some of those factors could include:

- Purpose of the well (e.g., piezometer, chemical sampling, groundwater extraction, geophysical logging)
- Anticipated depth to groundwater
- Single versus multiple water bearing zones
- Physical characteristics of the site soils and/or bedrock (e.g., density, tendency to heave, formation permeability)

- Chemical characteristics of the site soils (e.g., will soils be hazardous?)
- Chemical characteristics of the site groundwater (e.g. will groundwater be corrosive to well materials?)
- Logistical constraints (e.g., location of property boundaries, steep slopes, overhead power lines)
- Other site specific characteristics

The characterization of groundwater quality and site hydraulic characteristics must be directly relevant to the COPCs found on site and must directly support the remediation standards, remedial design, and/or groundwater model. The characterization of groundwater quality should provide useful quantification of the concentrations of COPCs at individual monitoring points across the site. Laboratory methods for analysis should be selected to conform with the DQOs (Section 2.4.1 and 2.4.2) and other pertinent regulatory requirements. Consistent with the West Virginia Requirements Governing Groundwater Standards (Legislative Rule 46CSR12), compliance with risk-based concentrations for inorganic parameters will be based on dissolved phase concentrations rather than total concentrations.

Useful resources to assist the LRS in development of the groundwater quality investigation program include:

- Aller, Linda, et al. 1989. Handbook of Suggested Practices for the Design and Installation of Ground-Water Monitoring Wells. National Water Well Association.
- Driscoll, F.G. 1986. Groundwater and Wells; 2nd Ed.; Johnson Filtration Systems, Inc.; Minnesota.
- USEPA. 1993. Subsurface Characterization and Monitoring Techniques - A Desk Reference Guide - Volumes 1 and 2. EPA/625/R-93/003a & b.
- USEPA. 1987. Handbook - Groundwater. EPA/625/6-87/016
- USEPA. 1991. Handbook - Ground Water Volume II - Methodology. EPA/625/6-90/016

2.4.7.2 Characterization of Groundwater Flow

The objective of the hydrogeologic characterization is to provide a quantification of the ability of the water-bearing unit at the site to transmit water and transport contaminants to a potential receptor. The level of detail required for evaluation of the site hydrogeological investigation will vary depending on the data requirements for risk assessment, remedial design and/or modeling. The characterization of groundwater flow usually proceeds from the simplest method to more complex methods. Listed below are some of the techniques available to the LRS for quantifying the site-specific hydrologic properties.

The usual methods of hydrologic characterization are:

- **Potentiometric Surface-Mapping**

A potentiometric surface map (e.g., groundwater contour map) is used to evaluate the direction of groundwater flow. Also, gradient calculations can be made from this map using either flow-net, flow-line or three-point calculations.

- **Hydraulic Conductivity and Porosity Evaluation Techniques**

Several methods are available to the LRS to estimate the hydraulic conductivity of the subsurface stratigraphic profile. Specific examples include:

- Literature Review of Hydrogeologic Parameters Values (e.g., porosity, hydraulic conductivity):

The least reliable estimation of hydraulic conductivity, is available from standard references (such as Freeze and Cherry, 1979, p29; Driscoll, 1986, p75). This estimation can be made from a visual estimate of grain-size or rock type, or from laboratory distribution analysis of the grain-size (ASTM D 421 and 422). As an indication of the probable ability of the subsurface at the investigation site to transmit groundwater, this estimation is suitable only for preliminary planning.

- Grain-Size Distribution:

The calculation of hydraulic conductivity from grain-size distribution (such as Freeze and Cherry, 1979, p350-351; Driscoll, 1986, p 738) is more reliable than estimation from literature values. The most relevant application for this calculation is in the sizing of well-screens and filter pack for proper well design and installation. Expansive use of these specific calculations to characterize the site-specific hydraulic conductivity is not typically adequate for making remedial design decisions; however, such estimates can be useful for planning.

- Laboratory Tests (Triaxial Chamber Tests - Vertical Permeability):

Laboratory tests of vertical hydraulic conductivity [ASTM D-5084] are useful indicators of the probable rate of vertical percolation of groundwater through the vadose zone; water-bearing zone, and/or aquitards. The calculations from these tests are more reliable than those of the grain-size distribution, but are generally limited to the assessment of vertical hydraulic conductivity.

- Time Lag Permeability Tests (Slug Tests)

Time lag permeability tests [ASTM Method D-4044, or other applicable guidance references (Hvorslev, 1951; Bouwer and Rice, 1976)] are single-point calculations of hydraulic conductivity based on the rate of recovery in response to an instantaneous change in the water level in the well. It is recommended that the tests be performed at multiple locations, as applicable, to evaluate the hydraulic variability across the site.

The results are more commonly used to find a likely average hydraulic conductivity of the site.

- **Aquifer Tests (Pump Tests)**

Aquifer testing (ASTM Method D-4050 or other applicable guidance) is the most reliable technique for calculating the hydraulic properties of the water bearing zone(s) underlying a site. Data collected from a properly designed aquifer test may provide quantitative and qualitative information such as:

- Quantitative
 - o Hydraulic properties: (e.g., transmissivity, hydraulic conductivity, specific capacity, specific yield)
 - o Zone of influence (if steady state conditions are achieved)
 - o Sustainable yield (if steady state conditions are achieved)
- Qualitative
 - o Aquifer type (e.g., confined, unconfined, leaky confined)
 - o Borehole storage, well efficiency
 - o Zone of influence (transient conditions)
 - o Heterogeneity/anisotropy
 - o Connectivity between multiple layers (as applicable)

The results of the aquifer tests can assist in defining modeling input parameters and remedial design criteria.

- **Tracer Tests**

Tracers are used as an effective means to evaluate the site hydraulic characteristics. The tracers can include dyes, salts, or trace elements. The tracer may be introduced into on-site groundwater (via monitoring wells, etc.). The presence of the tracer is monitored at designated points including extraction wells, springs/seeps, and other monitoring points. Tracer data are used to evaluate groundwater flow pathway, flow velocities, and other contaminant transport properties of the water-bearing zone (e.g. dispersion).

- **Modeling**

Although modeling may be done by a variety of means, modeling by computer is the most common. Computer models are to be performed by qualified individuals and are highly dependent on the quality of available data. Selection of the most appropriate model must be carefully considered by a qualified professional to best suit the amount of available data and to achieve the modeling objectives (i.e., future COPC fate and transport patterns, remedial design). [See Section 2.4.12].

The following lists indicate some of the standard sources of information useful in designing and characterization of hydrogeologic parameters.

- Driscoll, F.G. 1986 Groundwater and Wells. 2nd Ed. 1986. Johnson Filtration Systems, Inc. Minnesota. Chr.16.
- Freeze, R.A. and Cherry, J.A. 1979. Groundwater. Prentice-Hall.
- Kruseman, G.P. and deRiddler, N.A. 1991. Analysis and Evaluation of Pumping Test Data. 2nd Ed. ILRI Publication 47. The Netherlands.
- Lohman, S.W. 1972. Ground-Water Hydraulics. USGS-Prof. Paper 708. USGPO.
- Fetter, C.W., 1994. Applied Hydrogeology, 3rd Edition.
- Fetter, C.W., 1993. Contaminant Hydrogeology.

2.4.8 Surface Water and Sediment Sampling

Surface water and sediment sampling may be necessary if there is a possibility that contamination from the site at concentrations above risk-based levels could potentially migrate to a nearby surface water body (e.g., stream, spring, lake). The objective of the sampling will be to determine if the site has caused conditions that would pose an unacceptable risk to human-health, the environment and/or if applicable regulatory criteria have been exceeded as defined by 46 CSR 1.

Surface water bodies may have become contaminated as a result of spills, routine permitted or historical discharges, runoff from contaminated area(s), groundwater discharge or other routes. Selection of sampling locations should be based on the conceptual site model and anticipated contaminant transport mechanisms. In general, surface water samples should be collected in accordance with the instructions given in Appendix J. Sediment samples should be located at (and downstream of) the predicted locations where contaminants have entered and/or are entering the water body. The number of samples should be sufficient to characterize the extent of any potential contamination and to provide a sufficient database for risk assessments, if necessary. Samples also should be collected from upstream locations, and non-impacted “background” stations, if possible.

It is important to identify the final uses of the data prior to sample collection so all of the necessary data can be collected to evaluate risk or remedial design options. For example, parameters such as pH, hardness, total organic carbon (TOC), grain size, dissolved and/or total metals, Simultaneously Extracted Metals (SEM), Acid Volatile Sulfides (AVS) and pore water concentrations may be necessary to evaluate ecological risks in addition to chemical tests for the COPCs. The analytical laboratory should be contacted to discuss appropriate analytical detection limits, since evaluation criteria for ecological receptors may require the use of detection levels lower than those routinely specified by a particular method.

The following reference manuals provide guidance for design of a sampling program, as well as a description of various sampling techniques:

- USEPA. 1992. Guidance for Performing Site Inspections under CERCLA, Interim Final. USEPA, Hazardous Site Evaluation Division, Office of Solid Waste and Emergency Response, EPA/540-R-92-021. September 1992.
- USEPA. 1988. Guidance for Performing Remedial Investigations and Feasibility Studies under CERCLA, Interim Final. USEPA, Hazardous Site Evaluation Division, Office of Solid Waste and Emergency Response, EPA/540/G-89/004. October 1988.
- USEPA. 1992. NPDES Storm Water Sampling Guidance Document. USEPA, Office of Water, EPA/833-B-92-001. July 1992.
- NJDEP. 1992. Field Sampling Procedure Manual. New Jersey Department of Environmental Protection and Energy. May 1992.
- USEPA. 1991. Compendium of ERT Surface Water and Sediment Sampling Procedures, Surface Water Sampling SOP #2013, EPA/540/P-91-005, OSWER Directive 9360.4-03.

2.4.8.1 Surface Water

When evaluating risk or treatment alternatives with regard to surface, both contaminant and receiving stream characteristics must be considered. This may include examining maximum contaminant concentrations for evaluating acute impacts or average concentrations for determining exposures. The receptors, whether human or ecological, will have different exposure times and routes which must be taken into account. Also, worst case scenarios may need to be included in the analysis. For example, a small stream receiving contaminated groundwater recharge should tend to have higher concentrations during periods of low flow than after precipitation events (which may dilute the water samples). On the other hand, if the contamination is coming from surface water runoff, then the samples collected during periods of heavy rainfall may be representative of worst-case concentrations.

In addition to chemical data, flow velocity and/or discharge measurements will be necessary if it is important to estimate the mass of contamination that is entering the water body. Discharge measurements for streams can typically be made by measurements of flow velocity

and discharge area. Procedures for measurement of discharge rates can be found in most hydrology textbooks or USGS publications.

Surface water samples can be collected from different depths (e.g. surface, vertical mid-point, near bottom, composites, etc.) as appropriate for anticipated exposure scenarios. There are several types of sampling equipment and sampling techniques that can be used to collect water samples. The New Jersey Field Sampling Procedure Manual (NJDEP, 1992) contains a thorough description of sampling techniques/equipment, along with advantages and disadvantages of each. A few of the more common sampling techniques/equipment are as follows:

- Direct Dip
- Weighted Bottle Sampler
- Wheaton Dip Sampler
- Kemmerer Depth Sampler
- Beacon Bomb Sampler
- PACS Grab Sampler
- Pump

2.4.8.2 Sediment

Sediment sampling may be appropriate when:

- Contaminant properties suggest they may be present in only trace levels in the water column, but could accumulate to high concentrations in sediments;
- Sediments may act as a reservoir and source of contaminants to the water column;
- Sediments may accumulate contaminants over time, while contaminant levels in water are more variable; or
- Sediment contaminant levels could affect benthic organisms or other receptors of concern in aquatic ecosystems.

The sediment samples can be collected near the surface or at depth, as appropriate. However, risk evaluations generally are more concerned with the surficial sediments than deeper ones. There are several types of sampling equipment and sampling techniques that can be used to collect sediment samples. A few of the more common sampling techniques/equipment are as follows:

- Thin Wall Tube Auger

- Scoop/Trowel
- Sediment Corer
- Gravity Corer
- Bucket Auger
- Ponar Dredge
- Eckman Dredge
- PACS Sludge Getter
- Sludge Judge

In general, sampling equipment which minimizes or eliminates the loss of fine-grained material is preferred over such equipment as scoops/trowels, which tend to result in the loss of fine-grained material and therefore do not provide sediment samples that are representative of conditions to which biota would actually be exposed.

The New Jersey Field Sampling Procedure Manual (NJDEP, 1992 Compendium of ERT Surface Water and Sediment Sampling Procedures, Sediment Sampling SOP# 2016 (USEPA, 1991) contain thorough descriptions of sampling techniques/equipment, along with advantages and disadvantages of each and should be consulted for additional information.

2.4.9 Indoor Air Quality (IAQ)

IAQ may be a concern on some VRRRA sites, where the property development alternatives include reuse of existing buildings or construction of new buildings in areas of known or suspected contamination (e.g., volatile organics, etc.). In areas where existing buildings will be reused, an assessment of IAQ may sometimes be warranted. Modeling of future IAQ for new structures is difficult. However, engineering controls (e.g., vapor stop, subsurface ventilation, building structures without basement) can be incorporated into building designs to reduce the potential for future IAQ problems associated with buildings planned for areas with elevated radon and/or volatile organic compounds (VOC) containing soils.

IAQ is a complex occupational health issue. Building owners and occupants are concerned with low level exposure resulting from surrounding soil and groundwater contamination, outdoor air pollutants, contaminants present or generated from building materials or occupants, and biological agents such as bacteria and fungal spores. Generally, sick building syndrome can be caused by many factors including work environment, airborne contaminants and psychological issues.

The components of an IAQ assessment may include: (1) an evaluation of the building's ventilation system, (2) identifying sources of pollution from the surrounding area, (3) an

inspection of the facility for sources of indoor air pollution, and (4) precise instrumentation and proper analytical methods to measure factors affecting the air quality in the building. These components may include (but are not limited to) the following:

- VOCs
- Radon
- Temperature and Humidity
- Air movement / velocity
- Carbon dioxide
- Carbon monoxide
- Formaldehyde
- Ozone
- Micro-organisms (for example, fungus, bacteria)

While all of the parameters may be a concern in a real estate transaction, this section of the document focuses on those issues that present a concern primarily due to the condition of the land (e.g., radon and VOCs), as opposed to those resulting from the condition or management of the structure(s).

2.4.9.1 Volatile Organic Compounds

Hundreds of VOCs are found in indoor air at trace levels. VOCs may present an IAQ problem when individual organics or mixtures exceed normal background concentrations. The presence of VOCs may be due to many sources including: (1) those within the building (chemicals off-gassing from carpets, furniture and construction materials), and (2) contaminated soils or groundwater surrounding the structure(s). When investigating the presence and sources of VOCs within the indoor air of a building, careful examination of these conditions should be conducted.

Several direct-reading instruments are available that provide a low sensitivity "total" reading for different types of organics. Such estimates are usually presented in parts per million and are calculated with the assumption that all chemicals detected are the same as the one used to calibrate the instrument. A photoionization detector or flame ionization detector is an example of a direct-reading instrument used as a screening tool for measuring total VOCs (TVOCs). Direct-reading instruments do not provide sufficient sensitivity to differentiate normal from problematic mixtures of organics. However, instantaneous readouts may help to identify "hot spots," sources and pathways and can identify peak exposures if they happen to occur during the measurement period.

In addition to direct reading instruments, laboratory analyses of an air sample collection media (e.g., sorbent tubes, collection filters, and canisters) can provide an estimate of TVOCs in the air or the concentration of specific compounds. Generally, TVOCs or compound-specific concentrations determined by air sample collection and analysis provide more accurate results, but are unable to distinguish peak exposures.

High concentrations of individual VOCs may also cause IAQ problems. Individual VOCs can be measured in indoor air with varying degrees of sensitivity (i.e., measurement in parts per million to parts per billion) depending on the air sampling methodology used. Examples of collection media utilized by the sampling methodologies include sorbent tubes and evacuated canisters. Analysis involves gas chromatography followed by mass spectrometry.

Occupational exposure standards exist for many VOCs. No safety factors for applying these occupational limits to general IAQ are currently endorsed by USEPA and the National Institute for Occupational Safety and Health (NIOSH). Guidelines for public health exposure (as opposed to occupational exposure) for a few VOCs are available in the World Health Organization (WHO) Air Quality Guidelines for Europe. The Pan American Health Organization in Washington, D.C. is the regional office of the WHO for the United States (202) 974-3000. The WHO can also be accessed on the Internet at www.who.org. These guidelines address non-carcinogenic and carcinogenic effects. Measurement of trace organics may identify the presence of VOCs whose significance is difficult to determine. It may be helpful to compare levels in complaint areas to levels in outdoor air or non-complaint areas.

2.4.10 Tanks, Drums and Asbestos Containing Materials (ACM)

2.4.10.1 Tanks and Drums

Industrial and commercial operations often rely upon tanks, vessels, or drums for the storage of raw materials, product, or waste. These units generally consist of above or below grade tanks (buried or vaulted), process vessels (including reaction vessels, mixing or blending tanks) and barrels (commonly 55 gallon steel or fiberboard drums). Related pumps, piping or drains also may be present, and may contain residues similar to that found within the tanks or vessels.

As part of a VRRRA assessment of a former industrial or commercial facility, the location of, and assessment, of existing tanks, vessels, or drums are critical; given the short and long-term hazards which may now be associated with these units. The actual hazard(s) can vary greatly, and is largely dependent upon: (1) the materials previously stored within the units; (2) the condition of the units at the time operations ceased; and, (3) conditions affecting the units while idle (e.g. product deterioration).

Collecting the information necessary to assess and document current conditions consist of a number of inter-related tasks. The need for, extent and complexity of subsequent tasks is largely dependent upon the findings of the initial document review, and, as necessary, sampling of the contents of the units (to confirm documented conditions or gather baseline information when no information exists).

Two tasks necessary for the evaluation of a site include a document search of existing information and field verification of those conditions. This search could include both facility and public documents and would focus on process descriptions as well as chemical usage. This information, conducted prior to field investigation/verification would be used to:

1. Aid in the planning and development of a site-specific health and safety plan (e.g., HASP). Specifically, the information would be used to identify known chemical hazards (e.g. potential for explosive vapors to be present within sealed containers) as well as physical hazards which may be present (i.e. piping or vessels requiring destructive/confined space entries for inspection);
2. Provide background information for sampling purposes. Specifically, the information may be used for sample groupings of compatible materials as well as the segregation of unique (non-compatible materials). For instance, metal plating operations commonly utilize both acid and cyanide bearing solutions within the various plating baths (and within the same area). Mixing of these incompatible waste streams during sampling could have significant detrimental effect on site workers.
3. Assist in the identification of waste classification (for subsequent remedial purposes, as necessary); and

The second necessary task includes field verification of the background conditions. Based upon the complexity of the site, the field effort may include the following:

1. Confirmation of existing (remaining) tanks, vessels, or drums, including ancillary piping;
2. Visual inspection of the tanks vessels or drums, noting their current physical condition, contents and any other pertinent features (e.g., labels, pressure buildup); and,
3. Obtaining representative samples, as necessary.

At a number of old sites, incomplete or inaccurate records exist. Accordingly, the field investigation also is intended to locate and identify previously undocumented tank or drum staging areas. Some of these undocumented tanks could be associated with manufacturing support operations, such as maintenance, power distribution, or on-site waste treatment areas. In these instances, the items/chemicals of concern often include chemicals not associated with the manufactured product, but rather support operations and include hydraulic oils, fuels, refrigerants, etc. These areas could be dispersed throughout a facility. Accordingly, buried tanks or vessels abandoned in-place (where surface structures such as vents and dispensing stations have been removed) may not be readily apparent today without the use of subsurface geophysical equipment.

Given the wide and unique conditions possible within individual sites, which may be encountered, a comprehensive description of site inspection/sampling protocols is not included in this guidance. However, prior to any site activities, development of a site-specific Health and Safety/Sampling Plan is required. This Plan, developed in accordance with OSHA 1910

regulations would establish project health and safety protocols consistent with necessary sampling objectives. For tanks, vessels, and drums, these plans commonly include:

1. Mandatory use of personnel protective equipment for use with flammable, reactive, corrosive, shock sensitive or toxic chemicals;
2. Confined Space entry procedures (to gain entry for inspection/sampling purposes);
3. Hot Work Permits (for cutting and welding, when permissible); and,
4. Line-breaking procedures for opening pipelines of unknown condition.

2.4.10.2 Asbestos Containing Materials

Over the years, many building materials were made with asbestos. Materials commonly recognized as potential asbestos containing materials (ACMs) consist of thermal pipe insulation and spray-on fire proofing. Other materials may contain asbestos, such as flooring products, roofing materials, joint compound, plaster, ceiling products, wire insulation, and more. The USEPA, OSHA, state, and many local environmental and health and safety agencies have developed regulations governing the management of asbestos. These regulations may have an impact on some site investigations.

The USEPA's Asbestos National Emission Standards for Hazardous Air Pollutants (NESHAP) regulation is intended to minimize the release of asbestos fibers during activities that involve the handling or disturbance of ACM. The NESHAP regulation requires notification of the applicable regulatory authority prior to the demolition of any building/structure, or prior to renovation activities that will disturb ACM in the amount prescribed by either federal, state, or local regulations. In addition, OSHA's asbestos standards require the identification of ACM to protect workers during activities that may disturb ACM. In nearly all cases, compliance with these regulations will require an asbestos survey at VRRRA sites that include the presence of buildings/structures.

Most states and some local regulatory agencies require asbestos surveys to be performed by trained and licensed individuals. The asbestos survey conducted by the qualified individual should consist of the following items at a minimum:

- Location and description of identified potential ACM
- Assessment of the friability of the material
- Condition assessment
- Collection of an appropriate number of bulk samples of the material based on the quantity
- Analysis by a laboratory accredited for bulk asbestos fiber analysis by the National Institute of Standards and Technology under the National Voluntary Laboratory Accreditation Program (NIST/NVLAP)

- Quantity of the ACM

2.4.11 Decontamination

This section provides guidance regarding decontamination of field equipment including sampling devices as well as heavy equipment such as back hoes and drill rigs. Decontamination procedures will be an integral part of most if not all SAPs.

Decontamination is the process of removing or neutralizing contaminants which may have accumulated on field equipment. This process provides for protection of personnel, reduces or minimizes the transfer of contaminants between sampling locations (i.e., cross-contamination) or from contaminated zones to non-contaminated zones. Decontamination procedures are developed during the project planning stage (i.e., development of SAP).

The LRS is responsible for ensuring that the proper decontamination procedures are identified in the SAP. The field team leader is responsible for ensuring that the field decontamination procedures are implemented properly in the field.

The following references include information about decontamination alternatives.

- ASTM. 1990. Standard Practice for Decontamination of Field Equipment Used at Nonradioactive Waste Sites. D5088-90, (Vol. 4.08), ASTM, Philadelphia, PA.
- USEPA. 1985. Guide to Decontaminating Buildings, Structures, and Equipment at Superfund Sites. EPA/600/2-85/028.
- USEPA. 1992. RCRA Groundwater Monitoring Technical Enforcement Guidance Document (TEGD). Office of Waste Program Enforcement. OSWER Directive 9950.1.

2.4.11.1 Heavy Equipment

All heavy equipment such as drill rigs, back hoes, augers, and down hole tools should be decontaminated prior to drilling, excavation, or sampling activities are performed. “Dirty” equipment may result in false positive sampling results simply due to contamination from another site. Therefore, prior to performing any field activities, or prior to leaving the “hot zone” at the site, heavy equipment should be decontaminated. For augers and other down hole tools, decontamination should be performed between each sampling location.

Typically, decontamination of heavy equipment involves high-pressure water and/or steam cleaning. When necessary, the equipment can be cleaned with a scrub brush and soap-water solution prior to steam cleaning in order to remove visible signs of contamination (e.g., petroleum, oils, or tars).

Decontamination of heavy equipment will frequently be conducted in a designated area and over a pre-constructed decontamination pad, when containerization of the decontamination fluids is deemed appropriate. However, in some instances, decontamination may be conducted at the point where the heavy equipment was used, if such a procedure does not cause conditions to become more hazardous than they already are and/or create surficial concentration of contaminants in excess of risk-based concentrations.

A decontamination pad can be a lined pit or concrete/asphalt pad that is designed to accumulate the decontamination fluids. The objective of the decontamination pad is to collect the fluids without discharging to the ground surface, and to retrieve the fluids into a centralized location.

2.4.11.2 Sampling and Field Equipment

To better ensure that the chemical analyses represent actual field conditions, sampling equipment should be properly decontaminated prior to the field effort, during the sampling program (i.e., between sample locations or sample intervals), and at the conclusion of the field program. Preferably, dedicated sampling equipment (e.g., bailers) or disposable sampling equipment should be employed.

Soil, sediment, and water sampling equipment typically includes, but is not limited to, the following: split-barrel samplers (split-spoon samplers), bailers, bailing line, pumps and pump tubing, filtering equipment, trowels, bowls (for compositing or homogenizing soil samples) and beakers. Care should be taken to properly decontaminate this equipment or cross-contamination could occur.

The decontamination of sampling equipment should be designed based on the suspected contaminants of concern. Typically decontamination will include scrubbing with soapy water, and rinsing with tap water and distilled water. In some cases, dilute acid (e.g., dilute nitric acid) and/or solvents (methanol or hexane) may be used. However, in general, use of solvents should be avoided.

2.4.11.3 Field Analytical Equipment Decontamination

Field analytical equipment that may contact the sample media includes water level meters, pH or specific ion probes, thermometers, and borehole geophysical equipment. This equipment should be decontaminated prior to use, between sampling locations, and after the conclusion of the field program. Decontamination of this equipment should follow manufacturers recommended procedures and should prevent cross contamination.

2.4.12 Investigation Derived Waste

Site characterization field investigations may result in the generation and handling of soil, groundwater, drilling muds, decontamination products, personal protective equipment/garments, and other materials that may pose a risk to humans or the environment if not properly managed. Prior to initiating any type of field sampling activity, it is important to plan the handling of waste

products that will be generated. Not only is this important from an exposure or human/environmental protection basis, but improper or inefficient handling of such wastes could result in excessive costs or noncompliance with certain environmental regulations (e.g., RCRA, Toxic Substances Control Act [TSCA]). For these reasons, it is recommended that a IDW Management Plan (IDWMP) be incorporated into the SAP. The format of the IDWMP should reflect the complexity of the field investigation, the amount of information known about the site (i.e., what degree, if any, of contamination exists at the site based on historical use or previous sampling investigations), and the experience of the field team with respect to IDW management.

This section presents an overview of possible IDW management options, discusses the regulatory requirements of these options, and outlines general objectives for the management of IDW during site characterizations. This section is based on the document entitled “*Management of Investigation-Derived Wastes During Site Inspections*” (USEPA 540 G-91-009 - May 1991) and various other regulatory memorandums or federal guidance, including Department of Defense and other state guidelines.

2.4.12.1 IDW Management Considerations

IDW management considerations include identification of disposal and management options that are protective of human health and the environment, and comply with either state or federal laws. In general, best professional judgment should be exercised to determine whether an option is protective.

The LRS should consider the following when developing a plan to manage IDW:

- The potential degree of contamination that may be exhibited by the IDW;
- The potential exposure to human health or the environment to concentrations of contaminants in excess of the De Minimis or other appropriate standards;
- Safety and aesthetic factors associated with the disposition of the wastes (if the management option is to leave the IDW on site); and
- State or federal regulatory requirements for proper handling and treatment /disposal.

More information on RCRA wastes, including Land Disposal Restrictions (LDRs) and TSCA wastes can be found in the following references:

- USEPA. 1989 OSWER Directive 9347.3-05FS
- 40 CFR Part 260 (Hazardous Waste Management System: General)
- 40 CFR Part 261 (Identification and Listing of Hazardous Wastes)
- USEPA. 1990. PCB Guidance Manual, EPA/540/G-90/007, August.
- 40 CFR Part 761 (PCBs)

2.4.12.2 General Objectives for IDW Management

In addition to ensuring that the IDW management is protective of public health and the environment and conducted in accordance with applicable regulations, site managers need to consider two general objectives: (1) minimize the amount of IDW when possible; and (2) manage the IDW as part of the final remedial action for the site.

Minimizing the volume of IDW should be considered during the scoping of the field investigation and the development of the IDWMP. Potential ways to reduce the amount of IDW include the following:

Select field techniques which do not result in excessive IDW (e.g., soil gas surveys, Geoprobe^R sampling, direct push sampling techniques);

- Segregate wastes from “hot areas” from other areas which may not be contaminated; and
- Do not containerize IDW from background locations, which are known or suspected to be non-contaminated.

Managing the IDW as part of the final remedial action should consider the following:

- Backfill test pits and soil borings in areas where remediation is likely to occur (based on background information, field observations, or previous investigative data);
- If the IDW is containerized, manage the treatment/disposal as part of the remedial actions for the various site media.

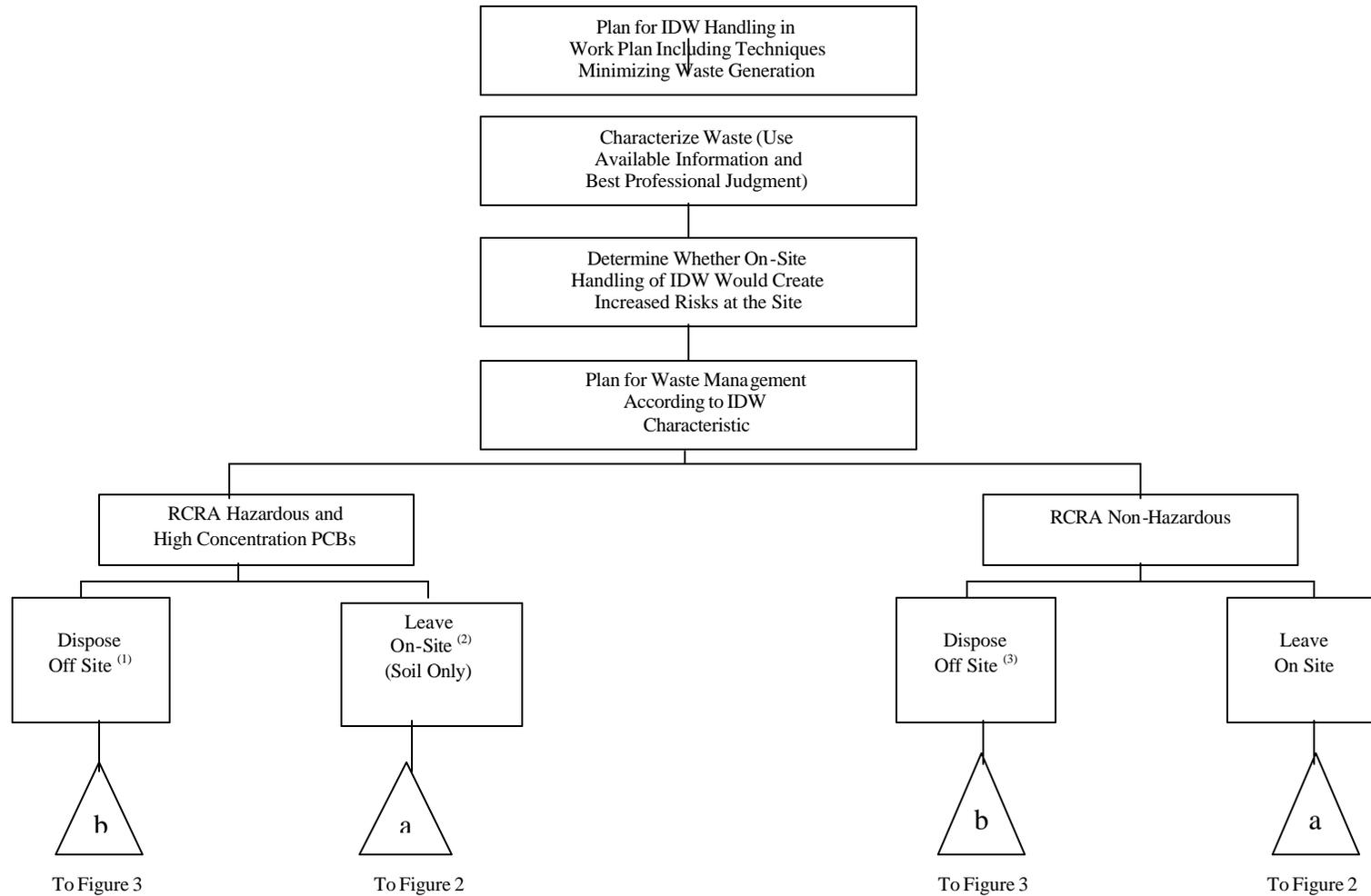
2.4.12.3 IDW Characterization

Management of IDW should be consistent with guidance provided in Management of Investigation-Derived Wastes During Site Inspections, (USEPA, 1991).

In order to determine the appropriate disposal option, it may be necessary to characterize the IDW via analytical testing. The most accurate way to characterize the waste is to obtain samples from the IDW. Using the analytical results from the site characterization program (without sampling the IDW) can also be employed in some cases.

Characterizing the IDW is more accurate and representative if the IDW is segregated during the field program (i.e., known contaminated IDW is contained separately from potentially contaminated IDW), and sampled accordingly. The objective is to obtain a sample that is representative of each waste source so the IDW is disposed properly and cost-effectively.

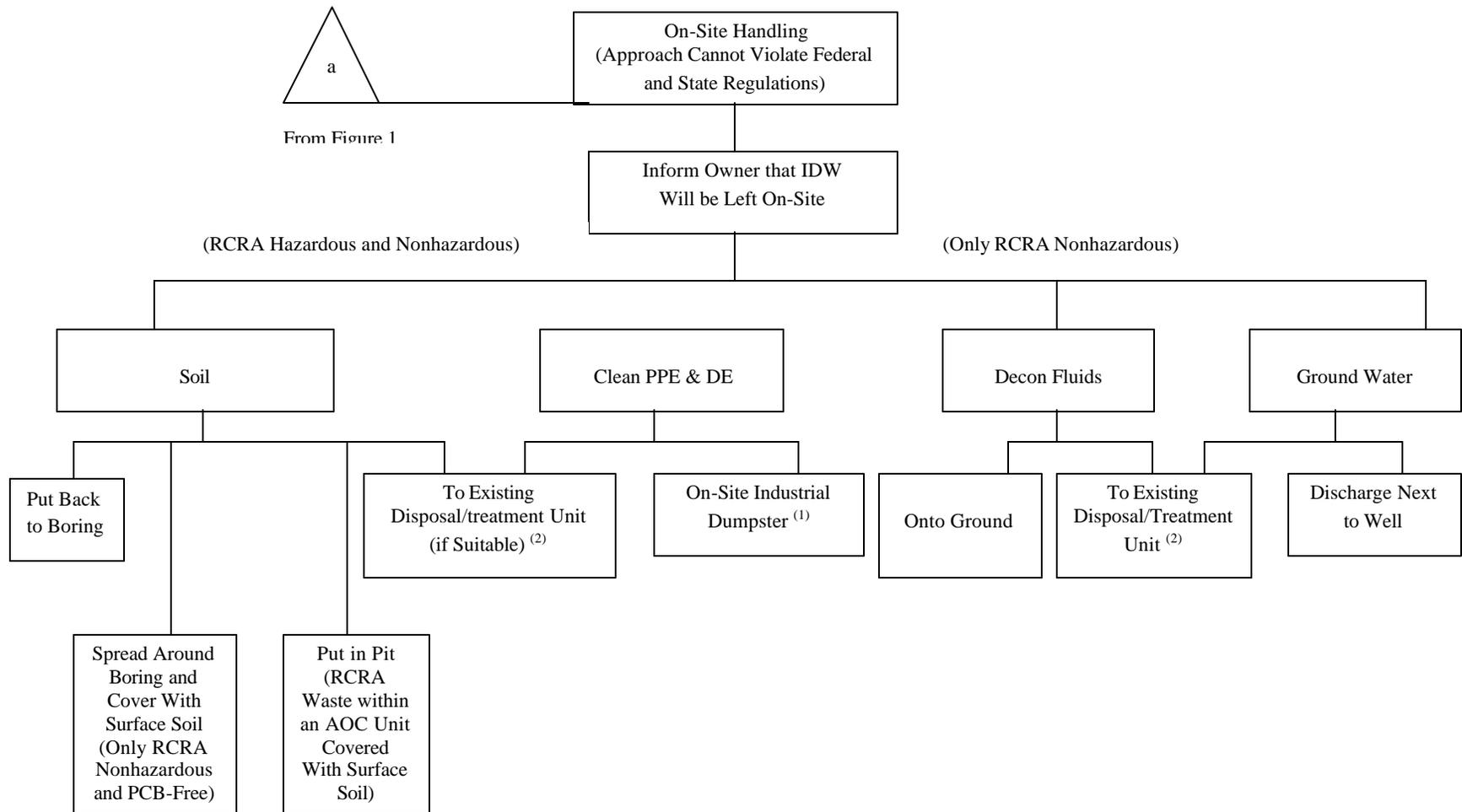
FIGURE 2-4: IDW MANAGEMENT DECISION TREE



- (1) Soil cuttings, ground water, and decontamination fluids creating increased hazards at the site should be disposed off-site. Before and after the site investigation, determine anticipated waste quantity and applicable regulations for waste generators.
- (2) If not prohibited by other legally enforceable requirements such as state ARARs.
- (3) Justified only in rare circumstances when a RCRA nonhazardous waste is a state hazardous waste and state legally enforceable requirements call for waste removal, or if leaving the waste on-site would significantly affect human health and the environment.

Source: USEPA 540G-91-009 – Management of Investigation Derived Wastes During Site Inspections – May 1991

FIGURE 2-5: IDW MANAGEMENT DECISION TREE

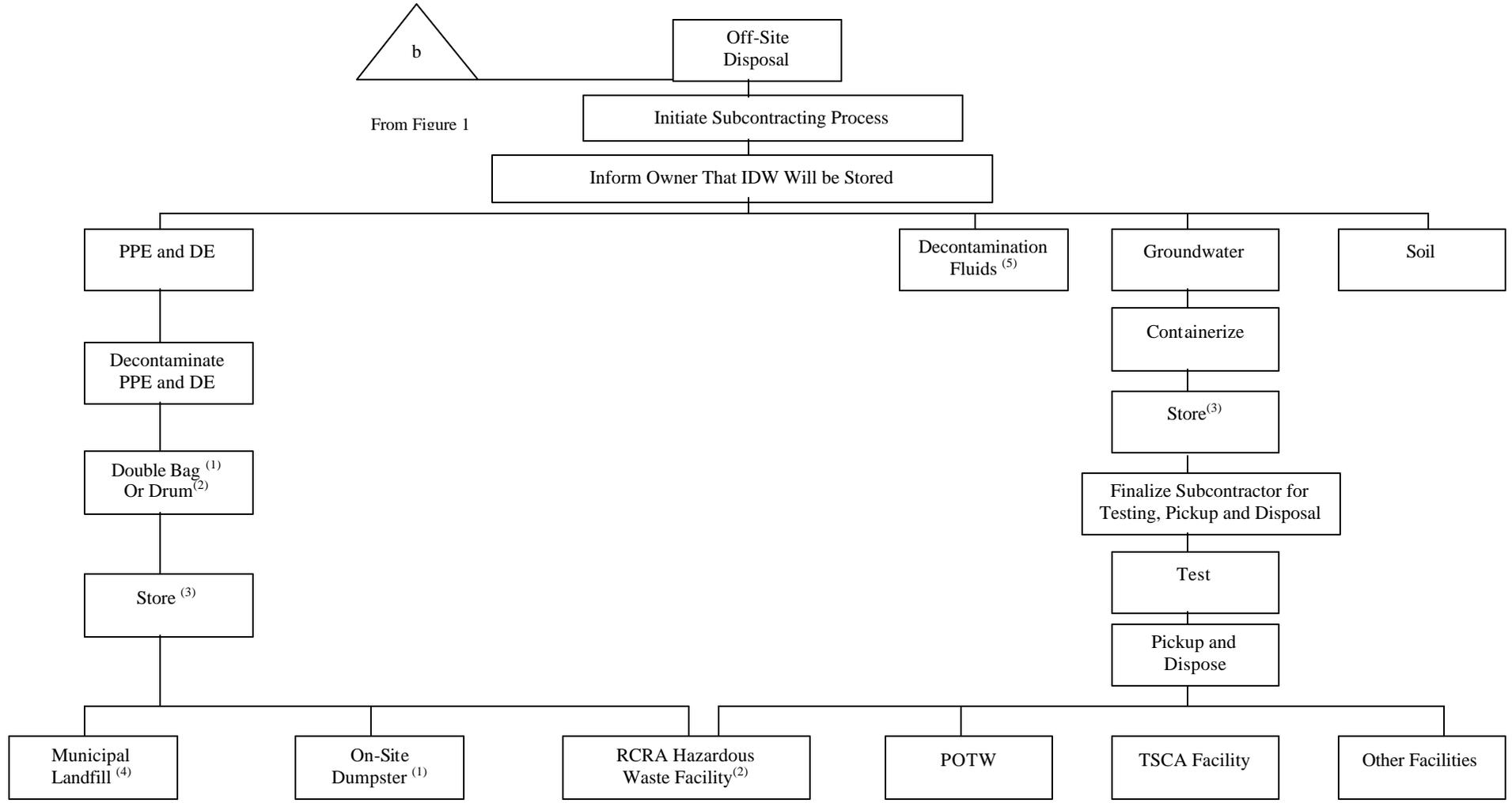


(1) Clean PPE and DE may also go to the nearest landfill or to a dumpster.

(2) If the receiving unit meets the off-site policy acceptability criteria.

Source: USEPA 540G-91-009 – Management of Investigation Derived Wastes During Site Inspections – May 1991

FIGURE 2-6: IDW MANAGEMENT DECISION TREE



- (1) Only RCRA nonhazardous waste.
- (2) Only RCRA hazardous waste generated in quantities greater than 100 kg/month when sent off-site.
- (3) In accordance with accumulation requirements for RCRA hazardous wastes.
- (4) Only if the conditionally exempt small quantity generator exception applies.
- (5) If the conditionally exempt small quantity generator exception applies, off-site disposal of decon fluids may not require subcontracting.

Source: USEPA 540G-91-009 – Management of Investigation Derived Wastes During Site Inspections – May 1991

Soil IDW that is containerized in drums can be efficiently characterized by compositing representative samples from each drum as long as the drums are segregated by area of concern. It makes no sense to obtain a composite sample from six “clean” drums and one “hot” drum since you may have a result that indicates that all of the IDW is contaminated when in reality, only one of the six drums is contaminated. Conversely, you may also falsely assume that all of the drums are non-hazardous when in fact one of the drums may be hazardous. This is why it is important to segregate “hot” wastes from other wastes.

Liquid IDW is usually containerized in either storage containers (1,000- and 2,500-gallon capacity containers are the norm) or 55-gallon drums. If drums are used to containerize the liquid IDW, either a composite sample should be collected; a sample of the suspected worst case drum, or the analytical results from the sampling program should be used to characterize the wastes. In the latter case, it is important to identify the monitoring well location on the drums so that the analytical results can be properly correlated.

Using the field program analytical results can also be used for soils, if the soils are segregated by area of concern or the soils are containerized in drums and properly identified by location.

The analytical suite for characterizing IDW should match the analytical suite for which you are testing. In cases where the LRS is unable to use generator knowledge or previous sample analyses to demonstrate the nature of any generated wastes, it will be necessary to analyze for RCRA hazardous waste constituents (ignitability, reactivity, corrosivity, and/or leachability/toxic compound leaching procedure [TCLP]) prior to shipment to an off site treatment/disposal facility.

2.4.12.4 IDW Disposal Options

The disposal option selected should be based on best professional judgment and consider the following:

- Volumes and types of wastes requiring disposal;
- Risks posed by disposing the IDW at the site without any containerization or characterization;
- Compliance with state and federal regulations;
- Whether the IDW can be managed as part of a future remedial action at the site;
- Public perception and safety;
- Compliance with transporter and disposal facility requirements.

To assess whether the IDW possesses a health or environmental risk, West Virginia De Minimis Standards can be used as a guide for this determination. The De Minimis Standards represent contaminant specific acceptable concentrations for soil and groundwater.

Figures 2-4 to 2-6 provide an IDW Management Decision Tree that can be used by the LRS to evaluate IDW disposal options (USEPA, 1991).

2.4.13 Modeling

The LRS may use modeling, to determine whether a source of contamination on the site will cause the applicable risk-based standards to be exceeded either at the property boundary, or at an off-site well. The purpose of this section is (1) to provide guidelines and references that will assist the LRS in selecting an appropriate groundwater model, (2) to discuss the type of information that must be submitted for review when requesting approval to use the proposed model, and (3) discuss how the results of the modeling should be reported.

2.4.13.1 Model Selection

The first thing that should be considered is whether it is even practical to attempt to model groundwater flow and/or contaminant transport at the site. Perhaps the most important part of the modeling process is choosing the correct model to use with the available data and site conditions.

In general, most computer models have been designed to simulate transport in porous media like silts, sands and gravel. These models cannot be effectively used to study a site where contaminants may have moved into fractured basalt or shale, or other formations that cannot be considered typical porous media. Sufficient data must be available to run the selected model.

If adequate site-specific data are not available, justifications will need to be made regarding the use of generic values or another approach must be considered. It may be more cost-effective at some sites to perform leaching tests or install monitoring wells than to do a modeling study.

There are many sources of published guidance to help model users with most aspects of modeling, such as: model selection, correct application, calibration, and verification. ASTM has published several documents dealing with several of these issues and provides acceptable guidance. See also references such as *Selection Criteria for Mathematical Models Used in Exposure Assessments: Ground-Water Models* (USEPA, 1988) and/or *Modeling of Soil Remediation Goals Based on Migration to Groundwater* (USEPA, 1991) for descriptions of available models. The National Research Council (1990) also provides an excellent discussion of the inherent limitations and uncertainties in using models to assist in the decision-making process.

The proposed models must be:

- Peer-reviewed

- Model-verified (shown to produce reliable and mathematically accurate results for all functions of the model)
- Consistent with actual physical conditions throughout the modeled area. The assumptions and limitations of the computer code, mathematical solution, technology used and computer code structure must be consistent with the actual physical conditions throughout the modeled area and the application of the model
- Used consistent with the model's documentation and all stated assumptions
- Calibrated to geologic, hydrogeologic, and physical conditions throughout the modeled area
- Field-validated (if possible) to determine if there exists a consistent comparisons between the modeled, or predicted, conditions and observed field conditions for the area being modeled.

The following analytical one-dimensional models are considered to be acceptable for modeling groundwater in the saturated zone, as long as they are used according to their limitations and intended uses:

- MULTIMED
- AT123D

The following numerical models are considered to be acceptable for modeling groundwater in the saturated zone, as long as they are calibrated to property conditions and are used according to their limitations and intended uses:

- FLOWPATH (2D)
- MODFLOW (3D)
- MT3D (in conjunction with MODFLOW)

The following models are considered to be acceptable for modeling in the vadose zone, as long as they are calibrated to site-specific conditions and are used according to their limitations and intended uses:

- VLEACH
- SESOIL
- MULTIMED

Selection of other models may be proposed by the LRS in the site characterization work plan.

2.4.13.2 Surface Water Modeling Selection

Computer modeling can be used to predict the instream concentrations of contaminants which are introduced into surface waters via storm water runoff (storm water models) or groundwater infiltration (surface water models). However, in many cases it is better to simply sample the surface water body directly. Like the previously described groundwater models, appropriate model selection is critical to the prediction of contaminant concentrations. Although detailed hydrologic and hydraulic analysis will not be necessary for all remediation sites, it may be required under certain circumstances. Hydrologic and hydraulic analysis may be utilized in conjunction with the activities described in Sections 2.4.5 and 2.4.8. Three types of analysis that may be needed are:

- Estimating peak discharges
- Hydraulic analysis
- Low Flow analysis
- Fate and transport analysis

Acceptable methodologies for these analyses are described in the following sections.

Estimating Peak Discharges

Estimating peak discharge for various storm events may be necessary on some remediation sites. Flood Insurance Studies (FISs) have been established for many communities by the Federal Emergency Management Agency (FEMA) as part of the National Flood Insurance Program. If a FIS exists for the local community, hydrologic data included in the study should be utilized as deemed appropriate. If FEMA data is not available for a particular watercourse, peak discharges may be estimated using SCS methodology.

One of the following hydrologic models may be used to estimate peak discharges:

- USDA, SCS. June 1986. Urban Hydrology for Small Watersheds, Technical Release 55 (TR-55).
- USDA, SCS. May 1982. Technical Release 20 (TR-20), Project Formulation - Hydrology.
- US Army Corps of Engineers. September 1990. Hydrologic Engineering Center, HEC-1, Flood Hydrograph Package.

Hydraulic Analysis

Hydraulic analysis may be used to:

- Estimate floodplain limits
- Determine the capacity of existing or proposed drainage systems
- Estimate the distance downstream that an accidental discharge may impact
- Estimate the time to impact at various locations downstream due to an accidental discharge

Hydraulic information contained in the FIS may be utilized as deemed appropriate. If FEMA data is not available, one of the following programs should be utilized:

- Federal Highway Administration. January 1994. Culvert Analysis, HY-8.
- U.S. Army Corps of Engineers. September 1990. Hydrologic Engineering Center. HEC-2, Water Surface Profiles.
- U.S. Army Corps of Engineers. April 1997. Hydrologic Engineering Center, HEC-RAS, River Analysis System.

The HY-8 Computer Program may be used to analyze culvert systems. This program determines capacities, headwater elevations, velocities, and other hydraulic characteristics of culvert systems. It should be noted that Manning's equation can be used to analyze channels or culverts under steady uniform flow conditions.

Either the HEC-2 or the HEC-RAS program may be used for a detailed backwater analysis of a stream. These programs are used for calculating water surface profiles as well as numerous other hydraulic parameters for steady gradually varied flow in natural or man-made channels.

Low Flow Analyses

Estimates of low flow values may be necessary to establish effluent limitations or to estimate the impact of discharges into a stream. Low flow values should be based on gage records of the stream under investigation or gage records on nearby streams with similar hydrologic characteristics. A Log-Pearson Type III analysis should be performed on the gage records to estimate the low flow value. Gages on nearby streams should be correlated with data from the stream under investigation if possible. Information on gages in the project area can be obtained from the USGS, Water Resource Division. Most USGS offices will perform the analysis for a small fee.

Toxicant Fate and Transport Models

There are numerous sources of information for guidance on selection, application and interpretation of surface water models such as the U.S. EPA's internet site and the Technical Support Document for Water Quality Based Toxics Control. Specific information on modeling

alternatives and many models are available from the US EPA Center for Exposure Assessment Modeling in Athens, Georgia. Most models are supported by guidance documents or technical manuals, which include detailed descriptions of the models assumptions and limitations, and the data and quality assurance necessary to operate the models.

Like the groundwater models, the surface water fate and transport models must also be:

- Peer-reviewed
- Model-verified
- Consistent with actual physical conditions throughout the modeled area
- Used in a manner consistent with the model's documentation and all stated assumptions
- Calibrated to hydrologic, geologic and physical conditions in the area
- Field-validated (if possible) to determine if there are consistent comparisons between predicted and observed parameters.

The following toxicant fate and transport models would be considered acceptable for use providing assumptions inherent in the models are accurate and they are used according to their limitations and intended uses.

- DYNTOX
- ESAMS-II
- WASP4
- HSPF
- SARA2
- MINTEQA2

Selection of other models may be proposed by the LRS in the site characterization work plan.

2.4.13.3 Model Approval

After a model has been selected and it has been determined what values will be used for most of the input parameters, the LRS should request WVDEP approval for use of this model at the site. The request should include a description of why this model is appropriate for the site including supporting documentation. The documentation must be sufficiently detailed and include relevant technical information about the model, such as:

- the model name, version number and date
- the names of the author(s) and company
- the intended use of the model as described by the author/company
- the governing mathematical equations and boundary conditions
- the assumptions used in the development of the model
- comparisons of the proposed model to other established models if available
- an example of a field application of the model should also be included.

2.4.13.4 Model Application

The purposes of a model could include:

- Predict if soil contamination will leach into the groundwater
- Predict if contaminants will migrate to the receptors of concern at concentration above acceptable levels
- Predict the most effective remedial alternative or design

After obtaining WVDEP approval, the application of the model can proceed. Since concentrations in a contaminant plume vary spatially and temporally, it is important to define a specific location and time for which the contaminant levels must be modeled. For example, the point of compliance may be 10 meters from the downgradient edge of the contaminated zone. A sufficient time period must be modeled to predict the maximum concentration of each contaminant at the point of exposure.

The LRS (or designated modeler) should use input data collected from the modeled area to calibrate the model to conditions throughout the modeled area. Whenever the modeler cannot rely upon input parameters collected from within the modeled area, the modeler must provide justification for why input parameters were not collected from within the modeled area, and demonstrate that the model is being calibrated and used properly. The modeler should validate the model predictions (if possible) with empirical data collected from the locations of the model output.

Modeling results should be presented in a summary report. This report should fully document the model and include model input parameters. In addition, site-specific sensitivity test results should be included for each parameter determined to be sensitive, using a range of input values considered to be reasonable for the site. The WVDEP may also ask for complete copies, either hard copies or electronic copies, of all input and output files used in your site-specific study.

2.5 Background Concentrations

The Rule specifies that where the De Minimis Standard is below natural background and where the Uniform and Site-Specific Risk-Based Standards are below anthropogenic background, that natural background may be used in place of the De Minimis Standard, and natural or anthropogenic background may be used in place of the Uniform and Site-Specific Risk-Based Standards.

2.5.1 Definition of Background

Natural background refers to the concentrations of elements that occur naturally in the earth, without any human interference. Background concentrations of naturally-occurring elements in both soils and groundwater vary greatly depending on soil types, geologic strata, and depositional environment. Anthropogenic background refers to concentrations of elements that occur over a widespread area as a result of human activities.

Methods to ascertain background levels are described in Appendix B. Alternatives to the methods for determining background levels described in this guidance should be approved by the director. No single method is appropriate for all contaminants, media, or sites, so a case-by-case evaluation is required and expert judgment is needed to design an appropriate strategy to determine background levels, particularly where anthropogenic sources are involved. A weight-of-evidence approach, where several independent lines of evidence are used to determine anthropogenic background, is preferred. For some sites, this may involve demonstrating that a release is confined to a “hot spot” or other aggregated area of contamination and has not become widely dispersed beyond a site, but that other human-activities unrelated to site activities have resulted in low levels of the contaminant being widely dispersed across as well as beyond the site. Unfortunately, it is extremely difficult to prove that a contaminant released at a site did not move to those other locations and is present due solely to activities unrelated to operations at the site.

Examples of methods to support a determination of anthropogenic background include the following:

1. Documentation of another area-wide source (outside the site) for the contaminant in soils, groundwater, or surface water. This approach is particularly useful for groundwater contamination where the flow rate and direction of the aquifer is well defined. Where ground water monitoring wells upgradient of the site indicate the presence of anthropogenic contaminants, these levels provide an indication of anthropogenic background. Caution should be used for aquifers that are not well defined, or contaminants that may move in an unexpected fashion (e.g., DNAPLs).
2. Statistical methods to compare upgradient and downgradient samples should account for spatial and temporal correlations among samples (See Gilbert, 1987).

3. Use of geostatistics or other spatial statistical approaches to demonstrate the extent of spread of a contaminant from the on-site source, relative to anthropogenic background.
4. Vertical and/or horizontal stratification of contaminant concentrations throughout a region, showing that anthropogenic sources contribute to elevated levels of the contaminant.
5. Chemical fingerprinting of releases, particularly where multiple contaminants or suitable tracer contaminants are involved, to demonstrate which contaminants are associated with a release versus off-site sources. Levels of contaminants in samples may provide evidence of an anthropogenic background level when patterns of chemical constituents associated with site-related releases are distinct from those found with releases associated with anthropogenic background. The presence of release-specific ratios of constituents, or specific tracer compounds in samples are examples of this approach. To be useful, the tracer compound(s) should have similar transport and fate characteristics as the contaminant of concern so that its distribution provides a reliable estimate of the distribution and concentration of the contaminant of concern.
6. Historical records of past releases documenting the source(s) of anthropogenic contaminants. Baseline data pre-dating on-site releases are particularly useful in this regard. Records of past releases provide supporting information.
7. Sampling of carefully selected areas outside the site to demonstrate that contaminants are widespread. Sample area selection criteria should be approved with the work plan in advance and should assure that site-related activities did not contribute to sample area contaminant levels.

2.5.1.1 Establishing Background for the De Minimis Standard

Published values of background concentrations for soil, sediment, groundwater, and surface water are to be used for the De Minimis Standard.

Soils

Natural background levels of many elements in soil are described in published literature. This information can be used for comparing natural background levels with the De Minimis Standard. Mean values from West Virginia soils are provided by Shacklette and Boerngen (1984), while Dragun and Chiasson (1991) give background levels for the Eastern United States. Table 2-3 provides background mean concentrations, standard deviations, and minimum and maximum values for a range of elements. West Virginia-specific information is primarily relied upon, and supplemented by Eastern United States (U.S.) data where the sample sizes in West Virginia are too small.

Sediments and Surface Water

Sediments and surface water are not a media considered in the De Minimis Standard, unless surface water is a drinking water source.

Groundwater

Until a compilation of information for background levels in groundwater for West Virginia is put together for use with the De Minimis Standard, refer to Subsection 2.5.1.2 of this guidance on establishing background for the Uniform and Site-Specific Risk-Based Standards.

2.5.1.2 Establishing Background for the uniform and Site-Specific Risk-Based Standards

Because background levels are greatly influenced by soil type and geologic strata, site-specific sampling is a more accurate method of determining an appropriate background value. The Uniform and Site-Specific Risk-Based Standards permit the use of anthropogenic background levels as the standard where anthropogenic background levels exceed the risk-based level. Methods to identify sample location and numbers of samples to collect for determining background in soils, groundwater, and surface water are discussed in Appendix B. Sediments may not be evaluated under the Uniform Standard.

2.5.1.2.1 Natural vs. Anthropogenic Background

Natural vs. anthropogenic background levels cannot always be easily established. This occurs because some contributors to anthropogenic background are decades, or centuries old. Examples include the use of arsenical pesticides in the early 1900s, the effects of mining from the 1800s onward, etc. As a result, it may not always be useful to try to determine whether background levels are natural or include some component of anthropogenic activity. However, it is appropriate to use any site-specific determination of background, whether it includes an anthropogenic component or not, for comparison to the Uniform and Site-Specific Risk-Based Standards.

Table 2-3: Natural Background Levels of Inorganics in Soil in West Virginia and Surrounding Areas

| Element | Sample Size | Mean (ppm) | Std. Dev. | Min | Max | Source |
|-------------------|--------------------|-------------------|------------------|-----------------|------------|-----------------------|
| Al | 10 | 63,300 | 10,000 | 50,000 | > 100,000 | Shacklette & Boerngen |
| As | 10 | 8.64 | 2.63 | 5.90 | 13.00 | Shacklette & Boerngen |
| B | 7 | 34 | 11.34 | 20 | 50 | Shacklette & Boerngen |
| Ba | 10 | 360 | 96.61 | 300 | 500 | Shacklette & Boerngen |
| Be ¹ | 10 | 0.75 | 0.82 | nd ² | 2.0 | Dragun & Chiasson |
| Ca | 10 | 1,500 | 800 | 400 | 2,500 | Shacklette & Boerngen |
| Co | 10 | 13.7 | 4.37 | 7.0 | 20.0 | Shacklette & Boerngen |
| Cr | 10 | 46 | 15.78 | 30 | 70 | Shacklette & Boerngen |
| Cu | 10 | 22.0 | 5.87 | 15.0 | 30.0 | Shacklette & Boerngen |
| F | 9 | 223 | 80.47 | 100 | 400 | Shacklette & Boerngen |
| Fe | 10 | 28,500 | 15,644 | 15,000 | 70,000 | Shacklette & Boerngen |
| Ga | 10 | 17.50 | 5.40 | 10.00 | 30.00 | Shacklette & Boerngen |
| Hg | 10 | 0.14 | 0.13 | 0.02 | 0.44 | Shacklette & Boerngen |
| K | 10 | 13,100 | 3,200 | 8,100 | 18,700 | Shacklette & Boerngen |
| La | 10 | 44.0 | 13.50 | 30.0 | 70.0 | Shacklette & Boerngen |
| Li | 10 | 34.3 | 8.98 | 21.0 | 49.0 | Shacklette & Boerngen |
| Mg | 10 | 3,200 | 1,300 | 2,000 | 5,000 | Shacklette & Boerngen |
| Mn | 10 | 770 | 368.3 | 300 | 1,500 | Shacklette & Boerngen |
| Na | 10 | 3,800 | 2,200 | 1,500 | 7,000 | Shacklette & Boerngen |
| Nb | 8 | 13 | 3.72 | 10 | 20 | Shacklette & Boerngen |
| Ni | 10 | 23.00 | 7.53 | 15.00 | 30.00 | Shacklette & Boerngen |
| Pb | 10 | 16.50 | 4.12 | 10.00 | 20.00 | Shacklette & Boerngen |
| Sb ^{1,4} | 131 | 0.76 | -- ³ | < 1.0 | 8.80 | Dragun & Chiasson |
| Sc | 10 | 9.10 | 2.77 | 5.00 | 15.00 | Shacklette & Boerngen |
| Se | 10 | 0.465 | 0.226 | < 0.1 | 0.80 | Shacklette & Boerngen |
| Sn ^{1,4} | 131 | 1.50 | -- ³ | < 0.1 | 10.0 | Dragun & Chiasson |
| Sr | 10 | 72.00 | 16.87 | 50.00 | 100.00 | Shacklette & Boerngen |
| Ti | 10 | 4,500 | 1,700 | 2,000 | 7,000 | Shacklette & Boerngen |
| U ^{1,4} | 130 | 2.70 | -- ³ | 0.29 | 11.0 | Dragun & Chiasson |
| V | 10 | 65 | 18.41 | 30 | 100 | Shacklette & Boerngen |
| Y | 10 | 28 | 9.19 | 20 | 50 | Shacklette & Boerngen |
| Yb | 10 | 3.1 | 0.74 | 2.0 | 5.0 | Shacklette & Boerngen |
| Zn | 10 | 60 | 18 | 40 | 98 | Shacklette & Boerngen |
| Zr | 10 | 220 | 58.69 | 150 | 300 | Shacklette & Boerngen |

Notes:

1. Values from Dragun & Chiasson were used due to small sample size in Shacklette & Boerngen paper. Dragun & Chiasson calculated the mean and standard deviation using the detection limit for values reported as "<[detection limit]", and using zero for values reported as "not detected".
2. nd indicates non-detect
3. -- indicates value not available
4. - Data for West VA were not available, therefore, values for Eastern U.S. soils were used.
5. Mean and standard deviation were calculated using one-half the detection limit for values reported as "<[detection limit]".

2.5.2 Comparison of Site Contaminant Concentrations to Background

The medium sampled influences the kind of statistical comparisons that can be made with background data. If samples are not taken randomly because they are purposely placed (such as with ground water well monitoring or air monitoring stations) the average station value is not an appropriate measure to test the statistical hypothesis for the one-tailed test. For this reason, determinations of background in groundwater are usually based on comparisons with upgradient wells. Statistical comparisons of downgradient versus upgradient well samples may include multiple comparison procedures, upper tolerance limits, or other methods approved by the director as described in 33CSR1.4.11. Consult Subsection 2.4.7 of this guidance for details on groundwater monitoring. Additional guidance for statistical comparison of groundwater data may be found in USEPA (1989, 1996a). Supplementary guidance for statistical comparisons of soil data may be found in USEPA (1996b,c).

Methods for comparison of site concentrations with background are given in Appendix B.

2.5.2.1 When Site Contaminant Concentrations are Less than Background

It is generally impractical and economically unjustified to remediate a site to levels below those which occur naturally. When site contaminant concentrations are determined to be less than background by appropriate statistical tests as described in Appendix B; no site remediation will be required.

2.6 Contaminants of Concern

The data collected at the site should be reviewed and determined to be sufficient for risk assessment purposes prior to deciding on the list of COPCs. As a general rule, the sampling data is likely to be sufficient if the samples are representative of the exposure area; the data quality conforms with the guidance in Subsection 2.4.1 and 2.4.2; the samples have been collected and handled in accordance with standard procedures for the collection methodology; and the samples were analyzed in accordance with appropriate laboratory methodologies and established protocols (See Subsection on 2.4).

Chemicals detected in at least one sample (even at levels below Practical Quantitation Limits (PQL) in a given medium at the site should be considered COPCs and should be carried through the screening assessment or risk assessment unless there is specific, justifiable rationale for dropping the contaminant from the risk characterization. The final list of contaminants which will be carried through the risk assessment after the selection process described below is conducted is termed the contaminants of concern (COCs). The selection of COCs should be evaluated considering conditions present at each individual site. The risk assessment portion of the report should document the process of identifying the COCs and list the chemicals that are identified for both the human health and ecological risk assessment. The specific basis for eliminating a chemical detected at the site from the list of contaminants of potential concern should be clearly documented. The following subsections (2.6.1 through 2.6.5) outline acceptable reasons for eliminating contaminants. Contaminants may be eliminated for other reasons upon approval by the Director.

2.6.1 Field or Laboratory Contaminants

Contamination may be introduced into a sample during sample collection, transport, or laboratory handling and analysis. A variety of quality control samples such as trip, equipment, laboratory calibration, and method blanks should be collected and analyzed to determine whether contaminants are being introduced by field or laboratory practices (See Subsection 2.4.1). A trip blank, which is made up in the laboratory and travels with the field team to the site and back before analysis, is subject to potential contamination from the air to which the containers are exposed. An equipment blank also travels to the field, but is opened at the site and used in the same manner as a sample (e.g., poured through sampling equipment), thus discriminating contamination due to field procedures. A laboratory calibration blank is placed in a laboratory instrument without the reagents with which site samples are treated, thus discriminating contamination due to the laboratory equipment or the water used in the laboratory. A method blank is manipulated and treated with the same reagents as the site samples, thus discriminating contamination due to the reagents or laboratory glassware.

A careful review of quality assurance and quality control data should be conducted as part of an investigation to avoid including chemicals attributable to sampling or laboratory activities in the risk assessment, while ensuring that chemicals which are site-related are not eliminated from further evaluation. When assessing the potential for sample contamination, USEPA (1989, 1992) recommends the following rule of thumb for common laboratory contaminants (e.g., acetone, 2-butanone, methylene chloride, toluene, and the common phthalate esters): consider sample results positive only if the concentration in the sample is more than ten (10) times the maximum detected in any blank; otherwise treat the sample as nondetect. If the contaminant in the blank is not one of these common laboratory contaminants, consider sample results positive only if the concentration in the sample is more than five (5) times the maximum detected in any blank; otherwise, treat the sample as nondetect. An exception to this rule may be if these contaminants are otherwise associated with the site based on their history of prior use at the site.

2.6.2 Low Concentrations and Low Frequency of Detection

Substances detected at low concentrations and low frequency may be omitted from the risk assessment process. The purpose of this criterion is to eliminate from the risk assessment any substance that is not present consistently enough or at high enough concentrations to contribute significantly to exposure.

2.6.2.1 Low Concentrations

For a chemical to be identified as a contaminant of concern, it must be present in a concentration above the detection limit of an appropriate method. Some compounds, e.g., those which biomagnify in the food chain or for which synergistic interactions have been reported, may cause health risks at levels below the detection limit of some standard methods, so care must be taken not to rule out COCs prematurely. The method detection limit (MDL) is the smallest concentration of a chemical, which can be accurately measured considering the instrumentation and background noise. As the chemical concentration approaches the MDL, the

level of confidence in accurate quantitation decreases. For use in risk characterization, the *Guidance for Data Usability in Risk Assessment (Part A) Final* (USEPA, 1992) recommends the use of the sample quantitation limit (SQL), which is the MDL adjusted to reflect sample specific action, or the MDL itself. Instrument detection limits should never be considered appropriate for use in the risk assessment. The practical quantitation limit (PQL), or the MDL multiplied by a factor of 2 to 5, may be appropriate unless the PQL is unusually high. Site specific conditions should be considered in determining which quantitation limit is used.

Data may be qualified due to concerns regarding chemical identification, chemical concentration, or both. One of the most commonly encountered types of data qualifiers are “J” values, used in the USEPA Contract Laboratory Program (CLP). The use of the “J” qualifier may indicate that the identification of the contaminant is uncertain or approximate or that the concentration of the contaminant in the sample is estimated. USEPA (1989) recommends the use of J-qualified data, but cautions that care should be exercised if the risk is being driven by the qualified data results.

2.6.2.2 Low Frequency of Detection

The frequency of detection will be evaluated at each site based upon the total number of samples collected, the sampling design, and the total area sampled. In order to establish that the frequency of detection is low, the total number of samples collected must be adequate to characterize the extent of contamination at the site. The number for what constitutes low frequency of detection will be a function of total sample size and, as such, it would not be appropriate to consider contaminants detected in one to two samples as low frequency when the total sample size was less than ten samples.

The samples included in the total sample size should be collected in the same medium with similar characteristics. For example, in soil samples, the samples used to develop frequency of detects should be collected at similar depths in areas where the soil has similar characteristics (e.g., soil collected in a flood plain would differ from that collected out of the flood plain).

When determining whether the frequency of detection of a particular contaminant is low, it is also important to consider the spatial relationship of that sample relative to other samples at the site. For example, a contaminant may only be detected in 2 out of 20 samples, but those two samples may represent a source area or “hot spot” which may need to be remediated to prevent degradation of other media (e.g., groundwater).

2.6.3 Unusually High Sample Quantitation Limits

Sample quantitation limits for a particular chemical in some samples may be unusually high due to one or more sample-specific problems (e.g., matrix interferences). Sometimes these values greatly exceed the positive results reported for the same chemical in other samples from the data set. The SQLs may be reduced by reanalyzing the sample or the samples may be excluded from the risk assessment if they cause the calculated exposure concentration to exceed the maximum detected concentration for a particular sample set (USEPA, 1989). If there are numerous problems with a data set such that quantitation limit's for the majority of the samples are elevated, reanalysis and possibly resampling is indicated.

2.6.4 Comparison to Background

Contaminants of potential concern associated with a site should be evaluated in relation to background conditions, either natural or anthropogenic, as appropriate. When chemicals are present at levels which are consistent with background then those chemicals need not be carried through the risk assessment process. This guidance addresses the determination of consistency with background in much greater detail in Subsection 2.5.

2.6.5 Evaluate Essential Nutrients

Chemicals that are: essential nutrients present at low concentrations, and toxic only at very high doses need not be considered further in the risk assessment. Examples of such chemicals are iron, magnesium, calcium, potassium, and sodium (USEPA, 1989).

2.6.6 Screen Against De Minimis or Benchmark Levels to Identify COCs

In an effort to streamline the investigation and cleanup of properties, the WVDEP has provided De Minimis human health screening levels for soil and groundwater media contaminants. The screening levels are provided for residential land and water use and industrial/commercial land use for soils. The screening levels are derived from the USEPA Region III Risk-Based Concentration Tables, although the industrial risk based concentrations have been modified to reflect a 1×10^{-5} carcinogenic risk and WV Groundwater Standards have been inserted when available.

A contaminant of potential concern in soil may be screened against the human health De Minimis standard to further refine the list of contaminants to be carried through the risk assessment (either using the Uniform Risk-Based or Site-Specific standard), only if leaching to groundwater, inhalation of volatiles, and ecological risk is not of concern (see Appendices D.4.5, D.4.3.1, and C-2 respectively). If a site-specific ecological risk assessment is performed, chemicals may be screened against available benchmarks to further refine the list of COCs. The more conservative of the human health De Minimis or the ecological benchmark shall be used as the screening value.

2.6.7 Additional Issues for Consideration

2.6.7.1 Chemical Species

It may be important to consider specific states of the chemicals when identifying COCs. Depending on the specific state of the chemical that is present at the site, there may be different health or environmental effects associated with the chemical. For example, differences in oxidation states of metals can result in changes in absorption or toxicity (e.g., hexavalent chromium is more toxic than trivalent chromium). In addition, some products may degrade over time and products of degradation may have different toxicity parameters (e.g., vinyl chloride is more toxic than trichloroethene). These factors should be considered when identifying contaminants of concern.

2.6.7.2 Groups of Compounds

Some of the data collected for a site may be presented as groups of compounds (e.g., TPH). Data on groups of chemicals is not generally useful in the risk assessment process. Toxicity information used to estimate risk is compound specific; therefore the estimation of risk associated with exposure to compounds that are identified as a group can be highly inaccurate or impossible, and as a result is not generally recommended. The individual contaminants are the COCs, but, to simplify discussion within the risk assessment, may be described as groups of compounds.

2.6.7.3 Tentatively Identified Compounds (TICs)

When gas chromatography-mass spectrometry (GC-MS) is used to analyze for the presence of organic compounds, the instrument is calibrated for authentic chemical standards. These standards represent the target compounds, which are being analyzed in the samples. A target compound in an environmental sample is identified by matching its mass spectrum and relative retention time to those obtained for the authentic standard during calibration. Quantitation of a target compound is achieved by comparison of its chromatographic peak area to that of an internal standard compound. When compounds are identified in the sample, but the GC-MS instrument was not specifically calibrated for those compounds, they are designated as TICs. The mass spectrum of the sample is compared to a computerized library of mass spectra, but since no standard was calibrated for the TIC, the identification is less certain than for target compounds. The *Guidance for Data Usability in Risk Assessment (Part A) Final* (USEPA, 1992) identifies several techniques which can be used to increase the confidence in identification and quantification of TICs.

- The TIC data should be reviewed by an analytical chemist trained in the interpretation of mass spectra and chromatograms;
- The identification of the TICs should be checked against the chromatographic retention indices or relative retention times. Use physical characteristics (e.g., boiling point or vapor pressure) to determine if identification is reasonable;
- The TICs should be compared to available site information regarding past use of the site and chemicals associated with prior uses of the site;
- TICs can be compared to known analytical response characteristics for similar compounds for better quantification.
- The sample could be re-analyzed using an authentic standard.

It is also advisable to evaluate whether the TIC is likely to be associated with other compounds detected at the site. The result may support the tentative identification or may aid in making a decision regarding the need to re-sample.

The TIC may also be classified as belonging to a particular class of compounds, such as polycyclic aromatic hydrocarbons, and may be discussed qualitatively in the risk assessment.

When dealing with TICs qualitatively, the impacts on cumulative site risk and overall uncertainty should be discussed. The data should be reviewed by an experienced analyst to obtain an “order of magnitude” estimate of the concentration, prior to any discussions of qualitative risk posed by TICs. The concentrations of TICs v. concentrations of identified compounds should be discussed in terms of the overall risk associated with the site.

2.7 Presentation of COCs in Tabular Format

Consider the creation of a separate table for each environmental medium (e.g., soil, sediment, groundwater, surface water, fish tissue, indoor air, outdoor air, soil gas) for which sampling data is available. Throughout the text and tables, present the data in a consistent manner (e.g., ug/L for groundwater and mg/kg for soils). Arrange the data chronologically, if appropriate (e.g., groundwater), and discuss in the text any apparent time trends in the data. Present the number of times detected/number of times analyzed (frequency of detection), range of sample quantitation limits, minimum and maximum concentrations, arithmetic mean and 95 percent upper confidence limit on the arithmetic mean (if applicable). If hot spots are identified, they may be presented in separate tables.

2.8 References

2.8.1 Preliminary Characterization

American Society for Testing Materials (ASTM). 1997. E 1527-97 Standard Practice for Environmental Site Assessment: Phase I Environmental Site Assessment Process.

2.8.2 Risk Assessment Data Requirements

United States Environmental Protection Agency (USEPA). 1989. Risk Assessment Guidance for Superfund Volume I. Human Health Evaluation Manual (Part A) Interim Final. Office of Solid Waste and Emergency Response. Washington, D.C. EPA/540/1-89/002. March 1989.

United States Environmental Protection Agency (USEPA). 1989. Risk Assessment Guidance for Superfund, Volume II: Environmental Evaluation. Office of Solid Waste and Emergency Response. Washington, D.C. EPA/540/1-89/001. March 1989.

United States Environmental Protection Agency (USEPA). 1992. Guidance for Data Usability in Risk Assessment (Part A) Final. Office of Emergency and Remedial Response. Washington, DC. Publication 9285.7-09A. April 1992.

United States Environmental Protection Agency (USEPA). 1995. Land Use in The CERCLA Remedy Selection Process. Elliott P. Laws, Asst. Administrator. OSWER Directive 9355.7-04. May 25, 1995.

United States Environmental Protection Agency (USEPA). 1997. Integrated Risk Information System On-line Database. Environmental Criteria and Assessment Office. Cincinnati, Ohio. www.epa.gov/docs/ordntrnt/ord/dbases/iris/index.html

2.8.3 Data Requirements for Remedial Action Design

United States Environmental Protection Agency (USEPA) Engineering Bulletin. 1992. Technology Pre-selection Data Requirements. EPA/540/S-92/009.

United States Environmental Protection Agency (USEPA). 1993. Guide for Conducting Treatability Studies under CERCLA. Office of Research and Development and Office of Emergency and Remedial Response. Cincinnati, Ohio. EPA/540/R-93/519-A.

2.8.4 Data Requirements for Modeling

Luckner, L and W.M. Schestakow. 1991. Migration Processes in the Soil and Groundwater Zone. Lewis Publishers, Inc. Chelsea, Michigan.

Tyler, L.D. and M.B. McBride. 1982. Mobility and Extractability of Cadmium, Copper, Nickel and Zinc in Organic and Mineral Soil Columns. Soil Science, 134(3), pp 198-205.

Korte, N.E., J Skopp, W.H. Fuller, E.E. Niebla and B.A. Aleshi. 1976. Trace Element Movement in Soils: Influence of Soil Physical and Chemical Properties. Soil Science, 122(6), pp 350-359.

2.8.5 Investigation Techniques for Sampling and Analysis Plans

Aller, Linda, et al. 1989. Handbook of Suggested Practices for the Design and Installation of Ground-Water Monitoring Wells. National Water Well Association.

American Society for Testing Materials (ASTM). 1991. Test Method (Field Procedure) for Instantaneous Change in Head (Slug Test) for Determining Hydraulic Properties of Aquifers. D-4044-91 (Vol. 04.08).

American Society for Testing Materials (ASTM). 1991. Test Method (Field Procedure) for Withdrawal and Injection on Well Tests for Determining Hydraulic Properties of Aquifer Systems D-4050-91 (Vol. 04.08).

American Society for Testing Materials (ASTM). 1993. Practice for Diamond Core Drilling for Site Investigation D-2113-83 - Reapproved 1993 (Vol. 04.08).

American Society for Testing Materials (ASTM). 1992. Test Method for Penetration Test and Split-barrel Sampling of Soils D-1586-84 - Reapproved 1992 (Vol. 04.08).

American Society for Testing Materials (ASTM). 1990. Test Method for Measurement of Hydraulic Conductivity of Saturated Porous Materials Using a Flexible Wall Permeometer. D-5084-90 (Vol. 04.09).

American Society for Testing Materials (ASTM). 1994. Test Method for Infiltration Rate of Soils (in Field) Using Double Ring Infiltrometer. D-3385-94 (Vol. 04.08).

American Society for Testing Materials (ASTM). 1995. Guide to Site Characterization for Environmental Purposes with Emphasis on Soil, Rock, Vadose Zone, and Groundwater. D-5730-95a (Vol. 04.09).

American Society for Testing Materials (ASTM). 1994. Practice for Thin-Walled Tube Sampling of Soils. D1587-94, (Vol. 4.08).

American Society for Testing Materials (ASTM). 1995. Practice for Soil Investigation and Sampling by Auger Borings. D1452-80 - Reapproved 1995 (Vol. 4.08).

American Society for Testing and Materials (ASTM). 1986. Standard Test Method for Deep, Quasi-Static, Cone and Friction-Cone Penetration Tests of Soil. D3441-86 (Vol. 4.08).

American Society for Testing Materials (ASTM). 1993. Standard Guide for Investigating and Sampling Soil and Rock. D420-93 (Vol. 4.08).

American Society for Testing and Materials (ASTM). 1990. Standard Practice for Decontamination of Field Equipment Used at Nonradioactive Waste Sites. D5088-90, (Vol. 4.08).

American Society for Testing Materials (ASTM). 1991. Guide for Soil Sampling from the Vadose Zone. D4700-91 (Vol. 4.08).

American Society for Testing Materials (ASTM). 1993. Draft Standard Guide for the Use of Hollow-Stem Augers for Geoenvironmental Exploration and Installation of Subsurface Water-Quality Monitoring Devices. D18.21 Ballot 93-03.

American Society for Testing Materials (ASTM). 1993. Draft Standard Guide for the Use of Direct Rotary Drilling for Geoenvironmental Exploration and Installation of Subsurface Water-Quality Monitoring Devices. D18.21 Ballot 93-03.

American Society for Testing Materials (ASTM). 1993. Draft Standard Guide for the Use of Air-Rotary Drilling for Geoenvironmental Exploration and

Installation of Subsurface Water-Quality Monitoring Devices. D18.21 Ballot 93-03, April 28, 1993.

Bouwer, H and R.C. Rice. 1976. A Slug Test for Determining Hydraulic Conductivity of Unconfined Aquifer with Completely or Partially Penetrating Wells. *Water Resources Research* 12(3), pp.423-428.

Bouwer, H. 1989. The Bouwer and Rice Slug Test - An Update. *Groundwater* 27(3), pp. 304-309.

Christy, T.M. and S.C. Spradlin. 1992. The Use of Small Diameter Probing Equipment for Contaminated Site Investigations. *Groundwater Management* 11:87-101 (6th NOAC).

Chiang, C.Y. et al., Characterization of Groundwater and Soil Conditions by Cone Penetrometry. In: *Proceedings (6th) NWWA/API Conference, Dublin, Ohio*. pp. 175-189.

Driscoll, F.G. 1986. *Groundwater and Wells: 2nd Ed.* Johnson Filtration Systems, Inc. Minnesota.

Freeze, R.A. and J.A. Cherry. 1979. Groundwater. Prentice-Hall.

Hvorslev, M.J. 1951. Time Lag and Soil Permeability in Ground-Water Observations, Bulletin 36. U.S. Army Corps of Engineers. Vicksburg, Mississippi.

Kruseman, G.P. and N.A. deRiddler. 1991. Analysis and Evaluation of Pumping Test Data. 2nd Edition. ILRI Publication 47. Netherlands.

Lohman, W.W. 1972. *Ground-Water Hydraulics*. USGS-Professional Paper 708. USGPO.

New Jersey Department of Environmental Protection (NJDEP). 1992. *Field Sampling Procedures Manual*. May, 1992.

United States Environmental Protection Agency (USEPA). 1983. *Methods of Chemical Analysis for Water and Waste*, EPA/600/4-79/020, 1983 rev.

United States Environmental Protection Agency (USEPA). 1984. *Geophysical Techniques for Sensing Buried Wastes and Waste Migration*. EPA/600/7-84/064.

United States Environmental Protection Agency (USEPA). 1985. *Guide to Decontaminating Buildings, Structures, and Equipment at Superfund Sites*. EPA/600/2-85/028.

United States Environmental Protection Agency (USEPA). 1992. *RCRA Groundwater Monitoring Technical Enforcement Guidance Document (TEGD)*. Office of Waste Program Enforcement. OSWER Directive 9950.1.

United States Environmental Protection Agency (USEPA). 1986. Test Methods for Evaluating Solid Waste, Office of Solid Waste and Emergency Response, Washington, DC, November 1986 revised January 1995, SW-846 Third Edition.

United States Environmental Protection Agency (USEPA). 1987. A Compendium of Superfund Field Operations Methods, Part 2. EPA/540/P-87/001 (OSWER Directive 9355.0-14).

United States Environmental Protection Agency (USEPA). 1987. Handbook - Groundwater. EPA/625/6-87/016.

United States Environmental Protection Agency (USEPA). 1988. Field Screening Methods Catalog: User's Guide. EPA/540/2-88/005.

United States Environmental Protection Agency (USEPA). 1988. Guidance for Performing Remedial Investigations and Feasibility Studies under CERCLA, Interim Final. USEPA, Hazardous Site Evaluation Division, Office of Solid Waste and Emergency Response, EPA/540/G-89/004. October.

United States Environmental Protection Agency (USEPA). 1988. Selection Criteria for Mathematical Models Used in Exposure Assessments: Ground-Water Models. EPA/600/8-91/038.

United States Environmental Protection Agency (USEPA). 1990. PCB Guidance Manual. EPA/540/G-90/007. August.

United States Environmental Protection Agency (USEPA). 1991. Subsurface Characterization for Subsurface Remediation. EPA/625/4-91/026.

United States Environmental Protection Agency (USEPA). 1991. Description and Sampling of Contaminated Soils: A Field Pocket Guide. EPA/625/12-91/002.

United States Environmental Protection Agency (USEPA). 1991. Second International Symposium, Field Screening Methods for Hazardous Waste and Toxic Chemicals, EPA/600/9-91/028.

United States Environmental Protection Agency (USEPA). 1991. Handbook - Ground - Water Volume II - Methodology. EPA/625/6-90/016.

United States Environmental Protection Agency (USEPA). 1991. Management of Investigation-Derived Wastes During Site Inspections. EPA/540/G-91/009.

United States Environmental Protection Agency (USEPA). 1991. Modeling of Soil Remediation Goals Based on Migration to Groundwater.

United States Environmental Protection Agency (USEPA). 1992. Guidance for Performing Site Inspections under CERCLA, Interim Final. USEPA, Hazardous Site Evaluation Division, Office of Solid Waste and Emergency Response, EPA/540/R-92/021. September.

United States Environmental Protection Agency (USEPA). 1992. Preparation of Soil Sampling Protocol: Techniques and Strategies NTIS PB-92220532.

United States Environmental Protection Agency (USEPA). 1992. Storm Water Management for Industrial Activities (EPA/832/R-92/006).

United States Environmental Protection Agency (USEPA). 1992. NPDES Stormwater Sampling Guidance Document. Office of Water. EPA/833/B-92/001. July 1992.

United States Environmental Protection Agency (USEPA). 1993. EPA Quality System Requirements for Environmental Programs (Draft) EPA/QA/R-1.

United States Environmental Protection Agency (USEPA). 1993. EPA Requirements for Quality Assurance Project Plans for Environmental Data Operations (Draft Final) EPA/QA/R-5.

United States Environmental Protection Agency (USEPA). 1993. Guidance for Planning Data Collection in Support of Environmental Decision Making Using the Data Quality Process. EPA/QA/G-4.

United States Environmental Protection Agency (USEPA). 1993. Guidance for Conducting Environmental Data Assessments (Draft). EPA/QA/G-9.

United States Environmental Protection Agency (USEPA). 1993. Data Quality Objective Process for Superfund-Interim Final Guidance. EPA/540/R-93/071.

United States Environmental Protection Agency (USEPA). 1993. Subsurface Characterization and Monitoring Techniques. A Desk Reference Guide. Vols. 1 and 2. EPA/625/R-93/003a.

United States Environmental Protection Agency (USEPA). 1993. Use of Airborne, Surface, and Borehole Geophysical Techniques at Contaminated Sites: A Reference Guide. EPA/625/R-92/007.

2.8.6 Background

Dragun, J. and A. Chiasson. 1991. Elements in North American Soils. Hazardous Materials Control Research Institute. Silver Spring, Maryland..

Gilbert, R.O. 1987. *Statistical Methods for Environmental Pollution Monitoring*. Van Nostrand Reinhold: New York, New York, 320 pp.

Shacklette, HT; Boerngen, JG. 1984. "Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States." US Geological Survey. USGS Professional Paper 1270.

Beckman, R. J., and R. D. Cook. 1983. Outliers. *Technometrics* 25:119-149.

USEPA. 1989. Statistical Analysis Of Groundwater Monitoring Data At RCRA Facilities. EPA/350-SW/89-026.

USEPA. 1996a. Superfund Guidance For Evaluating The Attainment Of Cleanup Standards. Volume 2. Groundwater. Center for Environmental Statistics. Available HTTP: <http://www.epa.gov/ces/pubs.html> .

USEPA. 1996b. Soil Screening Guidance: Users Guide. Office of Solid Waste and Emergency Response. Washington, DC 20460. EPA/540/R-96/018.

USEPA. 1996c. Superfund Guidance For Evaluating The Attainment Of Cleanup Standards. Volume 1. Soils and Solid Media. Center for Environmental Statistics. Available HTTP: <http://www.epa.gov/ces/pubs.html> .

2.8.7 Contaminants of Concern

United States Environmental Protection Agency (USEPA). 1989. *Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A), Interim Final*. Office of Emergency and Remedial Response (Washington, DC). EPA/540/1-89/002. December.

United States Environmental Protection Agency (USEPA). 1992. *Guidance for Data Usability in Risk Assessment (Part A) Final*. Office of Emergency and Remedial Response (Washington, DC). Publication 9285.7-09A, PB92-963356. April.

3.0 HUMAN HEALTH STANDARDS

3.1 Introduction

As described in the Voluntary Remediation and Redevelopment Rule (§60-3-9) the risk-based standards provide for the protection of human health and the environment relative to current and reasonably anticipated future land and water uses of the site. Risk-based standards are used to determine whether a remedial response action is necessary to identify target cleanup levels in the event that a remedial action is required, and to document that a level protective of human health and the environment exists or has been achieved for a site.

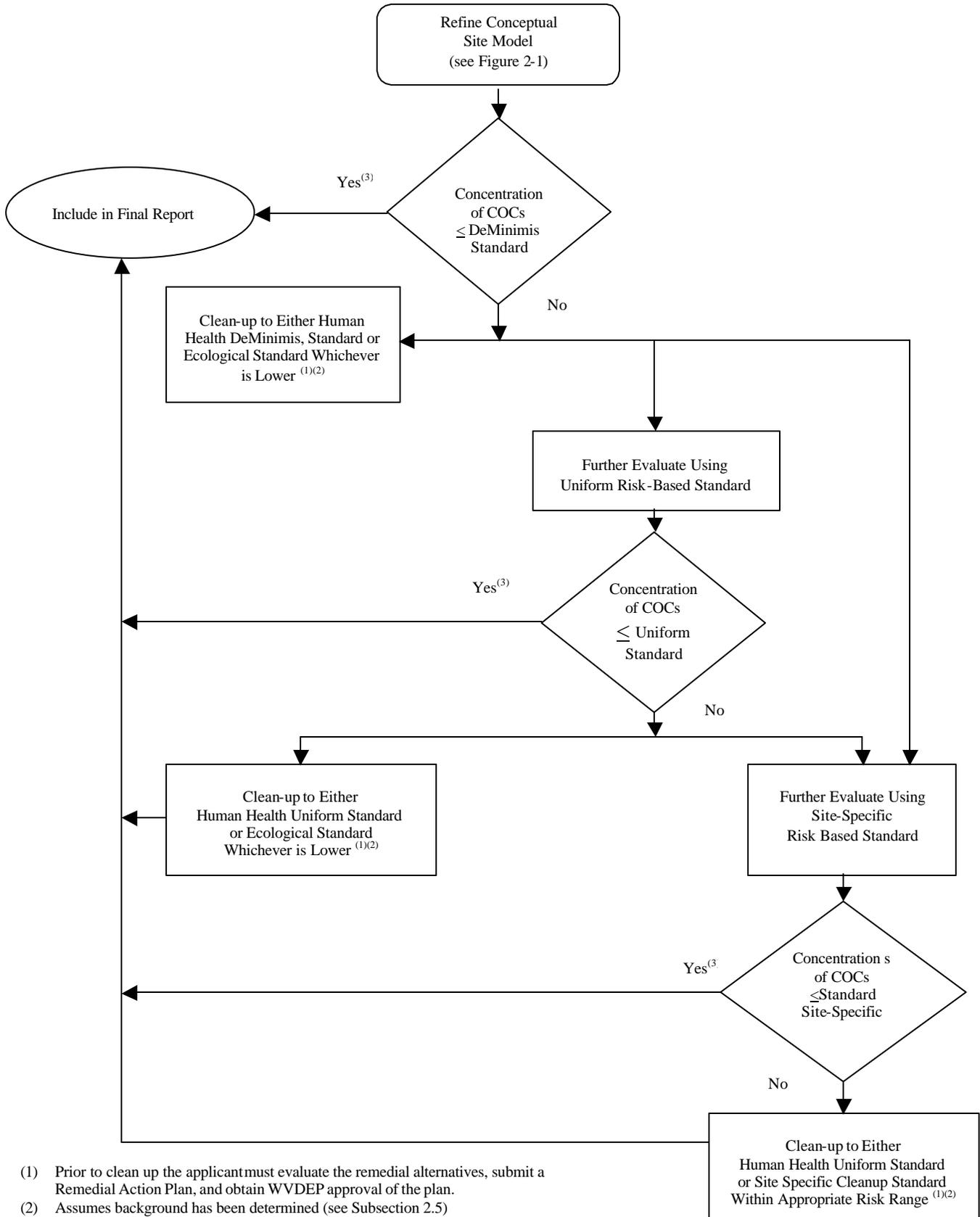
This section describes the human health standards to be applied to voluntary remediation sites consistent with §60-3-9 of the Rule. In conjunction with the ecological standards (Section 4.0 of this guidance), a variety of options may be applied to different contaminants at a site or to different portions of a site. The goal is to provide flexible standards so that an applicant may select the standard most appropriate for that particular site. The purpose of these standards is to develop risk-based soil, sediment, surface water, and groundwater remedial objectives for site-remediation.

Three options are available for developing risk-based human health standards at a site. A De Minimis standard (§60-3-9.2 of the rule) consists of pre-established numerical values from look-up tables (table 60-3B of the rule and Appendix C of this guidance). A Uniform Risk-Based Standard (§60-3-9.3 of the Rule) is determined by using established risk equations that incorporate either default or site-specific values for variable (formulae and default values are provided in Appendix D of this guidance). A Site-Specific Risk-Based Standard (§60-3-9.4 of the Rule) uses baseline and/or residual risk assessment to establish protective cleanup standards based on site-specific conditions and reasonably anticipated future land and water uses. As specified in the Rule (§60-3-9), where background levels of contaminants exceed the risk-based levels, background levels may be used as cleanup standards (see Subsection 2.5 and Appendix B of this guidance for procedures to determine background).

Section §60-3-8 of the Rule provides risk protocols for conducting baseline human health and ecological risk assessments, and for residual risk assessments (described in Section 5 of this guidance) of conditions following implementation of the proposed remedy. Procedures for probabilistic assessments are also addressed in §60-3-8 of the Rule and Section 6 and Appendix I of this guidance.

Figure 3-1 provides a flow diagram that illustrates the decision-making process for selecting a method of deriving human health standards for a site. This diagram should be used together with the Checklist for Determination of the Applicable Standard (Appendix C-1). This decision-making process should begin only after a conceptual site model has been developed (see Subsection 2.2.4 of this guidance) COCs have been identified (see Subsection 2.6 of this guidance for methods of COC selection), and sufficient data collected to characterize COC concentrations in media of concern. Subsection 2.3.1 of this guidance describes additional data requirements for risk assessment.

FIGURE 3-1: HUMAN HEALTH REMEDIATION STANDARD SELECTION PROCESS



- (1) Prior to clean up the applicant must evaluate the remedial alternatives, submit a Remedial Action Plan, and obtain WVDEP approval of the plan.
- (2) Assumes background has been determined (see Subsection 2.5)
- (3) Ecological receptors are not a factor or ecological standards are \geq Human Health Standards

For many simple sites with few contaminants, the De Minimis Standards may be sufficient to determine the need for remediation. For more complex site (e.g., sites where leaching of chemicals from soil to groundwater is a concern or sites where surface water has been impacted), or sites having contaminants which should not be included in the De Minimis evaluation (e.g., volatile chemicals in soil), the Uniform Risk-Based or Site-Specific Risk-Based Standards must be applied to all or portions of the site. Where site contamination is impacting surface waters, the following must be addressed:

- The De Minimis standard may not be used for any contaminant at a site where that contaminant is impacting surface water;
- The uniform and site-specific remediation standards for surface water are the applicable water quality standards found at 46CSR1 (surface water quality standards promulgated by the Environment Quality Board):
- An applicant may demonstrate compliance with surface water quality standards where site contamination is originating from a non-point source by following the Office of Water Resources’ “Monitoring Procedures to Determine Impact of Surface Water from Non-Point Source Site Remediation Projects.” (see Appendix J). The foregoing demonstration does not relieve the applicant of the requirement to assure that the adopted remediation standard is protective of human health and ecological receptors of concern.
- For any contaminants for which there is no water quality standard in 46CSR1, a remediation standard may be developed using the methodology for determining a site-specific standard in Section 3.4 of this guidance.

Subsection 1.1.3 of this guidance describes the options for selecting human health and ecological standards for a site. The following subsections provide more detailed guidance for using or developing human health standards.

3.2 Human Health De Minimis Risk-Based Standard

The De Minimis Standard is designed to be the quickest and easiest method from which remediation standards and objectives are derived, which also being protective of human health. As described in §60-4-9.2 of the Rule, the De Minimis standards apply to chemicals for which the primary exposure routes will be ingestion from soil, or ingestion. For soil, the De Minimis Standard will be either the risk-based concentrations (RBCs) (found in Table 60-3B of the Rule and reproduced in Appendix C-1 of this guidance) or the natural background levels of the contaminant, whichever is higher. RBC standards are provided in Appendix C for both residential and industrial land use scenarios. For groundwater, the De Minimis Standard will be the higher of the two, either the value from Appendix C or the natural background concentration.

No De Minimis Standards are available for surface water or sediments, and De Minimis Standards should not be applied to volatile organic chemicals (denoted in Appendix C by an “X” in the VOC column) in soils, or to chemicals that may have a primary route of exposure other than ingestion (e.g., inhalation after particulate release to air). Thus, development of an accurate

conceptual site model, as described in Subsection 2.2.4 of this guidance, is a critical step in determining eligibility of a site, or portions of a site for assessment using De Minimis Standards. Site eligibility for using the De Minimis Standards is determined by responses to the checklist for determining the appropriate standard included in Appendix C-1 of this guidance.

3.2.1 De Minimis Standards for Soil

The De Minimis Standard for both surface (<2 feet depth) and subsurface (>2 feet depth) soils is the higher of the following values:

- RBCs listed in Table 60-3B of the Rule and reproduced in Appendix C (for residential or industrial, as appropriate)
- Natural background levels (Subsection 2.5)

The De Minimis Standard may be selected for all or a portion of the contaminants and for all or a portion of the site, as appropriate. A land use covenant restricting residential land uses is required for portions of a site where the industrial De Minimis Standards are used.

Background concentrations of naturally-occurring constituents vary greatly depending upon the source of the soil matrix or depositional environment. When natural background is used as the De Minimis Standard, attainment may be demonstrated in several ways. A simple approach is to document that individual sample concentrations for a particular contaminant from a site are less than the upper tolerance limits (UTL) for that same analytes in natural background samples. Methods for calculating the UTL are provided in Appendix B. A comparison of the UTLs for analytes in natural background soils to the De Minimis Standards provided in Table 60-3B of the Rule indicates that natural background concentrations of arsenic, beryllium, iron, and may be greater than the De Minimis Residential Standards for those analytes. It is not practical to remediate a site to a standard less than background.

3.2.2 De Minimis Standards for Ground Water

The De Minimis Standard for ground water is the highest value of the following:

- Groundwater contaminant concentration limits established in Title 46 – Series 12 of the code of State Rules (46CSR12) (included in table 60-3B of the Rule and Appendix C of this guidance).
- RBC lookup values from Table 60-3B of the Rule and reproduced in Appendix C of this guidance.
- Natural background values for inorganics (see Subsection 2.5 and Appendix B).

3.2.3 Implementing the De Minimis Standards

For soils, after concentrations of COCs are determined, the values are compared to the RBC for residential or industrial land uses, as appropriate. If the 95th percentile upper confidence limit (UCL) on the mean (see Subsection 2.5 and Appendix B) or the maximum value of each COC is below the RBC value and ecological receptors are not of concern, then no further site assessment or remediation needs to occur (see Figure 3-1). If the natural background level exceeds the RBC value, the background level may be used as the standard. If the contaminant concentrations exceed both levels, the applicant has the option of reducing the contaminant concentrations to the higher of the background or RBC value, or determining an alternative remediation standard using the Uniform or Site-Specific Standards (Subsections 3.3 or 3.4 of this guidance). Sample locations that are clearly part of a different population should be evaluated separately (see Gilbert Appendix B – has statistical methods for 1987 determining outliers).

For groundwater, if COCs in each monitoring well are below the values listed in Appendix C, then no further assessment is necessary. If some inorganics exceed the RBCs but have the potential to be within background, then background groundwater concentrations should be evaluated using the procedures outlined in Subsection 2.5 and Appendix B. If background concentrations are higher than the RBC, background may be used for the standard. As with soils, if contaminant levels exceed RBCs or background, the applicant has the option of cleaning up to the De Minimis Standard or further evaluating the site using either the Uniform or Site-Specific Standard.

The De Minimis Standard may not be used for any contaminant at a site where the contaminant is impacting surface water.

3.3 Uniform Risk-Based Standard

The Uniform Risk-Based Standard described in §60-3-9.3 of the Rule relies on uniform, approved methodologies, exposure factors, and other input variables to calculate remediation standards. Site-specific variables may replace default variables with adequate technical justification. The remediation standards will be protective of human health based on current or reasonably anticipated future land and water use. Applicants who select the Uniform Standard need not meet the De Minimis Standard.

USEPA (1991a, 1996b,c) has developed standard default risk equations for typical exposure pathways; it is those exposure pathways that are considered in the Uniform Standard. The equations used in the Uniform Standard consider the following residential exposure pathways:

- Ingestion from drinking groundwater
- Ingestion from drinking surface water
- Inhalation of volatile from groundwater
- Inhalation of volatiles from surface water

- Ingestion of soil
- Inhalation of volatiles and particulates from soil
- Soil concentrations protective of groundwater

The equations used in the Uniform Standard considers the following industrial exposure pathways:

- Ingestion of soil
- Inhalation of volatiles and particulates from soil

For any land use, soil concentrations that are protective of groundwater must also be determined. Any major exposure pathways that were not listed (e.g., ingestion of water during recreational use of a surface water body) for this standard may need to be evaluated under the Site-Specific Standard.

The default assumptions and equations used to determine values for remediation levels under the Uniform Standard are found in Appendix D of this guidance. Site-specific information may be substituted for any of the default values listed provided that the justification for the site-specific value is adequately documented. Where significant risks occur from more than one pathway, cleanup levels determined from the Uniform equations should be adjusted to consider cumulative risks. It should be noted that the Uniform Risk-Based Standard equations in Appendix D are not appropriate for lead. Lead in drinking water must meet the WV Groundwater Standard. Lead in soils must meet either the De Minimis Standards or the method for deriving lead standards presented in Appendix F of this guidance may be used to derive a cleanup value.

As with the De Minimis Standard, not all sites may be appropriate for evaluation using the Uniform Standard approach. For example, Uniform Standard methods for assessment of contaminated sediments are not provided. An accurate conceptual site model is crucial in determining whether Uniform Standards will be sufficient to guide remediation decisions at a site (see Subsection 2.2.4 of this guidance for a description of conceptual site model development).

3.3.1 Uniform Standards for Groundwater

The Uniform Standard for groundwater includes consideration of inhalation and ingestion of groundwater. For sites where the groundwater is potable, the Uniform Standard is not likely to differ from the De Minimis Standards that are described in Subsection 3.2.2 of this guidance. The standard applied at such sites would be the higher of the MCL, the risk-based uniform standard based on ingestion exposures (see Appendix D), or the natural or anthropogenic background concentration for each COC.

For sites where chemical contaminants are present in groundwater, the methods described in Appendix D of this guidance should be used to derive a risk-based Uniform Standard.

For sites where potability or groundwater use may be an issue, §60-3-9.3.c of the Rule states that the Uniform Standard for groundwater shall be derived based on current or reasonably anticipated future land and water uses, the potential for migration of contaminants, and the usefulness of the aquifer as a source of drinking water. Groundwater that has a background total dissolved solids content greater than 2500 milligrams per liter (mg/L) is probably not useful as a source of drinking water. If it is suitably demonstrated that the groundwater is not and cannot serve as a source of drinking water and that the aquifer is not hydrogeologically connected to an aquifer being used for drinking water, the groundwater is not suitable as a source of drinking water.

3.3.2 Uniform Standards for Soil

The Uniform Standards for soil are based on USEPA's soil screening guidance (USEPA 1996b and c). In the soil screening guidance, screening levels are provided for two exposure routes, ingestion and inhalation. For volatile chemicals inhalation of vapors is considered, while for nonvolatile chemicals inhalation of particulates is included. The methods described in Appendix D of this guidance should be used to derive a risk-based Uniform Standard that accounts for exposures occurring by both ingestion and inhalation. Site-specific adjustments may include consideration of site data regarding the relative oral bioavailability of chemicals in soil (see Appendix E of this guidance), site data pertaining to the flux rates of volatile chemicals from soil, or site or regional data modifying assumptions about particulate releases to air.

The soil screening guidance also includes screening levels that provide varying degrees of protection for migration of chemicals from soil to groundwater. Two sets of values are provided based on dilution and attenuation factors (DAFs) of 20 and 1. Site specific DAFs may be developed with appropriate documentation. The standards for soil concentrations that are protective of groundwater were derived by USEPA using a complex model to predict contaminant migration from soil to groundwater in a two stage process: 1) release of contaminant in soil leachate; and 2) transport of the contaminant through the underlying soil and aquifer to a receptor well. The USEPA methodology is described in detail in Soil Screening Guidance: Technical Background Document (USEPA, 1996b). The USEPA document also provides guidance for making site-specific adjustments to the default standards.

In cases where risk-based or groundwater protection Uniform Standards are exceeded by anthropogenic background concentrations, the background value may be used to determine the need for remediation.

3.3.3 Establishing the Uniform Standards

For known or suspected carcinogens, acceptable clean-up levels may be established using Uniform Standards established at levels that represent an excess upper bound lifetime risk of between one in ten thousand (1×10^{-4}) to one in one million (1×10^{-6}). Special notification must be given for those non-brownfield sites where remediation levels will exceed the one in one hundred thousand (1×10^{-5}) level of risk for industrial sites and the 1×10^{-6} risk for residential sites (§60-3-9.3.d of the Rule). Risks should be characterized by the quantification of cumulative risks posed by multiple contaminants. For cumulative site risk, the cumulative site risk shall not exceed one in ten thousand (1×10^{-4}) (§60-3-9.3.g of the Rule).

For individual systemic toxicants, the Uniform Standards shall represent levels to which the human population could be exposed without appreciable risk of deleterious effect. For the Uniform Standard, the hazard quotient (HQ) shall not exceed one (1.0) for any individual or group of toxicants that act on the same target organ (§60-3-9.3.e of the Rule). Where multiple systemic toxicants affect the same target organ or act by the same method of toxicity, the Hazard Index (sum of the hazard quotients) shall not exceed 1.0. Where multiple systemic toxicants do not affect the same organ, the hazard index shall not exceed 10.0 (§60-3-9.4.b of the Rule). If the Hazard Index exceeds 1.0, further evaluations may be necessary as discussed in Section 3.4.1.3, Approach for Calculating NonCancer Risks. Consult the Integrated Risk Information System (IRIS) (www.epa.gov/ngispgm3/iris) or Health Effects Assessment Summary tables (HEAST) databases for the most recent information on target organs/systems affected by various chemicals.

If a contaminant exhibits both carcinogenic and noncarcinogenic effects, then the more conservative risk-based standard (i.e. the lower of the two values) shall be used as the remediation standard (§60-3-9.2e of the Rule).

Either natural or anthropogenic background concentrations may be used as the Uniform Standard. Background concentrations of anthropogenic constituents vary greatly depending upon regional sources and local conditions. The most critical consideration in developing an anthropogenic background will be to demonstrate that the anthropogenic levels found are from area-wide sources not related to site activities. Methods for determining background are provided in Subsection 2.5 and Appendix B of this guidance.

3.3.4 Uncertainty Analysis

It is important to specify the uncertainties associated with the assumptions made in developing the Uniform Standard to put the standard in proper perspective. Highly quantitative statistical uncertainty analysis is usually not practical or necessary. As in all environmental risk assessments, it is already known that uncertainty about the numerical results are generally large (i.e., on the range of an order of magnitude or greater). Consequently, it is more important to identify the key site-related variables and assumptions that contribute most to the uncertainty than precisely quantify the degree of uncertainty in the risk assessment (USEPA, 1989). USEPA (1989) suggests a format for qualitatively identifying uncertainty associated with risk calculations (see Exhibit 6-21, USEPA, 1989) which should be adequate for evaluating uncertainties associated with development of the Uniform Standard.

3.3.5 Attaining Compliance with the Uniform Standard

3.3.5.1 Soils

For soils, compliance with the Uniform Risk-Based Standards, or with the background level that has been equated to any of the standards, is achieved when a measure of the average contaminant concentration on the site or in the exposure unit is equal to or less than the standard. A measure of the average contaminant concentration is the appropriate value to compare to the standards because risk-based standards represent averages themselves (see USEPA 1996b).

Because average concentrations are uncertain, the 95% UCL on the mean concentration should be calculated for all soil samples within the site or exposure unit. The UCL can be calculated following the procedure given in USEPA (1992a). The UCL, which provides a measure of the average concentration that accounts for its uncertainty, is compared to the standard. If the UCL is less than the standard, then remediation is complete. Note that this procedure does not require each sample location on the site to have a contaminant concentration less than the standard. From a risk basis, it is only necessary for the average contaminant concentration to be less than the standard. Sample locations that are clearly part of a different population should be evaluated separately (see guidance such as USEPA, 1996a).

3.3.5.2 Groundwater

Because of the site-specific factors related to groundwater, there are several methods that can be used to demonstrate compliance with the Uniform Standard. Approvable compliance demonstrations include comparison of highest level in any well to the standard, statistical comparison of results from select wells to the standard, or other reasonable methods as approved by the Director. When an acceptable demonstration is made that site levels are below the Uniform risk-based standard, the applicant has attained compliance and no additional remediation will be required.

The following is a list of factors to consider when deciding upon the method to be used to demonstrate compliance:

- In most situations, it is recommended that a statistical evaluation of the groundwater be conducted. An approved and acceptable method is to calculate a one-sided 95% upper confidence level on the mean on selected wells.
- Selection of wells to be used is supported by the site characterization and any relevant risk assessment. The wells must be part of the same population (e.g., wells within a plume of contamination).
- If there is an insufficient number of wells (or samples) to do statistical evaluation, then the results from each well may need to be compared to the standard. In this case, all results would need to be below the standard to demonstrate compliance. It should be noted that the applicant has the right to install additional wells to be able to do a statistical evaluation.
- If areas of contamination (i.e. plume) exist that are at least an order of magnitude higher than surrounding concentrations, those may need to be evaluated separately.
- The applicant may use other statistical methods or evaluated techniques provided they are shown to be appropriate and adequate and are approved by the Director.

3.4 Site-Specific Risk-Based Standard

The Site-Specific Standard (described in §60-3-9.4 of the Rule) relies on a baseline or residual risk assessment conducted by the applicant, as described in §60-3-8.4 of the Rule. All sites or portions of sites qualify for the Site-Specific Standard, but some sites may be more easily or economically remediated using De Minimis or Uniform Standard methods. Site-specific remediation standards must take into account current and reasonably anticipated future land and water use expectations and the use of institutional or engineering controls, if applicable.

Critical review of the site conceptual model is the first step in determining whether a baseline risk assessment needs to be conducted. As described in Subsection 2.2.4 of this guidance, the site conceptual model describes potential receptors and potentially complete exposure pathways. The complexity of the conceptual model, i.e., the kinds of affected media, number of complete exposure pathways, and exposure scenarios will determine the need for a baseline risk assessment. At this stage the conceptual site model, list of complete exposure pathways, and list of COCs should be reviewed to determine if any revisions are needed based on currently available information.

Prior to undertaking a baseline risk assessment, the adequacy of available data to support a risk assessment should be determined. Data requirements for risk assessment are listed in Subsection 2.3.1 of this guidance. Guidance on developing data quality objectives (DQOs) for risk assessment purposes may be found in Guidance for Data Usability in Risk Assessment (Part A) Final (USEPA, 1992b). Particular attention should be paid to whether DQOs have been met (see Subsection 2.4.1 of this guidance). DQOs from the site assessment should be reviewed and refined for the risk assessment process.

The following subsections provide guidance for conducting the baseline risk assessment, followed by guidance for implementing the Site-Specific Standard. For point estimates, more detailed guidance is provided in Appendix H. Probabilistic risk assessments are discussed in Appendix I.

3.4.1 Baseline Risk Assessment

The guidance provided in this subsection may be applied to both baseline risk assessment and to residual risk assessments. The primary source of guidance for the conduct of baseline risk assessments is found in the Risk Assessment Guidance for Superfund: Volume I – Human Health Evaluation Manual (Part A) (USEPA, 1989). When evaluating risks of exposure to lead, USEPA's childhood lead exposure model should be used for residential land uses or other land uses where young children may be exposed frequently. For industrial / commercial land uses, USEPA's adult lead exposure model should be used. Alternative models with appropriate documentation may be used for evaluating lead exposures to adults with the approval of the Director. Lead risk assessment guidance is provided in Appendix F of this document. The methods described in the following subsections do not apply to evaluating contamination by radionuclides. Until specific guidance is issued by WVDEP, evaluation of radionuclide contamination should be conducted in accordance with current USEPA guidance.

3.4.1.1 Exposure Assessment

The objective of the exposure assessment is to estimate the type and magnitude of exposures to the chemicals of potential concern that are present at or migrating from a site. Exposure assessments are conducted to estimate the magnitude of potential human exposures, the frequency and duration of these exposures and the pathways by which humans are potentially exposed. Exposures are quantified only for complete exposure pathways. A complete exposure pathway includes four elements:

- A source of a chemical (s)
- A mechanism of release, retention or transport of a chemical in a given medium (e.g., air, water, soil)
- A point of human contact with the exposure medium (i.e., exposure point) and
- A route of exposure at the point of contact (i.e., ingestion, inhalation or dermal contact).

Without the potential for all of these elements, an exposure pathway is not complete and does not need to be quantified.

For complete exposure pathways, reasonable maximum exposure (RME) estimates for both current and future land-use assumptions are to be determined. Current exposure estimates are used to determine whether a threat exists based on existing exposure conditions at the site. Future exposure estimates are used to provide decision-makers with an understanding of potential future exposures and threats under reasonably anticipated future land and water uses. Future exposure assessments include a qualitative estimate of the likelihood of such exposures occurring. In addition to deterministic estimates of the RME, probabilistic exposure estimates may be conducted in accordance with §60-3-8.7 of the Rule and Section 6 and Appendix I of this guidance to better quantify uncertainty and assist in risk management decisions at the site.

An exposure assessment involves a broad three-step process:

STEP 1: Characterizing Exposure Setting

- Analyzing physical setting (climate, vegetation, groundwater hydrology, surface water, soil type, etc.)
- Identifying exposed and potentially exposed populations (from site visit, population surveys, topographic, housing or other maps, etc.) For such things as location relative to the site, activity patterns, and the presence of sensitive subpopulations.
- Determining current land use and reasonably anticipated future land use (and current and future population characteristics)

STEP 2: Identifying Exposure Pathways

- Estimating chemical sources, releases and receiving media
- Identifying all complete pathways of exposure. Each exposure pathway describes a unique mechanism by which a population may be exposed.
- Identifying COCs for each medium
- Evaluating fate and transport in release media (i.e., accumulation, transformation, transport, etc.)

STEP 3: Quantifying Pathway-Specific Exposure (based upon exposure concentrations and intake variables)

- Estimate the exposure concentrations separately for groundwater, surface water, sediments, soil, food, and air, as appropriate. Because of the uncertainty associated with any estimate of exposure concentration, the upper confidence limit (the 95% upper confidence limit) on the arithmetic average will be used for this variable. It is necessary to verify whether the data is normal or log normal and calculate the UCLS accordingly, using the procedures outlined in Appendix B.
- Estimating contaminant intakes for each exposure route for each complete pathway.

Most of Steps 1 and 2 will be completed during development of the site conceptual model, and will be reviewed and updated as the exposure assessment is started. Step 3 will be the most time consuming aspect of the exposure assessment. Exposure is defined as the contact of a human with a chemical contaminant. The magnitude of the exposure is determined by measuring or estimating the intake or dose of the contaminant in milligrams (mg) of chemical per kilogram body weight (kg). These intake estimates require site-specific data on chemical concentrations that people are likely to encounter to be combined with estimates of intake rates for the medium the chemical is in, e.g., the amount of soil ingested by a child each day. The generic formula for determining contaminant intakes follows:

Equation 3-1:

$$\text{Intake (mg/kg body weight-day)} = \frac{C * CR * EFD * 1/AT}{BW}$$

Where:

- C = chemical concentration (e.g., mg/L water)
- CR = contact rate with the medium containing the chemical (e.g., L/day)
- EFD = exposure frequency (days per year) and duration (years) (EF * ED yields days for units)
- BW = body weight (kg) – 70 kg

AT = averaging time (days) – (e.g., 70 yr for carcinogens,
30 yr for noncarcinogens)

The intakes that may need to be determined include, but are not limited to:

- Intake from drinking ground and/or surface water
- Intake from incidental ingestion of surface waters while swimming
- Intake from dermal contact with ground and/or surface water
- Incidental ingestion of soil, sediment and/or dust
- Dermal intake of soil, sediment and/or dust
- Intake through inhalation of vapor-phase chemicals
- Intake through inhalation of particulate phase chemicals
- Intake through foods

Detailed formulas and assumptions for calculating exposure by many of these pathways is provided in Appendix D. USEPA has issued numerous documents providing guidance and data for use in exposure assessments. These documents are the preferred sources of methods and assumptions to be used by the applicant in site-specific risk assessments.

- United States Environmental Protection Agency (USEPA). 1989. Risk Assessment Guidance for Superfund: Volume I - Human Health Evaluation Manual (Part A). Office of Emergency and Remedial Response. EPA/540/1-89/002.
- United States Environmental Protection Agency (USEPA). 1991a. Risk Assessment Guidance for Superfund: Volume I - Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals) Interim. Office of Emergency and Remedial Response. Publication 92585.7-01B. December.
- United States Environmental Protection Agency (USEPA). 1992c. Dermal Exposure Assessment: Principles and Applications. Office of Research and Development, Washington, DC. EPA/600/8-91/011B.
- United States Environmental Protection Agency (USEPA). 1991b. Standard Default Exposure Factors for Superfund. Office of Emergency and Remedial Response. OSWER Directive 9285.6-03.

A massive compilation of data for use in exposure assessment can be found in USEPA's Exposure Factors Handbook. The August 1997 version is available on-line at <http://www.epa.gov/ordntrnt/ORD/WebPubs/exposure/index.html>. Adobe Acrobat reader is

needed to read these PDF documents. The USEPA web site should be checked for availability of more recent versions.

- United States Environmental Protection Agency (USEPA). 1997. Exposure Factors Handbook. Volume I-General Factors. EPA/600/P-95/002Ba.
- United States Environmental Protection Agency (USEPA). 1997. Exposure Factors Handbook. Volume II-Food Ingestion Factors. EPA/600/P-95/002Bb.
- United States Environmental Protection Agency (USEPA). 1997. Exposure Factors Handbook. Volume III-Activity Factors. EPA/600/P-95/002Bc.

When estimating ingestion intakes, it may be appropriate to adjust the intake estimate to account for different degrees of absorption from different exposure media. The need for such an adjustment arises when the exposure medium differs from the exposure medium in toxicity studies relied on for the toxicity assessment (see Subsection 3.4.1.2 of this guidance). Such adjustments are most commonly performed to account for reduced absorption of chemicals from soil compared to absorption of dissolved chemical administered in water. Detailed guidance for making relative bioavailability adjustments is provided in Appendix E of this document.

Exposure assessments for contaminants where significant risk is contributed via food ingestion should consider uptake from soil into plants or uptake into fish, as appropriate. Contaminants such as certain organic mercury compounds or organic compounds with a high octanol-water partition coefficient may exhibit dramatic increases in concentration in seafood; thus, the food ingestion pathway may dominate risk estimates. Where subsistence hunting and fishing represent a significant part of the diet of subpopulations, appropriate biomagnification factors should be incorporated into risk equations.

Summary of Exposure Information Needed

- Estimated intakes (chronic, subchronic, and short-term, as appropriate) for each contaminant of concern
- Important exposure modeling assumptions, including:
 - contaminant concentrations at exposure points
 - frequency and duration of exposure
 - absorption assumptions (bioavailability, etc.)
 - characterization of uncertainties

List which exposure pathways can reasonably contribute to the exposure of the same individuals over the same time period.

Uncertainties Related to Exposure Assessment

The exposure assessment should conclude with a discussion of sources of uncertainty in the estimate. Common sources of uncertainty include:

- Adequacy and completeness of the sampling data (e.g., how well the data represents actual site conditions, or whether or not all COCs have been identified)
- Identification of all potential exposure pathways and routes of exposure
- Accuracy of exposure assumptions (i.e., how well assumptions reflect actual exposure conditions).

The nature and likely magnitude of effect of each source of uncertainty on exposure estimates should be discussed.

3.4.1.2 Toxicity Assessment

The purpose of a toxicity assessment is to evaluate the potential for substances of potential concern to cause adverse health effects in exposed persons and to define, as thoroughly as possible, the relationship between the extent of exposure to a hazardous substance and the likelihood and severity of any adverse health effects. Standard procedures for a toxicity assessment include identifying toxicity values for carcinogenic and noncarcinogenic effects and summarizing other relevant toxicity information. WVDEP relies on toxicity values, developed and verified by USEPA, to describe the dose-response relationship. If verified toxicity values for a COC are not available from USEPA, the applicant should consult with WVDEP prior to relying on other sources of toxicity values. Complete copies of all references used to support alternate toxicity values must be provided to WVDEP upon request.

USEPA-derived toxicity values used in risk assessments are termed carcinogenic slope factors (CSFs), reference doses (RfDs), and reference concentrations (RfCs). Oral slope factors are used to estimate the incremental lifetime risk of developing cancer corresponding to ingested doses calculated in the exposure assessment. Some chemicals also have inhalation slope factors that are used to assess inhaled chemicals. The potential for noncarcinogenic health effects of ingested chemicals is typically evaluated by comparing estimated daily intakes with RfDs, which represent daily intakes at which no adverse effects are expected to occur over a lifetime of exposure. Both CSFs and RfDs are specific to the route of exposure (e.g., ingestion [oral] exposure).

Currently, there are no CSFs or RfDs for dermal exposure; therefore, route-to-route extrapolation is necessary to assess dermal exposure. This process is described in Appendix E. No toxicity values are available for lead. Instead, USEPA relies on benchmark values for blood lead levels that are health protective. Exposures are assessed by comparing the blood lead benchmark values with blood lead levels predicted by pharmacokinetic models that estimate blood lead levels resulting from specified doses of lead. These models are described in Appendix F of this guidance.

The primary source for USEPA-derived toxicity values is USEPA's IRIS data base. This computerized database contains verified toxicity values in addition to up-to-date health risk and USEPA regulatory information for many substances commonly detected at hazardous waste sites. USEPA's Health Effects Assessment Summary Tables (USEPA 1993) also provide USEPA-derived toxicity values that may or may not be verified at the time of publication.

Because toxicity information may change rapidly and quickly become outdated, care should be taken to find the most recent information available. IRIS is updated monthly, provides verified RfDs and slope factors, and is the USEPA's preferred source of toxicity information. Information in IRIS supersedes all other sources. Only if values are unavailable in IRIS for the contaminant of concern should other information sources be consulted. Toxicity values which have been withdrawn from IRIS may be used in the risk assessment provided a discussion is included on the uncertainty associated with using these values. The USEPA web server has published the updated IRIS list. It is available at: http://www.epa.gov/ngispgm3/iris/Substance_List.html or call the IRIS User Support at (513) 569-7254.

Toxicity Information Needed

For each COC included in the risk assessment, a toxicity profile or a tabular representation of the information should be provided that includes the following elements:

- carcinogenicity of the chemical (e.g., oral and/or inhalation slope factors verified by USEPA, critical study(ies) upon which the slope factors are based (including the exposure/dosing medium), weight of evidence and carcinogenicity classification, and type of cancer observed for all Class A carcinogens)
- systemic toxicity of the chemical, (e.g., chronic and subchronic RfDs and RfCs, the critical effect associated with each RfD and RfC (e.g., kidney damage). critical study(ies) upon which the RfD and/or RfC is based (including the exposure/dosing medium), uncertainty factors and modifying factors used in deriving each RfD/RfC, and "degree" of confidence in each RfD (i.e., high, medium, or low))
- pharmacokinetic data that may affect the extrapolation from animals to humans for both the RfD and the slope factor
- the degree of absorption from various media
- uncertainties in any route-to-route extrapolations

Sample table needed.

For a more detailed evaluation of the toxicity of a compound, ATSDR profiles may be included in an appendix.

Noncarcinogenic Assessment

Currently USEPA derives RfDs by applying uncertainty factors to a no observed adverse effect level (NOAEL) or from a lowest observed adverse effect level (LOAEL) for each chemical. Another method of deriving RfDs is called the benchmark dose (BMD) approach (USEPA, 1995). The BMD is a dose of a chemical that is predicted to result in a specified amount of increased response compared to unexposed controls. In the BMD approach a dose-response model is applied to toxicity data. Toxicity information used to IRIS to derive BMDs should be obtained from the IRIS database, if available. A statistical lower bound on the BMD (termed the BMDL) may be used as a substitute for the traditional NOAEL or LOAEL method of deriving RfDs.

Inhalation RfCs

The inhalation RfC is analogous to the oral RfD and is likewise based on the assumption that thresholds exist for certain toxic effects. The inhalation RfC considers toxic effects for both the respiratory system (portal-of-entry) and for effects peripheral to the respiratory system (extrarespiratory effects). It is expressed in units of mg/cu.m. In general, the RfC is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily inhalation exposure of the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. Methods for deriving RfCs can be found in Interim Methods for Development of Inhalation Reference Doses (USEPA, 1990) and Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry (USEPA, 1994a)

Carcinogenicity Assessment

The carcinogenic assessment includes three aspects for the substance in question; the weight-of-evidence judgment of the likelihood that the substance is a human carcinogen, and quantitative estimates of risk from oral exposure and from inhalation exposure. The quantitative risk estimates are presented in three ways. The slope factor is the result of application of a low-dose extrapolation procedure and is presented in milligrams per kilogram-day [mg/kg-day]⁻¹. The unit risk is the quantitative estimate in terms of either risk per micrograms per liter (µg/L) drinking water or risk per micrograms per cubic meter (µg/m³) air breathed. The third form in which risk is presented is a drinking water or air concentration providing cancer risks of 1 in 10,000, 1 in 100,000 or 1 in 1,000,000. The rationale and methods used by USEPA to develop the carcinogenicity information in IRIS are described in The Risk Assessment Guidelines of 1986 (EPA/600/8-87/045) and in the IRIS Background Document. IRIS summaries developed since the publication of USEPA's more recent Proposed Guidelines for Carcinogen Risk Assessment also utilize those guidelines where indicated (Federal Register 61(79):17960-18011, April 23, 1996.

Uncertainties Related to Toxicity Assessment

Sources of uncertainty in the toxicity assessment should be identified. Typical sources of uncertainty include:

- Using dose-response information from effects observed at high doses to predict the adverse health effects that may occur following exposure to the low levels expected from human contact with the agent in the environment
- Using dose-response information from short-term exposure studies to predict the effects of long-term exposures, and vice-versa
- Using dose-response information from animal studies to predict effects in humans
- Using dose-response information from homogeneous animal populations or healthy human populations to predict the effects likely to be observed in the general population consisting of individuals with a wide range of sensitivities

The likelihood and relative magnitude of each source of uncertainty should be discussed. For example, USEPA states that the range of possible values around RfDs is “perhaps an order of magnitude” (USEPA, 1995).

3.4.1.3 Risk Characterization

Risk characterization is the final step of the baseline human health risk assessment process. In this step the exposure and toxicity assessments are integrated into quantitative and qualitative expressions of risk. To characterize the potential noncarcinogenic effects of ingested chemicals, comparisons are made between projected intakes of contaminants and RfDs. For inhalation exposures, exposure point concentrations in air may be directly compared to the RfC. To characterize the potential carcinogenic effects, the probabilities that an individual will develop cancer over a lifetime of exposure are estimated from projected intakes and CSFs. The major assumptions, scientific judgments, and the uncertainties embodied in the risk assessment should also be presented.

Cancer and noncancer health risks are estimated, assuming long-term exposure to chemicals detected at the site. The risk characterization methods described in USEPA guidance (USEPA 1989) are used to calculate upper-bound excess lifetime cancer risks for potential carcinogens and hazard indices for chemicals with noncancer health effects. Following USEPA guidance, numerical estimates of risk should be rounded to one significant figure to reflect the level of certainty associated with calculated risks. Risks associated with exposures to lead may be assessed following the methods described in Appendix F.

Risk characterizations are completed in the following broad steps:

STEP 1: Organize Outputs of Exposure and Toxicity Assessments

- Exposure duration
- Absorption adjustments
- Consistency check

STEP 2: Quantify Pathway Risks for Each Substance

- Estimate cancer risk for each contaminant
- Estimate the Noncancer HQ for each contaminant
- Calculate total cancer risk for each exposure pathway
- Calculate noncancer HI for each exposure pathway

STEP 3: Combine Risks Across Pathways that affect the same individual(s) over the same time periods:

- Sum Cancer Risks
- Sum Hazard Indices

STEP 4: Assess and Present Uncertainty

- Site-specific factors
- Toxicity assessment factors

STEP 5: Consider Site-Specific Human Health or Exposure Studies

- Compare adequate studies with results of risk assessment

STEP 6: Summarize Results of the Baseline Risk Assessment

Approach for Calculating Cancer Risks

Quantifying total excess cancer risk requires calculating risks associated with exposure to individual carcinogens and aggregating risks associated with simultaneous exposure to multiple carcinogenic chemicals. Cancer risks for a single carcinogen are calculated by multiplying the

carcinogenic chronic daily intake (CDI)³⁴ of the chemical by its carcinogenic slope factor as follows:

$$\text{Risk} = \text{CDI} \times \text{CSF}$$

Where:

- Risk = a unitless probability (e.g., 1×10^{-6}) of an individual developing cancer over a 70-year lifetime
- CDI = chronic daily intake averaged over 70 years (mg/kg-day)
- CSF = carcinogenic slope factor, expressed in $(\text{mg}/\text{kg}\text{-day})^{-1}$.

A 1×10^{-6} cancer risk represents a one-in-one-million additional probability that an individual may develop cancer over a 70-year lifetime as a result of the exposure conditions evaluated. This linear equation is valid only at risk levels less than approximately 1×10^{-2} . Where risks associated with chemical exposures are greater than this level, USEPA (1989) recommends using the following equation:

$$\text{Risk} = 1 - \exp(-\text{CDI} \times \text{CSF})$$

where:

- Risk = a unitless probability of an individual developing cancer over a 70-year lifetime
- Exp = the exponential
- CDI = chronic daily intake averaged over 70 years (mg/kg-day)
- CSF = carcinogenic slope factor, expressed in $(\text{mg}/\text{kg}\text{-day})^{-1}$.

Because cancer risks are assumed to be additive, risks associated with simultaneous exposure to more than one carcinogen in a given medium are aggregated to determine a total cancer risk for each exposure pathway (USEPA 1989). Where multiple exposure pathways exist, total cancer risks for each pathway are then summed for reasonable combinations of exposure pathways to determine the total cancer risk for the population of concern. Refer to Appendices D, H, and I for additional exposure pathway equations.

Approach for Calculating Noncancer Risks

In contrast with carcinogenic effects, potential noncancer effects are not expressed as a probability. Instead, these effects are expressed as the ratio of the estimated exposure over a specified time period to the RfD derived for a similar exposure period (e.g., CDI: chronic RfD). This ratio is termed a hazard quotient and is calculated as follows:

$$\text{HQ} = \text{CDI}/\text{RfD}$$

Where:

- HQ = hazard quotient
- CDI = chronic daily intake
- RfD = reference dose.

The estimated exposure and the reference toxicity factor are expressed in the same units and represent the same exposure period (i.e., chronic, subchronic, or shorter-term). Where available, toxicity factors for the noncarcinogenic effects of carcinogenic chemicals must also be included. If the chemical-specific CDI exceeds the RfD (i.e., the HQ is greater than 1), noncancer adverse health effects may be a concern. Exposures resulting in a HQ that is less than or equal to 1 are very unlikely to result in noncancer adverse health effects.

Hazard quotients for individual chemicals are summed for each exposure pathway to determine a noncancer hazard index as follows:

$$HI = CDI_1/RfD_1 + CDI_2/RfD_2 + \dots + CDI_i/RfD_i$$

Where:

| | | |
|------------------|---|---|
| HI | = | hazard index |
| CDI _i | = | chronic daily intake for the i th toxicant |
| RfD _i | = | reference dose for the i th toxicant |

Where multiple exposure pathways exist, HIs for each exposure pathway are then summed for reasonable combinations of exposure pathways to determine a total hazard index.

Typically, where a HI exceeds 1, an additional analysis is considered in accordance with USEPA guidance (USEPA 1989). Specifically, the target organ of the effect used as the basis for the reference toxicity factor (i.e., the critical effect) is reviewed. Where this review indicates the potential for the primary contributors to the calculated total HIs to operate via different target organs or mechanisms of toxicity, this potential is noted and the calculated HIs may be modified to reflect values associated with specific target organs or effects. The uncertainty factor associated with each toxicity factor is also reviewed to determine the validity of summing HQs.

Approach for Lead

For lead, the blood lead concentration estimates predicted by chemical-specific models are compared with benchmark values as described in Appendix F. USEPA's most recent methods for evaluating lead exposures should also be considered. To assess the potential health risks associated with lead exposures, the predicted blood lead concentrations derived using lead-specific models are compared with blood lead benchmark values. For assessing childhood lead exposures, USEPA uses a pharmacokinetic model to predict blood lead concentrations associated with specified lead doses (USEPA, 1994b). The benchmark value is that the predicted 95th percentile blood lead concentration for children be less than 10 micrograms per deciliter ($\mu\text{g}/\text{dL}$), the minimum blood lead concentration in young children at which the Center for Disease Control (CDC) recommend some type of follow-up (CDC, 1991). Use of this goal is interpreted as meaning that there is a 95-percent probability that an exposed child would have a blood lead concentration less than the target concentration of 10 $\mu\text{g}/\text{dL}$ (USEPA, 1994b). For assessing adult lead exposures, USEPA has selected protection of fetuses born to women exposed under the conditions assumed in the model as the target population for protection (USEPA, 1996g). Based on this determination, the benchmark value for blood lead in the fetus is also 10 $\mu\text{g}/\text{dL}$.

Approach for Radionuclides **[Reserved]**

3.4.1.4 Uncertainty Analysis

A description of the minimum requirements for the uncertainty analysis is provided for the Uniform Standard (see Subsection 3.3.4). For the site-specific standard, the uncertainties need to be much more explicitly analyzed. Risk managers, decision makers, and the public need to be aware of the uncertainties in the analysis in order to avoid becoming overly dependent upon quantitative representations of results and to assure that nonquantifiable values are also considered properly.

This section provides a brief overview of the goals of uncertainty, components of uncertainty analyses, some proposed methods to determine uncertainties, and recommended sources that more fully discuss uncertainty analysis. Uncertainty commonly surrounds the likelihood, magnitude, distribution, and implications of risks. As a critical dimension in the characterization of risk, uncertainties must be considered in terms of magnitude, sources, and character. There are three sources of uncertainty in risk assessments:

- *Inherent randomness* (Stochasticity). This type of uncertainty can be estimated but not reduced because it is a characteristic of the system being assessed.
- *Imperfect or Incomplete Knowledge of Things That Could be Known* (Ignorance). This is the “easiest” type of uncertainty to reduce or eliminate as it becomes less as the general knowledge bases about contaminant expand.
- *Error* (Mistakes in execution of assessment activities). This type of uncertainty can only be estimated.

Some additional reasons why uncertainties are desirable to have identified and addressed:

- Uncertain information from different sources of different quality must be combined for the assessment.
- Decisions need to be made about whether or how to expend resources to acquire additional information.
- Biases may result in so-called “best estimates” that in actuality are not very accurate.
- Important factors and potential sources of disagreement in a problem can be identified.
- Addressing uncertainties increases the likelihood that the results of an assessment will be used in an appropriate manner.

Table 3-1 illustrates common types of uncertainty that surround exposure assessments. A table such as this should be used to summarize the main sources of risk.

Table 3-1: Three Types of Uncertainty and Associated Sources and Examples for Exposure Assessment

| Type of Uncertainty | Sources | Examples |
|----------------------------|--|--|
| Scenario Uncertainty | Descriptive errors Aggregation errors Judgment errors Incomplete analysis | Incorrect or insufficient information Spatial or temporal approximations Selection of an incorrect model Overlooking an important pathway |
| Parameter Uncertainty | Measurement errors Sampling errors Variability Surrogate data | Imprecise or biased measurements Small or unrepresentative samples In time, space or activities Structurally – related chemicals |
| Model Uncertainty | Modeling errors | Excluding relevant variables |

Source: United States Environmental Protection Agency (USEPA). 1996d. Exposure Factors Handbook. Volume I-General Factors.

Part of the uncertainty analysis is to address the limitations of uncertainty analysis in risk assessments. These include, but are not limited to:

- Truly unexpected risks
- Unknown frequencies of risk to real events
- Cognitive biases that affect judgments about uncertainty, as well as risk
- The pressures caused by social, cultural, and institutional forces upon analysis and interpretation of uncertainty, and risk in general

Additional information on uncertainty analysis may found in the Exposure Factors Handbook. Volume I - General Factors, 1996d, USEPA, EPA/600/8-89/043; available on <http://www.epa.gov/docs/ordntrnt/ORD/WebPubs/exposure/index.html>

3.4.2 Implementing Site-Specific Risk-Based Standards

For individual known or suspected carcinogens, the remediation standard must be set to represent an excess upper-bound lifetime cancer risk of between one in ten thousand (1×10^{-4}) to one in one million (1×10^{-6}). Public notification is required if calculated residual cancer risks exceed the one in one million level (1×10^{-6}) for residential land use or the one in one hundred thousand (1×10^{-5}) level for industrial land use.

For individual systemic toxicants, remedial standards shall represent levels to which the human population could be exposed without appreciable risk of deleterious effect. For individual systemic toxicants, remedial standards shall represent levels where the hazard quotient shall not exceed 1.0 (two significant digits of accuracy) (§60-3-9.4.b of the Rule).

Where multiple systemic toxicants affect the same target organ or act by the same method of toxicity, the hazard index (sum of the hazard quotients) shall not exceed 1.0. Where multiple systemic toxicants do not affect the same organ the hazard index shall not exceed 10.0 (§60-3-9.4.b of the Rule). If the Hazard Index exceeds 1.0, further evaluations may be necessary as discussed in Section 3.4.1.3, Approach for Calculating Noncancer Risks.

3.4.2.1 Site-Specific Risk-Based Standards for Groundwater

Site-Specific Risk-Based remedial standards for groundwater shall be established using at least the following considerations:

- Potential receptors based on the current and reasonably anticipated future use of groundwater (§60-3-9.4.e.1 of the Rule).
- The potential for groundwater to serve as a drinking water source (§60-3-9.4.e.2 of the Rule), based on:
 - The total dissolved solids content greater than 2500 milligrams per liter (mg/L)
 - or-
 - It can be demonstrated to the director that the aquifer is not being used and cannot be used for drinking water, and
 - The aquifer is not hydrologically connected to an aquifer being used for drinking water;
- The site-specific sources of contaminants (§60-3-9.4.e.3 of the Rule);
- Natural environmental conditions affecting the fate and transport of contaminants (e.g., natural attenuation) (§60-3-9.4.e.4 of the Rule);
- Institutional and engineering controls (§60-3-9.4.e.5 of the Rule).

3.4.2.2 Site-Specific Risk-Based Standards for Surface Water and Sediments

Remediation standards for surface water and sediments should be established using at least the following considerations:

- Potential receptors based on the current and reasonably anticipated future use of the site
- The site-specific sources of contaminants
- Natural environmental conditions affecting the fate and transport of contaminants (e.g., natural attenuation)

- Institutional and engineering controls

Site-Specific Risk-Based Standards for surface water and sediments are likely to be based on recreational exposures.

3.4.2.3 Site-Specific Risk-Based Standards for Soil

Remedial standards for soil shall be established using at least the following considerations ((§ 60-3-9.4.f of the Rule):

- Potential receptors based on the current and reasonably anticipated future use of the site
- The site-specific sources of contaminants
- Natural environmental conditions affecting the fate and transport of contaminants (e.g., natural attenuation)
- Institutional and engineering controls

3.5 References

Federal Register. 1996. Proposed Guidelines for Carcinogen Risk Assessment . 61(79):17960-18011, April 23.

United States Centers for Disease Control (CDC). 1991. Preventing Lead Poisoning in Young Children. United States Department of Health and Human Services. October.

United States Environmental Protection Agency (USEPA). 1986. Superfund Public Health Evaluation Manual. Office of Emergency and Remedial Response. EPA 540/1-86/060.

United States Environmental Protection Agency (USEPA). 1989. Risk Assessment Guidance for Superfund: Volume I- Human Health Evaluation Manual (Part A). Office of Emergency and Remedial Response. EPA/540/1-89/002.

United States Environmental Protection Agency (USEPA). 1990. Interim Methods for Development of Inhalation Reference Doses. EPA/600/8-88/066F.

United States Environmental Protection Agency (USEPA). 1991a. Risk Assessment Guidance for Superfund: Volume I- Human Health Evaluation Manual (Part B, Development of Risk-based Preliminary Remediation Goals) Interim. Office of Emergency and Remedial Response. Publication 92585.7-01B. December.

United States Environmental Protection Agency (USEPA). 1991b. Standard Default Exposure Factors for Superfund. Office of Emergency and Remedial Response. OSWER Directive 9285.6-03.

United States Environmental Protection Agency (USEPA). 1992a. Risk Assessment Guidance for Superfund. Volume I: Human Health Evaluation Manual (Part A) - Supplemental Guidance: Calculating the Concentration Term. Office of Emergency and Remedial Response (Washington, DC). OSWER Directive 9285.6-03; Publication 9285.7-081; NTIS PB92-963373. May.

United States Environmental Protection Agency (USEPA). 1992b. Guidance for Datas Usability in Risk Assessment (Part A) Final. Office of Emergency and Remedial Response. Publication 9285.7-09A. April.

United States Environmental Protection Agency (USEPA). 1992c. Dermal Exposure Assessment: Principles and Applications. Office of Research and Development, Washington, DC. EPA/600/8-91/011B.

United States Environmental Protection Agency (USEPA). 1994a. Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry. EPA/600/8-90/066F. October.

United States Environmental Protection Agency (USEPA). 1994b. Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children. Office of Emergency and Remedial Response. Pub. No. 9285.7-15-1, EPA/540/R-93/081. February.

United States Environmental Protection Agency (USEPA). 1995. The Use of the Benchmark Dose Approach in Health Risk Assessment. Office of Research and Development. EPA/630/R-94/007. February.

United States Environmental Protection Agency (USEPA). 1996a. Superfund Guidance for Evaluating the Attainment of Cleanup Standards. Volume I: Soils and Solid Material. Center for Environmental Statistics. <http://www.epa.gov/ces/sfvoll/contents.htm>

United States Environmental Protection Agency (USEPA). 1996b. Soil Screening Guidance: Technical Background Document. Office of Solid Waste and Emergency Response. EPA/540/R-95/128. May.

United States Environmental Protection Agency (USEPA). 1996c. Soil Screening Guidance: User's Guide. Office of Solid Waste and Emergency Response. EPA/540/R-96/018. Publication 9355.4-23. July.

United States Environmental Protection Agency (USEPA). 1997a. Exposure Factors Handbook. Volume I - General Factors. EPA/600/P-95/002Ba.

United States Environmental Protection Agency (USEPA). 1997b. Exposure Factors Handbook. Volume II - Food Ingestion Factors. EPA/600/P-95/002Bb.

United States Environmental Protection Agency (USEPA). 1996f. Exposure Factors Handbook (Draft). Volume III-Activity Factors. EPA/600/P-95/002Bc.

United States Environmental Protection Agency (USEPA). 1996g. Recommendations of the Technical Review Workgroup for Lead for an Interim Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil. December.

4.0 ECOLOGICAL RISK-BASED STANDARDS

Under §60-3-9 of the Rule, remediation standards must adequately protect human health as well as the environment. The mechanism set forth to ensure compliance with the latter is the ecological assessment protocol (§60-3-9.1.b). This subsection outlines the procedures and requirements for preparing and conducting an ecological assessment. The procedures and requirements are not meant to be inclusive or comprehensive. Applicants undertaking an ecological assessment are directed to consult the references listed in Table 4-1 for general guidance and background information. However, it should be noted that requirements and stipulations outlined in this guidance and the Rule must take precedence in order to ensure compliance with the voluntary remediation program.

Table 4-1: Recommended Guidance Sources for the Execution of Ecological Risk Assessments

| |
|---|
| United States Environmental Protection Agency (USEPA). 1992. Framework for Ecological Risk Assessment. EPA/630/R-92/001. February |
| USEPA. 1997. Ecological Risk Assessment Guidance for Superfund: Process for designing and Conducting Ecological Risk Assessments. Interim Final. June 5. |
| USEPA . 1998. Guidelines for Ecological Risk Assessments 63 CFR 26846-26922 (1998) |
| USEPA; Region 3. 1991. EPA Region III Guidance on Handling Chemical Concentration Data Near the Detection Limit in Risk Assessments. Interim Final. November 4. |
| USEPA; Region 3. 1994. Use of Monte Carlo Simulations in Risk Assessment. EPA/903/F-94/001. February. |

An ecological assessment, much like its counterpart in human health risk assessment, has separate components of increasing detail and specificity. However, unlike the procedures for human health risk assessment all applicants are expected to perform a De Minimis Ecological Screening Evaluation. If the results of the De Minimis analysis indicate the presence of potential receptors of concern and complete pathways of exposure, then the applicant may elect to either undertake a Uniform Ecological Evaluation or proceed directly to the development of Site-Specific Ecological Risk-Based Standards. The conceptual site model (CSM) developed in Subsection 2.2.4 of this guidance provides the basis for the design of the ecological risk evaluation/assessment.

The three types of evaluation that constitute the ecological assessment protocols are developed in greater detail below:

- De Minimis Ecological Screening Evaluation—This is the first step in the ecological assessment process (§60-3-9.1.b.1). The De Minimis Screen is intended to determine whether ecological receptors of concern are exposed to site-related stressors. The De Minimis Ecological Screening Evaluation differs from the human health De Minimis Standard in that no quantitative standards are involved other than a comparison to water quality standards for

aquatic life (46 CSR 1). It is intended to simply evaluate whether any potential pathways of exposure to site contaminants exist. If exposure pathways exist and ecological receptors of concern are present, the criteria outlined in §60-3-9.5.a of the Rule should be used to further evaluate whether assessment is needed under the Uniform or Site-Specific Standards (See Figure 4-1). All applicants are expected to perform a De Minimis Ecological Screening Evaluation.

- **Uniform Ecological Evaluation**—If the De Minimis Ecological Screening Evaluation indicates that further assessment of ecological risk is needed, an applicant may elect to proceed to a Uniform Ecological Evaluation (§60-3-9.1.b.2). In this analysis, contaminant concentrations in soil and sediments are compared to generic benchmarks, approved by WVDEP, that reflect no significant ecological risk to specific receptors of concern. Contaminant concentrations in surface water are compared to surface water quality standards for the protection of aquatic life. If no surface water quality standard for the protection of aquatic life exists for a particular contaminant, the procedure outlined in 46 CSR 1, section 9 (CSR, 1996) may be used to develop benchmark values as comparison criteria. As in the human health Uniform Standard, if the benchmark values for media other than surface water are less than natural or anthropogenic background, the background concentrations are used as the comparison criteria. If a contaminant's concentration exceeds the comparison criterion, then the applicant may choose to remediate the environmental media using the criterion concentration as a remediation standard or develop a site-specific ecological risk-based value.
- **Ecological Site-Specific Risk-Based Standards**—If a valid exposure pathway exists and ecological receptors of concern (See Rule Sec. 2.14) are present, the applicant may choose to develop site-specific risk-based standards. This may be performed as a baseline ecological risk assessment where the specific attributes and parameters of the site and the receptor(s) of concern are used to determine their ecological risk from the contaminants. If the risk associated with the contaminant(s) exceeds the acceptable risk (as per 60.8.1.e), it may be necessary to remediate the site using site-specific values as remediation standards. As in the human health Site-Specific Standard, if the calculated values are less than natural or anthropogenic background, the background concentrations are used as the remediation standards. In addition, surface water quality standards for aquatic life must be met.

Local conditions may be considered to decide whether a site is degrading an aquatic habitat (§60-3-9.5.a.3). In cases where a site does not present an ecological risk over and above “local conditions” and further release of contaminants into the aquatic environment has been stopped, there will be no need for further evaluation beyond completion of the De Minimis checklist (Appendix C-2).

If no complete exposure pathway exists and the site does not meet any of the other criteria outlined above, then no further ecological analysis or remediation, based on ecological risk, is required (§60-3-9.5.c). If, however, the site meets any of the listed criteria in §60-3-9.5.a

and exposure pathways can be demonstrated to exist between the site contamination and any ecological receptors of concern, then the applicant may elect to undertake a Uniform Ecological Evaluation or proceed directly to the development of Site-Specific Ecological Standards. A flow chart illustrating this decision process is provided in Figure 4-2.

FIGURE 4-1

DEMINIMIS ECOLOGICAL SCREENING
EVALUATION (REVISED)

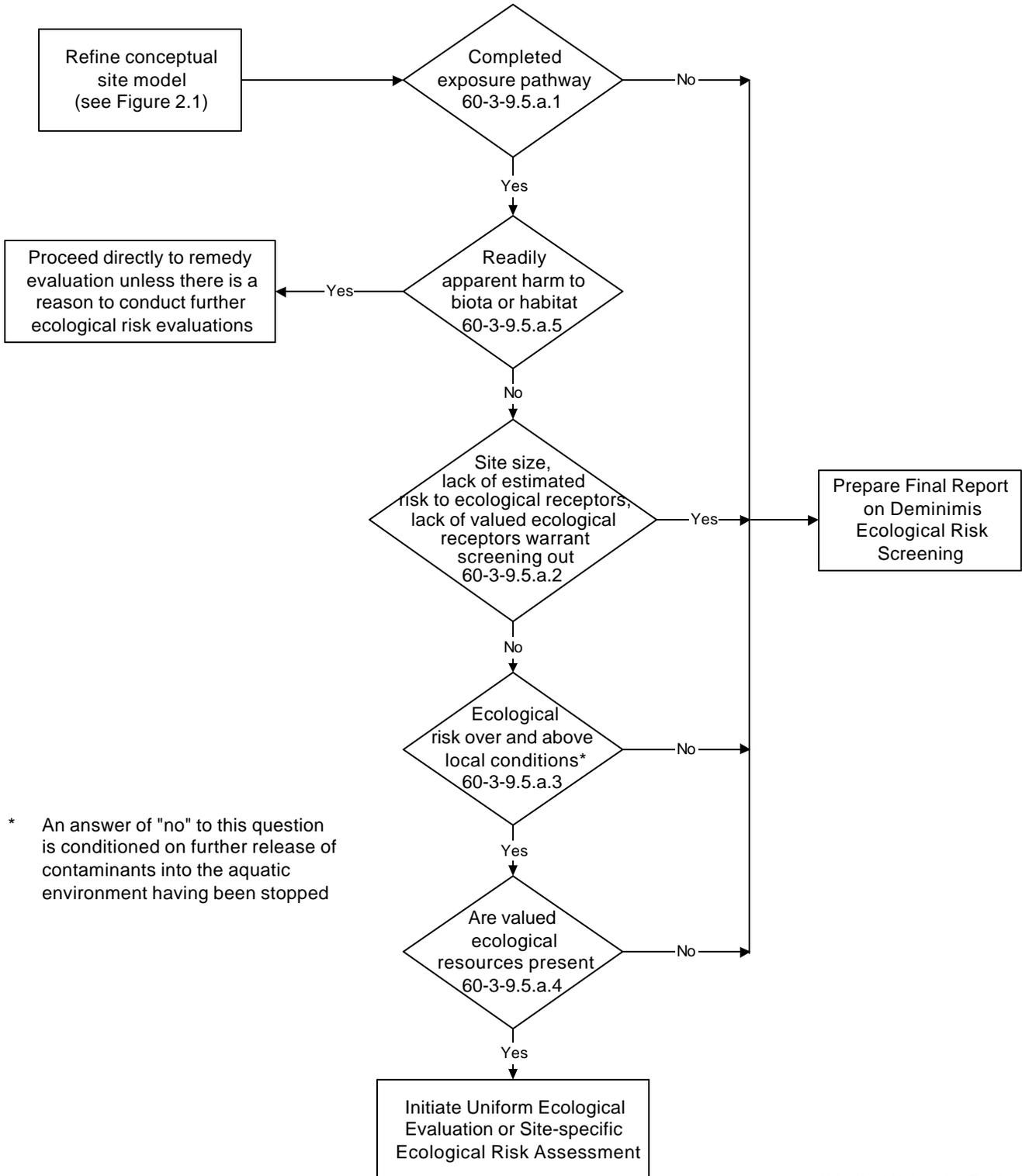
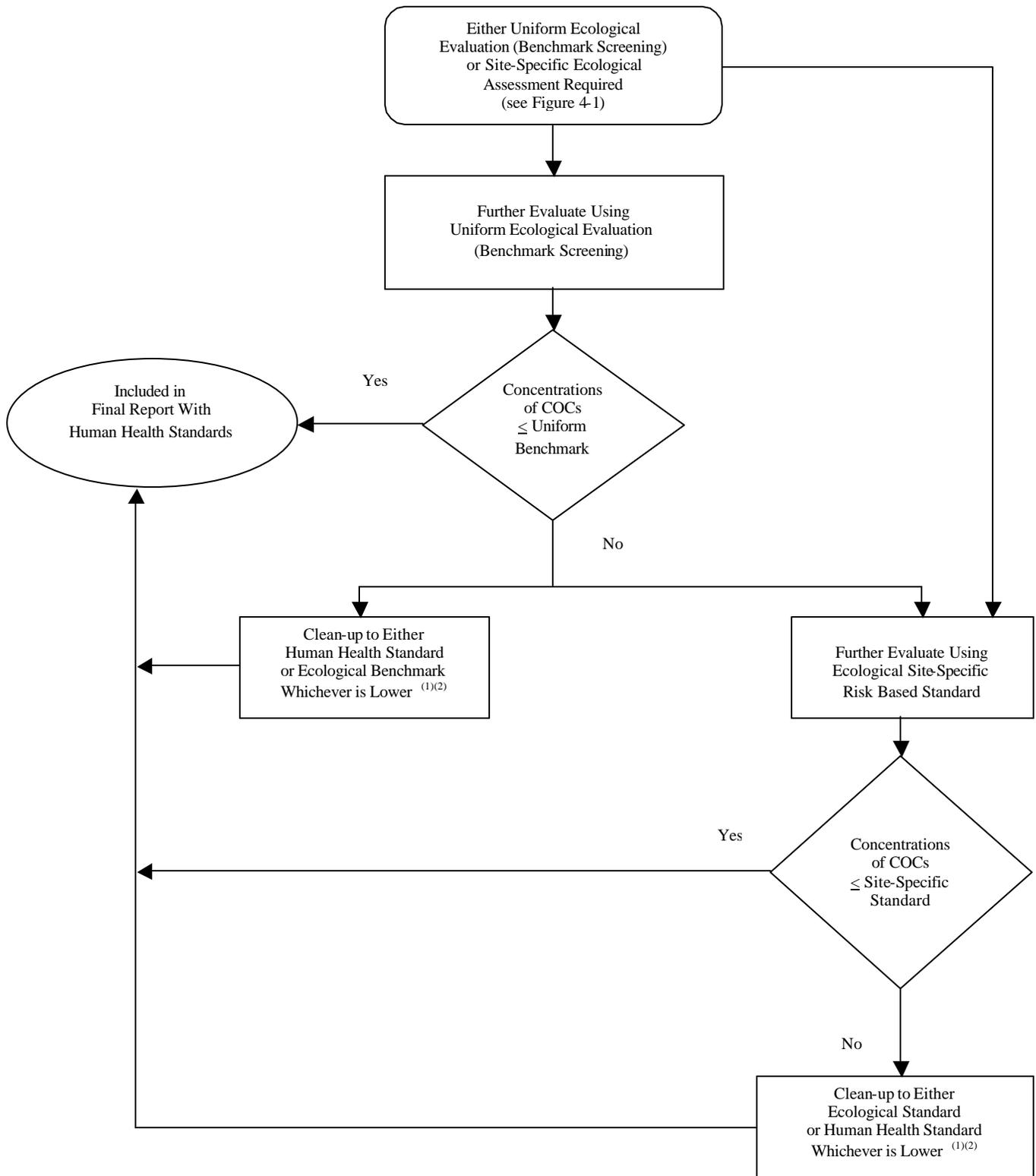


FIGURE 4-2: ECOLOGICAL RISK ASSESSMENT



- (1) Prior to clean-up the applicant must evaluate the remedial alternatives, submit a Remedial Action Plan, and obtain WVDEP approval of the plan.
- (2) Assumes background has been determined (see Subsection 2.5)

4.1 De Minimis Ecological Screening Evaluation

A De Minimis Ecological Screening Evaluation includes an assessment of the physical and ecological characteristics of the site and the nature and extent of contamination to determine if there are complete exposure pathways to ecological receptors of concern. If there are no complete exposure pathways between contaminants of concern in environmental media and ecological receptors of concern, it can be concluded that contaminants at the site pose no significant ecological risk (§60-3-9.1.b.1 of the Rule). Decisions associated with the De Minimis Ecological Screening Evaluation are illustrated in Figure 4-1.

At the screening stage of the ecological assessment process, the goal is to confirm the presence of a contaminant release, an ecological receptor of concern, and an exposure pathway. Actual site concentrations will not be a consideration, at this screening stage unless a valid exposure pathway can be demonstrated. Site contamination can be identified concurrently with the requirements for site characterization and the human health risk assessments.¹ Receptor and pathway identification specific to the ecological evaluation must be performed to fulfill the mandated screening requirements. A checklist is provided in Appendix C-2 to aid in completion of the screening.

If the site does not pass the De Minimis Ecological Screening Evaluation then additional ecological risk evaluations are necessary at that site (see Sections 4.2, 4.3, and 4.4 of this Guidance Manual). Failure to pass the De Minimis Ecological Screening Evaluation is not equivalent to a finding that there is an ecological risk at a particular site; such a failure is only a finding that additional evaluation is required.

This section of the Guidance Manual focuses on the use of the “Checklist to Determine the Applicable Ecological Standard” provided in Appendix C-2. The checklist process and logic are illustrated in Figure 4-1. This checklist is based on Section 60-3-9.5.a of the Rule. The checklist is divided into five sections, each corresponding to one or more sections of the Rule, as follows:

- Step 1. Determine whether a De Minimis Ecological Screening Evaluation is appropriate for your site (see 60-3-9.5.a.1)
- Step 2. Identify any readily apparent harm or exceedances of Water Quality Standards (see 60-3-9.5.a.5)
- Step 3. Identification of contamination associated with ecological habitats (see 60-3-9.5.a.2 and 60-3-9.5.a.3)
- Step 4. Characterize the potential ecological habitat (see 60-3-9.5.a.4)

¹ It is important to note that although contaminant analysis for ecological assessments may be conducted concurrently with the human health assessment, special considerations must be taken into account. For example, ecological benchmarks are sometimes lower than the corresponding human health-based standards. Therefore, it would be prudent to ensure that the sample detection limit for a given contaminant is appropriate. Furthermore, the distribution of the contamination should be evaluated not only with regard to human exposures, but also exposures to potential ecological receptors of concern.

- Step 5. Identify any potential ecological receptors of concern (see 60-3-9.5.a.4 and the definition of “ecological receptors of concern” at Section 2.14 of the Rule)

This section of the guidance addresses Steps 1, 2, and 3 in the following three subsections. Step 4 and 5 are addressed together in a fourth succeeding subsection. The last subsection discusses the reporting requirements for this screening process and checklist. Examples outlining the application of this screening process and the checklist are provided at the end of this section.

4.1.1 Determination of a Potential Complete Exposure Pathway

An exposure pathway is a direct or indirect physical association between a contaminant originating from the site and an ecological receptor of concern. An exposure pathway should be considered complete if an ecological receptor of concern is reasonably expected to contact a contaminant from the site via exposure to any environmental medium, including biota. Therefore, the presence of a complete exposure pathway will require a source and mechanism of contaminant release to the environment, an environmental transport medium, a point of potential contact between an ecological receptor of concern and the environmental medium, and a feasible exposure route at the contact point. Assumptions regarding contaminant transport or fate should be conservative and prudent to ensure that all relevant exposure pathways are evaluated.

Contaminated media for consideration in the De Minimis Ecological Screening Evaluation includes soil, sediments, surface water and biota. Groundwater may also be an important medium of exposure through uptake of shallow groundwater by deep-rooted plants and in the transport of contaminants into a surface water body. Table 4-2 outlines the type of exposure routes that must be considered in identifying potential complete exposure pathways.

Table 4-2: Expected Routes of Exposure Based on the Medium of Contamination

| Media | Direct Receptor Exposure | Indirect Media Exposure |
|---------------|---|--|
| Soil | Dermal contact Ingestion Gas/particulate inhalation Plant uptake | Leaching to groundwater Runoff to surface water and sediments Food chain contamination |
| Sediments | Direct contact Ingestion Plant uptake | Transport to surface water Bulk transport downstream Food chain contamination |
| Surface Water | Direct contact Ingestion Ventilation Plant uptake | Bulk Transport downstream Saturation and capillary transport to soil Absorption in sediments Food chain contamination |
| Groundwater | Plant Uptake (shallow groundwater) | Discharge to surface water |
| Biota | Ingestion | |

If there has not been a release to the environment at or from the site, the De Minimis Ecological Screening Evaluation can be concluded based on the lack of contaminated media, and, therefore, an exposure point. If no habitat exists that could be affected by site-related contamination, the De Minimis Ecological Screening Evaluation can also be concluded based on the lack of any potential ecological receptors of concern (§60-3-9.5.a.1).

To fulfill the requirements of the ecological De Minimis Ecological Screening Evaluation, a demonstration must be made for the presence or lack of pathways of exposure between the contamination and the ecological receptor(s) of concern. A Certificate of Completion will only be granted if none of the following conditions are found to apply:

- A contaminant stressor has migrated off-site and has become widely distributed in the environment (§60-3-9.5.b.1).
- Wildlife or ecological resources (receptors) of concern are exposed or have the potential for exposure to stressors (contaminants), either on or off-site (§60-3-9.5.b.2).
- Remediation of contamination at the site has the potential to expose ecological receptors of concern to adverse impacts (§60-3-9.5.b.3).
- There is a potential for indirect or cumulative impacts to ecosystems of concern (§60-3-9.5.b.4).
- Rare or sensitive species of concern are potentially at risk (§60-3-9.5.b.5).
- Adverse ecological effects have been observed in otherwise high quality habitats (§60-3-9.5.b.6).
- Projected land use involves the presence of sensitive ecosystems (§60-3-9.5.b.7). See Section 2.2.3.2.

4.1.2 Identifying Readily Apparent Harm

Sites which have been the cause of readily apparent ecological harm, or sites where there is a significant risk of harm to biota or habitats do not pass the De Minimis Ecological Screening Evaluation. If any one of the following criteria are observed at the site, then readily apparent harm is found:

- Visual evidence of stressed biota attributable to the release at the site, including, but not limited to, fish kills or abiotic conditions
- Visible presence of oil, tar, or other non-aqueous phase contaminant in soil over an area greater than two acres, or over an area equal to or greater than 1,000 square feet in sediment.

A risk of ecological harm would exist if it were reasonable to forecast any of these conditions as occurring in the future due to site-related constituents of concern.

For sites with readily apparent harm or the risk of such harm, both the Rule and practical experience suggest that further ecological evaluation may be redundant and unproductive. It may be more appropriate for the respondent to postpone further ecological evaluations until some remediation has been implemented and the readily apparent harm has been controlled or, at least, mitigated. In cases of readily apparent ecological harm, the purposes of VRRRA would be best served, in most cases, by prompt remedial action to control the source and to address the impacted media to the maximum and quickest extent. As a minimum, the respondent should proceed promptly to remedy selection and implementation.

4.1.3 Identifying Contamination Associated with Ecological Habitats

Although a release to the environment may have occurred or natural habitat is located on or near the site, the De Minimis Ecological Screening Evaluation can be concluded at this stage if the following two conditions are met:

- Environmental media associated with the onsite and adjacent habitat have been sampled and analyzed, and the site-related contaminants have not been detected above background concentrations
- Site-related constituents are not currently migrating to aquatic habitats, including wetlands

If both of these conditions are not met, or if site contamination has not been investigated, the respondent must proceed with identification of potential ecological habitats and receptors of concern

4.1.4 Identifying Potential Ecological Habitats and Receptors of Concern

Ecological receptors of concern are defined as specific ecological communities, populations, or individual organisms protected by federal, state, or local laws and regulations or those local populations which provide important natural or economic resources, functions, and values.

As discussed in Subsection 4.1.1, if no habitat exists that could be affected by contamination related to the site, the De Minimis screening can be concluded based on the lack of any potential receptors of concern (§60-3-9.5.a.1). If, however, natural habitats exist, progress toward identifying receptors of concern should begin with a description of each habitat. Descriptions of all potential habitats should address the following:

- General type of habitat
- Location of the habitat relative to the rest of the (site considering potential transport pathways)
- Area and topography of the defined habitats

- Predominant physical and geographical features
- Dominant plant and animal species known to occur at the site
- Soil and sediment types
- Human encroachment and interactions, including historical disturbances
- Evidence of Natural disturbance

Once it has been established that natural habitats exist and they have been described and characterized, it is necessary to identify potential assessment endpoints. The criteria for selecting assessment endpoints, upon which receptor selection will depend, are based on the management goals developed for the site. The Management goals for the De Minimis Ecological Screening Evaluation should address the protection of ecological receptors of concern.

The presence of ecological receptors of concern will depend on the habitat on and near the site. Those receptors residing or otherwise utilizing the valued environments listed in Subsection 2.2.3.2 shall be identified as ecological receptors of concern. If such habitat is identified within or near the site, a complete exposure pathway may exist and it will be necessary proceed with further ecological risk assessment. The applicant may elect to either undertake a Uniform Ecological Evaluation or proceed directly to the development of Ecological Site-Specific Risk-Based Standards. Note that there may be additional requirements that apply under federal law in the case of threatened or endangered species. The requirements are not preempted by the Voluntary Remediation Program.

State and regional wildlife agencies, local governments, interest groups, and universities are available to provide technical assistance in the identification of potential receptors. The West Virginia Division of Natural Resources² and the regional offices of the U.S. Fish and Wildlife Service³ maintain wildlife databases including information on threatened and endangered species. Other sources that may be helpful in these determinations are listed in Table 4-3. An onsite investigation should follow the initial habitat analysis. The purpose of the onsite investigation is to verify that the previously identified habitat can support potential ecological receptors of concern and to ensure that other potential receptors were not overlooked. The results of this investigation should be documented for inclusion in the work plan. The final selection of receptors, along with criteria and rationale, must be included in the final report as part of the listing of technical standards pursuant to §60-3-6.1.g of the Rule.

² Wildlife Database Manager, WV Division of Natural Resources, P.O. Box 67, Elkins, WV 26241, phone (304) 637-0245.

³ U.S. Fish and Wildlife Service, P.O. Box 1278, Elkins, WV 26241, phone (304) 636-6586.

Table 4-3: Reference Sources for Species Distribution Information

| |
|--|
| <p>WV-DNR Natural Heritage Database (Rare, Threatened, and Endangered Species)</p> <p>http://www.heritage.tnc.org/nhp/us/wv</p> <p>WV-DNR (Game Species)</p> <p>Bucklew, A. R., Jr., and G. A. Hall. 1994. The West Virginia Breeding Bird Atlas. University of Pittsburgh Press. Pittsburgh, PA. 215 p.</p> <p>Hall, G. 1983. West Virginia Birds: Distribution and Ecology. Carnegie Museum of Natural History. Pittsburgh, PA. 180 p.</p> <p>Green, N. B., and T. K. Pauley. 1987. Reptiles and Amphibians in West Virginia. University of Pittsburgh Press. Pittsburgh, PA. 241 p.</p> <p>Stauffer, J. R. Jr., J. M. Boltz and L. R. White. 1995. The Fishes of West Virginia. Academy of Natural Sciences of Philadelphia. Philadelphia, PA. 389 p.</p> <p>Allen, T. 1997. The Butterflies of West Virginia and Their Caterpillars. University of Pittsburgh Press. Pittsburgh, PA. 388 p.</p> <p>Mussels of West Virginia (In Preparation, contact Janet Clayton, WV-DNR)</p> <p>Strausbaugh, P. D., and E. L. Core. 1973. Flora of West Virginia. Seneca Books, Inc. Grantsville, WV. 1079 p.</p> <p>WVU Herbarium (County-by-County Database in Preparation)</p> |
|--|

4.1.5 Reporting Requirements

A report on the execution of the De Minimis Ecological Screening Evaluation must be included in the final report in accordance with §60-3-11.3 of the Rule. If the assessment is completed prior to the submission of the work plan, it should be included in support of proposed future assessments and remediation activities (§60-3-10.5.b). The report should be structured to address the questions presented in the checklist provided in Appendix C-2 for determining the applicable standard. It should also include any validated sampling data, a description of the habitat characterization and identification of any assessment endpoints, measurement endpoints and receptors of concern. The report should also describe the presence or absence of exposure pathways.

4.2 Uniform Ecological Evaluation

A Uniform Ecological Evaluation (as described in §60-3-9.1.b.2 of the Rule) is a generic evaluation of the potential effect a site’s contamination may have on identified ecological receptors of concern. It is a screening analysis that compares the site-specific concentration of a contaminant with WVDEP-approved standards or criteria in order to determine whether it represents a potential threat to ecological communities associated with the site.

4.2.1 Benchmarks and Generic Exposure Models for Uniform Ecological Evaluation

The Uniform Ecological Evaluation involves comparing the concentrations of stressors in environmental media with generic standards or benchmarks. These standards are intended to protect the most sensitive ecological receptor(s) of concern as defined in the management goals. Selection of suitable reference concentrations is discussed below.

Sources of appropriate ecological benchmarks, as outlined in §60-3-9.6 of the Rule, are listed in Table 4-4. The receptors of concern used in this analysis should be those identified in the De Minimis Ecological Screening Evaluation. Methods for determining anthropogenic background levels are outlined in Subsection 2.5 and Appendix B of this guidance. Derivation of applicant derived benchmarks are outlined in Subsection 4.2.4.

Table 4-4: Approved Sources and Methods for the Derivation of Medium-Specific Ecological Benchmarks^A

| Media | Benchmark/Toxicity Data Sources |
|--|--|
| Surface Soil ^b and Sediment | Anthropogenic Background Levels Direct Contact Benchmarks ^c Applicant Derived Values for Direct Contact ^d |
| Surface Water | Federal Ambient Water Quality Criteria State Water Quality Criteria Anthropogenic Background Levels Applicant Derived Values ^d |
| Groundwater ^e | Federal Ambient Water Quality Criteria State Water Quality Criteria Anthropogenic Background Levels Applicant Derived Values ^c |

^a Sources are listed in §60-3-9.6 of the Rule.

^b Surface soil constitutes the layer no greater than 2 ft below the surface.

^c This category is limited to benchmark values available from sources outlined in Table 4-4 and Appendix G.

^d This method is only to be used if no state or federal criteria exist. See Subsection 4.3.4 of this guidance.

^e Groundwater should only be considered if it is expected to affect a surface water body of concern (§60-3-9.6.c).

4.2.2 Applicant-Derived Benchmarks for Uniform Ecological Evaluation

If no criterion or appropriate benchmark exists for a given stressor, it is the responsibility of the applicant to derive an appropriate benchmark. The benchmark value must be based on either the bounded NOAEL or LOAEL derived from peer-reviewed sources for the contaminant

stressor specific to the contaminated medium and the receptor of concern (60-3-9.6.b.2). This usually referred to as the toxicity reference value (TRV). The TRV is a level of exposure that represents the maximum at which no significant ecological risk exists. A particular TRV is specific both to the receptor and stressor. It is empirical in that it is based on a specific dose-response relationship derived from experimental observations. TRVs for typical ecological receptors are available.

Approved sources for TRVs are listed in Table 4-5 and Appendix G. For receptors that must be protected on an individual basis (e.g., special status species), the TRV is the bounded NOAEL for the respective receptor and stressor. If the receptor is to be protected at the population level, the TRV is the dose that is likely to induce a population-level response. Criteria for the evaluation of an appropriate TRV are listed in Table 4-6. Benchmark values may be developed using the formulas provided in Figures 4-3 through 4-6.

With appropriate documentation, site-specific input parameters for the equations are preferred over default values. If there are numerous receptors of concern, then the screening criteria should be established based on the receptor whose exposure and toxicological sensitivity results in the lowest benchmark screening value. For surface water, the benchmark criterion is usually the TRV (in mg/l) that is protective of all aquatic receptors. If the most sensitive receptor exposed to surface water is terrestrial, then the model in Figure 4-3 should be used. For other environmental media, models and inputs are provided in Figures 4-4 through 4-6.

Table 4-5: Acceptable References for the Derivation of Benchmark Values^A

| |
|--|
| EPA AQUIRE database (www.epa.gov/earth100/records/a00120.html) |
| EPA IRIS Database(www.epa.gov/ngispgm3/iris/) |
| EPA HEAST Database |
| EPA ASTER Database (www.epa.gov/earth100/records/a00122.html) |
| EPA PHYTOTOX Database ^b |
| EPA Terrestrial Toxicity Database (TERRATOX) ^b |
| USFWS Technical Reports |
| Oak Ridge National Laboratory Toxicological Benchmark Technical Reports (www.hsrdr.ornl.gov/ecorisk/reports.html) |
| Other EPA documents acceptable to DEP |
| ATSDR Toxicological Profiles (www.atsdr.cdc.gov/toxpro2.html) |
| Other peer-reviewed publications ^c |
| Data developed in accordance with a peer-reviewed scientific testing protocol and approved by DEP |

^a These references are listed as acceptable under §60-3-8.1.c.2 of the Rule. No priority should be inferred from the order they are presented.

^b Access available through ECOTOX (www.epa.gov/superfund/oerr/r19/ecotox)

^c Additional references for benchmark values are provided in Appendix G.

Table 4-6: Criteria For The Evaluation Of TRVs

- Does the nature of the response have a direct impact on the measurement endpoint?
- Is the response the most sensitive effect to be expected?
- Is the mode of exposure consistent with the conceptual model?
- Is the TRV specific to the stressor as it occurs in the medium on-site?
- Is the expected response associated with the TRV consistent with the routes of exposure?
- Is the TRV relevant to the receptor and its habitat conditions on-site?
- Were appropriate allowances made for interspecies comparisons?
 - Application of uncertainty factors
 - Use of secondary interspecies application models
 - Comparable considerations of bioavailability relative to the exposure model

4.2.3 Risk Characterization based on the Uniform Ecological Evaluation

Risk characterization in the Uniform Ecological Evaluation involves comparing the contaminant concentrations (either the 95% upper confidence limit on the mean [UCL] or the maximum value) to the appropriate benchmark values specific to the receptors of concern. If a contaminant's concentration in an environmental medium is less than the benchmark, it may be assumed that it represents no significant ecological risk and no further action need be considered. If the contaminant's concentration in the medium exceeds the benchmark, there is a potential for unacceptable risk. While field survey data are valuable for understanding current environmental conditions, they are not used under the Uniform Standard to determine adverse effects to ecological receptors of concern. The use of field survey data is appropriate under the Site-Specific Standard.

When more than one chemical is present, the potential for additive, synergistic or antagonistic effects should be discussed. This discussion will usually be qualitative, except for cases where quantitative estimates of relative toxicity are available, e.g., dioxins, PCBs, or organophosphates. Interactions among chemicals are considered most likely when chemicals are known to affect the same toxic endpoint, e.g., reproductive effects. If multiple chemicals are present which have the same toxicity endpoint and toxicity data are available, the concentrations should be summed and compared to a single benchmark that has been approved by the WVDEP. If a substantial number of chemicals with similar toxic endpoints are present and toxicity data are not available, the potential for interactions should be discussed even if no benchmarks are exceeded.

If field survey data show readily apparent harm where several chemicals are involved, benchmarks selected should consider interactive or synergistic effects.

If a stressor exceeds a benchmark concentration, then there are two alternatives available to the applicant: 1) the benchmark is accepted as the remediation standard for that stressor, or 2) the applicant may undertake a site-specific ecological evaluation to determine a remediation standard unique to the particular site.

Figure 4-3A: Equations for the Derivation of Benchmarks for Surface Water Specific to Terrestrial Receptors

$$SWSTL = \frac{TRV}{IR}$$

Where:

SWSTL = Mean surface water screening threshold limit (mg/l)

TRV = Toxicity reference value (mg/kg bw day)

IR = Intake rate (l/kg bw day)

Figure 4-3: Equations for the Derivation of Benchmarks for Soil

Where the TRV is derived from water exposures which assumed 100% bioavailability, the following equations are to be used:

Where the TRV is derived from soil exposures

$$SSTL = TRV / IR$$

Organic Contaminants:

$$SSTL = \frac{TRV \times \left(k_{oc} \times f_{oc} + \left[\frac{(q_w + q_l \times H')}{(1-n) \times \rho_s} \right] \right)}{IR}$$

Inorganic Contaminants:

$$SSTL = \frac{TRV \times 10^{7-pH-pKa}}{IR}$$

Where¹:

- SSTL = Mean soil screening threshold limit (mg/kg soil dw)
- TRV² = Toxicity reference value (mg/kg bw day)
- k_{oc}³ = Water-organic carbon partition coefficient (l/kg soil dw)
- f_{oc} = Fraction of organic carbon (kg/kg; default 0.0165⁶)
- θ_w = Water filled pore space (l/l; default 0.3)
- θ_a = Air filled pore space (l/l; default 0.13)
- H' = Henry's law constant (unitless)
- n = Soil porosity (l/l; default 0.43)
- ρ_s = Particle density (kg/l; default 2.65)
- pH⁴ = Soil pH (default 4.7⁶)
- pKa = Log equilibrium constant for hydroxide formation
- IR = Intake rate (kg dw/kg bw day)

Intake Rates:

For plants: IR = 1

$$\text{For passerines: } IR = \frac{0.398 \times W^{0.85}}{W}$$

$$\text{For herbivorous mammals: } IR = \frac{0.577 \times W^{0.727}}{W}$$

$$\text{For predatory mammals: } IR = \frac{0.235 \times W^{0.822} \times BAF}{W}$$

$$\text{For predatory birds: } IR = \frac{0.648 \times W^{0.651} \times BAF}{W}$$

Where:

- W = body mass (g)
- BAF = Biomagnification Factor⁵

Figure 4-3: Equations for the Derivation of Benchmarks for Soil Cont'd

Notes:

¹ Unless otherwise stated, all default values were taken from the USEPA's Soil Screening Guidance (1994)

² The TRV used should be the lowest for all terrestrial receptors of concern associated with the site.

³ The k_{oc} may be estimated from the contaminant's octanol-water partition coefficient (k_{ow}) using the following equation:

$$\text{Log}(k_{oc}) = 2.8 \times 10^{-4} + 0.983 \times \text{Log}(k_{ow})$$

⁴ Median values for 181 West Virginia Soils (Jenks. 1969)

⁵ BMFs are chemical and receptor-specific parameters.

Figure 4-4: Equations for the Derivation of Benchmarks for Sediment

Organic Contaminant:

$$SdSTL = ATV \times f_{oc} \times k_{oc}$$

Inorganic Contaminant:

$$SdSTL = ATV \times 10^{7-pH-pKa}$$

Where¹:

SdSTL = Mean sediment screening threshold limit (mg/kg sediment dw)

ATV² = Aquatic Toxicity Value (mg/l)

k_{oc} ³ = Water-organic carbon partition coefficient (l/kg)

f_{oc} = Fraction of organic carbon (default 0.20)

pH = Sediment pH (default ?)

pKa = Log equilibrium constant for hydroxide formation

¹ Unless otherwise stated, all default values were taken from the USEPA's Sediment Quality Criteria (1993)

² If available, use the appropriate ecological ambient water quality criteria. Otherwise, use the lowest TRV (in mg/l) for all aquatic receptors of concern associated with the site.

³ Refer to figure 4-3, note 3 for the derivation of the K_{oc} from the contaminant's K_{ow} .

Figure 4-5: Equations for the Derivation of Benchmarks for Groundwater

$$GwSTL = \frac{Dr}{Tr} \times ATV$$

Where:

GwSTL = Mean groundwater screening threshold limit (mg/L)

Dr = Groundwater discharge rate (L/day)

Tr¹ = Surface water turnover rate (L/day)

ATV² = Aquatic Toxicity Value (mg/L)

¹ If the surface water body is a creek or river, then substitute the mean flow volume (l/day) for the turnover rate.

² If available, use the appropriate ecological ambient water quality criteria. Otherwise, use the lowest TRV for all aquatic receptors of concern associated with the site.

4.2.4 Reporting Requirements for the Uniform Ecological Evaluation

The results of the Uniform Ecological Evaluation are to be included in the final report in accordance with §60-3-11.3 of the Rule. If the assessment is completed prior to the submission of the work plan, it should be included in support of proposed assessment and remediation activities (§60-3-10.b). The report should identify ecological receptors of concern and media contamination upon which exposure pathways are based. It should also list appropriate benchmarks and discuss their sources and derivations. Comparisons of contaminant concentrations to their benchmarks should be presented in tabular form for each medium. The report on the Uniform Ecological Evaluation should also include a clear discussion of the screening results and an analysis of the uncertainty associated with any of the quantified values.

4.3 Ecological Site-Specific Risk-Based Standards

The development of ecological risk-based site-specific standards is analogous to developing a baseline ecological risk assessment. Applicants may choose to develop remediation standards through this process instead of relying on the benchmark standards derived in the Uniform Ecological Evaluation. The process for ecological risk assessment generally follows the guidance's listed in Table 4-1 and involves problem formulation, exposure analysis, ecological effects, and risk characterization. These steps are described in the subsequent sections.

4.3.1 Problem Formulation

The problem formulation component addresses the management goals through the definition of the assessment and measurement endpoints, identification of the receptor(s) of concern, and the development of the CSM and the analysis plan. The process for defining the endpoints and receptors is discussed in Subsection 4.1.1 under the De Minimis Ecological Screening Evaluation. The following is an example of the development of management goals and assessment and measurement endpoints.

| | |
|------------------------|--|
| Management goal: | Maintenance of fish communities |
| Assessment endpoint: | Maintenance of a benthic community that can serve as a prey base for local fish populations |
| Measurement endpoints: | <ul style="list-style-type: none">• concentrations of chemicals of concern in sediment and water column relative to levels reported in scientific literature to be harmful• toxicity observed in a whole sediment bioassay at levels considered significant according to test protocol; and• benthic invertebrate community structure / productivity relative to reference areas |

Measurement endpoints should be weighted, giving the most weight to the measurement endpoint that best represents the assessment endpoint, allowing it to have the greater influence on the conclusions of the risk assessment. Attributes to be considered which help to define how well a measurement endpoint represents the assessment endpoint include: 1) strength of association between assessment and measurement endpoints, 2) data quality, and 3) study design and execution. This process is described in Menzie et. al., 1996.

4.3.1.1 Quantifying Measurement Endpoints

In the De Minimis and Uniform Evaluations, measurement endpoints were considered qualitatively to identify the ecological receptors of potential concern. In the development of site-specific standards, it will be necessary to establish quantitative limits on the measurement endpoints to characterize the relationship between the contaminants of concern and the receptor population effects. The methods employed will be specific to the particular situation being considered. A review of the scientific literature and guidance documents listed in Table 4-1 will provide examples that may be applicable.

4.3.1.2 Refinement of the Conceptual Site Model (CSM)

The CSM is a series of working hypotheses regarding how the contaminant(s) interact with the ecological receptor(s) of concern. Refinement of the CSM will help in quantifying the measurement endpoints. Examples of criteria that the conceptual site model should address include the following:

- Is the model sufficiently quantitative to associate the stressor to the measurement endpoint via the receptor?
- Does the model directly reflect the habitat of consideration?
- Does the model account for all media and all potential routes of exposure?
- Does the model adequately reflect the concerns inherent in the management goals?

Based on the results of the CSM, an analysis plan should be formulated. The analysis plan is the practical description of the methods and strategies that will be used to meet data requirements of the conceptual model(s). It should include the types of media and biota to be sampled, the contaminants to be analyzed for as well as the potential ecological habitats and their characterization requirements. The analysis plan will ensure that there is adequate site-specific information to perform the risk analysis as well as providing a useful tool in the identification of data gaps for the subsequent uncertainty analysis.

Although the establishment of the measurement endpoints, the CSM, and the analysis plan should be done early in the assessment process, they must be considered amendable and open to modifications during the course of the site investigations as new information develops. Flexibility is essential in problem formulation to ensure completion of a precise and cost-effective site-specific ecological assessment.

Further discussion of the development of a conceptual site model is provided in Subsection 2.2.4 of this guidance document.

4.3.2 Quantitative Exposure Analysis

The quantification of receptor exposure to a stressor requires the numerical description of both the nature of the contaminant and the impact it has on the receptor as it interacts with that environment. The former is defined by contaminant fate and transport models (See Subsection 2.4.13) and the latter by the risk characterization.

In selecting pathways for evaluation, the applicant may take into account the availability of toxicity information in the scientific literature. There is a paucity or complete absence of scientific information on several pathways (e.g., inhalation and dermal contact for a large number of contaminants and a majority of potential receptors of concern (see Table 4-2). Such pathways need not be evaluated if the applicant can document a lack of quantitative information in the scientific literature, however, a qualitative assessment should be discussed in the risk

characterization and uncertainty subsections (see 4.3.4 and 4.3.6). Field surveys maybe used to help determine whether COPCs are having an adverse effect on receptors. However, the WVDEP will continue to monitor this area and, if appropriate information becomes available, the applicant will be advised.

Sometimes both exposure and effects are assessed directly using media toxicity testing or biological field surveys. The application of these direct toxicity analyses is most commonly used for assessments of lower and middle trophic organisms. For higher trophic receptors, it is usually neither practical nor economical to determine the actual toxicity. Therefore, it becomes necessary to model the potential impact based on the exposure the receptor is likely to incur and the toxicity threshold above which an adverse effect may be sustained. Considerations for this type of assessment are discussed below.

4.3.2.1 Biological Field Surveys

Field surveys are a method used to determine whether evidence of an adverse impact can be identified and correlated with contaminant concentrations within the environment. The scope of the survey is based on the measurement endpoints established in the problem formulation phase. The performance of a biological field survey involves the cataloging of wildlife present within the habitat under evaluation. Within this context, the field survey should be performed with sufficient detail and statistical precision to permit a quantitative comparison with a reference site that is similar to the site under investigation in all respects possible with the exception of the contaminant(s) under investigation. The detail required should be sufficient not only to determine if there has been any adverse impact, but also to reasonably attribute such impacts to the appropriate cause.

4.3.2.2 Direct Toxicity Determinations

The determination of potential risk may also be made through the application of direct toxicity testing. This is most common in the assessment of surface waters and sediments, although it may be applied to other environmental media. In direct toxicity testing, an indigenous or sentinel species is exposed to samples of the site media, usually under laboratory conditions, and the toxicity of the medium is determined based upon its effect on a measurement endpoint (e.g. lethality, reproduction, malformations, etc.). Examples of this type of direct toxicity analysis would include the *Daphnia* survival/reproduction assay for surface water or the 10-day *Hyalella* or *Chironomus* toxicity test for benthic macroinvertebrates in sediment (SETAC. 1993). Care must be taken to ensure that the results of direct toxicity testing are applicable to the overall risk characterization of the site. This is best accomplished by comparing the results to a reference site that is similar to the site under investigation in all respects possible with the exception of the contaminant(s) of potential concern.

4.3.2.3 Receptor Exposure Models

Receptor exposure models are mathematical constructs used to estimate the amount of a contaminant to which a specific receptor or population of receptors is likely to be exposed. The two major considerations in receptor exposure models are direct contact with contaminated

media and indirect contact through contaminated foodstuffs. Parameters used as variables in the fate, transport and exposure models should ideally be derived from site-specific observations. Where this is not practical, default assumptions, approved by WVDEP, may be used.

- Direct Exposure to Contaminated Media**—A receptor of concern will be exposed to a contaminant if it is found in direct contact with a contaminated medium. The receptor exposure model determines the actual dose of the stressor that the receptor is expected to receive. For animals, direct exposure usually occurs through a combination of dermal contact, respiration, and ingestion⁴. For plants, exposure occurs through deposition, stomatal infusion and/or evapotranspirative uptake. The specific exposure is the product of the amount of environmental medium contacted, the contaminant's concentration in the environment and the proportion of the contaminant that is likely to be absorbed by the receptor. Attenuation factors may also be used if the affected habitat only accounts for a portion of the receptor's total range, or if absorption of the chemical stressor is expected to be less than complete. When evaluating absorbance efficiencies, it is important to consider this parameter relative to the bases of the comparative TRV and not just that of the absolute absorbance. The total direct exposure to a stressor is the sum of all specific exposures by all pathways. Model equations for the determination of direct exposure are listed in Figures 4-6, 4-7 and 4-8.

Figure 4-6: Model Equations for Direct Ingestion Exposure

| | |
|--|--|
| Soil and Sediments: | $D_{is} = [C_s] \times IR_s \times Ba_s \times 10^{-6}$ |
| Water ¹ : | $D_{iw} = [C_w] \times IR_w \times Ba_w$ |
| Where: | <p> D_{is} = Exposure dose from ingestion of soil or sediment (mg/kg day) D_{iw} = Exposure dose from ingestion of water (mg/kg day) $[C_s]$ = Concentration of contaminant in soil/sediment (mg/kg) $[C_w]$ = Concentration of contaminant in water (mg/l) IR_s = Soil/sediment ingestion rate (mg/kg day) IR_w = Water ingestion rate (l/kg day) Ba_s = Proportional Bioavailability from soil/sediment Ba_w = Proportional Bioavailability from water </p> |
| <p>¹ This model is to be applied to terrestrial receptors only. For aquatic receptors, water ingestion is considered a component of direct contact.</p> | |

⁴ Exposure to a contaminant through drinking water will be considered a direct exposure for the purpose of this analysis.

Figure 4-7: Calculation Model for the Exposure of Receptors Through the Ingestion of Biota

$$D_f = \sum_{k=1}^m ([C_k] \times Ba_k \times IR_k \times 1000)$$

Where:

| | | |
|---------|---|---|
| D_f | = | Average daily dose (mg/kg day) |
| m | = | Number of contaminated food types |
| $[C_k]$ | = | Average contaminant concentration in food k (mg/kg) |
| Ba_k | = | Proportion absorbed from foodstuff k |
| IR_k | = | Daily intake rate of item k (g/kg day) |

In some situations, it may not be possible to directly determine the concentration of a contaminant within a receptor's food item(s). In these cases, it will be necessary to estimate the concentration based on the foodstuff/prey's exposure and a Biomagnification factor. The model for this type of estimate is provided in Figure 4-10. Biomagnification factors are empirical estimates that possess a high degree of uncertainty particularly when applied in situations different than those in which they were derived or over multiple trophic levels. Biomagnification factors tend to be very conservative and should only be considered when site-specific data cannot be obtained. Sources for bioaccumulation factors, biomagnification factors and food chain multipliers are limited but available from various USEPA guidance's as well as the scientific literature.

Figure 4-8: Models for the Estimation of Biota Contamination Based on Medium Contamination Concentration

$$[C_k] = [C_m] \times BMF$$

Where:

| | | |
|---------|---|--|
| $[C_k]$ | = | Contaminant concentration in foodstuff/prey item (mg/kg) |
| $[C_m]$ | = | Contaminant concentration in environmental media (mg/kg) |
| BMF | = | Biomagnification factor |

Determination of BMF:

$$BMF = BAF \times FCM$$

Where:

| | | |
|------------|---|--|
| BAF | = | Bioaccumulation factor: $BAF = \frac{[C_{T1}]}{[C_m]}$ |
| $[C_{T1}]$ | = | Contaminant concentration in first trophic level |
| FCM | = | Food chain multiplier (contaminant-specific) |

Total Exposure Profiles—The total exposure is the sum of total direct and total indirect exposure. It is the value (or distribution) that will be used in the risk characterization analysis. Estimates of total exposure are to be reported in terms of central tendency (mean or median) as well as plausible upper-bound estimates (e.g., 95th percentile) pursuant to § 60-3-8.7.h of the Rule.

4.3.3 Ecological Response Analysis

The ecological response analysis is the phase where comparative toxicity values are generated in order to evaluate the risk from exposure. Its primary function is to provide a standard against which the contamination exposure under investigation may be measured. The standard should represent a level of exposure that is considered allowable or acceptable. Evaluation on the suitability of the standard is based on the values inherent in the management goals and should be detailed within the analysis plan.

If the risk analysis is to be based upon either a biological field survey or direct toxicity analyses, then an acceptable habitat standard must be established to which the results are to be compared (pursuant to 60-8.1.3.2). In most cases, the results are compared to a reference area that represents an ecologically acceptable condition and is similar in all respects possible with the exception of the contaminant(s) of potential concern. Alternately, the site may be compared to a hypothetical construct of what would be expected under acceptable circumstances, although this method tends to be highly uncertain.

If the risk characterization is to be based on exposure modeling, then the effects analysis must provide a threshold dosage that the specific receptor of concern may be exposed to without incurring an unacceptable risk for an adverse effect. This is usually referred to as the TRV. Approved sources for TRVs are listed in Table 4-4. For receptors that must be protected on an individual basis (e.g., special status species), the TRV is the bounded NOAEL for the respective receptor and stressor. If the receptor is to be protected as the population level, the TRV is the dose that is likely to induce a population-level response. Criteria for the evaluation of the applicability of a TRV are listed in Table 4-6.

If probabilistic methodologies are to be employed in the response analysis, then the estimations developed as part of a probabilistic method must fall within the bounds of the dose-response curve. Determinations based on unbounded estimates of toxicity should be avoided.

4.3.4 Risk Characterization

Risk characterization is the phase of the risk assessment where a value is placed on the potential impact that a stressor has on the ecological environment. This value is an expression of the risk based on the evaluation of the measurement endpoints. In most cases, the risk is expressed in a Boolean fashion; that being whether an acceptable risk exists or not. The definition of acceptability is evaluated on the assessment endpoints based on the parameters established in the management goals. If the risk characterization demonstrates that conditions for a site exceed the bounds of acceptable risk, then, under the Rule (§ 60-3-9.7.b), remediation may be necessary prior to the granting of a status of no further action required. Decisions on the

appropriate remediation measures required in order for the site to conform to the management goals should be determined on a weight-of-evidence basis. If an adverse effect can be demonstrated to have occurred and that effect can be attributed to the contaminant, then it may be necessary to consider remediation at the site.

4.3.4.1 Risk Characterization Based on Biological Field Surveys

When characterizing risk based on biological field surveys, the biological condition of the site is compared to the reference established in the ecological effects analysis. This comparison must meet two specific considerations in order for a risk to be attributed to a specific contaminant. The first is whether any differences observed between the site under investigation and the reference represents an adverse impact. This may require a level of professional judgment since no two habitats are ever identical. The determination of an adverse effect is best based on quantifiable differences in the character of the habitats such as significant differences in biodiversity or productivity. The second consideration is whether any detectable adverse effect can be directly attributable to the contaminant in question. This must be determined through a process of elimination where all other potential factors that could affect the habitat are ruled out until a characteristic adverse effect can be reasonably attributed to the presence of the contaminant.

4.3.4.2 Risk Characterization Based on Direct Toxicity Testing

Risk characterization based on direct toxicity testing is similar to that of characterization by the biological field survey in that it is based on comparison to a reference situation either real or hypothetical, that is within the definition of acceptable as defined by the management goals. Here, the effects analysis defines a rate of toxic response that is the threshold for acceptable risk. If the medium toxicity from the site under investigation statistically exceeds that level, then the risk of an adverse effect is deemed unacceptable. Similarly as with the biological field survey method, it is necessary to ascribe the causative stressor through a process of elimination. However, unlike field surveys, it is much easier to ascribe a threshold concentration based on the results of the toxicity tests and concurrent medium contamination analysis. This can then be used as a site-specific benchmark to evaluate other portions of the site that have not been directly tested for toxicity.

4.3.4.3 Risk Characterization Based on Exposure Models

Risk characterization using exposure models entails comparing the site-specific exposure results against the TRV derived in the effects analysis to determine whether there is a significant ecological risk. This comparison is to be made regardless of whether single-point or probabilistic methods are employed. For point-estimate analyses, this is accomplished by calculating a hazard quotient for each stressor and receptor. The format for the calculation of single point estimates is detailed in Figure 4-9. For probabilistic determinations, the receptor response threshold (or distribution) is compared to the approximated response corresponding to the 90th percentile of the exposure distribution (§60-3-9.7.c).

Figure 4-9 Model Calculations for the Determination of Hazard Indices

$$HI_n = \frac{(D_{ds} + D_{dw} + D_{is} + D_{iw} + D_f + D_o)_n}{TRV_n}$$

Where:

HI_n = Hazard index for contaminant n

D_{ds} = Exposure resulting from direct contact with soil/sediment (mg/kg day)

D_{dw} = Exposure resulting from direct exposure to water (mg/kg day)

D_{is} = Exposure resulting from ingestion of soil/sediment (mg/kg day)

D_{iw} = Exposure resulting from ingestion of water (mg/kg day)

D_f = Exposure resulting from ingestion of foodstuffs (mg/kg day)

D_o = Exposure resulting from any other significant route (mg/kg day)

TRV_n = Toxicity reference value for contaminant n (mg/kg day)

If the ratio of the exposure concentration to the TRV (or the approximate receptor response to the threshold response) is less than 1 for receptor, it can be concluded that no significant ecological risk exists for that receptor. If, however, the hazard quotient is greater than 1, then an unacceptable risk is deemed to exist under the Rule (§60-3-9.7.b), and remediation may be necessary.

4.3.5 Remediation Standards Based on Ecological Risk

If it is found that a particular receptor/stressor interaction represents a significant ecological risk, then it will be necessary to establish site-specific, risk-based standards. This is accomplished by calculating a concentration for the stressor in an environmental medium that corresponds to an exposure level for the receptors of concern that does not exceed the lowest TRV. For surface water, the remediation benchmark is equivalent to the TRV for the identified receptors of concern with the highest HI. This value may be compared to a daily average concentration for the entire water body and should not necessarily be applied as a “not-to-exceed” value.

4.3.6 Uncertainty Analysis

The uncertainty analysis identifies the uncertainty associated with the various steps of the risk assessment process. This information is vital for interpreting the results of the risk assessment in the remedial decision making process. Descriptions of uncertainty should be as complete and detailed, as possible and should cover both quantitative and qualitative aspects of the assessment process. A partial list of potential sources of uncertainty that may be included in this analysis is provided in Table 4-7. Table 4-8 identifies specific considerations to be included in the uncertainty analysis for the ecological risk assessment final report. Additional guidance on uncertainty analysis may be found in USEPA guidances listed in Table 4-1.

If probabilistic methodologies were employed in the risk assessment, the uncertainty associated with the selection of the data distribution, compensation for potential correlations, and the bounded limits of the inputs should be addressed in the uncertainty analysis. Furthermore, the results of all sensitivity analyses should be included and discussed with regard to the uncertainty inferred from the distribution of the results. Further information on the reporting of uncertainty associated with probabilistic models may be found in Appendix H of this guidance.

4.3.7 Reporting Requirements

At the completion of the site-specific risk assessment, the applicant should be able to communicate to WVDEP a reasonable estimate of ecological risks, indicate the overall degree of confidence in the risk estimates, cite lines of evidence supporting the risk estimates, and interpret the ecological adversity. This information is to be outlined in the final report as required under §60-3-11.2 of the Rule. It is important that the risk assessment results be presented in a manner that is clear, transparent, reasonable, and consistent to facilitate its use in making risk management decisions. Specific aspects particular to the ecological risk assessment process are listed in Table 4-8.

Table 4-7: Potential Sources of Uncertainty

| Source | Considerations |
|---|---|
| Habitat Characterization | <ul style="list-style-type: none"> • Theoretical or empirical basis for the inclusion or exclusion of regions as habitats • Identification of species present in identified habitats • Evaluation of the significance of the habitat to potential receptors • Characterization of physical attributes of habitat • Characterization of ecological attributes of habitat |
| Stressor Distribution | <ul style="list-style-type: none"> • Selection of stressors of concern • Sensitivity and errors associated with media sampling • Data gaps in sampling (spatial, temporal, media types) • Identification of pathways for stressor transport |
| Endpoint and Receptor Selection | <ul style="list-style-type: none"> • Assumption and uncertainty in statistical models of stressor distribution • Presence or absence of threatened or endangered species • Basis for the selection of measurement endpoints |
| Exposure Models (including fate and transport modeling) | <ul style="list-style-type: none"> • Significance of the measurement endpoint to the quality of habitat • Causal association of the receptor to the endpoint • Ecological significance of receptor(s) • All quantifications of the ecological models employed • Applicability of selected models to site-specific conditions • Quantification limits of selected exposure models • Basis for the selection of default assumptions in the quantitative models • Error associated with site-specific parameters and input variables |
| Response Models | <ul style="list-style-type: none"> • Basis and applicability of response models to specific receptors of concern • Basis for the selection of default assumptions in the quantitative models • Applicability of quantified toxicity values and other input variables • Extrapolation of toxicological response to population and measurement endpoints • Confidence in the accuracy of the dose-response relationship |

Table 4-8: Critical Items to be Included in the Final Report on site-specific Ecological Risk Assessment

- Results and basis for the problem formulation
- Description of and rationale for the management goals, assessment and measurement endpoints, and receptor selection
- Presentation of the conceptual model and the assessment endpoints.
- Discussion of the major data sources and analytical procedures used.
- Review of the exposure and response analyses.
- Description of the risks to receptors, including quantitative risk estimates.
- Review and summary of major areas of uncertainty and the approaches used to address the uncertainty.
 - Discussion of generally accepted scientific positions on issues of inherent uncertainty (e.g., inter-species extrapolation of toxicity information).
 - Identification of major data gaps and, where appropriate, indication of whether gathering additional data would significantly reduce uncertainty.
 - Discussion of science policy judgments or default assumptions used to bridge information gaps, and the basis for these assumptions.

4.4 References

Jenks, E. M. 1969. Some chemical characteristics of the major soil series of West Virginia. West Virginia University Agricultural Experiment Station Bulletin 582T. Morgantown, WV.

Menzie, M., Henning, M.H., Cura, J., Finkelstein, K., Gentile, J., Maughan, J., Mitchell, D., Petron, S., Potocki, B., Svirsky, S., and Tyler, P. 1996. Special Report of the Massachusetts Weight-of Evidence Approach for Evaluating Ecological Risks. Human and Ecological Risk Assessment. Vol. 2, No. 2., pp. 277-304.

Society of Environmental Toxicology and Chemistry (SETAC). 1993. Guidance Document on Sediment Toxicity Tests and Bioassays for Freshwater and Marine Environments. Editors: Hill, I.R., Matthiessen, P. and Heimbach, F. November.

Suter, G.W., Barnhouse, L.W., Bartell, S.M., Mill, T., Mackay, D., and Paterson, S. 1993. Ecological Risk Assessment. Lewis Publishers.

United States Environmental Protection Agency (USEPA). 1989. Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference. USEPA, Environmental Research Laboratory, Corvallis, OR. EPA/600/3-89/013.

United States Environmental Protection Agency (USEPA). 1993. Technical basis for Establishing Sediment Quality Criteria for Non-Ionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning. EPA-822-R-93-011. Office of Water. Washington, DC. September

USEPA. 1993. Wildlife Exposure Factors Handbook. EPA/600/R-93/187a (Volume I) and EPA/600/R-93/187b (Volume II).

USEPA. 1994. Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates. EPA/600/R-94/024. June 1994.

USEPA. 1996a. Soil Screening Guidance: Technical Background Document. EPA/540/R-95/128. Office of Solid Waste and Emergency Response. Washington, DC. May.

5.0 RESIDUAL RISK ASSESSMENT

As stated in §8.6 of the Rule, a residual risk assessment (RRA) may be conducted considering conditions that will be present at the site following implementation of a proposed remedy. The RRA should consider and evaluate both human health and ecological risk. Included in the RRA is an assessment of the risks under current and reasonably anticipated future land and water use scenarios under the following conditions.

- The exposure conditions that will be present following remediation and the concentrations of untreated waste constituents or treatment residuals remaining at the conclusion of any excavation, treatment, or off-site disposal (§8.6.b.1); and/or
- The exposure conditions that will result following implementation of any institutional or engineering controls necessary to manage risks from treatment residuals or untreated hazardous constituents (§8.6.b.2).

The RRA must follow the same basic procedures outlined in Sections 3 and 4 of this guidance document, except that the conditions used to define the site must reflect post-remediation conditions, including site-specific numeric remediation standards and site-specific exposure conditions that incorporate any engineering and institutional controls proposed as part of the remedial action. It is not necessary to develop a RRA at sites where any one of the three risk-based standard indicates no further action is the proposed remedy.

At some sites, the RRA may be the only risk assessment performed to obtain a certificate of completion. Examples may include, but are not limited to:

- Sites where the applicant has already implemented a remedial action (e.g., a removal action has taken place and the risk assessment can now be performed using concentrations of contaminants remaining after the removal action) or
- Sites where harm is readily apparent (described in Section 4, ecological standards) and the applicant has elected not to perform a risk assessment but proceed directly to the remedy evaluation.

6.0 PROBABILISTIC RISK ASSESSMENT

Probabilistic risk assessments may be completed for the site-specific standard for both human health and ecological risk assessment. Probabilistic methods may be applied to describe parameter values required in transport and fate modeling, environmental media concentration data, exposure parameters, and toxicity estimates for both human and ecological populations. Combining probabilistic descriptions of some or all of these parameters will yield a probabilistic risk characterization. Techniques to perform probabilistic risk assessments are provided in Appendix I.

Probabilistic techniques are not necessary for all risk assessments, therefore, a tiered approach is recommended. In all cases, both human and ecological risk assessment, a deterministic calculation should be done first. If the results of this calculation show that either 1) site risks are so low so as to warrant no remediation, or 2) site risks are so high that there is no question but that remediation must be performed, then a probabilistic risk assessment is most likely unnecessary. However, if the deterministic risk assessment shows that site risks are in the range where the decision whether to remediate or not falls, then a probabilistic risk assessment can be of use to better define the range of likely risks and their uncertainty. In these instances, a probabilistic risk assessment can assist in making risk management decisions.

7.0 REMEDY SELECTION AND EVALUATION

7.1 General

It is anticipated that sites entered into the VRRRA program will vary greatly in terms of size, nature and extent of contamination, human health and ecological risks, physical conditions, and other pertinent factors. The process of remedy selection and evaluation must, therefore, be flexible to facilitate appropriate responses to the full range of sites and management issues. As long as the selected remedy or remedies satisfy the evaluation criteria established in §60-3-9.8 of the Legislative Rules, there is no intent to restrict the range and remedies considered or the process of remedy selection. In some cases, it may not be necessary to consider a variety of candidate remedies so long as the selected remedy meets the criteria of §60-9.8a of the rule.

The approaches to remedy identification and selection provided in this section are offered as guidance only. There is no regulatory mandate to apply the methods outlined in this section.

The guidance related to remedy identification and selection is organized in two parts, as follows:

- Remedy identification with a bibliography of information sources on various types or categories
- Remedy evaluation discussing the criteria established in the Legislative Rule with a bibliography of information sources on remedy evaluation.

Natural attenuation is discussed, in terms of the specific regulatory criteria for approval of this approach, in subsection 7.5 of this guidance (Section 7.5).

The Voluntary Remediation Agreement (VRA) and the Workplan must demonstrate that the selected remedy or combination of remedies have been evaluated in relation to the criteria established in the Legislative Rule. If the site is divided into multiple units for the purpose of remediation, the remedy for each unit must be evaluated in relation to these criteria. The VRA and Workplan are not required to describe the selection process or the remedies considered and the reasons for their selection or nonselection. However, discussions of the remedy selection process, the candidate remedies considered, and the evaluation of each candidate remedy may be appropriate components of the VRA or Workplan to assist in demonstrating that the selected remedy or combination of remedies is appropriate for that particular site.

The guidance provided in this section is not intended to restrict the range of candidate remedies considered and/or selected for any particular site as long as the selected remedy meets the evaluation criteria. Specifically, there is no intent to restrict or discourage use of innovative methods. Similarly, there is no intent to recommend or give preference to any particular remedy, category of remedies or to any product, service or vendor.

7.2 Identification of Candidate Remedies

The first step in remedy selection and evaluation is the identification of candidate remedies based on the analysis of the nature and extent of contamination and the cleanup objectives.

Table 7-1 provides a partial list of candidate remedies by environmental media. Although these lists are not complete, they do indicate the wide range of remedies available for each media. Table 7-1 should be viewed with the following notes or comments in mind:

- Many of the table entries represent categories of remedies, with different treatment reagents, microbes, process units, or methods available to address various site conditions, contaminants, and contaminant concentrations.
- Some candidate remedies will have beneficial impact on more than one environmental media. For example, the pumping and treating of groundwater may reduce soil contamination levels if contaminants can migrate to the saturated zone. Similarly, in situ chemical or biological treatments may address both soil and groundwater contamination.
- Many of the treatment processes identified in Table 7-1 are marketed and supported by process, reagent, and/or equipment vendors. Typically, each process is supported by multiple vendors. Further, specialized consultants and laboratories offer services related to process evaluation, reagent or microbe selection, and treatment formula development.
- There are a variety of data sources available to assist in remedy selection and evaluation. These sources include government publications (federal and state agencies), reference books by commercial publishers and associations, buyer's guides in industry magazines, and internet-accessible electronic data bases.

Subsection 7-6 presents a partial bibliography of published and electronic data sources to assist in remedy identification and evaluation.

7.3 Initial Screening of Candidate Remedies

An initial screening should be conducted to select a short list of appropriate alternatives for evaluation from the universe of remediation technologies. Based on the available information, only those technologies that apply to the media or source of contamination should pass the initial screening and be evaluated. The use of presumptive remedy guidance, where available, can in many cases provide immediate focus to the selection of alternatives. Presumptive remedies such as landfill caps (Ref EPA Presumptive Remedy Document) involve the use of remedial technologies that have been consistently selected in the past at similar sites or for similar contaminants.

Table 7-1: Partial Listing of Potential Candidate Remedies by Media

| Soils | Groundwater | Surface Water and Leachate |
|---|--|--|
| <ul style="list-style-type: none"> • No action • Natural attenuation (passive or intrinsic remediation) • Excavation and off-site disposal with treatment (typically hazardous waste) • Excavation and off-site disposal without treatment (typically non-hazardous waste) • On-site, ex situ thermal treatment • On-site, ex situ chemical treatment • On-site, ex situ fixation/stabilization • On-site, ex situ biological treatment • Soil vapor extraction • Passive soil venting • Soil washing • Soil flushing • Cap/cover over source area • Containment around source area • In situ chemical treatment • In situ biological treatment • In situ fixation/stabilization | <ul style="list-style-type: none"> • No action • Natural attenuation (passive or intrinsic remediation) • In-well aeration • Air sparging • Dual phase vacuum extraction and treatment • Extraction pumping and chemical treatment • Extraction pumping and biological treatment • Extraction pumping and physical treatment • In situ biological treatment • In situ chemical treatment • Funnel-and-gate technology • Vertical barriers • Inceptor trenches • Rock fracturing and enhanced groundwater collection (with appropriate treatment) • Cap and cover • Containment (e.g., slurry wall, tight sheeting, etc.) | <ul style="list-style-type: none"> • No action • Natural attenuation (passive or intrinsic remediation) • Collection and chemical treatment • Collection and biological treatment • Collection and physical treatment (e.g., filtration, aeration) • Cut-off wall or flow barrier upgradient of source • Surface water flow diversion • Cap/cover over source area • Containment around source area |

7.3.1 Screening Criteria

Candidate remedies should be screened initially against the following broad criteria:

- **Applicability and Appropriateness to Site**

Consider the specific contaminants present and their extent; the impacted media; the size of the site; the nature, extent, and status of the sources of contamination; and the physical condition of the site to identify potential remedies that appear to be applicable and appropriate to the specific site. Give further consideration only to those candidate remedies that are considered to be appropriate and applicable to the specific site.

- **Technical Feasibility**

Consider the steps and procedures required to implement each potential remedy in relation to site-specific conditions (site size, topography, current land use, future land use – if known, drainage routes, surface conditions and materials, subsurface conditions, and other factors) to assess the technical feasibility, practicality, and probability of success of applying that remedy to the specific site. Also consider the performance history (beneficial impact, implementation problems and other relevant information) of the candidate remedy at other sites with similar characteristics. Give further consideration only to those candidate remedies that are evaluated as technically feasible at the specific site.

7.3.2 Screening Method

The initial screening should be conducted in accordance with the following general methodology:

- Pass/fail evaluation of each candidate remedy against each screening criterion subsection 7.3.1.
- Further consideration only of remedies “passing” the two criteria – no further consideration of remedies “failing” either criteria.
- The initial screening should be considered as brief, focused, and informal, and is not required to be reported.

The candidates passing both screening criteria (i.e. the short list remedies) qualify for further evaluation.

7.4 Evaluation of Short-List Remedies

This section sets out seven criteria for those technologies passing the initial screening criteria.

7.4.1 Evaluation Criteria

For those remedies passing the screening criteria, this section describes the seven criteria used for further evaluation.

1. Effectiveness in Protecting Human Health and the Environment

Each remedy is evaluated for the ability to eliminate, reduce or control the identified exposure pathways. Short and long term impacts and potential cross-media impacts are identified and evaluated.

The remedy is evaluated relative to attainment of the identified remediation standard goals.

During assessment of this evaluation criteria, additional requirements to implement each remedy including institutional and/or engineering controls are identified.

2. Long-Term Reliability to Achieve Standards

Evaluation of long-term reliability of each remedy includes the following:

Assessment of Residual Risks

- Magnitude
- Type (treatment residuals and/or residual contamination)
- Assessment of Reliability
- To meet cleanup goals
- Nature and extent of long-term management
- Long-term monitoring requirements
- Operation and maintenance requirements
- Identification of difficulties and uncertainties associated with implementation
- Component replacement requirements
- Duration of institutional and/or engineering controls

3. Short-Term Risks Posed by Implementation

Each remedy is evaluated to identify short-term risks during implementation (construction phase through achievement of cleanup goals) by consideration of the following:

Assessment of Risks to Workers

- Identification of risks
- Identification of risk mitigation methods

Assessment of Risks to Site Neighbors and the Community

- Identification of risks
- Identification of risk mitigation methods

Assessment of Risks to the Environment

- Identification of environmental impacts
- Identification and assessment of mitigation measures
- Identification of unavoidable impacts

Assessment of Time Required for Remediation Implementation

- Identification of time frame for construction
- Identification of time frame to achieve cleanup goals

4. Acceptability to the Affected Community

An assessment of the acceptability to the affected community involves identifying the affected community (if any), potential issues of concern, and review of comments received under §60-3-7.4 of the Rule (only applies to brownfield applicants). Although there is no requirement in the rule, applicants are encouraged to seek community input in reviewing remedial alternatives that may potentially cause offsite impacts. If permitting is a requirement, this also needs to be considered. As appropriate, mitigation measures are identified and evaluated.

5. Implementability and Technical Practicability

The implementability and technical practicability of each remedy are assessed by evaluation of the following.

Assessment of Technical and Engineering Feasibility

- Technical difficulties and unknowns
- Reliability of technology
- Ease of implementation
- Monitoring requirements

Assessment of Administrative Feasibility

- Permit requirements
- Consistency with other applicable regulations

Assessment of Availability of Services, Equipment and Materials

- Availability of treatment, storage, and disposal services
- Availability of equipment
- Requirements for specialized equipment

- Availability of workers
- Availability of technology

6. Cost Evaluation

Consider the following elements as they are applicable to the remedy being evaluated:

Capital costs

- Engineering costs including process development, design services, and related support activities
- Process equipment including ancillary equipment and process control devices
- Labor, materials, and equipment to install or construct the remedy including earthwork, foundations, structures, and utilities (including cap, containment, or other site work items, if appropriate)
- Contractors overheads, allowances for general tools and supplies, and profit
- Site costs during construction such as support facilities, utilities, fencing, and security
- Permits and other fees
- Construction management including procurement of equipment and contracted services and construction supervision
- General administrative costs
- Health and safety items

Operating Costs

- Operating and maintenance labor
- Maintenance parts and supplies
- Treatment reagents and/or other operating supplies
- Operating utilities
- Health and safety items
- Required reporting
- Site management and administration costs during the remedy operating period

Monitoring and reporting costs e.g.

- Sampling and analysis of results as required or appropriate
- Collection and analysis of perimeter and/or environmental monitoring samples
- Collection and analysis of progress and/or confirmatory samples

In many cases, candidate remedies will have variations in the projected timing of expenditures. This is most likely to be the case when remedy implementation extends into the future for several years or more. If the differences in the amounts and timing of these expenditures are significant, it may be appropriate to calculate the present worth of the stream of expenditures for each candidate remedy. Under these circumstances, present worth calculations will provide a more useful and valid economic comparison of the remedies being evaluated.

Present worth calculations provide estimates of the current values of future expenditures by considering both the time-value of money (i.e., the effective discount on money deposited now at interest to meet future obligations) and the increases of future costs due to inflation. Present worth calculations are done using standard methods and formulas. Textbooks in financial analysis and engineering economics provide detailed explanations of present worth calculations and formulas (subsection 7.6.5). Pocket calculators programmed with the appropriate financial analysis formulas to facilitate these calculations are available at modest cost; such calculators come with clear, step-by-step instructions to assist the user.

Making present worth evaluations requires estimation of future interest and inflation rates. This can be simplified by recognizing that the object of these calculations is the comparison of alternate remedies, so consistency in using the factors is more important than the actual factors applied. A useful approximation can often be developed by applying a risk-free, inflation-free interest rate and a zero inflation rate to the present worth formulas. Specific information on project interest rates and inflation rates can be found in general business publications. Local bankers and/or librarians may be able to assist in developing this information.

7. Net Environmental Benefits

An evaluation of the net environmental benefits of a remedy includes the following:

Consideration of the projected reduction in quantity, toxicity, mobility, and risk.

Consideration of potential site reuse.

- Restrictions
- Time frame for reuse

7.4.2 Evaluation Method

Each candidate remedy should be evaluated against the seven criteria specified in the Rule as described above. This evaluation may be done for a short list of candidate remedies. It may be useful to provide a concise report of the alternatives considered and the evaluation conducted to support the demonstration that the selected remedy meets the human health and environmental protection criteria. The remedy meeting the effectiveness in protection criteria, achieving remediation standards, and with the lowest overall cost (including present worth calculation, if appropriate) should be selected unless there are extenuating circumstances favoring the selection of another candidate remedy. The Rule leaves remedy selection to the discretion of the remediating party as long as the selected remedy meets the protectiveness criteria for both human health and the environment.

The remediating party is required only to identify the selected remedy and demonstrate that it meets the protectiveness goals. There is no requirement to report the selection process or to establish a formal remedy evaluation and selection process comparable to the Feasibility Studies required for "Superfund" or the Corrective Measures Studies required for RCRA Corrective Actions.

7.5 Inclusion of Natural Attenuation in Remedy Evaluation

Section §60-3-9.9 of the Rule allows for submission of a remediation plan which includes the natural attenuation of contaminants of concern contained in soils, sediments, and/or groundwater for the entire site or portions of the site. The Rule provides conditions which must be met and/or demonstrated for the Department to approve natural attenuation as a viable remedy. This section provides guidance for the regulated community to compile the evidenced needed for such a strategy.

Section §60-3-9.9 of the Rule specifies several environmental criteria which must be demonstrated before the WVDEP will approve a natural attenuation remediation plan. These conditions include:

1. That the contaminants of concern have the capacity to degrade or attenuate under site-specific conditions (§60-3-9.9a).
2. That the contaminant plume in groundwater or soil volume is not increasing in size (§60-3-9.9.b).
3. That all sources of contamination and free product have been controlled or removed, where applicable (§60-3-9.9b).
4. The contaminant migration will not result in the violation of applicable groundwater standards (46CSR1) at any existing or reasonably foreseeable receptor (§60-3-9.9d).
5. A groundwater discharge to a surface water body will not result in contaminant concentrations at the sediment/water interface that result in violations to the surface water standards (46CRS12) (§60-3-9.9f).

Natural attenuation of inorganic and organic compounds in soils, sediments, and groundwater can occur by a number of mechanisms, primarily biological and physical. Physical mechanisms for natural attenuation include sorption, dilution, volatilization, and dispersion. Biological mechanisms include biodegradation, which results in the destruction of contaminants by aerobic and anaerobic microorganisms. To support remediation by natural attenuation, it must be scientifically demonstrated that attenuation of site contaminants is occurring at rates sufficient to be protective of human health and the environment. Much of the information needed for a natural attenuation strategy will be collected as part of the site characterization phase and the investigation of the extent of contaminant migration. However, some of the evidence needed to support natural attenuation is quite specific, and therefore, for efficiency, collection of data to support natural attenuation as a remedial option should be considered as part of the early phases of the investigation (see for additional guidance OSWER Directive 9200.4-17, "Use of Monitoring Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites).

7.5.1 Developing Evidence in Support of Natural Attenuation

There are several steps to take in gathering the evidence needed to support natural attenuation. These steps are directed towards pursuing three technical lines of evidence:

1. Documented mass loss of contaminants.
2. Presence and distribution of geochemical and biochemical indicators.
3. Direct microbiological evidence.

The following paragraphs outline the steps to be taken to gather the necessary evidence, and provide guidance for completion. This guidance is primarily geared toward natural attenuation in groundwater, however, the same principles apply to the natural attenuation of contaminants in soils. Depending on the location and depth of the soil contamination, it may be necessary to utilize institutional or engineering controls to: 1) prevent potential receptor exposure to contaminated soils from the site, and/or 2) mitigate soil that acts as a contaminant source groundwater.

7.5.1.2 Review Available Site Data for Evidence of Natural Attenuation

Historical data of contaminant concentrations can provide some of the most defensible evidence for natural attenuation if there has been a mass loss of contaminants at the site. In addition, the existing data may provide evidence for both geochemical and biochemical indicators of intrinsic bioremediation (i.e., presence of daughter products, byproducts of microbial respiration, loss of electron acceptors, etc.). This step serves to define data needs and the locations of additional monitoring points as well as determining the likelihood of exposure pathway completion.

7.5.1.3 Develop a Preliminary Conceptual Site Model

The conceptual site model is a presentation and explanation of the contaminant distribution in site groundwater in relation to contaminant fate and transport processes. This model should include:

- The location of the source(s) of contamination. As stated above, the source(s) of contamination must be controlled or removed, where practicable. If the source(s) of contamination cannot be controlled or removed, the effect of the continuing source(s) on contaminant fate and transport relative to the rate of the natural attenuation processes must be considered in the conceptual model.
- The relative distribution of the COCs, both vertically and horizontally, in soil and groundwater.
- The location of potential human and ecological receptors.

- Site specific characteristics which make the site amenable to natural attenuation.
- An estimate of the contaminant transport velocity and direction of groundwater flow. §60-3-9.9c of the Rule requires that the travel time and direction of contaminant migration be predicted with reasonable certainty.
- Estimation of the length of time necessary to achieve site specific remedial objectives.

Further discussion of the development of a conceptual site model is discussed in Section 2.2.4 as well as in Weidemeier et al. (1995) and Feenstra et al. (1996).

7.5.1.4 Additional Data Requirements

The data required to support a natural attenuation remedial technology are specific to the site and the type of contaminants present. Table 7-2 lists a number of soil and groundwater parameters used to support natural attenuation; an explanation of each of these parameters is contained in several publications (ASTM 1996; Wiedemeier et al., 1995; Wiedemeier et al. 1996a; Wiedemeier et al., 1996; RTDF Bioconsortium Guidance Handbook, www.rtdf.org/public/bioremed/default.htm). These data should be evaluated for a number of monitoring points located:

- upgradient of the source area in a non-contaminated area;
- in the source area;
- downgradient of the source area in the dissolved phase contaminant plume; and
- downgradient of the plume.

Upgradient, or in some cases sidegradient, groundwater monitoring wells can be used to quantify background concentrations for a number of the parameters being evaluated. For sites having more than one aquifer or a significant vertical component of flow, monitoring well locations should also be selected to adequately represent the vertical profile.

The analytical data collected during site characterization activities can be evaluated to better define the biodegradation kinetics (i.e. first order decay rate). An understanding of the biodegradation kinetics is a necessary component for quantifying input parameters used in models that incorporate natural attenuation equations. The biodegradation kinetics are site specific; dependent on the contaminant type, microbiological community, and available nutrients. Contaminant type is important since various chemicals degrade at faster rates than others. Additionally, chemicals degrade under aerobic and/or anaerobic conditions at varying rates. A microbiological community is required for biodegradation within an environment that is favorable for organism growth (note: pH values outside of the 6-8 range and high levels of certain chemicals may slow community growth or be toxic to the microorganisms). The available nutrients involve naturally occurring or engineered electron acceptors (i.e. dissolved

oxygen, nitrogen, sulfate, iron, carbon dioxide) and electron donors (i.e., carbon sources) that are used by the microorganisms to break down the contaminants of concern through respiration.

Methods for calculating the first order decay rates are presented in Weidemeier 1995 and Buscheck and Alcantar, 1995). Alternatively, literature values of first order decay rates may be obtained but must be clearly documented, justified, and qualified as subjective (Wilson et al, 1996; Howard et al 1991; Rifai, et al 1995).

Table 7-2: Parameters Used to Assess Natural Attenuation

| Field Parameters | Inorganics | Organics | Dissolved Gases | Micro-biological | Physical | Hydro-geological |
|------------------|--|---------------------------------------|-----------------|---|---------------------|---|
| Dissolved oxygen | Ammonia/TKN | VOCs (cis & trans isomers identified) | Methane | PLFA (Phospholipid Fatty Acid Analysis) | Grain size analysis | Subsurface and surficial geology including lithology, stratigraphy, and structure |
| Redox potential | Chloride | Semi VOCs CO ₂ | Ethane | | Porosity | |
| Conductivity | Sulfide | TOC | Ethene | Total heterotrophic and contaminant-specific bacterial plate counts | | Hydraulic gradient |
| Temperature | Sulfate | COD/BOD | | | | |
| pH | Nitrate | Alkalinity (carbonate & bicarbonate) | | | | |
| | Nitrite | | | | | |
| | Ortho-Phosphate | | | | | |
| | Iron (total & dissolved - field filtered) | | | | | |
| | Manganese (total and dissolved-field filtered) | | | | | |

Microcosm studies are conducted only when the microbiological and chemical evidence for natural attenuation at the site is inconclusive. Wiedemeier et al. (1995), discusses protocols for setup and analysis of microcosm studies. Biodegradation rates obtained from microcosm studies are often much faster than the actual field rates (Rifai et al., 1995). Therefore, results of microcosm studies are generally used qualitatively to demonstrate that the biodegradation processes are occurring in the field, and not to develop biodegradation rates for modeling.

7.5.1.5 Collect Additional Data in Support of Natural Attenuation

Since in situ biodegradation can proceed under both aerobic and anaerobic conditions, sampling soils and groundwater for natural attenuation parameters must be performed in a manner that does not change the redox potential (E_h) of these materials. In general, exposure to oxygen and agitation of the samples must be minimized. Use of low flow purge and sample methods with submersible or peristaltic pumps and flow-through sampling cells are

recommended (ASTM, 1992). Under no circumstances should bailers be used for this type of sampling. A more complete discussion of groundwater and soil characterization for natural attenuation can be found in Weidemeier et al. (1995: In prep.: 1996a)

7.5.1.6 Refine Conceptual Site Model

After the site data has been compiled and evaluated relative to natural attenuation processes, the conceptual site model should be refined to more accurately reflect the fate and transport processes affecting the contaminants of concern. This data analysis should include an evaluation of the geological, chemical, and biological factors that affect the rate and extent of natural attenuation. The refined conceptual site model can be used as a basis for analytical or numerical modeling designed to simulate the migration and attenuation of contaminants. It is mandatory that a natural attenuation strategy for a site be protective of human health and the environment, therefore, conservative model input parameters should be used. All input parameters should be clearly defined and justified.

7.5.2 Simulation of Natural Attenuation

Two classes of mathematical models (screening and advanced) can be used to demonstrate that natural attenuation is a viable remedial option. Simple analytical screening models are primarily designed to determine the feasibility of using natural attenuation as part of a remedial strategy. At smaller sites with apparently limited impacts, it may be appropriate to use a screening model as the primary groundwater model to simulate natural attenuation, and predict the extent and duration of contaminant migration. One such model is BIOSCREEN, which has been developed and endorsed by the US Air Force (Newell et al., 1996; internet www.epa.gov/ada/csmos.html).

Sites with complex hydrogeology or multiple contaminant source areas may require the use of an advanced numerical groundwater contaminant fate and transport model to simulate natural attenuation and predict the extent and duration of contaminant migration. Examples of advanced numerical models include: Bioplume II, III, RT3D, BIOMOD3-D, and BioF&T3-D.

Bioplume II is a two dimensional numerical groundwater flow model developed to simulate oxygen limited aerobic decay. Bioplume II is most applicable to sites with relatively simple geology (homogeneous and isotropic porous media) which are contaminated with petroleum hydrocarbons. A new version of Bioplume II (Bioplume III) currently under development will be able to simulate more complex microbial processes, multiple chemical species, and aerobic/anaerobic processes (Newell et al., 1995, Rifai et al., 1987).

More advanced two and three dimensional numerical models include RT 3D (<http://terrassa.pnl.gov:2080/bioprocess/rt3d.html>) BIOMOD 3-D and BioF&T 3-D (Scientific Software Group, e-mail: info@scisoftware.com). RT3D and Biomod 3-D are typically used in conjunction with the USGS finite-difference groundwater flow model, MODFLOW 3-D. These models are capable of simulating groundwater flow and contaminant transport in the saturated and unsaturated zones in heterogeneous, anisotropic porous media or fractured media. Each of these models simulate complex microbial processes based on oxygen limited, anaerobic, first-order, or Monod type biodegradation kinetics, as well as anaerobic or first-order sequential

degradation involving multiple daughter species. Given the capabilities of RT3D, BIOMOD 3-D and BioF&T 3-D, these models can be used at sites with the most complex hydrogeology (e.g., interbedded sands and clay, fractured bedrock, and multiple aquifers) and complex contaminant distribution (e.g., multiple source areas and non-aqueous phase contamination), and are applicable to most contaminants (e.g., petroleum hydrocarbons, chlorinated solvents, explosives, and heavy metals).

7.5.3 Conduct an Exposure-Pathway Analysis

After calculating the rate of natural attenuation and predicting the future concentration and extent of the contaminant plume, it is necessary to evaluate whether the plume has the potential to impact receptors before contaminant concentrations have degraded to the applicable groundwater and/or surface water standards. Both ecological and human receptors need to be identified as well as points of exposure under current and future land, surface water, and groundwater use scenarios. Before the agency can accept a proposal for natural attenuation, the applicant must demonstrate that the contaminant migration will not result in the exceedance of any groundwater standards at any existing or reasonably foreseeable human receptor, or the exceedance of any surface water standard if the receptor is a surface water body. The standards for surface waters are contained in 46CSR1 and the groundwater quality standards are contained in 46CSR12.

The location of potential receptors can be ascertained in a number of ways, such as:

1. A search of state and local records for the locations of private and public drinking water wells within the expected path of plume migration.
2. A request from a public water surveyor for a listing of their service area within the expected path of plume migration.
3. A survey of streams and rivers within the expected path of plume migration.
4. Contact the local, county, or state planning boards to determine potential future land uses of adjacent properties within the expected path of plume migration.
5. Field survey / resident interviews.

If the contaminant plume is, or will be, migrating onto adjacent properties, the applicant must demonstrate that either the properties are served by an existing public water supply which uses surface water or hydraulically isolated groundwater; or the applicant has obtained written consent from the property owners allowing contaminant migration onto their property. This is important even if adjacent properties are currently vacant, because 60CSR3 requires consideration of potential receptors in the reasonably foreseeable future.

If the contaminant plume is expected to intercept surface waters, the groundwater discharge beyond the sediment/water interface cannot exhibit contaminant concentrations that

would result in violations of standards for surface waters contained in 46CSR1. This can be determined through one or more of the following techniques:

- Install groundwater monitoring wells at the upgradient boundary of the surface water body.
- Model the expected effect of the groundwater discharge using mass balance modeling techniques.
- Other methods/strategies acceptable to the department.

The choice of the method(s) used to assess potential surface water impacts must be considered on a case by case basis dependent upon site-specific issues, such as the ability to gain access to offsite properties, the potential for a regional impact or other downgradient sources, potential upstream sources, seasonal conditions, etc.

7.5.4 Develop a Long-Term Monitoring Plan

A long-term monitoring plan is used to monitor plume migration and to verify that natural attenuation is ongoing and its rate is sufficient to preclude impact to receptors. This monitoring plan should include periodic sampling of wells in the different areas of the site, for example: a) upgradient of the source area in a non-contaminated area; b) in the source area; c) downgradient of the source area in the dissolved contaminant plume; e) downgradient of the plume; and (f) surface water collection points. Downgradient compliance monitoring points need to include one or more monitoring wells at least one year's advective time of travel upgradient of any potential receptor, and at least one monitoring well no farther away from the leading edge of the contaminated groundwater five years advective travel time. These wells should be sampled for all of the parameters used to support the natural attenuation strategy for the site including parent and daughter compounds, dissolved gasses, electron donors and electron acceptors. Information regarding the long-term monitoring plan, analytical suite, sampling frequency, etc., is discussed in Wiedemeier, et al, 1995.

7.6 References

7.6.1 Remedy Selection, Contaminant-Specific

A Compendium of Technologies Used in the Treatment of Hazardous Wastes, EPA/625/8-87/014, US Environmental Protection Agency, Center for Environmental Research Information, Cincinnati, OH, 1987.

A Guide to the Assessment and Remediation of Underground Petroleum Releases. Publication 1628, American Petroleum Institute, Washington, DC, 1989.

Guidance on Remedial Actions for Superfund Sites with PCB Contamination, EPA/540/6-90/007, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC, 1990.

Presumptive Remedies: Site Characterization and Technology Selection for CERCLA Sites with Volatile Organic Compounds in Soils. EPA/540/F-93/048, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC, 1993.

Technology Selection Guide for Wood Treater Sites. Publications 9355.0-46FS and 9355.0-46, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC, 1993.

7.6.2 Remedy Selection, Technology-Specific

Engineering Bulletin: Soil Washing Treatment. EPA/540/2-90/017, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC and Office of Research and Development, Cincinnati, OH, 1990.

Engineering Bulletin: Thermal Desorption Treatment. EPA/540/2-91/008, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC and Office of Research and Development, Cincinnati, OH, 1991.

Engineering Bulletin: In-Situ Steam Extraction Treatment. EPA/540/2-91/005, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC and Office of Research and Development, Cincinnati, OH, 1991.

Engineering Bulletin: In-Situ Soil Vapor Extraction Treatment. EPA/540/2-91/006, US Environmental Protection Agency. Office of Emergency and Remedial Response, Washington, DC and Office of Research and Development, Cincinnati, OH, 1991.

Engineering Bulletin: Slurry Biodegradation. EPA/540/2-90/016, US Environmental Protection Agency. Office of Emergency and Remedial Response, Washington, DC and Office of Research and Development, Cincinnati, OH, 1990.

Handbook for Stabilization/Solidification of Hazardous Wastes. EPA/540/2-90/001, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, 1986.

Engineering Bulletin: Mobile/Transportable Incineration Treatment. EPA/540/2-90/014, US Environmental Protection Agency. Office of Emergency and Remedial Response, Washington, DC and Office of Research and Development, Cincinnati, OH, 1990.

Handbook on In-Situ Treatment of Hazardous Waste-Contaminated Soils. EPA/540/2-90/002, US Environmental Protection Agency, Risk Reduction Engineering Laboratory, Cincinnati, OH, 1990.

Engineering Bulletin: Chemical Dehalogenation Treatment: APEG Treatment. EPA/540/2-90/015, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, and Office of Research and Development, Cincinnati, OH, 1990.

Engineering Bulletin: Chemical Oxidation Treatment. EPA/540/2-91/025, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, and Office of Research and Development, Cincinnati, OH, 1990.

Ground Water and Leachate Treatment Systems. EPA/625/R-94/005, US Environmental Protection Agency, Office of Research and Development, Washington, DC, 1995.

Remediation of Contaminated Sediments. EPA/625/6-91/028, US Environmental Protection Agency, Office of Research and Development, Washington, DC, 1991.

Pump-and-Treat Ground Water Remediation. EPA/625/R-95/005, US Environmental Protection Agency, Office of Research and Development, Washington, DC, 1996.

Land Use in the CERCLA Remedy Selection Process. OSWER Directive No. 9355.7-04, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC, 1995.

7.6.3 Remedy Evaluation

Remediation Technologies Screening Matrix and Reference Guide. EPA/542/B-93/05, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC, 1993.

Technology Screening Guide for Treatment of CERCLA Soils and Sludges. EPA/540/2-88/004, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC, 1988.

Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA, Interim Final. EPA/540/G-89/004, OSWER Directive 9355.3-01, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC, 1988.

Guidance on Conducting Non-Time Critical Removal Actions Under CERCLA. EPA/540/R-93/057, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, 1993

Summary of Treatment Technology Effectiveness for Contaminated Soil. EPA/540/2-89/53, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, 1991.

Superfund Engineering Issue: Issues Affecting the Applicability and Success of Remedial/Removal Incineration Projects. EPA/540/2-91/004, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, and Office of Research and Development, Cincinnati, OH, 1991.

Guidance on Remedial Actions for Contaminated Ground Water at Superfund Sites. EPA/540/G-88/003, US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, 1988.

7.6.4 Electronic Databases

US Environmental Protection Agency RREL Treatability Database. Computer disk available from Risk Reduction Engineering Laboratory, Cincinnati, OH, 1990.

Vendor Information System for Innovative Treatment Technologies (VISITT). US Environmental Protection Agency, Office of Solid Waste and Emergency Response, PO Box 42419, Cincinnati, OH, 45242-0419.

Clean-up Information Bulletin Board System (CLU-IN). US Environmental Protection Agency, <http://www.clu-in.com>

7.6.5 Cost Analysis / Economics

Blank, Lehlend T. and Anthony Tarquin. Engineering Economy. 3rd ed. McGraw-Hill. New York. 1989.

Collier, Courtland A. and William B. Ledbetter. Engineering Cost Analysis. Harper and Row Publishers. New York. 1982.

Newnan, Donald G. Engineering Economic Analysis. 3rd ed. Engineering Press, Inc., San Jose, CA. 1988.

Fabrycky, W. J. and G. J. Thuesen. Economic Decision Analysis. 2nd ed. Prentice-Hall, Inc., Englewood Cliffs, NJ. 1980.

7.6.6 Natural Attenuation

ASTM Designation: D4448-85a, 1922. Standard Guide for Sampling Groundwater Monitoring Wells. In: ASTM Standards on Groundwater and Vadose Zone Investigations: Drilling, Sampling, Well Installation and Abandonment Procedures (1996), pp. 50-63.

ASTM Designation: RENE, 1996. ASTM Guide for Remediation by Natural Attenuation.

Bioremediation of Chlorinated Solvents Consortium of the Remediation Technologies Development Forum (RTDF), 1996. Guidance Handbook on Natural Attenuation of Chlorinated Solvents, <http://www.rtdf.org>.

Buscheck, T. E., and C. M. Alcantar, 1995. Regression Techniques And Analytical Solutions To Demonstrate Intrinsic Bioremediation. In: Proceedings of the 1995 Battelle 3rd International In Situ and On-Site Bioreclamation Symposium, San Diego, CA., April, pp. 109-116.

Domenico, P.A., 1987. An Analytical Model For Multidimensional Transport Of A Decaying Contaminant Species, *J Hydro*, 91:49-58.

Feenstra, S., J. A. Cherry, and B.L. Parker, 1996. Conceptual Models For The Behavior Of Dense Non-Aqueous Phase Liquids (DNAPLs) In The Subsurface. In: Dense Chlorinated Solvents and Other DNAPLs in Groundwater. J. F. Pankow and J. A. Cherry (eds.), Waterloo Press, Portland, Oregon.

Howard, P.H., R.S. Boethling, W.F. Jarvis, W.M. Meyland, and E.M. Michalenko, 1991. Handbook of Environmental Degradation Rates, Lewis Publishers, Chelsea, MI.

Newell, C.J., J.W. Winters, H.S. Rifai, R.N. Miller, J. Gonzales, and T.H. Wiedemeier, 1995. Modeling Intrinsic Remediation With Multiple Electron Acceptors: Results Form Seven Sites. In: Proceedings of the Petroleum Hydrocarbons and Organic Chemicals in Groundwater Conference, Houston, TX, November, National Groundwater Association, pp. 33-48.

Newell, C.J., R.K. McLeod, and J.R. Gonzales, 1996. BIOSCREEN Natural Attenuation Decision Support System, Version 1.3, US Air Force Center for Environmental Excellence, Brooks AFB, San Antonio, TX.

Rifai, H.S., P. B. Dedient, R.C. Borden, and J.F. Haasbeek, 1987. BIOPLUME II-Computer Model Of Two-Dimensional Transport Under The Influence Of Oxygen Limited Biodegradation In Groundwater, User's Manual, Ver. 1.0, Rice University, Houston, TX.

Rifai, H.S., R.C. Borden, J.T. Wilson, and C.H. Ward, 1995. Intrinsic bioattenuation for subsurface restoration. In: Proceedings of the 1995 Battelle 3rd International In Situ and On-Site Bioreclamation Symposium, San Diego, CA, April, pp. 1-29.

US Environmental Protection Agency (USEPA) 1997. Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites. OSWER Directive 9200.4-17, Office of Solid Waste and Emergency Response. Washington, DC, November.

Wiedemeier, T.H., J.T. Wilson, D.H. Kampbell, R.N. Miller, and J.E. Hansen, 1995. Technical Protocol For Implementing Intrinsic Remediation With Long-Term Monitoring For Natural Attenuation Of Fuel Contamination Dissolved In Groundwater. San Antonio, TX: US Air Force Center for Environmental Excellence.

Wiedemeier, T.H., M.A. Swanson, D.E. Moutoux, J.T. Wilson, D.H. Kampbell, J.E. Hansen, and P. Haas, 1996a. Overview Of The Technical Protocol For Natural Attenuation Of Chlorinated Aliphatic Hydrocarbons In Groundwater Under Development For The US Air Force Center for Environmental Excellence, Symposium on Natural Attenuation of Chlorinated Organics in Groundwater, Dallas, TX, pp. 35-39.

Wiedemeier, T.H., M.A. Swanson, J.T. Wilson, D.H. Kampbell, R.N. Miller, and J.E. Hansen, 1996b. Approximation Of Biodegradation Rate Constants For Monoaromatic Hydrocarbons (BTEX) In Groundwater, Groundwater Monitoring and Remediation, pp. 186-194.

Wiedemeier, T.H., M.A. Swanson, D.E. Moutoux, J.T. Wilson, D.H. Kampbell, J.E. Hansen, P. Haas, and F.H. Chapel, 1996C. Technical Protocol For Natural Attenuation Of Chlorinated Solvents In Groundwater, US Air Force Center for Environmental Excellence, San Antonio, TX.

Wilson, J.T., D.H. Kampbell, and J.W. Weaver, 1996. Environmental Chemistry And The Kinetics Of Biotransformation Of Chlorinated Organic Compounds In Groundwater, EPA Symposium on Natural Attenuation of Chlorinated Organics in Groundwater, Dallas, TX, pp. 124-127.

Zheng, C. 1990. MT3D: A Modular Three-Dimensional Transport Model for Simulation of Advection, Dispersion and Chemical Reaction of Contaminants in Groundwater Systems, Waterloo Hydrogeologic, Inc., Waterloo, Ontario, Canada.

8.0 REMEDIAL ACTION WORKPLAN

8.1 Purpose

The purpose of this workplan is to describe the remedy to be employed at a site and a statement of work and schedule for the remediation. Where various remedial alternatives were considered, the workplan should address the rationale for remedy selection which includes but is not limited to a description of information used in the decision making process, a discussion of potential remediation alternatives, and any uncertainty or risks which exist.

8.2 Information Required

The workplan must address, directly or by reference, the investigation conducted by the applicant to further determine the nature and extent of actual or threatened releases that led to the preparation of the workplan. It will also describe assessments to be performed to further characterize the site or contaminants before remedial action is initiated. Risk assessment conducted to show the appropriateness of remedy selection should be documented in detail as described in Sections 5.0 and 6.0. The statement of work to accomplish the remediation and an implementation schedule must be submitted and must be carried out in accordance with the risk protocol and remediation standards in the rule (§60-3-9). The sampling plan to be implemented following remediation to determine the adequacy of the remediation program must also be addressed in the workplan.

8.3 Remediation Standards

Remediation standards may be attained through one or more remediation activities that can include treatment, removal, engineering or institution controls, natural attenuation and innovative or other demonstrated measures. Remediation standards are to be defined where appropriate for soil, sediment, surface water, and groundwater (§60-3-9.7.d.1). These standards are to be established using the following considerations as described in §60-3-9.

- potential receptors of concern based on the current and reasonably anticipated use of the site;
- site-specific sources of contaminants;
- natural environmental conditions affecting the fate and transport of contaminants, such as natural attenuation processes, as determined by approved scientific methods; and
- institutional and engineering controls.

The remediation standards or combination of standards selected by each applicant for the protection of human health (§60-3-9.1.a) and the ecological receptors (§60-3-9.1.b) must be described including the rationale for the selection of each standard.

8.4 Remediation Measures

Specific remediation measures to be implemented for the site must be addressed. These may include treatment, removal, engineering or institutional controls, natural attenuation and innovative or other demonstrated measure which may be utilized should be defined.

8.4.1 Selection of Alternatives

Where various remedial alternatives were considered, the remedial workplan must address the remedial action selected to achieve the goal of cost effective protection of human health and the environment. Describe:

- the effectiveness of the remedy in protecting human health and the environment;
- the reliability of the remedial action in achieving the standards over the long term;
- the short term risks to the affected community, those engaged in the remedial action effort, and to the environment;
- the acceptability of the remedial action to the affected community;
- the implementability and technical practicability of the remedial action from an engineering perspective;
- the cost effectiveness of the action; and
- the net environmental benefits of the action.

8.4.2 Natural Attenuation

Where the remedy selected is based upon natural processes of degradation and attenuation of contaminants, the remedial workplan must include a description of relevant site-specific conditions, including written documentation of projected groundwater use in the contaminated area based on current state or local government planning efforts; the technical basis for the request; and any other information requested by the Director. The applicant must also demonstrate that all conditions described in §60-3-9.9 of the rule have been satisfied (§60-3-9.9a - §60-3-9.9i).

8.4.3 Uncertainty or Risks

The remedial workplan will include a discussion of any risk or uncertainty associated with selection and implementation of remedial alternatives. It will fully describe any assumptions made in the selection of remediation alternatives and the reason that assumptions

are acceptable and defensible. The workplan will also describe the risks and uncertainties associated with remediation and defend the acceptability of the risks.

8.5 Remediation

The following items which provide details of the remediation activity, must be included in the Remedial Action Workplan:

- statement of work to be conducted to accomplish the proposed remediation;
- schedule for completion of remedial actions;
- verification sampling plan to determine the adequacy of the remediation; and
- any additional information or supporting plans.

8.6 Submittal

Workplans and reports required by the VRA shall be submitted to the Director by the applicant or the applicant's LRS. The Director may approve or disapprove the workplan within 30 days of receipt based on quality and completeness. Disapproval of a workplan must be communicated to the applicant within 5 days of the disapproval with a list of reasons for disapproval and additional information needed. If a workplan is disapproved, the applicant must either resubmit the workplan or formally terminate the Voluntary Remediation Agreement. If workplans are not approved or disapproved within the 30 days, the workplan will be deemed approved. Time limitations on the submission and approval of the Remedial Action Workplan are to be set out in the VRA.

9.0 FINAL REPORT

When all applicable standards developed for the site have been met and all requirements of the VRA have been satisfied, the final report may be prepared and submitted. Sites may be subdivided for the purpose of preparing the final report.

9.1 Contents

The final report shall include:

- all data and information needed to document and verify that all applicable standards have been met and that all activities specified in the Voluntary Remediation Agreement have been completed;
- the site location including the street address, legal description (including lot and block numbers), and site location map;
- the names, addresses, telephone numbers, and facsimile transmission numbers for the current owners and operators of the site, the owners and/or operators conducting the remediation (if different), and the licensed remediation specialist; and
- a description of the ongoing work, such as site cover or treatment system operation and maintenance, groundwater or surface water monitoring, and planned activities and schedules for the aforementioned.

9.2 Appendices

The following information should be attached as appendices to the final report:

- copies of appropriate documents confirming that the institutional controls such as deed restrictions or land use covenants have been properly recorded if they are part of the remediation program (including a site map showing the area(s) subject to the institutional controls) and
- supporting documentation, such as sample collection records, field monitoring data, laboratory reports, relevant correspondence, chain of custody forms, and permits.

9.3 Additional Documentation

Earlier reports, plans, and/or other relevant documents may be incorporated into the final report by reference providing that a complete bibliographic reference is furnished and that the items have been previously submitted to the Division. The use of maps, drawings, photographs, tables, and other aids to visualization and data presentation is encouraged.

9.4 Certification

The completeness and accuracy of the final report will be certified, in writing, by an authorized agent of the applicant and by the Licensed Remediation Specialist. The form of the certification shall be as follows:

I hereby certify that the information presented in this report is, to the best of my knowledge and belief, true, accurate, and complete, having been prepared under a system and organization designed to produce true, accurate, and complete information.

APPENDIX A: CHECKLIST FOR CONCEPTUAL SITE MODEL DEVELOPMENT

This checklist is to be submitted with the application and should incorporate information available at the time of submittal.

Step 1. Define Site Characteristics

- 1.1 Check geologic setting characteristics that apply (“yes” situation found at/near site)
 - fractured rock fill material none as listed above
 - alluvial aquifer karst

- 1.2 Depth to ground water: _____ feet.
 Is the underlying aquifer
 - confined perched unconfined don’t know

- 1.3 General direction of ground water flow across the site:
 - NW N NE E SE S SW W

- 1.4 Local surface water bodies:
 - wetlands spring/seep stream river lake pond/impoundment
 Surface water distance(s) from site _____ miles(s)

- 1.5 Are there known discharge points/springs from the underlying aquifer?
 - yes no
 Distance from site to known discharge points _____ miles(s)

- 1.6 Determine average soil characteristics for usual site conditions:
 - 1. Soil type (check appropriate)
 - clay silt sand gravel
 - 2. Is the average soil or water pH less than or equal to 3 or greater than or equal to 9?
 - yes no

- 1.7 Have any of the following activities occurred at the site?
 - surface mining deep mining injection or extraction wells
 - monitoring wells

Step 2. Define the Contaminant Characteristics

2.1 Basic Contaminant Information

| <u>Contaminant Category</u> | <u>Petroleum</u> | <u>Metals</u> | <u>Other Inorg.</u> | <u>SVOCs</u> | <u>VOCs</u> | <u>PCBs</u> | <u>Pests.</u> | <u>Other</u> |
|-----------------------------|------------------|---------------|---------------------|--------------|-------------|-------------|---------------|--------------|
| Surface Impoundments | - | - | - | - | - | - | - | - |
| Above ground drums | - | - | - | - | - | - | - | - |
| Buried drums | - | - | - | - | - | - | - | - |
| AST | - | - | - | - | - | - | - | - |
| UST | - | - | - | - | - | - | - | - |
| Piles | - | - | - | - | - | - | - | - |
| Landfill | - | - | - | - | - | - | - | - |
| Open dump | - | - | - | - | - | - | - | - |
| Other _____ | - | - | - | - | - | - | - | - |

- 2.2 Indication of Suspected Contamination
- unusual level of vapors erratic behavior of product dispensing equipment
 - release detection results indicate a release discovery of holes in a storage tank
 - spill/release other (specify) _____

- 2.3 Visible evidence of contamination (check all that apply):
- contaminant stained or contaminant saturated soil or backfill
 - ponded contaminants
 - free products or sheen on ponded water
 - free product or sheen on the groundwater surface
 - free product or sheen surface water
 - visual evidence of stressed biota (fish kills, stressed vegetation, etc.)
 - visible presence of oil, tar, or other non-aqueous phase contaminant $\geq 1,000$ sq. ft
 - other (specify) _____

- 2.4 Are there any interim remedial actions that have or will take place?
- yes no (if "yes", fill out 2.5)

2.5 Interim remedial actions (check all that apply):

| | Planned | Initiated | Completed | Not Applicable |
|--|--------------------------|--------------------------|--------------------------|--------------------------|
| Regulated substance removed from storage tanks | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> |
| Containment of contamination | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> |
| Contaminated soil excavated | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> |
| Free product recovered | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> |
| Temporary water supplies provided | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> |
| Other (specify) _____ | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> | <input type="checkbox"/> |

Step 3. Define Exposure Media and Transport Pathways

3.1 Identify media affected (or potentially affected by) contaminants:

Contaminant

- _____ air groundwater surface water soil sediments biota
- _____ air groundwater surface water soil sediments biota
- _____ air groundwater surface water soil sediments biota

3.2 Identify contaminant release mechanisms (check all that apply):

Contaminant

- _____ leaching volatilization fugitive dust erosion / runoff
- _____ leaching volatilization fugitive dust erosion / runoff
- _____ leaching volatilization fugitive dust erosion / runoff

3.3 Groundwater use:

Is the groundwater connected to or part of an aquifer that serves as a source of drinking water?

- yes no (if "yes", groundwater is of concern)

Are there reasonably expected future groundwater uses based on state or local planning?

- yes no (if "yes", groundwater is of concern)

- 3.4 Local water supplies:
- | | | |
|----------------------------------|----------------------------------|-------------------------------|
| Industrial / municipal | <input type="checkbox"/> surface | <input type="checkbox"/> well |
| Residential | <input type="checkbox"/> surface | <input type="checkbox"/> well |
| Agricultural | <input type="checkbox"/> surface | <input type="checkbox"/> well |
| Water supply distance from site: | _____miles(s) | |

3.5 Local surface water (check all that apply):

- Use
- | | |
|---------------------------|--------------------------|
| Domestic supply | <input type="checkbox"/> |
| Recreation | <input type="checkbox"/> |
| Irrigation/stock watering | <input type="checkbox"/> |
| Industrial supply | <input type="checkbox"/> |
| Not currently used | <input type="checkbox"/> |
| Fisheries | <input type="checkbox"/> |
| Other _____ | <input type="checkbox"/> |

3.6 Local groundwater use (check all that apply):

- Use
- | | |
|-----------------------------|--------------------------|
| Domestic supply | <input type="checkbox"/> |
| Irrigation / stock watering | <input type="checkbox"/> |
| Industrial supply | <input type="checkbox"/> |
| Not currently used | <input type="checkbox"/> |
| Other _____ | <input type="checkbox"/> |

3.7 Check if the following exposure pathways are applicable under foreseeable use of the site:

- | | |
|---|---|
| <input type="checkbox"/> soil ingestion | <input type="checkbox"/> surface water ingestion |
| <input type="checkbox"/> inhalation of soil particles/vapors | <input type="checkbox"/> dermal contact with surface water |
| <input type="checkbox"/> dermal contact with soil | <input type="checkbox"/> groundwater ingestion |
| <input type="checkbox"/> consumption of plants | <input type="checkbox"/> consumption of terrestrial animals |
| <input type="checkbox"/> consumption of aquatic organisms | <input type="checkbox"/> other _____ |
| <input type="checkbox"/> inhalation of vapors released from groundwater | |

APPENDIX B: DETERMINING BACKGROUND CONCENTRATIONS

B.1 Choosing Sample Locations

Background concentrations must be determined by sampling areas not affected by site contamination. The selection of a sampling area for background samples is a site-specific decision. The samples should be collected in an unbiased fashion (i.e., not from any areas that are expected to have particularly high or low concentrations). To the extent practical in selecting locations for samples to determine the background levels, the following criteria should be considered as appropriate for soils, sediments, and groundwater. Additional criteria for each media are given below.

1. The samples must be taken up-wind/up-stream and/or up-gradient from suspected or known contamination from the site under study or other sites that are suspected or known to be contaminated.
2. The samples should be taken from areas beyond the contamination boundary but within 5 miles.
3. Samples should be taken from areas that have the same basic characteristics as the medium of concern at the site. The samples should be taken from the same geologic strata as is found at the site.
4. Depth intervals similar to that from which samples will be collected at the site are also to be analyzed. More than one sample at each depth interval and medium within a stratum should be collected.

The same sampling and analysis procedures must be used for the proposed background areas as were used on the site. To the extent practical, the applicant should include a complete and detailed description of the anthropogenic impact history of the areas selected, the basis for concluding anthropogenic contaminants in these areas are not site-related and a justification for their selection as representative of anthropogenic impacts to the site.

B.1.1 Soils

Areas chosen to represent background should be of the same soil type or geologic stratum, and should have no large-scale spatial variations. If the site exhibits large-scale spatial variations, it should be subdivided into more homogeneous subsections and matching background areas should be found for each subsection.

B.1.2 Sediments

For sediments, background samples should be matched for particle size distribution, acid volatile sulfides, total organic carbon, and water content; this may require identifying matched watersheds, or sediment sampling sites sufficiently far downstream to dilute any site influence on

sediment contaminant levels. If the site exhibits large-scale spatial variations, it should be subdivided into more homogeneous subsections and matching background areas should be found for each subsection. Where closely matched sediments cannot be found, the impact should be described in the uncertainty analysis.

B.1.3 Groundwater

Determination of background in groundwater is usually based on comparisons with upgradient wells of similar geologic setting not affected by the site.

B.2 Choosing Sample Size

The minimum number of samples required to establish background levels is site-specific, and, although the methods are not media-specific, samples required may vary for soil vs. groundwater. Gilbert (1993) suggests 10 or more as a “reasonable number” of background measurements, but does not give any justification for this number. Ohio EPA (1991) proposes 7 samples for an initial survey of background, based on the assumption that the desired confidence and the relative standard deviation are equal. Both of these guidelines are only “rules of thumb”. The number of samples required should depend on the desired standard error on the mean and on the variance of the background distribution. If the background concentrations are highly variable, then more samples will be required to achieve a particular standard error on the mean. EPA 1989 and 1996 describe in detail the statistical parameters for choosing sample sizes.

B.3 Report Requirements for Site-Specific Background

The applicant must identify how site background was established and for which media (soil, sediments, groundwater, and/or surface water). The investigative methods used must be identified (e.g. monitoring wells, soil borings, water samples). The sample locations need to be shown on a map (enclosed with the results). The tabular presentation of sample results will facilitate review. The presentation of the results will include, but is not limited to:

1. Description of media sampled (e.g., soil, water, sediments)
2. List of background investigation analytical methods
3. Description of methods used in collecting background data (e.g., soil borings, existing literature.
4. Background sample location map and rationale for sample locations
5. Description of sampling procedure and sampling equipment used
6. Description of monitoring well and/or soil boring installations (if appropriate)

7. Description of field screening procedures used and tabulated results of the field screening procedures.
8. Description of blanks and controls used
9. Presentation of background data in tabular form (media, parameters, concentrations, depth of samples, etc.
10. Statistical evaluation of background results
11. Documentation procedures, waste disposal data and manifests, and chain-of-custody forms

All the samples taken for the intent of determining background levels are to be included in the final report. Statistical analyses must include all data that are not known to be in error, and the source of data quality errors must be described fully for any data which are excluded. The sampling protocols must be the same as will be applied to the samples collected at the site.

B.4 Statistical Methods for Comparison of Site Concentrations With Background

A number of statistical methods have been recommended for comparing site and background concentrations. These methods are independent of the media sampled (soil, sediment, groundwater). These methods include the following:

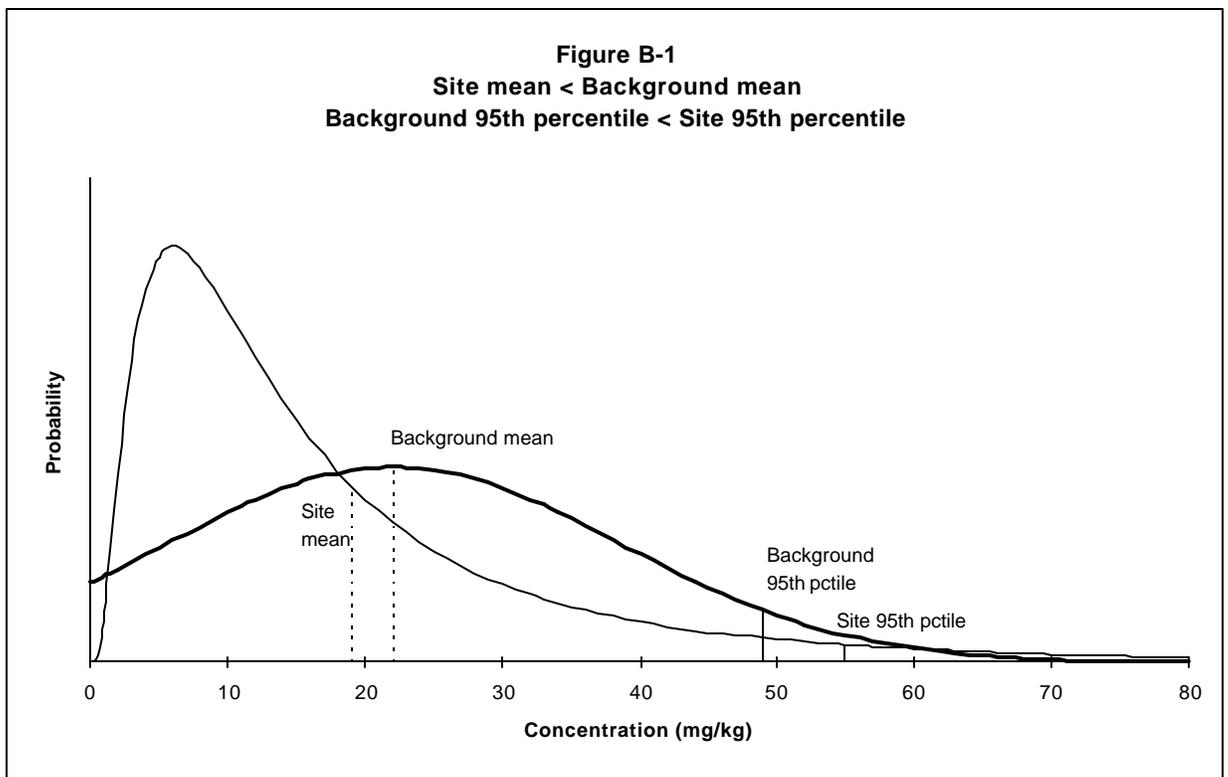
- Comparisons of distributions of site and background concentrations (e.g., quantile test, Wilcoxon rank sum test),
- Comparisons of site and background means (e.g., t-test), and
- Comparisons of high concentrations (e.g., hot measurement comparison, using 95% upper tolerance limit on 95th percentile to represent hot measurement).

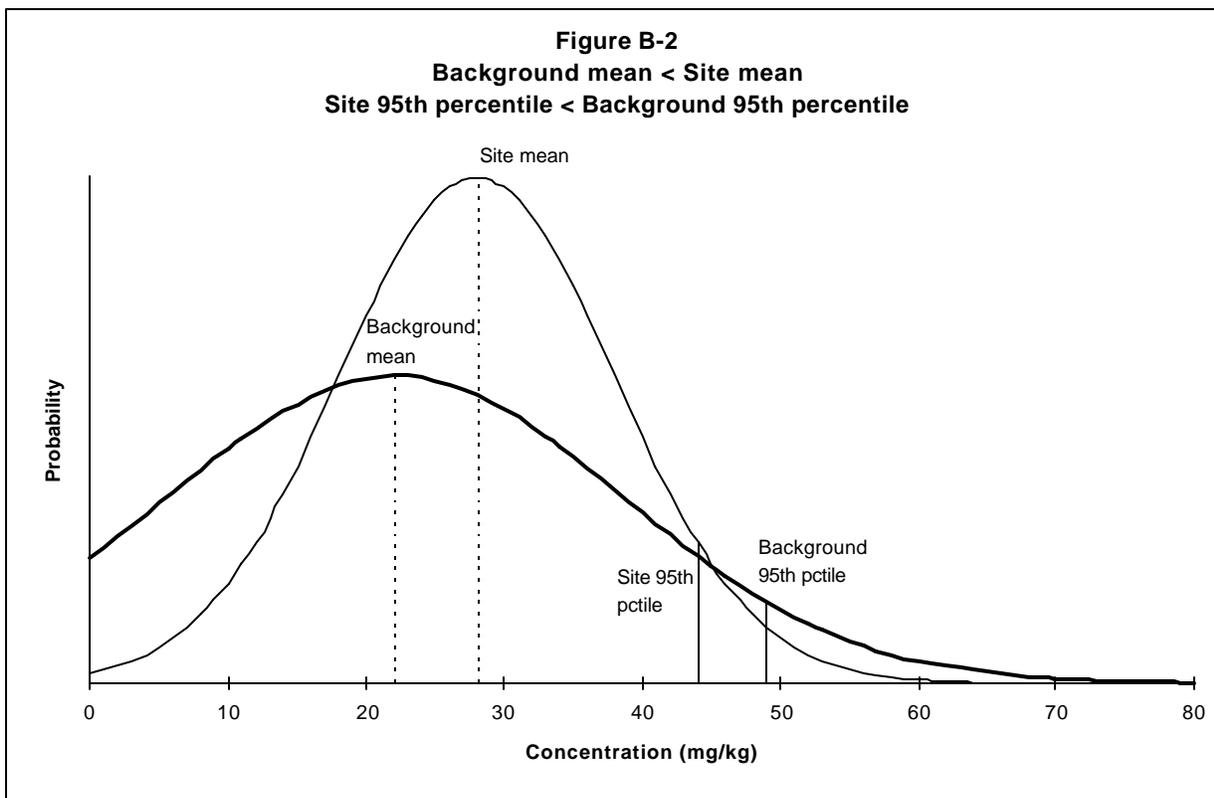
A number of documents describe the various methods, such as USEPA (1989), ASTM (1993), Ohio EPA (1991), and Gilbert (1993, 1987). The statistical tests described in these sources, like most statistical tests, are designed to show that two distributions (or two quantities representing distributions) are different. Failure to show that two distributions are different, however, does not necessarily imply they are the same. If the test fails to show a statistically significant difference, there are two possibilities:

1. The distributions are the same, or
2. The distributions are different, but the test did not have enough power, i.e., there were not enough samples to demonstrate a statistically significant difference.

Using these kinds of tests, there is no way to distinguish between these two possibilities. Consequently, these tests cannot show that two distributions are the same.

This guidance discusses two methods of comparing site data to background. In order to determine whether the site data fall within the range of background concentrations, it is most appropriate to use both a comparison of means and a comparison of individual site concentrations with an upper tolerance limit (UTL) background concentration. Both comparisons are recommended because failure of either alone can indicate that some portion of the site concentrations exceed background. Figures B-1 and B-2 shows sample distributions for site and background. Figure B-1 illustrates a situation where the site mean is less than the background mean, but greater than 5% of site concentrations exceed background in the upper “tail” of the distribution. Figure B-2 illustrates a situation where the site mean exceeds the background mean, but less than 5% of site concentrations exceed background in the upper tail of the distribution. These represent situations where site concentrations may exceed background even though one of the statistical tests is passed.





The following terminology is used in this guidance:

- **Sample mean.** The sample mean is the arithmetic average calculated from a sample consisting of a number of observations.
- **True mean.** The true mean is the mean of the underlying distribution from which the sample is drawn. The true mean is unknown, but it can be estimated by the sample mean. The precision of this estimate improves as the sample size increases.
- **Standard error.** The standard error on the mean is a measure of the uncertainty in the estimate of the true mean. The standard error is defined as (SD/ \sqrt{N}) , where SD is the sample standard deviation and N is the number of observations.
- **Distribution of the mean.** The distribution of the mean describes the uncertainty in the sample mean as an estimate of the true mean. There are many plausible values for the true mean, which is unknown, and probability of each of these values is given by the distribution of the mean. The spread of this distribution is determined by the standard error on the mean.

B.4.1 Comparison of Means

A two tiered approach is recommended here. At sites for which both site and background concentrations are well characterized, so that there is little uncertainty in the two means, the Tier

1 method may be used. As discussed in Section B.5.1.1, this method is a simple comparison of means, where complicated statistical calculations are not required. If background concentrations are well characterized, but site concentrations are not as well characterized, so that there is significant uncertainty in the site mean, the Tier 2 method is applicable. This method, presented in Section B.5.1.2, is more complicated but can be used in a wider range of situations.

Both methods depend on the definition of an acceptable difference, represented by the symbol Δ , between the true site mean and the true background mean. Selection of an appropriate value for Δ is discussed in Section B.5.1.4. Section B.5.1.5 discusses how all of the methods encourage more complete characterization of both site and background concentrations. A flow chart for comparing the site mean to the background mean is provided in Figure B-3.

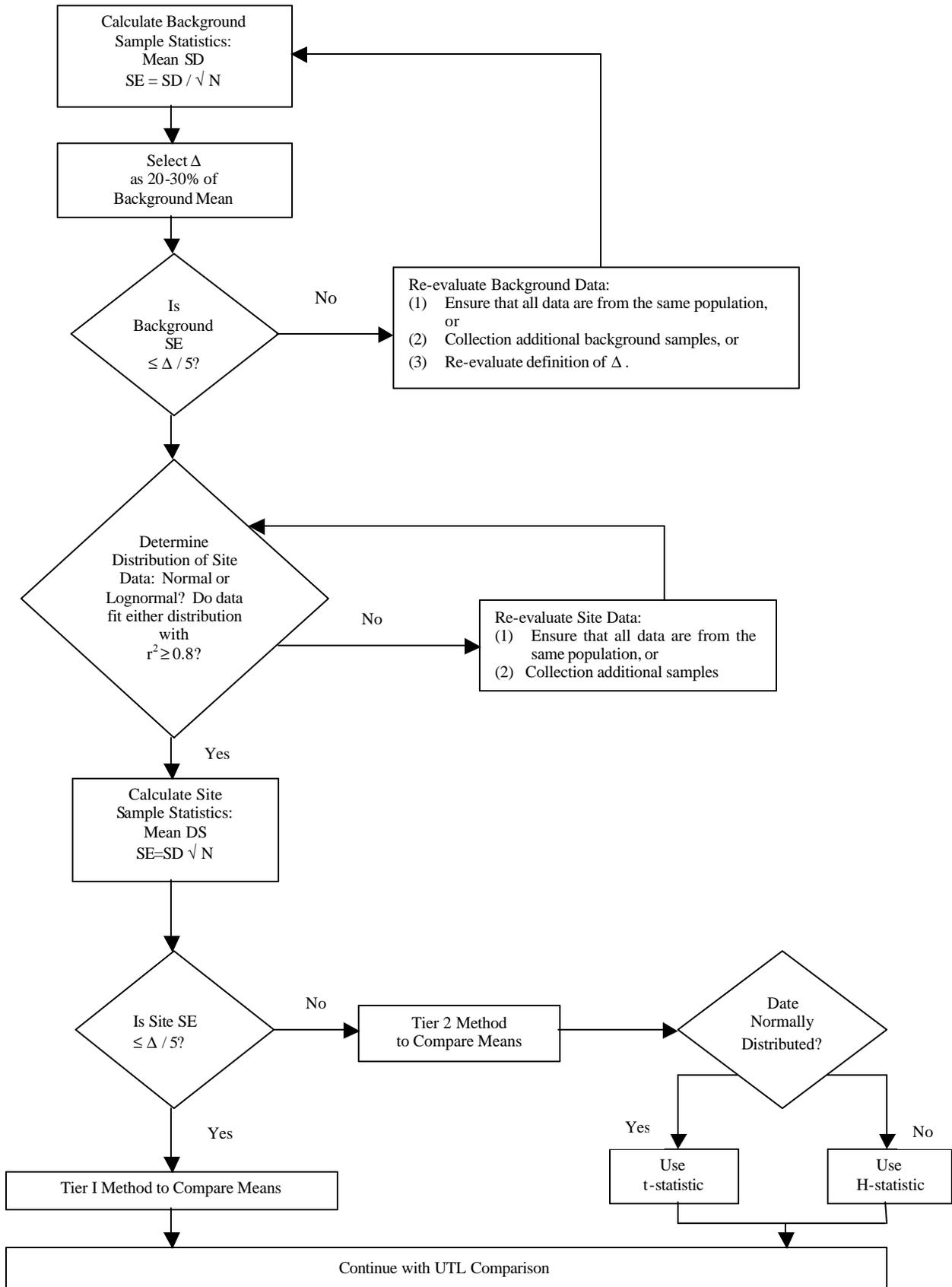
B.4.1.1 Tier 1 Method for Comparing Means

The Tier 1 method depends on two critical assumptions: both the site mean and the background mean are known precisely enough that it is not necessary to consider uncertainty in the means. In other words, it is assumed that the true means are equal to the sample means. If these assumptions are made, then the appropriate test is a simple comparison of sample means. If the site mean is less than or equal to the background mean plus Δ , then the two means are effectively the same, so site and background concentrations can be considered equivalent.

For the two assumptions to be justified, the standard errors on both the site mean and the background mean must be small compared to Δ (e.g., both standard errors should be less than $\Delta/5^5$).

⁵Standard error less than $\Delta/5$ is used throughout this guidance as an example of a reasonable criterion for ignoring the uncertainty in the mean. A different criterion could be used without changing the ideas presented here.

**FIGURE B-3
COMPARISON OF SITE MEAN TO BACKGROUND MEAN**



Otherwise, the true site mean could be substantially higher than the sample mean, or the true background mean could be substantially lower than the sample mean, or both. In either case, the simple comparison of the sample means would not show conclusively that the true means are effectively the same. Consequently, if the standard errors on the means are not small compared to Δ , the Tier 1 method should not be used.

For example, consider the site and background data sets described by the summary statistics in Table B-1.

Table B-1: Summary Statistics for Example Data Sets

| | Site Data | Background Data |
|---------------------------------|------------------|------------------------|
| N | 100 | 25 |
| Sample Mean (ppm) | 27 | 25 |
| Sample Standard Deviation (ppm) | 10 | 5 |
| Standard Error (ppm) | 1.0 | 1.0 |

If Δ , the acceptable difference between the site and background means, is defined as 20% of the background mean (5 ppm, in this case), then both the site and background data sets meet the criterion that the standard error is less than or equal to $\Delta/5$. In this case the site mean is less than the background mean plus Δ (i.e., $25 + 5 = 30$, $27 < 30$), so the conclusion is that the site and background means are equivalent for risk assessment purposes.

B.4.1.2 Method for Comparing Means

The Tier 2 method requires less restrictive assumptions than Tier 1, but the statistical test is more complicated as a result. This method depends on the assumption that the background mean is known precisely enough that it is not necessary to consider the uncertainty in the mean (i.e., the standard error on the background mean is less than $\Delta/5$). The site mean, however, is represented by a probability distribution that incorporates the uncertainty. This is necessary when the standard error on the site mean is not small compared to Δ (i.e., the standard error on the site mean exceeds $\Delta/5$).

In order to show that the site mean is effectively the same as the background mean, the following probability (P^*) must be calculated:

$$P^* = P[\mu_s \leq (\mu_b + \Delta)]$$

where:

P^* is the probability that the true site mean is less than or equal to the true background mean plus Δ ,

μ_s is the true site mean, represented by a probability distribution,

μ_b is the true background mean, assumed to be known exactly, and

Δ is the acceptable difference between μ_s and μ_b .

If the value of P* is sufficiently high (e.g., 80%), then the site and background means are shown to be effectively the same.

P* can be calculated because it is assumed that μ_b is known exactly and the distribution of μ_s is known. If the site data are normally distributed, or if the central limit theorem holds⁶, then the site mean follows the *t* distribution. If the site data are lognormally distributed and the central limit theorem does not hold (data too skewed/not enough observations), then the *H*-statistic must be used to determine the distribution of the site mean (Land, 1975). In either case, it is possible to determine the value of P* by calculating the necessary value of *t* or *H* and using a look-up table to determine the level of confidence that value corresponds to (see End Notes 1 and 2). End Note 3 discusses how to determine if data are normally or lognormally distributed.

As an example, consider the site and background data sets described by the summary statistics in Table B-2. The background data set here is identical to the one in Table B-1, but the site data set has fewer samples and a larger standard deviation. Assuming a Δ of 5 ppm, the Tier 1 method would not be applicable in this case because the standard error on the site mean is not small compared to Δ .

Table B-2: Summary Statistics for Example Data Sets

| | Site Data | Background Data |
|---------------------------------|------------------|------------------------|
| N | 20 | 25 |
| Sample Mean (ppm) | 27 | 25 |
| Sample Standard Deviation (ppm) | 15 | 5 |
| Standard Error (ppm) | 3.4 | 1.0 |

If the central limit theorem applies, P* can be calculated from the *t*-statistic, as explained in End Note 1. The value of the *t*-statistic corresponding to the critical value of $\mu_b + \Delta$ can be calculated as:

$$t = \frac{(\mu_b + \Delta - \bar{x}_s) \cdot \sqrt{N_s}}{SD_s}$$

where:

\bar{x}_s is the sample mean calculated from the site data (27 ppm),

SD_s is the sample standard deviation calculated from the site data (15 ppm),

N_s is the number of observations in the site sample (20),

t is the *t*-statistic,

μ_b is the true background mean, assumed to be exactly (25 ppm), and

Δ is the acceptable difference between the true site mean and the true background mean (5 ppm)

⁶The central theorem states that the distribution describing the uncertainty in the mean of a sample drawn from any distribution approaches normality as the number of observations increases. In general, when the sample variance is unknown, the sample mean follows the *t* distribution. However, if the distribution from which the observations are drawn is sufficiently skewed, and there are an insufficient number of observations, the sample mean will not be *t*-distributed.

From this equation, t is 0.89. For 19 degrees of freedom ($N-1$), this value of t corresponds to the 81st percentile from the distribution of the site mean. As a result, P^* is 0.81. In other words, there is an 81% probability that the true site mean is less than or equal to the true background mean plus Δ . If the required level of confidence is 80%, then it can be concluded that the site and background means are effectively the same.

B.4.1.3 Selection of Δ

All of the methods discussed here depend on the selection of an appropriate Δ . The choice of Δ is a risk management decision. One possibility is to define Δ , which should be chemical-specific, as a percentage of the background mean. For example, if Δ is 20% of the background mean, then an acceptable site mean would be no more than 20% higher than the background mean.

If Δ is too small, then a very large data set would be required to show that the means are effectively the same with any reasonable degree of confidence. For example, consider the case in which $\Delta = \text{zero}$. If the site and background data sets are drawn from the same distribution (so that the means are identical), then it would never be possible to show that $\mu_s \geq \mu_b + \Delta$ with greater than 50% confidence (i.e., $P^* \geq 50\%$). If 80% or 90% confidence is required, then Δ must exceed zero.

B.4.1.4 Required Characterization of Site and Background Concentrations

Both of the recommended methods encourage more complete characterization of both site and background concentrations. The Tier 1 method requires that the uncertainty in both the background and site means be small compared to Δ . This condition can only be met if both site and background concentrations are well characterized. The number of samples required depends on the value of Δ and on the variance of the underlying distributions. A distribution with high variance requires more samples to reduce the uncertainty in the mean.

The Tier 2 method requires that the uncertainty in the background mean be small compared to Δ , which means that background must be well characterized. In addition, this method rewards a more complete characterization of the site, which increases the precision of the estimate of the true site mean. Assuming that site and background are nearly equivalent, P^* will increase as the precision in the estimate of the true site mean increases, showing more conclusively that the site and background means are effectively the same.

B.4.2 Comparison of Individual Samples to an Upper Tolerance Limit

Individual data points from a site should be compared with a value that represents the upper end of the range of background concentrations, with the criteria that a large percentage of them, e.g. 95%, should fall within the range of background. (It would be inappropriate to compare each data point to the background mean, because as many as 50% of the data points could exceed this value even if all site data fell within the range of background). The level that individual data points are compared to is termed an upper tolerance limit (UTL). An upper

tolerance limit (UTL) is usually specified as the 95th percent upper confidence limit on the 95th percentile of the distribution describing the data, where the 95th percentile is the value below which 95% of the data fall. Conceptually, this means that there is a 95 percent certainty, or probability, that 95 percent of the concentrations fall below the UTL. Or, if multiple sets of samples are taken from the same area and the 95th percentile of each sample set is assessed, then 95% of the 95th percentiles would fall below the UTL. A flow chart for comparison of individual site data to background is provided in Figure B-4.

B.4.2.1 Calculating the Upper Tolerance Limit on Normally Distributed Data

The upper tolerance limit on a normally distributed data set is calculated with the k statistic, as described in USEPA (1989) and Gilbert (1987, 1993). The formula is:

$$UTL = \bar{x} + k \cdot s$$

where \bar{x} is the sample mean, s is the sample standard deviation, and k is the k statistic, which is a function of sample size, the percentile for which a UTL is to be estimated (95th in this case), and the confidence limit on this percentile (95th% upper confidence limit). Values of the k statistic are tabulated in USEPA (1989), Table A.4.

B.4.2.2 Calculating the Upper Tolerance Limit on Lognormally Distributed Data

The upper tolerance limit on a lognormally distributed data set is calculated with the k statistic, as described in USEPA (1989) and Gilbert (1987, 1993):

$$UTL = \exp(\bar{x} + k \cdot s)$$

where \bar{x} and s are the mean and standard deviation, respectively, of the log-transformed concentrations, and k is the k statistic, which is a function of sample size, the percentile for which a UTL is to be estimated (95th in this case), and the confidence limit on this percentile (95th% upper confidence limit). Values of the k statistic are tabulated in USEPA (1989), Table A.4.

B.4.2.3 Estimated Upper Limit for Background

The Ohio EPA (1996) estimates an upper limit for background as the mean plus 2 standard deviations:

$$UL_{background} = \bar{x} + 2 \cdot s$$

where \bar{x} is the sample mean and s is the sample standard deviation. When the number of background samples (N) is greater than 70, use of this formula will yield an upper limit that is higher (less conservative) than the UTL, since the value of the k – statistic is less than 2 for $N > 70$.

End Note 1 Calculation of P* from t-Statistic

If the central limit theorem applies, the distribution of the site mean can be approximated by the t distribution. The percentile from this distribution corresponding to the critical value of $\mu_b + \Delta$ can be calculated as:

$$\bar{x}_s + t \cdot \frac{SD_s}{\sqrt{N_s}} = \mathbf{m} + \Delta$$

where:

\bar{x}_s is the sample mean calculated from the site data.

SD_s is the sample standard deviation calculated from the site data,

N_s is the number of observations in the site sample,

t is the t-statistic

μ_b is the true background mean, assumed to known exactly, and

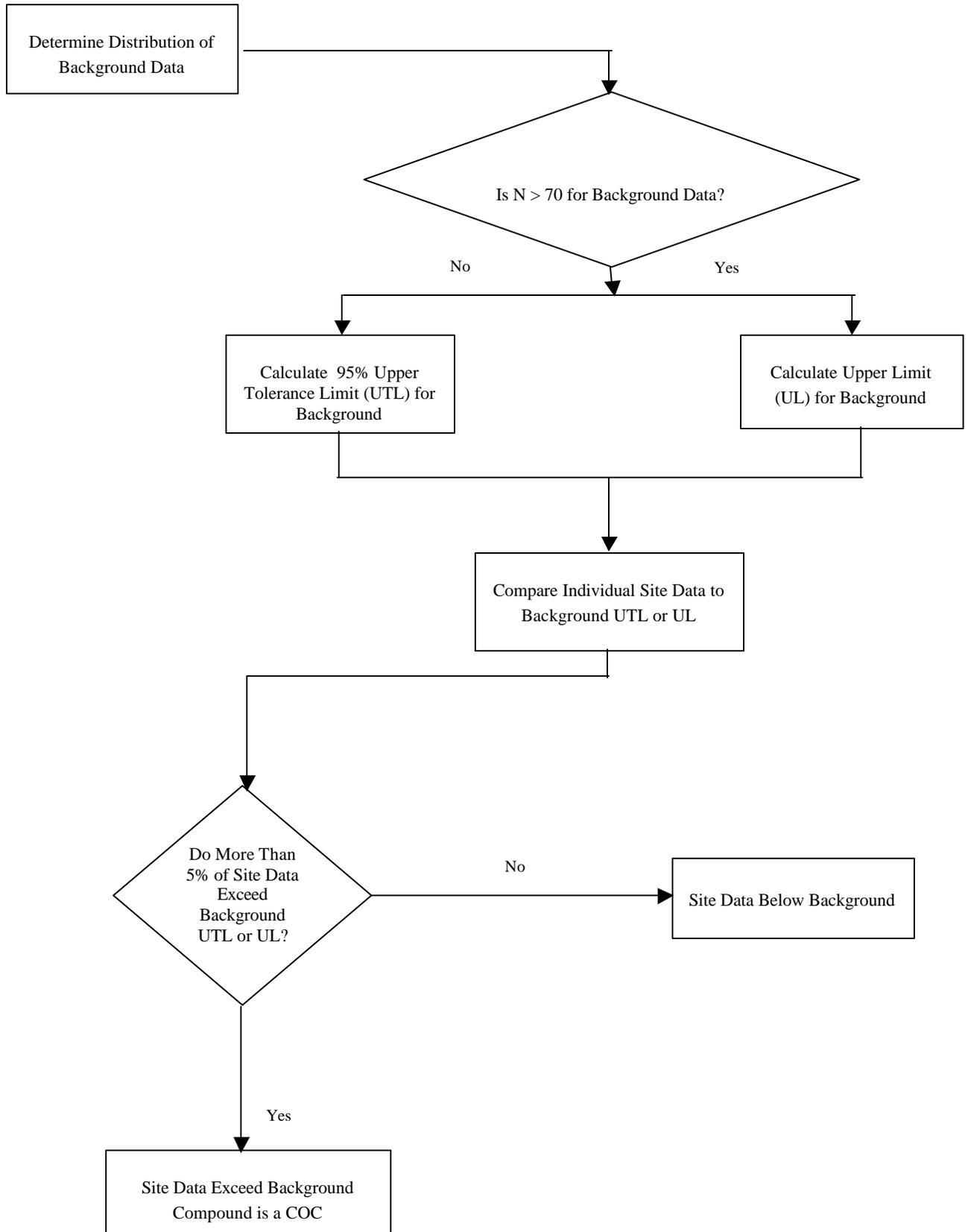
Δ is the acceptable difference between the true site mean and the true background mean.

All of these quantities are known except for t , which can be calculated as:

$$t = \frac{(\mathbf{m} + \Delta - \bar{x}_s) \cdot \sqrt{N_s}}{SD_s}$$

This value of t can be looked up in a table, for the t distribution with $N_s - 1$ degrees of freedom, to determine what level of confidence it corresponds to, which gives the value of P*.

FIGURE B-4
COMPARISON OF INDIVIDUAL SITE DATA TO BACKGROUND UTL



End Note 2 Calculation of P* from H-Statistic

If the site data re lognormally distributed, the percentile from the distribution of the site mean corresponding to the critical value of $\mu_b + \Delta$ can be calculated from the H-statistic (Land, 1975):

$$e^{\left(\log GM_s + \frac{(\log GSD_s)^2}{2} + \frac{(\log GM_s) \cdot H}{\sqrt{N_s - 1}} \right)} = \mathbf{m}_b + \Delta$$

where:

Log refers to the natural logarithm,

GM_s and GSD_s are the sample geometric mean and sample geometric standard deviation calculated from the site data,

N_s is the number of observations in the site sample,

H is the H-statistic,

μ_b is the true background mean, assumed to be exactly, and

Δ is the acceptable difference between the true site mean and the true background mean.

All of these quantities are known except for H, which can be calculated as:

$$H = \frac{\sqrt{N_s - 1} \cdot \left(\log(\mathbf{m}_b + \Delta) - \log GM_s - \frac{(\log GSD_s)^2}{2} \right)}{\log GM_s}$$

The resulting H-statistic can be looked up in a table to determine the level of confidence it corresponds to, which gives the value of P*.

End Note 3 Determining if a Distribution is Normal or Lognormal

In order to determine whether a set of sample data is normally or lognormally distributed, two plots should be constructed. One is a plot of the concentrations vs. their z score, and the second is a plot of the logarithms of the concentrations vs. their Z-score (Figures B-5 and B-6). Data that are normally distributed will plot as a straight line on the first figure, while data that are lognormally distributed will plot as a straight line on the second figure. A linear regression is done for each plot to fit a straight line to the data, and the R-squared value (coefficient of determination) is calculated. The R-squared value is an indication of how well the data fit a straight line. The R-squared values are compared and the plot with the higher R-squared value is considered to be the distribution that best describes the data. Most environmental data fit either a normal or log-normal distribution (Ott, 1990). If the data do not fit either distribution with an R-squared value of at least 0.70, it may be that: 1) not enough samples were collected, or 2) the samples may represent different areas of the site (i.e., different spatial distributions) and may have been inappropriately combined into one data set. In this case it is important to determine the need for additional sample collection and review the sources of the data before proceeding with any analysis.

The probability plots discussed above can be constructed in Excel by the following method:

- List the sample concentrations in ascending order.
- Calculate the natural log of each concentration.
- Determine the rank of each concentration.
- Determine the cumulative probability for each data point. This is equivalent to the Rank (n+1).
- Determine the Z-score for each data point from its cumulative probability. In Excel, the Z-score can be calculated using the NORMSINV function:

$$Z\text{-score} = \text{NORMSINV}(\text{Cumulative Probability})$$

- Plot the Z-score vs. the sample concentration and the Z-score vs. the log-transformed concentrations.

Figure B-5

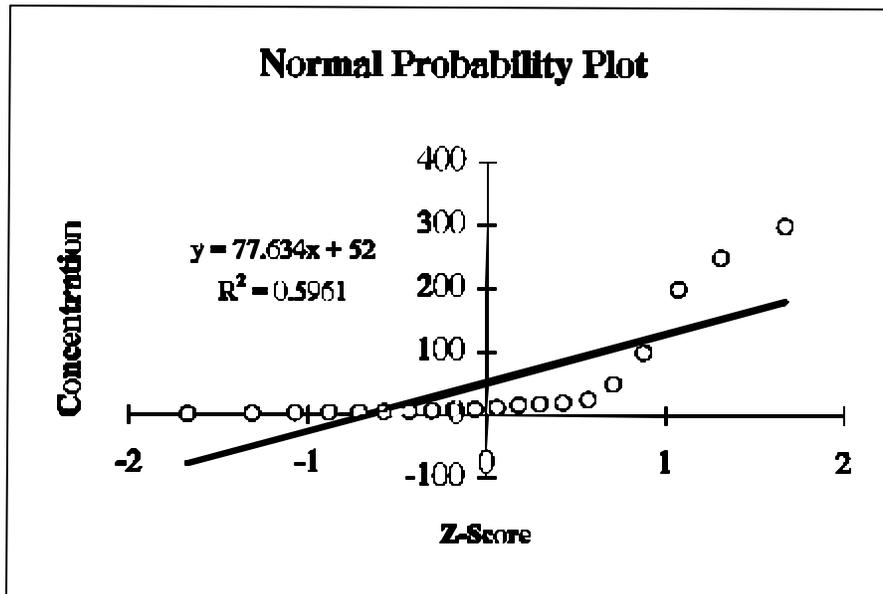
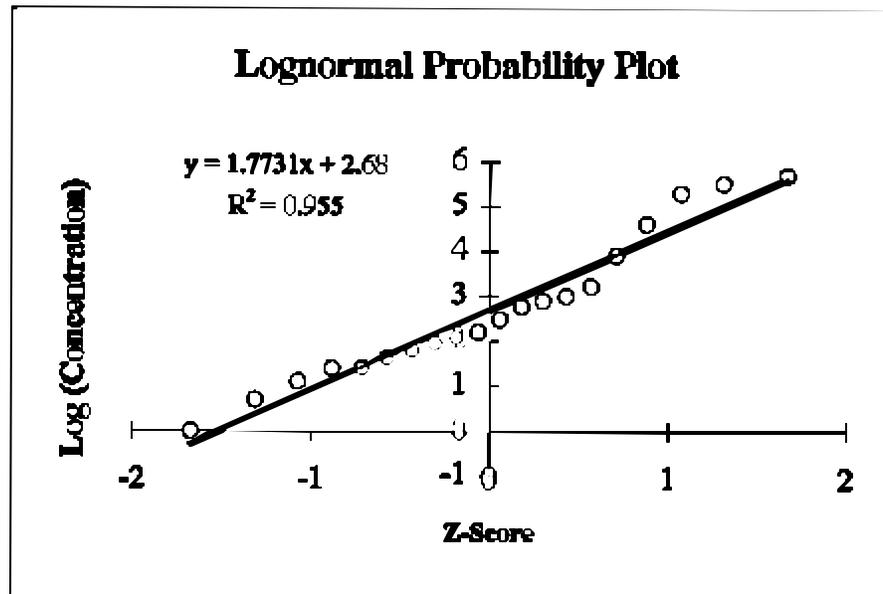


Figure B-6



References

ASTM, 1993. ASTM Guide to the Comparison of Hazardous Waste Site and Background Soil Data

Gilbert, R.O. 1987. *Statistical Methods for Environmental Pollution Monitoring*. Van Nostrand Reinhold: New York, New York, 320 pp.

Gilbert, R.O. 1993. Battelle, Pacific Northwest Laboratories, memo to Beverly Ramsey, describing recommended process for comparison of site data with background concentrations at the Rocky Flats Plant. July 30.

Land, CE. 1975. Tables of Confidence Limits for Linear Functions of the Normal Mean and Variance. Selected Tables in Mathematical Statistics. III: 385-419.

Ohio Environmental Protection Agency. 1991. *How Clean is Clean Policy*, Division of Emergency and Remedial Response, July 26.

Ohio EPA, 1996. Phase II Property Assessments. Voluntary Action Program, Rule OAC 3745-300-07. Division of Emergency and Remedial Response. December 16.

Ott, W.R., 1990. A physical explanation of the lognormality of pollutant concentrations. J. Air Waste Management Assoc. 40:1378:1383.

US Environmental Protection Agency (USEPA). 1989. *Methods for Evaluating the Attainment of Cleanup Standards – Volume I: Soils and Soils Media*. Office of Policy, Planning and Evaluation. EPA/230/02-89-942.

USEPA, Office of Emergency and Remedial Response (Washington, DC). 1992. “Risk Assessment Guidance for Superfund. Volume I: Human Health Evaluation Manual (Part A) – Supplemental Guidance: Calculating the Concentration Term” OSWER Directive 9285.6-03; Publication 9285.7-081; NTIS PB92-963373. May.

USEPA. 1992. Statistical Methods for Evaluating the Attainment of Cleanup Standards: Volume III: Reference Based Standards for Soils and Media. EPA/230/R-94/004. December.

USEPA. 1996. Superfund Guidance for Evaluating the Attainment of Cleanup Standards. Volume I. Soils and Solid Media. Center for Environmental Statistics. Available HTTP: <http://www.epa.gov.ces/pubs.htm>

APPENDIX C-1: DETERMINATION OF THE APPLICABLE HUMAN HEALTH STANDARD

Step 1. Determine Whether the De Minimis Standard is Appropriate for Your Site

The De Minimis Standard applies to chemicals for which the primary exposure routes will be ingestion of soil or groundwater. For soil, the De Minimis Standard is either the risk-based concentration (RBC) (Table 60-3b of the Rule and reproduced in Appendix C of this Guidance) or the natural background levels of the contaminant, whichever is higher. Evaluating a site based on the De Minimis Standard consists of aggregating site data and comparing either maximum concentrations detected, or the 95% upper confidence limit of the arithmetic mean (UCLM) concentration to established RBCs. If site concentrations do not exceed the RBC or site-specific background, then no further evaluation or remediation of the site is required. The De Minimis approach is limited to particular compounds (e.g., it does not apply to volatile compounds), and is appropriate only for residential or industrial exposure scenarios. Below are several questions that will help to determine whether your site may be evaluated under the De Minimis Standard.

Check yes or no for each of the following questions:

- 1.1 Is more than one medium contaminated?
_ yes _ no
- 1.2 Are there more than 10 chemicals present at the site?
_ yes _ no
- 1.3 Are the chemicals at the site known to volatilize or to leach to groundwater?
_ yes _ no
- 1.4 Is future use of the site expected to be other than residential or industrial?
_ yes _ no
- 1.5 Does the site present attractive habitat for wildlife (e.g. wetlands)?
_ yes _ no

If all questions were answered with a “no”, the De Minimis approach is likely to be appropriate for your site and the Worksheet C-1 should be completed. If there are any “yes” responses to the questions above, the De Minimis standard may not be appropriate for the site and more site-specific characterization is needed. If several or all questions were answered with a “yes”, the De Minimis standard is probably not appropriate for your site and options under the Uniform Standard or Site-Specific Assessment should be evaluated.

Step 2. Determine Whether the Uniform Standard is Appropriate for Your Site

The Uniform Standard is based on the use of WVDEP approved methodologies to calculate remediation standards. Equations and default input parameters are provided in Appendix D. Advantages to using the Uniform Standard include the fact that this methodology can be used to determine remediation standards for some chemicals not included under the De Minimis Standard (e.g., volatile compounds), and that with adequate documentation, site-specific information can be incorporated into the calculations. The disadvantages of the approach defined under the Uniform Standard are that exposure scenarios and potential exposure pathways included in these calculations are limited. Specifically, in order to evaluate a site based on the Uniform Standard, land use must be either residential or industrial, and potential exposure pathways for the site are limited to:

- residential ingestion from drinking groundwater
- residential ingestion from drinking surface water
- residential inhalation of volatiles from groundwater
- residential inhalation of volatiles from surface water
- residential ingestion of soil
- residential inhalation of volatiles and particulates from soil
- industrial ingestion of soil
- industrial inhalation of volatiles and particulates from soil

Below are several questions that will assist in determining whether application of the Uniform Standard is appropriate for your site.

Check yes or no for each of the following questions:

- 2.1 Is future use of the site potentially other than residential or industrial use?
_ yes _ no
- 2.2 Is direct contact with impacted surface water (e.g., swimming or wading) likely to occur at your site?
_ yes _ no
- 2.3 Do potentially impacted sediments exist at your site that you feel should not be held to residential or industrial soil cleanup standards?
_ yes _ no
- 2.4 Do home vegetable gardens potentially exist in the vicinity of your site and is homegrown produce potentially impacted by site-related chemicals?
_ yes _ no

- 2.5 Is there any potential for human ingestion of fish that are impacted by site-related chemicals?
 _ yes _ no
- 2.6 Are there any dairy farms or livestock grazing areas within the area of impact of your site?
 _ yes _ no
- 2.7 Is impacted groundwater or surface water used for irrigation or any use other than drinking water?
 _ yes _ no

In addition, if you plan to use site specific modeling in determining exposure point concentrations for media at your site, you should use the site specific assessment.

If you have answered “yes” to any of the questions 2.1 through 2.7 above, then there are potential pathways for human exposure to site-related chemicals that are not addressed in the methodology provided for determining a Uniform Standard. Therefore, a site-specific assessment of human health risks may be more appropriate for your site.

Worksheet C-1. Compare Site Data to Chemical-Specific De Minimis RBC Values

| Contaminant | Max. Soil Conc. mg/kg | UCLM soil mg/kg | Soil Ingestion RBCs Residential/Industrial mg/kg | |
|--------------------|------------------------------|------------------------|---|--|
| | | | | |
| | | | | |
| Contaminant | Max. GW Conc. ug/L | UCLM GW ug/L | GW RBCs ug/L | |
| | | | | |
| | | | | |
| | | | | |

UCLM = 95% upper confidence limit on the arithmetic mean

RBC = Risk Based Concentrations provided in Table 60-3-b of the Rule

If the RBC value exceeds the UCLM site concentration value for a contaminant, attainment is complete for that contaminant. If the UCLM concentrations for all site contaminants are less than the corresponding RBC value, no remediation is required. If the site values exceed the RBC values, additional assessment of the site is required. Refer to Figure 3-1 of this Guidance for a flow diagram regarding how to proceed.

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | Risk-Based Concentrations | | | | | | | | | | | | | | |
|-----------------------------|----------|---------------------------|----------------|-----------------|----------------|-----------------|---|-----------------|---|-------------|-------|-------|---------|--------------|-------|---|
| | | Soil Ingestion | | | | | | | | | | | | Ground Water | | |
| | | RfDo mg/kg/d | I | RfDi mg/kg/d | | CPSo kg d/mg | I | CPSi kg d/mg | I | V O C | µg/L | mg/kg | C | mg/kg | C | |
| Acephate | 30560191 | 4.00E-03 | I | | | 8.70E-03 | I | | | | 7.7 | C | 6600 | C | 73 | C |
| Acetaldehyde | 75070 | | | 2.57E-03 | I | | | 7.70E-03 | I | | 94 | N | | | | |
| Acetochlor | 34256821 | 2.00E-02 | I | | | | | | | | 730 | N | 41000 | N | 1600 | N |
| Acetone | 67641 | 1.00E-01 | I | | | | | | | | 3700 | N | 200000 | N | 7800 | N |
| Acetone cyanohydrin | 75865 | 7.00E-02 | H | 4.00E-02 | A | | | | | | 2600 | N | 140000 | N | 5500 | N |
| Acetonitrile | 75078 | 6.00E-03 | I | 1.43E-02 | A | | | | | | 220 | N | 12000 | N | 470 | N |
| Acetophenone | 98862 | 1.00E-01 | I | 5.71E-06 | W ¹ | | | | | x | 0.042 | N | 200000 | N | 7800 | N |
| Acifluorfen | 62476599 | 1.30E-02 | I | | | | | | | | 470 | N | 27000 | N | 1000 | N |
| Acrolein | 107028 | 2.00E-02 | H | 5.71E-06 | I | | | | | | 730 | N | 41000 | N | 1600 | N |
| Acrylamide | 79061 | 2.00E-04 | I | | | 4.50E+00 | I | 4.55E+00 | I | | 0.015 | C | 13 | C | 0.14 | C |
| Acrylic acid | 79107 | 5.00E-01 | I | 2.86E-04 | I | | | | | | 18000 | N | 1000000 | N | 39000 | N |
| Acrylonitrile | 107131 | 1.00E-03 | H | 5.71E-04 | I | 5.40E-01 | I | 2.38E-01 | I | | 0.12 | C | 110 | C | 1.2 | C |
| Alachlor | 15972608 | 1.00E-02 | I | | | 8.00E-02 | H | | | | 2 | S | 720 | C | 8 | C |
| Alar | 1596845 | 1.50E-01 | I | | | | | | | | 5500 | N | 310000 | N | 12000 | N |
| Aldicarb | 116063 | 1.00E-03 | I | | | | | | | | 37 | N | 2000 | N | 78 | N |
| Aldicarb sulfone | 1646884 | 1.00E-03 | I | | | | | | | | 37 | N | 2000 | N | 78 | N |
| Aldrin | 309002 | 3.00E-05 | I | | | 1.70E+01 | I | 1.71E+01 | I | | 0.004 | C | 3.4 | C | 0.038 | C |
| Ally | 74223646 | 2.50E-01 | I | | | | | | | | 9100 | N | 510000 | N | 20000 | N |
| Allyl alcohol | 107186 | 5.00E-03 | I | | | | | | | | 180 | N | 10000 | N | 390 | N |
| Allyl chloride | 107051 | 5.00E-02 | W ¹ | 2.86E-04 | I | | | | | | 1800 | N | 100000 | N | 3900 | N |
| Aluminum | 7429905 | 1.00E+00 | E ¹ | | | | | | | | 37000 | N | 1000000 | N | 78000 | N |
| Aluminum phosphide | 20859738 | 4.00E-04 | I | | | | | | | | 15 | N | 820 | N | 31 | N |
| Amdro | 67485294 | 3.00E-04 | I | | | | | | | | 11 | N | 610 | N | 23 | N |
| Ametryn | 834128 | 9.00E-03 | I | | | | | | | | 330 | N | 18000 | N | 700 | N |
| m-Aminophenol | 591275 | 7.00E-02 | H | | | | | | | | 2600 | N | 140000 | N | 5500 | N |
| 4-Aminopyridine | 504245 | 2.00E-05 | H | | | | | | | | 0.73 | N | 41 | N | 1.6 | N |
| Amitraz | 33089611 | 2.50E-03 | I | | | | | | | | 91 | N | 5100 | N | 200 | N |
| Ammonia | 7664417 | | | 2.86E-02 | I | | | | | | 1000 | N | | | | |
| Ammonium sulfamate | 7773060 | 2.00E-01 | I | | | | | | | | 7300 | N | 410000 | N | 16000 | N |
| Aniline | 62533 | | | 2.86E-04 | I | 5.70E-03 | I | | | | 10 | N | 10000 | C | 110 | C |
| Antimony and compounds | 7440360 | 4.00E-04 | I | | | | | | | | 6 | S | 820 | N | 31 | N |
| Antimony pentoxide | 1314609 | 5.00E-04 | H | | | | | | | | 18 | N | 1000 | N | 39 | N |
| Antimony potassium tartrate | 304610 | 9.00E-04 | H | | | | | | | | 33 | N | 1800 | N | 70 | N |
| Antimony tetroxide | 1332316 | 4.00E-04 | H | | | | | | | | 15 | N | 820 | N | 31 | N |
| Antimony trioxide | 1309644 | 4.00E-04 | H | | | | | | | | 15 | N | 820 | N | 31 | N |
| Apollo | 74115245 | 1.30E-02 | I | | | | | | | | 470 | N | 27000 | N | 1000 | N |
| Aramite | 140578 | 5.00E-02 | H | | | 2.50E-02 | I | 2.49E-02 | I | | 2.7 | C | 2300 | C | 26 | C |
| Arsenic | 7440382 | 3.00E-04 | I | | | | | | | | 11 | N | 610 | N | 23 | N |
| Arsenic (as carcinogen) | 7440382 | | | | | 1.50E+00 | I | 1.51E+01 | I | | 0.045 | C | 38 | C | 0.43 | C |
| Arsine | 7784421 | | | 1.43E-05 | I | | | | | | 0.52 | N | | | | |
| Assure | 76578148 | 9.00E-03 | I | | | | | | | | 330 | N | 18000 | N | 700 | N |
| Asulam | 3337711 | 5.00E-02 | I | | | | | | | | 1800 | N | 100000 | N | 3900 | N |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations Soil Ingestion | | | | | | | |
|-----------------------------------|----------|-----------------|-----------------|-----------------|-----------------|----------------|---|----------------------|----------------------|----------|--------|--------|-------|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Residential mg/kg | | | | | |
| Atrazine | 1912249 | 3.50E-02 | I | | 2.22E-01 | H | | 3 | S | 260 | C | 2.9 | C | |
| Avermectin B1 | 65195553 | 4.00E-04 | I | | | | | 15 | N | 820 | N | 31 | N | |
| Azobenzene | 103333 | | | | 1.10E-01 | I | 1.08E-01 | I | | 0.61 | C | 520 | C | |
| Barium and compounds | 7440393 | 7.00E-02 | I | 1.43E-04 | A | | | 2000 | S | 140000 | N | 5500 | N | |
| Baygon | 114261 | 4.00E-03 | I | | | | | 150 | N | 8200 | N | 310 | N | |
| Bayleton | 43121433 | 3.00E-02 | I | | | | | 1100 | N | 61000 | N | 2300 | N | |
| Baythroid | 68359375 | 2.50E-02 | I | | | | | 910 | N | 51000 | N | 2000 | N | |
| Benefin | 1861401 | 3.00E-01 | I | | | | | 11000 | N | 610000 | N | 23000 | N | |
| Benomyl | 17804352 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N | |
| Bentazon | 25057890 | 2.50E-03 | I | | | | | 91 | N | 5100 | N | 200 | N | |
| Benzaldehyde | 100527 | 1.00E-01 | I | | | | | x | 610 | N | 200000 | N | 7800 | N |
| Benzene | 71432 | | | 1.71E-03 | E ¹ | 2.90E-02 | I | 2.90E-02 | I | x | 5 | S | 2000 | C |
| Benzenethiol | 108985 | 1.00E-05 | H | | | | | 0.37 | N | 20 | N | 0.78 | N | |
| Benzidine | 92875 | 3.00E-03 | I | | 2.30E+02 | I | 2.35E+02 | I | | 0.00029 | C | 0.25 | C | |
| Benzoic acid | 65850 | 4.00E+00 | I | | | | | 150000 | N | 1000000 | N | 310000 | N | |
| Benzotrichloride | 98077 | | | | 1.30E+01 | I | | 0.0052 | C | 4.4 | C | 0.049 | C | |
| Benzyl alcohol | 100516 | 3.00E-01 | H | | | | | 11000 | N | 610000 | N | 23000 | N | |
| Benzyl chloride | 100447 | | | | 1.70E-01 | I | | x | 0.062 | C | 340 | C | 3.8 | C |
| Beryllium and compounds | 7440417 | 5.00E-03 | I | | 4.30E+00 | I | 8.40E+00 | I | | 4 | S | 13 | C | |
| Bidrin | 141662 | 1.00E-04 | I | | | | | 3.7 | N | 200 | N | 7.8 | N | |
| Biphenthrin (Talstar) | 82657043 | 1.50E-02 | I | | | | | 550 | N | 31000 | N | 1200 | N | |
| 1,1-Biphenyl | 92524 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N | |
| Bis(2-chloroethyl)ether | 111444 | | | | 1.10E+00 | I | 1.16E+00 | I | x | 0.0092 | C | 52 | C | |
| Bis(2-chloroisopropyl)ether | 39638329 | 4.00E-02 | I | | 7.00E-02 | H | 3.50E-02 | H | x | 0.26 | C | 820 | C | |
| Bis(chloromethyl)ether | 542881 | | | | 2.20E+02 | I | 2.17E+02 | I | x | 0.000049 | C | 0.26 | C | |
| Bis(2-chloro-1-methylethyl)ether | 0 | | | | 7.00E-02 | W ¹ | 7.00E-02 | W ¹ | | 0.96 | C | 820 | C | |
| Bis(2-ethylhexyl)phthalate (DEHP) | 117817 | 2.00E-02 | I | | 1.40E-02 | I | | | | 4.8 | C | 4100 | C | |
| Bisphenol A | 80057 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N | |
| Boron (and borates) | 7440428 | 9.00E-02 | I | 5.71E-03 | H | | | 3300 | N | 180000 | N | 7000 | N | |
| Boron trifluoride | 7637072 | | | 2.00E-04 | H | | | 7.3 | N | | | | | |
| Bromodichloromethane | 75274 | 2.00E-02 | I | | 6.20E-02 | I | | x | 0.17 | C | 920 | C | | |
| Bromoethene | 593602 | | | | | | 1.10E-01 | H | x | 0.096 | C | | | |
| Bromofom (tribromomethane) | 75252 | 2.00E-02 | I | | 7.90E-03 | I | 3.85E-03 | I | x | 2.4 | C | 7200 | C | |
| Bromomethane | 74839 | 1.40E-03 | I | 1.43E-03 | I | | | x | 8.7 | N | 2900 | N | | |
| 4-Bromophenyl phenyl ether | 101553 | 5.80E-02 | O ¹ | | | | | | 2100 | N | 120000 | N | 4500 | N |
| Bromophos | 2104963 | 5.00E-03 | H | | | | | | 180 | N | 10000 | N | 390 | N |
| Bromoxynil | 1689845 | 2.00E-02 | I | | | | | | 730 | N | 41000 | N | 1600 | N |
| Bromoxynil octanoate | 1689992 | 2.00E-02 | I | | | | | | 730 | N | 41000 | N | 1600 | N |
| 1,3-Butadiene | 106990 | | | | | | 9.80E-01 | I | x | 0.011 | C | | | |
| 1-Butanol | 71363 | 1.00E-01 | I | | | | | | 3700 | N | 200000 | N | 7800 | N |
| Butyl benzyl phthalate | 85687 | 2.00E-01 | I | | | | | | 7300 | N | 410000 | N | 16000 | N |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | Ground Water µg/L | Soil Ingestion | | | | | | |
|------------------------------|----------|-----------------|-----------------|-----------------|-----------------|-------------|---------------------------|----------------------|-------------------------|----------------|---|-------|-----|------|-----|---|
| | | | | | | | Industrial2 mg/kg | Residential mg/kg | | | | | | | | |
| Butylate | 2008415 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N | | | |
| sec-Butylbenzene | 135988 | 1.00E-02 | E ¹ | | | | x | 61 | N | 20000 | N | 780 | N | | | |
| tert-Butylbenzene | 104518 | 1.00E-02 | E ¹ | | | | x | 61 | N | 20000 | N | 780 | N | | | |
| Butylphthalyl butylglycolate | 85701 | 1.00E+00 | I | | | | | 37000 | N | 1000000 | N | 78000 | N | | | |
| Cacodylic acid | 75605 | 3.00E-03 | H | | | | | 110 | N | 6100 | N | 230 | N | | | |
| Cadmium and compounds | 7440439 | 5.00E-04 | I | 5.71E-05 | W ¹ | | | 6.30E+00 | I | 5 | S | 1000 | N | 39 | N | |
| Caprolactam | 105602 | 5.00E-01 | I | | | | | 18000 | N | 1000000 | N | 39000 | N | | | |
| Captafol | 2425061 | 2.00E-03 | I | | 8.60E-03 | H | | 7.8 | C | 6700 | C | 74 | C | | | |
| Captan | 133062 | 1.30E-01 | I | | 3.50E-03 | H | | 19 | C | 16000 | C | 180 | C | | | |
| Carbaryl | 63252 | 1.00E-01 | I | | | | | 3700 | N | 200000 | N | 7800 | N | | | |
| Carbofuran | 1563662 | 5.00E-03 | I | | | | | 40 | S | 10000 | N | 390 | N | | | |
| Carbon disulfide | 75150 | 1.00E-01 | I | 2.00E-01 | I | | x | 1000 | N | 200000 | N | 7800 | N | | | |
| Carbon tetrachloride | 56235 | 7.00E-04 | I | 5.71E-04 | E ¹ | 1.30E-01 | I | 5.25E-02 | I | x | 5 | S | 440 | C | 4.9 | C |
| Carbosulfan | 55285148 | 1.00E-02 | I | | | | | 370 | N | 20000 | N | 780 | N | | | |
| Carboxin | 5234684 | 1.00E-01 | I | | | | | 3700 | N | 200000 | N | 7800 | N | | | |
| Chloral | 75876 | 2.00E-03 | I | | | | | 73 | N | 4100 | N | 160 | N | | | |
| Chloramben | 133904 | 1.50E-02 | I | | | | | 550 | N | 31000 | N | 1200 | N | | | |
| Chloranil | 118752 | | | | 4.03E-01 | H | | 0.17 | C | 140 | C | 1.6 | C | | | |
| Chlordane | 57749 | 6.00E-05 | I | | 1.30E+00 | I | 1.29E+00 | I | | 2 | S | 44 | C | 0.49 | C | |
| Chlorimuron-ethyl | 90982324 | 2.00E-02 | I | | | | | 730 | N | 41000 | N | 1600 | N | | | |
| Chlorine | 7782505 | 1.00E-01 | I | | | | | 3700 | N | 200000 | N | 7800 | N | | | |
| Chlorine dioxide | 10049044 | | | 5.71E-05 | I | | | 2.1 | N | | | | | | | |
| Chloroacetaldehyde | 107200 | 6.90E-03 | O ¹ | | | | | 250 | N | 14000 | N | 540 | N | | | |
| Chloroacetic acid | 79118 | 2.00E-03 | H | | | | | 73 | N | 4100 | N | 160 | N | | | |
| 2-Chloroacetophenone | 532274 | | | 8.57E-06 | I | | | 0.31 | N | | | | | | | |
| 4-Chloroaniline | 106478 | 4.00E-03 | I | | | | | 150 | N | 8200 | N | 310 | N | | | |
| Chlorobenzene | 108907 | 2.00E-02 | I | 5.71E-03 | A | | x | 100 | S | 41000 | N | 1600 | N | | | |
| Chlorobenzilate | 510156 | 2.00E-02 | I | | 2.70E-01 | H | 2.70E-01 | H | | 0.25 | C | 210 | C | 2.4 | C | |
| p-Chlorobenzoic acid | 74113 | 2.00E-01 | H | | | | | 7300 | N | 410000 | N | 16000 | N | | | |
| 4-Chlorobenzotrifluoride | 98566 | 2.00E-02 | H | | | | | 730 | N | 41000 | N | 1600 | N | | | |
| 2-Chloro-1,3-butadiene | 126998 | 2.00E-02 | A | 2.00E-03 | H | | x | 14 | N | 41000 | N | 1600 | N | | | |
| 1-Chlorobutane | 109693 | 4.00E-01 | H | | | | x | 2400 | N | 820000 | N | 31000 | N | | | |
| Chlorodibromomethane | 124481 | 2.00E-02 | I | | 8.40E-02 | I | x | 0.13 | C | 680 | C | 7.6 | C | | | |
| 1-Chloro-1,1-difluoroethane | 75683 | | | 1.43E+01 | I | | x | 87000 | N | | | | | | | |
| Chlorodifluoromethane | 75456 | | | 1.43E+01 | I | | x | 87000 | N | | | | | | | |
| Chloroethane | 75003 | 4.00E-01 | E ¹ | 2.86E+00 | I | | x | 8600 | N | 820000 | N | 31000 | N | | | |
| 2-Chloroethyl vinyl ether | 110758 | 2.50E-02 | O ¹ | | | | x | 150 | N | 51000 | N | 2000 | N | | | |
| Chloroform | 67663 | 1.00E-02 | I | | 6.10E-03 | I | 8.05E-02 | I | x | 0.15 | C | 9400 | C | 100 | C | |
| Chloromethane | 74873 | | | | 1.30E-02 | H | 6.30E-03 | H | x | 1.40 | C | 4400 | C | 49 | C | |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | Ground Water µg/L | Soil Ingestion | | | |
|--|----------|-----------------|-----------------|-----------------|-----------------|-------------|---------------------------|----------------------|-------------------------|----------------|---|--------|---|
| | | | | | | | Industrial2 mg/kg | Residential mg/kg | | | | | |
| 4-Chloro-2,2-methylaniline hydrochloride | 3165933 | | | 4.60E-01 | H | | | 0.15 | C | 120 | C | 1.4 | C |
| 4-Chloro-2-methylaniline | 95692 | | | 5.80E-01 | H | | | 0.12 | C | 99 | C | 1.1 | C |
| beta-Chloronaphthalene | 91587 | 8.00E-02 | I | | | | | 2900 | N | 160000 | N | 6300 | N |
| o-Chloronitrobenzene | 88733 | | | 2.50E-02 | H | | x | 0.42 | C | 2300 | C | 26 | C |
| p-Chloronitrobenzene | 100005 | | | 1.80E-02 | H | | x | 0.59 | C | 3200 | C | 35 | C |
| 2-Chlorophenol | 95578 | 5.00E-03 | I | | | | | 180 | N | 10000 | N | 390 | N |
| 2-Chloropropane | 75296 | | 2.86E-02 | H | | | x | 170 | N | | | | |
| Chlorothalonil | 1897456 | 1.50E-02 | I | 1.10E-02 | H | | | 6.1 | C | 5200 | C | 58 | C |
| o-Chlorotoluene | 95498 | 2.00E-02 | I | | | | x | 120 | N | 41000 | N | 1600 | N |
| Chlorpropham | 101213 | 2.00E-01 | I | | | | | 7300 | N | 410000 | N | 16000 | N |
| Chlorpyrifos | 2921882 | 3.00E-03 | I | | | | | 110 | N | 6100 | N | 230 | N |
| Chlorpyrifos-methyl | 5598130 | 1.00E-02 | H | | | | | 370 | N | 20000 | N | 780 | N |
| Chlorsulfuron | 64902723 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N |
| Chlorthiophos | 60238564 | 8.00E-04 | H | | | | | 29 | N | 1600 | N | 63 | N |
| Chromium III and compounds | 16065831 | 1.00E+00 | I | 5.71E-07 | W ¹ | | | 37000 | N | 1000000 | N | 78000 | N |
| Chromium VI and compounds | 18540299 | 5.00E-03 | I | | | | | 180 | N | 10000 | N | 390 | N |
| Chromium (total) | | | | | | | | 100 | S | | | | |
| Cobalt | 7440484 | 6.00E-02 | E ¹ | | | | | 2200 | N | 120000 | N | 4700 | N |
| Copper and compounds | 7440508 | 4.00E-02 | E ¹ | | | | | 1500 | N | 82000 | N | 3100 | N |
| Crotonaldehyde | 123739 | 1.00E-02 | W ¹ | | | | | 0.035 | C | 30 | C | 0.34 | C |
| Cumene | 98828 | 4.00E-02 | I | 2.57E-03 | H | | | 1500 | N | 82000 | N | 3100 | N |
| Cyanides: | 0 | | | | | | | | | | | | |
| Barium cyanide | 542621 | 1.00E-01 | W ¹ | | | | | 3700 | N | 200000 | N | 7800 | N |
| Calcium cyanide | 592018 | 4.00E-02 | I | | | | | 1500 | N | 82000 | N | 3100 | N |
| **Chlorine cyanide | 506774 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N |
| Copper cyanide | 544923 | 5.00E-03 | I | | | | | 180 | N | 10000 | N | 390 | N |
| Cyanazine | 21725462 | 2.00E-03 | H | 8.40E-01 | H | | | 0.08 | C | 68 | C | 0.76 | C |
| Cyanogen | 460195 | 4.00E-02 | I | | | | | 1500 | N | 82000 | N | 3100 | N |
| Cyanogen bromide | 506683 | 9.00E-02 | I | | | | | 3300 | N | 180000 | N | 7000 | N |
| Cyanogen chloride | 506774 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N |
| Free cyanide | 57125 | 2.00E-02 | I | | | | | 200 | S | 41000 | N | 1600 | N |
| Hydrogen cyanide | 74908 | 2.00E-02 | I | 8.57E-04 | I | | | 730 | N | 41000 | N | 1600 | N |
| Potassium cyanide | 151508 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N |
| Potassium silver cyanide | 506616 | 2.00E-01 | I | | | | | 7300 | N | 410000 | N | 16000 | N |
| Silver cyanide | 506649 | 1.00E-01 | I | | | | | 3700 | N | 200000 | N | 7800 | N |
| Sodium cyanide | 143339 | 4.00E-02 | I | | | | | 1500 | N | 82000 | N | 3100 | N |
| Thiocyanate | 0 | 2.00E-02 | E ¹ | | | | | 730 | N | 41000 | N | 1600 | N |
| Zinc cyanide | 557211 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N |
| Cyclohexanone | 108941 | 5.00E+00 | I | | | | x | 30000 | N | 1000000 | N | 390000 | N |
| Cyclohexamine | 108918 | 2.00E-01 | I | | | | | 7300 | N | 410000 | N | 16000 | N |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | | | | | | | | |
|--|----------|-----------------|-----------------|-----------------|-----------------|-------------|---------------------------|----------------------|----------------|---|----------------------|---|--------|---|--------|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Soil Ingestion | | Residential mg/kg | | | | | |
| Cyhalothrin/Karate | 68085858 | 5.00E-03 | I | | | | | | | | 180 | N | 10000 | N | 390 | N |
| Cypermethrin | 52315078 | 1.00E-02 | I | | | | | | | | 370 | N | 20000 | N | 780 | N |
| Cyromazine | 66215278 | 7.50E-03 | I | | | | | | | | 270 | N | 15000 | N | 590 | N |
| Dacthal | 1861321 | 1.00E-02 | I | | | | | | | | 370 | N | 20000 | N | 780 | N |
| Dalapon | 75990 | 3.00E-02 | I | | | | | | | | 200 | S | 61000 | N | 2300 | N |
| Danitol | 39515418 | 2.50E-02 | I | | | | | | | | 910 | N | 51000 | N | 2000 | N |
| DDD | 72548 | | | | 2.40E-01 | I | | | | | 0.28 | C | 240 | C | 2.7 | C |
| DDE | 72559 | | | | 3.40E-01 | I | | | | | 0.20 | C | 170 | C | 1.9 | C |
| DDT | 50293 | 5.00E-04 | I | | 3.40E-01 | I | 3.40E-01 | I | | | 0.20 | C | 170 | C | 1.9 | C |
| Decabromodiphenyl ether | 1163195 | 1.00E-02 | I | | | | | | x | | 61 | N | 20000 | N | 780 | N |
| Demeton | 8065483 | 4.00E-05 | I | | | | | | | | 1.5 | N | 82 | N | 3.1 | N |
| Diallate | 2303164 | | | | 6.10E-02 | H | | | x | | 0.17 | C | 940 | C | 10 | C |
| Diazinon | 333415 | 9.00E-04 | H | | | | | | | | 33 | N | 1800 | N | 70 | N |
| Dibenzofuran | 132649 | 4.00E-03 | E ¹ | | | | | | | | 150 | N | 8200 | N | 310 | N |
| 1,4-Dibromobenzene | 106376 | 1.00E-02 | I | | | | | | x | | 61 | N | 20000 | N | 780 | N |
| 1,2-Dibromo -3-chloropropane | 96128 | | | 5.71E-05 | I | 1.40E+00 | H | 2.42E-03 | H | x | 0.2 | S | 41 | C | 0.46 | C |
| 1,2-Dibromoethane | 106934 | | | 5.71E-05 | H | 8.50E+01 | I | 7.70E-01 | I | x | 0.05 | S | 0.67 | C | 0.0075 | C |
| Dibutyl phthalate | 84742 | 1.00E-01 | I | | | | | | | | 3700 | N | 200000 | N | 7800 | N |
| Dicamba | 1918009 | 3.00E-02 | I | | | | | | | | 1100 | N | 61000 | N | 2300 | N |
| 1,2-Dichlorobenzene | 95501 | 9.00E-02 | I | 4.00E-02 | A | | | | | x | 600 | S | 180000 | N | 7000 | N |
| 1,3-Dichlorobenzene | 541731 | 8.90E-02 | O ¹ | | | | | | | x | 600 | S | 180000 | N | 7000 | N |
| 1,4-Dichlorobenzene | 106467 | | | 2.29E-01 | I | 2.40E-02 | H | | | x | 75 | S | 2400 | C | 27 | C |
| 3,3'-Dichlorobenzidine | 91941 | | | | | 4.50E-01 | I | | | | 0.15 | C | 130 | C | 1.4 | C |
| 1,4-Dichloro-2-butene | 764410 | | | | | | | 9.30E+00 | H | x | 0.0011 | C | | | | |
| Dichlorodifluoromethane | 75718 | 2.00E-01 | I | 5.71E-02 | A | | | | | x | 390 | N | 410000 | N | 16000 | N |
| 1,1-Dichloroethane | 75343 | 1.00E-01 | H | 1.43E-01 | A | | | | | x | 810 | N | 200000 | N | 7800 | N |
| 1,2-Dichloroethane (EDC) | 107062 | | | 2.86E-03 | E ¹ | 9.10E-02 | I | 9.10E-02 | I | x | 5 | S | 630 | C | 7 | C |
| 1,1-Dichloroethylene | 75354 | 9.00E-03 | I | | | 6.00E-01 | I | 1.75E-01 | I | x | 7 | S | 95 | C | 1.1 | C |
| 1,2-Dichloroethylene (cis) | 156592 | 1.00E-02 | H | | | | | | | x | 70 | S | 20000 | N | 780 | N |
| 1,2-Dichloroethylene (trans) | 156605 | 2.00E-02 | I | | | | | | | x | 100 | S | 41000 | N | 1600 | N |
| 1,2-Dichloroethylene (mixture) | 540590 | 9.00E-03 | H | | | | | | | x | 55 | N | 18000 | N | 700 | N |
| 2,4-Dichlorophenol | 120832 | 3.00E-03 | I | | | | | | | | 110 | N | 6100 | N | 230 | N |
| 2,4-Dichlorophenoxyacetic Acid (2,4-D) | 94757 | 1.00E-02 | I | | | | | | | x | 70 | S | 20000 | N | 780 | N |
| 4-(2,4-Dichlorophenoxy)butyric Acid | 94826 | 8.00E-03 | I | | | | | | | | 290 | N | 16000 | N | 630 | N |
| 1,2-Dichloropropane | 78875 | | | 1.14E-03 | I | 6.80E-02 | H | | | x | 5 | S | 840 | C | 9.4 | C |
| 2,3-Dichloropropanol | 616239 | 3.00E-03 | I | | | | | | | | 110 | N | 6100 | N | 230 | N |
| 1,3-Dichloropropene | 542756 | 3.00E-04 | I | 5.71E-03 | I | 1.75E-01 | H | 1.30E-01 | H | x | 0.077 | C | 330 | C | 3.7 | C |
| Dichlorvos | 62737 | 5.00E-04 | I | 1.43E-04 | I | 2.90E-01 | I | | | | 0.23 | C | 200 | C | 2.2 | C |
| Dicofol | 115322 | | | | | 4.40E-01 | W ¹ | | | | 0.15 | C | 130 | C | 1.5 | C |
| Dicyclopentadiene | 77736 | 3.00E-02 | H | 5.71E-05 | A | | | | | x | 0.42 | N | 61000 | N | 2300 | N |
| Diieldrin | 60571 | 5.00E-05 | I | | | 1.60E+01 | I | 1.61E+01 | I | | 0.0042 | C | 3.6 | C | 0.04 | C |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | | | | |
|--------------------------------------|----------|-----------------|-----------------|-----------------|-----------------|----------------|---------------------------|----------------------|--|---|---------|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Soil Ingestion Residential mg/kg | | | |
| Diesel emissions | 0 | | 1.43E-03 | I | | | 52 | N | | | | |
| Diethyl phthalate | 84662 | 8.00E-01 | I | | | | 29000 | N | 1000000 | N | 63000 | N |
| Diethylene glycol, monobutyl ether | 112345 | | 5.71E-03 | H | | | 210 | N | | | | |
| Diethylene glycol, monoethyl ether | 111900 | 2.00E+00 | H | | | | 73000 | N | 1000000 | N | 160000 | N |
| Diethylforamide | 617845 | 1.10E-02 | H | | | | 400 | N | 22000 | N | 860 | N |
| Di(2-ethylhexyl)adipate | 103231 | 6.00E-01 | I | | 1.20E-03 | I | 400 | S | 48000 | C | 530 | C |
| Diethylstilbestrol | 56531 | | | | 4.70E+03 | H | 0.000014 | C | 0.012 | C | 0.00014 | C |
| Difenzoquat (Avenge) | 43222486 | 8.00E-02 | I | | | | 2900 | N | 160000 | N | 6300 | N |
| Diflubenzuron | 35367385 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N |
| 1,1-Difluoroethane | 75376 | | 1.14E+01 | I | | x | 69000 | N | | | | |
| Diisopropyl methylphosphonate (DIMP) | 1445756 | 8.00E-02 | I | | | | 2900 | N | 160000 | N | 6300 | N |
| Dimethipin | 55290647 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N |
| Dimethoate | 60515 | 2.00E-04 | I | | | | 7.3 | N | 410 | N | 16 | N |
| 3,3'-Dimethoxybenzidine | 119904 | | | | 1.40E-02 | H | 4.8 | C | 4100 | C | 46 | C |
| Dimethylamine | 124403 | | 5.71E-06 | W ¹ | | | 0.21 | N | | | | |
| 2,4-Dimethylaniline hydrochloride | 21436964 | | | | 5.80E-01 | H | 0.12 | C | 99 | C | 1.1 | C |
| 2,4-Dimethylaniline | 95681 | | | | 7.50E-01 | H | 0.09 | C | 76 | C | 0.85 | C |
| N-N-Dimethylaniline | 121697 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N |
| 3,3'-Dimethylbenzidine | 119937 | | | | 9.20E+00 | H | 0.0073 | C | 6.2 | C | 0.069 | C |
| N,N-Dimethylformamide | 68122 | 1.00E-01 | H | 8.57E-03 | I | | 3700 | N | 200000 | N | 7800 | N |
| 1,1-Dimethylhydrazine | 57147 | | | | 2.60E+00 | W ¹ | 3.50E+00 | W ¹ | 0.026 | C | 22 | C |
| 1,2-Dimethylhydrazine | 540738 | | | | 3.70E+01 | W ¹ | 3.70E+01 | W ¹ | 0.0018 | C | 1.5 | C |
| 2,4-Dimethylphenol | 105679 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N |
| 2,6-Dimethylphenol | 576261 | 6.00E-04 | I | | | | 22 | N | 1200 | N | 47 | N |
| 3,4-Dimethylphenol | 95658 | 1.00E-03 | I | | | | 37 | N | 2000 | N | 78 | N |
| Dimethyl phthalate | 131113 | 1.00E+01 | H | | | | 370000 | N | 1000000 | N | 780000 | N |
| Dimethyl terephthalate | 120616 | 1.00E-01 | I | | | | 3700 | N | 200000 | N | 7800 | N |
| 1,2-Dinitrobenzene | 528290 | 4.00E-04 | H | | | | 15 | N | 820 | N | 31 | N |
| 1,3-Dinitrobenzene | 99650 | 1.00E-04 | I | | | | 3.7 | N | 200 | N | 7.8 | N |
| 1,4-Dinitrobenzene | 100254 | 4.00E-04 | H | | | | 15 | N | 820 | N | 31 | N |
| 4,6-Dinitro-o-cyclohexyl phenol | 131895 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N |
| 2,4-Dinitrophenol | 51285 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N |
| Dinitrotoluene mixture | 0 | | | | 6.80E-01 | I | 0.099 | C | 84 | C | 0.94 | C |
| 2,4-Dinitrotoluene | 121142 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N |
| 2,6-Dinitrotoluene | 606202 | 1.00E-03 | H | | | | 37 | N | 2000 | N | 78 | N |
| Dinoseb | 88857 | 1.00E-03 | I | | | | 7 | S | 2000 | N | 78 | N |
| di-n-Octyl phthalate | 117840 | 2.00E-02 | H | | | | 6 | S | 41000 | N | 1600 | N |
| 1,4-Dioxane | 123911 | | | | 1.10E-02 | I | 6.1 | C | 5200 | C | 58 | C |
| Diphenamid | 957517 | 3.00E-02 | I | | | | 1100 | N | 61000 | N | 2300 | N |
| Diphenylamine | 122394 | 2.50E-02 | I | | | | 910 | N | 51000 | N | 2000 | N |
| 1,2-Diphenylhydrazine | 122667 | | | | 8.00E-01 | I | 7.70E-01 | I | 0.084 | C | 72 | C |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations Soil Ingestion | | | | | | | | |
|--|----------|-----------------|-----------------|-----------------|-----------------|-------------|---|----------------------|----------------------|--------|--------|-------|---|----|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Residential mg/kg | | | | | | |
| Diquat | 85007 | 2.20E-03 | I | | | | 20 | S | 4500 | N | 170 | N | | | |
| Direct black 38 | 1937377 | | | 8.60E+00 | H | | 0.0078 | C | 6.7 | C | 0.074 | C | | | |
| Direct blue 6 | 2602462 | | | 8.10E+00 | H | | 0.0083 | C | 7.1 | C | 0.079 | C | | | |
| Direct brown 95 | 16071866 | | | 9.30E+00 | H | | 0.0072 | C | 6.2 | C | 0.069 | C | | | |
| Disulfoton | 298044 | 4.00E-05 | I | | | | 1.5 | N | 82 | N | 3.1 | N | | | |
| 1,4-Dithiane | 505293 | 1.00E-02 | I | | | | 370 | N | 20000 | N | 780 | N | | | |
| Diuron | 330541 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N | | | |
| Dodine | 2439103 | 4.00E-03 | I | | | | 150 | N | 8200 | N | 310 | N | | | |
| Endosulfan | 115297 | 6.00E-03 | I | | | | 220 | N | 12000 | N | 470 | N | | | |
| Endothall | 145733 | 2.00E-02 | I | | | | 100 | S | 41000 | N | 1600 | N | | | |
| Endrin | 72208 | 3.00E-04 | I | | | | 2 | S | 610 | N | 23 | N | | | |
| Epichlorohydrin | 106898 | 2.00E-03 | H | 2.86E-04 | I | 9.90E-03 | I | 4.20E-03 | I | 6.8 | C | 5800 | C | 65 | C |
| 1,2-Epoxybutane | 106887 | | | 5.71E-03 | I | | | | | 210 | N | | | | |
| Ethephon (2-chloroethyl phosphonic acid) | 16672870 | 5.00E-03 | I | | | | 180 | N | 10000 | N | 390 | N | | | |
| Ethion | 563122 | 5.00E-04 | I | | | | 18 | N | 1000 | N | 39 | N | | | |
| 2-Ethoxyethanol acetate | 111159 | 3.00E-01 | A | | | | 11000 | N | 610000 | N | 23000 | N | | | |
| 2-Ethoxyethanol | 110805 | 4.00E-01 | H | 5.71E-02 | I | | 15000 | N | 820000 | N | 31000 | N | | | |
| Ethyl acrylate | 140885 | | | 4.80E-02 | H | | 1.4 | C | 1200 | C | 13 | C | | | |
| EPTC (S-Ethyl dipropylthiocarbamate) | 759944 | 2.50E-02 | I | | | | 910 | N | 51000 | N | 2000 | N | | | |
| Ethyl acetate | 141786 | 9.00E-01 | I | | | | 33000 | N | 1000000 | N | 70000 | N | | | |
| Ethylbenzene | 100414 | 1.00E-01 | I | 2.86E-01 | I | | x | 700 | S | 200000 | N | 7800 | N | | |
| Ethylene cyanohydrin | 109784 | 3.00E-01 | H | | | | 11000 | N | 610000 | N | 23000 | N | | | |
| Ethylene diamine | 107153 | 2.00E-02 | H | | | | 730 | N | 41000 | N | 1600 | N | | | |
| Ethylene glycol | 107211 | 2.00E+00 | I | | | | 73000 | N | 1000000 | N | 160000 | N | | | |
| Ethylene glycol, monobutyl ether | 111762 | | | 5.71E-03 | H | | 210 | N | | | | | | | |
| Ethylene oxide | 75218 | | | 1.02E+00 | H | 3.50E-01 | H | 0.066 | C | 56 | C | 0.63 | C | | |
| Ethylene thiourea (ETU) | 96457 | 8.00E-05 | I | | | 1.19E-01 | H | 0.57 | C | 480 | C | 5.4 | C | | |
| Ethyl ether | 60297 | 2.00E-01 | I | | | | x | 1200 | N | 410000 | N | 16000 | N | | |
| Ethyl methacrylate | 97632 | 9.00E-02 | H | | | | 3300 | N | 180000 | N | 7000 | N | | | |
| Ethyl p-nitrophenyl phenylphosphorothioate | 2104645 | 1.00E-05 | I | | | | 0.37 | N | 20 | N | 0.78 | N | | | |
| Ethyl nitrosourea | 759739 | | | 1.40E+02 | W ¹ | | 0.00048 | C | 0.41 | C | 0.0046 | C | | | |
| Ethylphthalyl ethyl glycolate | 84720 | 3.00E+00 | I | | | | 110000 | N | 1000000 | N | 230000 | N | | | |
| Express | 10120 | 8.00E-03 | I | | | | 290 | N | 16000 | N | 630 | N | | | |
| Fenamiphos | 22224926 | 2.50E-04 | I | | | | 9.1 | N | 510 | N | 20 | N | | | |
| Fluometuron | 2164172 | 1.30E-02 | I | | | | 470 | N | 27000 | N | 1000 | N | | | |
| Fluoride | 7782414 | 6.00E-02 | I | | | | 4000 | S | 120000 | N | 4700 | N | | | |
| Fluoridone | 59756604 | 8.00E-02 | I | | | | 2900 | N | 160000 | N | 6300 | N | | | |
| Flurprimidol | 56425913 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N | | | |
| Flutolanil | 66332965 | 6.00E-02 | I | | | | 2200 | N | 120000 | N | 4700 | N | | | |
| Fluvalinate | 69409945 | 1.00E-02 | I | | | | 370 | N | 20000 | N | 780 | N | | | |
| Folpet | 133073 | 1.00E-01 | I | | | 3.50E-03 | I | 19 | C | 16000 | C | 180 | C | | |
| Fomesafen | 72178020 | | | 1.90E-01 | I | | 0.35 | C | 300 | C | 3.4 | C | | | |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | Risk-Based Concentrations | | | | | | | | | | | | | | |
|---|----------|---------------------------|-----------------|-----------------|-----------------|-------------|----------|----------------------|----------------------|---|----------|--------------|---------|---|--------|---|
| | | Soil Ingestion | | | | | | | | | | Ground Water | | | | |
| | | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | µg/L | Industrial2 mg/kg | Residential mg/kg | | | | | | | |
| Fonofos | 944229 | 2.00E-03 | I | | | | | | | | 73 | N | 4100 | N | 160 | N |
| Formaldehyde | 50000 | 2.00E-01 | I | | | | 4.55E-02 | I | | | 7300 | N | 410000 | N | 16000 | N |
| Formic Acid | 64186 | 2.00E+00 | H | | | | | | | | 73000 | N | 1000000 | N | 160000 | N |
| Fosetyl-al | 39148248 | 3.00E+00 | I | | | | | | | | 110000 | N | 1000000 | N | 230000 | N |
| Furan | 110009 | 1.00E-03 | I | | | | | | | | 37 | N | 2000 | N | 78 | N |
| Furazolidone | 67458 | | | | | 3.80E+00 | H | | | | 0.018 | C | 15 | C | 0.17 | C |
| Furfural | 98011 | 3.00E-03 | I | 1.43E-02 | A | | | | | | 110 | N | 6100 | N | 230 | N |
| Furium | 531828 | | | | | 5.00E+01 | H | | | | 0.0013 | C | 1.1 | C | 0.013 | C |
| Furmecyclo | 60568050 | | | | | 3.00E-02 | I | | | | 2.2 | C | 1900 | C | 21 | C |
| Glufosinate-ammonium | 77182822 | 4.00E-04 | I | | | | | | | | 15 | N | 820 | N | 31 | N |
| Glycidaldehyde | 765344 | 4.00E-04 | I | 2.86E-04 | H | | | | | | 15 | N | 820 | N | 31 | N |
| Glyphosate | 1071836 | 1.00E-01 | I | | | | | | | | 700 | S | 200000 | N | 7800 | N |
| Haloxypop-methyl | 69806402 | 5.00E-05 | I | | | | | | | | 1.8 | N | 100 | N | 3.9 | N |
| Harmony | 79277273 | 1.30E-02 | I | | | | | | | | 470 | N | 27000 | N | 1000 | N |
| HCH (alpha) | 319846 | | | | | 6.30E+00 | I | 6.30E+00 | I | | 0.011 | C | 9.1 | C | 0.1 | C |
| HCH (beta) | 319857 | | | | | 1.80E+00 | I | 1.80E+00 | I | | 0.037 | C | 32 | C | 0.35 | C |
| HCH (gamma) Lindane | 58899 | 3.00E-04 | I | | | 1.30E+00 | H | | | | 0.2 | S | 44 | C | 0.49 | C |
| HCH-technical | 608731 | | | | | 1.80E+00 | I | 1.79E+00 | I | | 0.037 | C | 32 | C | 0.35 | C |
| Heptachlor | 76448 | 5.00E-04 | I | | | 4.50E+00 | I | 4.55E+00 | I | x | 0.4 | S | 13 | C | 0.14 | C |
| Heptachlor epoxide | 1024573 | 1.30E-05 | I | | | 9.10E+00 | I | 9.10E+00 | I | x | 0.2 | S | 6.3 | C | 0.07 | C |
| Hexabromo benzene | 87821 | 2.00E-03 | I | | | | | | | x | 12 | N | 4100 | N | 160 | N |
| Hexachlorobenzene | 118741 | 8.00E-04 | I | | | 1.60E+00 | I | 1.61E+00 | I | x | 1 | S | 36 | C | 0.4 | C |
| Hexachlorobutadiene | 87683 | 2.00E-04 | H | | | 7.80E-02 | I | 7.70E-02 | I | x | 0.14 | C | 730 | C | 8.2 | C |
| Hexachlorocyclopentadiene | 77474 | 7.00E-03 | I | 2.00E-05 | H | | | | | x | 50 | S | 14000 | N | 550 | N |
| Hexachlorodibenzo-p-dioxin mixture | 19408743 | | | | | 6.20E+03 | I | 4.55E+03 | I | | 0.000011 | C | 0.0092 | C | 0.0001 | C |
| Hexachloroethane | 67721 | 1.00E-03 | I | | | 1.40E-02 | I | 1.40E-02 | I | x | 0.75 | C | 4100 | C | 46 | C |
| Hexachlorophene | 70304 | 3.00E-04 | I | | | | | | | | 11 | N | 610 | N | 23 | N |
| Hexahydro-1,3,5-trinitro-1,3,5-triazine | 121824 | 3.00E-03 | I | | | 1.10E-01 | I | | | | 0.61 | C | 520 | C | 5.8 | C |
| 1,6-Hexamethylene diisocyanate | 822060 | | | 2.86E-06 | I | | | | | | 0.10 | N | | | | |
| n-Hexane | 110543 | 6.00E-02 | H | 5.71E-02 | I | | | | | x | 350 | N | 120000 | N | 4700 | N |
| Hexazinone | 51235042 | 3.30E-02 | I | | | | | | | | 1200 | N | 67000 | N | 2600 | N |
| Hydrazine, hydrazine sulfate | 302012 | | | | | 3.00E+00 | I | 1.71E+01 | I | | 0.022 | C | 19 | C | 0.21 | C |
| Hydrogen chloride | 7647010 | | | 5.71E-03 | I | | | | | | 210 | N | | | | |
| Hydrogen sulfide | 7783064 | 3.00E-03 | I | 2.85E-04 | I | | | | | | 110 | N | 6100 | N | 230 | N |
| Hydroquinone | 123319 | 4.00E-02 | H | | | | | | | | 1500 | N | 82000 | N | 3100 | N |
| Imazalil | 35554440 | 1.30E-02 | I | | | | | | | | 470 | N | 27000 | N | 1000 | N |
| Imazaquin | 81335377 | 2.50E-01 | I | | | | | | | | 9100 | N | 510000 | N | 20000 | N |
| Iprodione | 36734197 | 4.00E-02 | I | | | | | | | | 1500 | N | 82000 | N | 3100 | N |
| Iron | 7439896 | 3.00E-01 | E ¹ | | | | | | | | 11000 | N | 610000 | N | 23000 | N |
| Isobutanol | 78831 | 3.00E-01 | I | | | | | | | x | 1800 | N | 610000 | N | 23000 | N |
| Isophorone | 78591 | 2.00E-01 | I | | | 9.50E-04 | I | | | | 71 | C | 60000 | C | 670 | C |
| Isopropalin | 33820530 | 1.50E-02 | I | | | | | | | | 550 | N | 31000 | N | 1200 | N |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations Soil Ingestion | | | | | |
|---|-----------|-----------------|-----------------|-----------------|-----------------|-------------|---|----------------------|----------------------|----|-------|----------------|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Residential mg/kg | | | |
| Isopropyl methyl phosphonic acid | 1832548 | 1.00E-01 | I | | | | 3700 | N | 200000 | N | 7800 | N |
| Isoxaben | 82558507 | 5.00E-02 | I | | | | 1800 | N | 100000 | N | 3900 | N |
| Kepone | 143500 | | | 1.80E+01 | E ¹ | | 0.0037 | C | 3.2 | C | 0.035 | C |
| Lactofen | 77501634 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N |
| Lead | 7439-92-1 | | | | | | 15 | S | 1000 | R1 | 400 | R ² |
| Linuron | 330552 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N |
| Lithium | 7439932 | 2.00E-02 | E ¹ | | | | 730 | N | 41000 | N | 1600 | N |
| Londax | 83056996 | 2.00E-01 | I | | | | 7300 | N | 410000 | N | 16000 | N |
| Malathion | 121755 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N |
| Maleic anhydride | 108316 | 1.00E-01 | I | | | | 3700 | N | 200000 | N | 7800 | N |
| Maleic hydrazide | 123331 | 5.00E-01 | I | | | | 18000 | N | 1000000 | N | 39000 | N |
| Malononitrile | 109773 | 2.00E-05 | H | | | | 0.73 | N | 41 | N | 1.6 | N |
| Mancozeb | 8018017 | 3.00E-02 | H | | | | 1100 | N | 61000 | N | 2300 | N |
| Maneb | 12427382 | 5.00E-03 | I | | | | 180 | N | 10000 | N | 390 | N |
| **Manganese and compounds | 7439965 | 2.30E-02 | I | 1.43e-05 | I | | 840 | N | 47000 | N | 1800 | N |
| Mephosfolan | 950107 | 9.00E-05 | H | | | | 3.3 | N | 180 | N | 7 | N |
| Mepiquat chloride | 24307264 | 3.00E-02 | I | | | | 1100 | N | 61000 | N | 2300 | N |
| Mercuric chloride | 7487947 | 3.00E-04 | I | | | | 11 | N | 610 | N | 23 | N |
| Mercury (inorganic) | 7439976 | 3.00E-04 | H | 8.57E-05 | H | | 2 | S | 610 | N | 23 | N |
| Mercury (methyl) | 22967926 | 1.00E-04 | I | | | | 3.7 | N | 200 | N | 7.8 | N |
| Merphos | 150505 | 3.00E-05 | I | | | | 1.1 | N | 61 | N | 2.3 | N |
| Merphos oxide | 78488 | 3.00E-05 | I | | | | 1.1 | N | 61 | N | 2.3 | N |
| Metalaxyl | 57837191 | 6.00E-02 | I | | | | 2200 | N | 120000 | N | 4700 | N |
| Methacrylonitrile | 126987 | 1.00E-04 | I | 2.00E-04 | A | | 3.7 | N | 200 | N | 7.8 | N |
| Methamidophos | 10265926 | 5.00E-05 | I | | | | 1.8 | N | 100 | N | 3.9 | N |
| Methanol | 67561 | 5.00E-01 | I | | | | 18000 | N | 1000000 | N | 39000 | N |
| Methidathion | 950378 | 1.00E-03 | I | | | | 37 | N | 2000 | N | 78 | N |
| Methomyl | 16752775 | 2.50E-02 | I | | | | 910 | N | 51000 | N | 2000 | N |
| Methoxychlor | 72435 | 5.00E-03 | I | | | | 40 | S | 10000 | N | 390 | N |
| 2-Methoxyethanol acetate | 110496 | 2.00E-03 | A | | | | 73 | N | 4100 | N | 160 | N |
| 2-Methoxyethanol | 109864 | 1.00E-03 | H | 5.71E-03 | I | | 37 | N | 2000 | N | 78 | N |
| 2-Methoxy-5-nitroaniline | 99592 | | | 4.60E-02 | H | | 1.5 | C | 1200 | C | 14 | C |
| Methyl acetate | 79209 | 1.00E+00 | H | | | | 37000 | N | 1000000 | N | 78000 | N |
| Methyl acrylate | 96333 | 3.00E-02 | A | | | | 1100 | N | 61000 | N | 2300 | N |
| 2-Methylaniline hydrochloride | 636215 | | | 1.80E-01 | H | | 0.37 | C | 320 | C | 3.5 | C |
| 2-Methylaniline | 95534 | | | 2.40E-01 | H | | 0.28 | C | 240 | C | 2.7 | C |
| Methyl chlorocarbonate | 79221 | 1.00E+00 | W ¹ | | | | 37000 | N | 1000000 | N | 78000 | N |
| 4-(2-Methyl-4-chlorophenoxy) butyric acid | 94815 | 1.00E-02 | I | | | | 370 | N | 20000 | N | 780 | N |
| 2-Methyl-4-chlorophenoxyacetic acid | 94746 | 5.00E-04 | I | | | | 18 | N | 1000 | N | 39 | N |
| 2-(2-Methyl-14-chlorophenoxy)propionic acid | 93652 | 1.00E-03 | I | | | | 37 | N | 2000 | N | 78 | N |
| Methylcyclohexane | 108872 | | | 8.57E-01 | H | | 31000 | N | | | | |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | | | | | | | | |
|--|----------|-----------------|-----------------|-----------------|-----------------|-------------|---------------------------|----------------------|-------|---|---------|---|---------|---|--------|---|
| | | | | | | | Ground Water µg/L | Soil Ingestion | | | | | | | | |
| | | | | | | | Industrial2 mg/kg | Residential mg/kg | | | | | | | | |
| Methylene bromide | 74953 | 1.00E-02 | A | | | x | 61 | N | 20000 | N | 780 | N | | | | |
| Methylene chloride | 75092 | 6.00E-02 | I | 8.57E-01 | H | 7.50E-03 | I | 1.64E-03 | I | x | 5 | S | 7600 | C | 85 | C |
| 4,4'-Methylene bis(2-chloroaniline) | 101144 | 7.00E-04 | H | | | 1.30E-01 | H | 1.30E-01 | H | | 0.52 | C | 440 | C | 4.9 | C |
| 4,4'-Methylenebisbenzeneamine | 101779 | | | | | 2.50E-01 | W1 | | | | 0.27 | C | 230 | C | 2.6 | C |
| 4,4'-Methylene bis(N,N'-dimethyl)aniline | 101611 | | | | | 4.60E-02 | I | | | | 1.5 | C | 1200 | C | 14 | C |
| 4,4'-Methylenediphenyl isocyanate | 101688 | | | 5.71E-06 | I | | | | | x | 0.035 | N | | | | |
| Methyl ethyl ketone | 78933 | 6.00E-01 | I | 2.86E-01 | I | | | | | x | 1900 | N | 1000000 | N | 47000 | N |
| Methyl hydrazine | 60344 | | | | | 1.10E+00 | W1 | | | | 0.061 | C | 52 | C | 0.58 | C |
| Methyl isobutyl ketone | 108101 | 8.00E-02 | H | 2.29E-02 | A | | | | | | 2900 | N | 160000 | N | 6300 | N |
| Methyl methacrylate | 80626 | 8.00E-02 | H | | | | | | | | 2900 | N | 160000 | N | 6300 | N |
| 2-Methyl-5-nitroaniline | 99558 | | | | | 3.30E-02 | H | | | | 2 | C | 1700 | C | 19 | C |
| Methyl parathion | 298000 | 2.50E-04 | I | | | | | | | | 9.1 | N | 510 | N | 20 | N |
| 2-Methylphenol (o-cresol) | 95487 | 5.00E-02 | I | | | | | | | | 1800 | N | 100000 | N | 3900 | N |
| 3-Methylphenol (m-cresol) | 103394 | 5.00E-02 | I | | | | | | | | 1800 | N | 100000 | N | 3900 | N |
| 4-Methylphenol (p-cresol) | 106445 | 5.00E-03 | H | | | | | | | | 180 | N | 10000 | N | 390 | N |
| Methyl styrene (mixture) | 25013154 | 6.00E-03 | A | 1.14E-02 | A | | | | | x | 60 | N | 12000 | N | 470 | N |
| Methyl styrene (alpha) | 98839 | 7.00E-02 | A | | | | | | | x | 430 | N | 140000 | N | 5500 | N |
| Methyl tertbutyl ether (MTBE) | 1634044 | 5.00E-03 | E1 | 8.57E-01 | I | | | | | x | 180 | N | 10000 | N | 390 | N |
| Metolacrol (Dual) | 51218452 | 1.50E-01 | H | | | | | | | | 5500 | N | 310000 | N | 12000 | N |
| Metribuzin | 21087649 | 2.50E-02 | I | | | | | | | | 910 | N | 51000 | N | 2000 | N |
| Mirex | 2385855 | 2.00E-04 | I | | | 1.80E+00 | W1 | | | | 0.037 | C | 32 | C | 0.35 | C |
| Molinate | 2212671 | 2.00E-03 | I | | | | | | | | 73 | N | 4100 | N | 160 | N |
| Molybdenum | 7439987 | 5.00E-03 | I | | | | | | | | 180 | N | 10000 | N | 390 | N |
| Monochloramine | 10599903 | 1.00E-01 | I | | | | | | | | 3700 | N | 200000 | N | 7800 | N |
| Naled | 300765 | 2.00E-03 | I | | | | | | | | 73 | N | 4100 | N | 160 | N |
| 2-Naphthylamine | 91598 | | | | | 1.30E+02 | E1 | | | | 0.00052 | C | 0.44 | C | 0.0049 | C |
| Napropamide | 15299997 | 1.00E-01 | I | | | | | | | | 3700 | N | 200000 | N | 7800 | N |
| Nickel and compounds | 7440020 | 2.00E-02 | I | | | | | | | | 100 | S | 41000 | N | 1600 | N |
| Nitrapyrin | 1929824 | 1.50E-03 | W1 | | | | | | | | 55 | N | 3100 | N | 120 | N |
| Nitrate | 14797558 | 1.60E+00 | I | | | | | | | | 10000 | S | 1000000 | N | 130000 | N |
| Nitric oxide | 10102439 | 1.00E-01 | W1 | | | | | | | | 3700 | N | 200000 | N | 7800 | N |
| Nitrite | 14797650 | 1.00E-01 | I | | | | | | | | 1000 | S | 200000 | N | 7800 | N |
| 2-Nitroaniline | 88744 | 6.00E-05 | W1 | 5.71E-05 | H | | | | | | 2.2 | N | 120 | N | 4.7 | N |
| 3-Nitroaniline | 99092 | 3.00E-03 | O1 | | | | | | | | 110 | N | 6100 | N | 230 | N |
| 4-Nitroaniline | 100016 | 3.00E-03 | O1 | | | | | | | | 110 | N | 6100 | N | 230 | N |
| Nitrobenzene | 98953 | 5.00E-04 | I | 5.71E-04 | A | | | | | x | 3.4 | N | 1000 | N | 39 | N |
| Nitrofurantoin | 67209 | 7.00E-02 | H | | | | | | | | 2600 | N | 140000 | N | 5500 | N |
| Nitrofurazone | 59870 | | | | | 1.50E+00 | H | 9.40E+00 | H | | 0.045 | C | 38 | C | 0.43 | C |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations Soil Ingestion | | | | | | | |
|--|----------|-----------------|-----------------|-----------------|-----------------|-------------|---|----------------------|----------------------|---|-------|---|--------|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Residential mg/kg | | | | | |
| Nitrogen dioxide | 10102440 | 1.00E+00 | W ¹ | | | | 37000 | N | 1000000 | N | 78000 | N | | |
| Nitroguanidine | 556887 | 1.00E-01 | I | | | | 3700 | N | 200000 | N | 7800 | N | | |
| 4-Nitrophenol | 100027 | 6.20E-02 | O ¹ | | | | 2300 | N | 130000 | N | 4800 | N | | |
| 2-Nitropropane | 79469 | | | 5.71E-03 | I | | 9.40E+00 | H | 210 | N | | | | |
| N-Nitrosodi-n-butylamine | 924163 | | | 5.40E+00 | I | | 5.60E+00 | I | 0.012 | C | 11 | C | 0.12 | C |
| N-Nitrosodiethanolamine | 1116547 | | | 2.80E+00 | I | | 0.024 | C | 20 | C | 0.23 | C | | |
| N-Nitrosodiethylamine | 55185 | | | 1.50E+02 | I | | 1.51E+02 | I | 0.00045 | C | 0.38 | C | 0.0043 | C |
| N-Nitrosodimethylamine | 62759 | | | 5.10E+01 | I | | 4.90E+01 | I | 0.0013 | C | 1.1 | C | 0.013 | C |
| N-Nitrosodiphenylamine | 86306 | | | 4.90E-03 | I | | 14 | C | 12000 | C | 130 | C | | |
| N-Nitroso di-n-propylamine | 621647 | | | 7.00E+00 | I | | 0.0096 | C | 8.2 | C | 0.091 | C | | |
| N-Nitroso-N-methylethylamine | 10595956 | | | 2.20E+01 | I | | 0.0031 | C | 2.6 | C | 0.029 | C | | |
| N-Nitrosopyrrolidine | 930552 | | | 2.10E+00 | I | | 2.13E+00 | I | 0.032 | C | 27 | C | 0.3 | C |
| m-Nitrotoluene | 99081 | 1.00E-02 | H | | | | 61 | N | 20000 | N | 780 | N | | |
| o-Nitrotoluene | 88722 | 1.00E-02 | H | | | | 61 | N | 20000 | N | 780 | N | | |
| p-Nitrotoluene | 99990 | 1.00E-02 | H | | | | 61 | N | 20000 | N | 780 | N | | |
| Norflurazon | 27314132 | 4.00E-02 | I | | | | 1500 | N | 82000 | N | 3100 | N | | |
| NuStar | 85509199 | 7.00E-04 | I | | | | 26 | N | 1400 | N | 55 | N | | |
| Octabromodiphenyl ether | 32536520 | 3.00E-03 | I | | | | 110 | N | 6100 | N | 230 | N | | |
| Octahydro-1357-tetranitro-1357-tetrazocine | 2691410 | 5.00E-02 | I | | | | 1800 | N | 100000 | N | 3900 | N | | |
| Octamethylpyrophosphoramidate | 152169 | 2.00E-03 | H | | | | 73 | N | 4100 | N | 160 | N | | |
| Oryzalin | 19044883 | 5.00E-02 | I | | | | 1800 | N | 100000 | N | 3900 | N | | |
| Oxadiazon | 19666309 | 5.00E-03 | I | | | | 180 | N | 10000 | N | 390 | N | | |
| Oxamyl | 23135220 | 2.50E-02 | I | | | | 200 | S | 51000 | N | 2000 | N | | |
| Oxyfluorfen | 42874033 | 3.00E-03 | I | | | | 110 | N | 6100 | N | 230 | N | | |
| Pacllobutrazol | 76738620 | 1.30E-02 | I | | | | 470 | N | 27000 | N | 1000 | N | | |
| Paraquat | 1910425 | 4.50E-03 | I | | | | 160 | N | 9200 | N | 350 | N | | |
| Parathion | 56382 | 6.00E-03 | H | | | | 220 | N | 12000 | N | 470 | N | | |
| Pebulate | 1114712 | 5.00E-02 | H | | | | 1800 | N | 100000 | N | 3900 | N | | |
| Pendimethalin | 40487421 | 4.00E-02 | I | | | | 1500 | N | 82000 | N | 3100 | N | | |
| Pentabromo-6-chloro cyclohexane | 87843 | | | 2.30E-02 | H | | 2.9 | C | 2500 | C | 28 | C | | |
| Pentabromodiphenyl ether | 32534819 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N | | |
| Pentachlorobenzene | 608935 | 8.00E-04 | I | | | | 4.9 | N | 1600 | N | 63 | N | | |
| Pentachloronitrobenzene | 82688 | 3.00E-03 | I | | 2.60E-01 | H | 0.041 | C | 220 | C | 2.5 | C | | |
| Pentachlorophenol | 87865 | 3.00E-02 | I | | 1.20E-01 | I | 1 | S | 480 | C | 5.3 | C | | |
| Permethrin | 52645531 | 5.00E-02 | I | | | | 1800 | N | 100000 | N | 3900 | N | | |
| Phenmedipham | 13684634 | 2.50E-01 | I | | | | 9100 | N | 510000 | N | 20000 | N | | |
| Phenol | 108952 | 6.00E-01 | I | | | | 22000 | N | 1000000 | N | 47000 | N | | |
| m-Phenylenediamine | 108452 | 6.00E-03 | I | | | | 220 | N | 12000 | N | 470 | N | | |
| p-Phenylenediamine | 106503 | 1.90E-01 | H | | | | 6900 | N | 390000 | N | 15000 | N | | |
| Phenylmercuric acetate | 62384 | 8.00E-05 | I | | | | 2.9 | N | 160 | N | 6.3 | N | | |
| 2-Phenylphenol | 90437 | | | 1.94E-03 | H | | 35 | C | 30000 | C | 330 | C | | |
| Phorate | 298022 | 2.00E-04 | H | | | | 7.3 | N | 410 | N | 16 | N | | |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | | | | | |
|-----------------------------------|----------|-----------------|-----------------|-----------------|-----------------|-------------|---------------------------|----------------------|----------------|------|----------------------|-------|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Soil Ingestion | | Residential mg/kg | | |
| Phosmet | 732116 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N | |
| Phosphine | 7803512 | 3.00E-04 | I | 8.57E-05 | I | | 11 | N | 610 | N | 23 | N | |
| Phosphoric acid | 7664382 | | I | 2.86E-03 | I | | 100 | N | | | | | |
| Phosphorus (white) | 7723140 | 2.00E-05 | I | | | | 0.73 | N | 41 | N | 1.6 | N | |
| p-Phthalic acid | 100210 | 1.00E+00 | H | | | | 37000 | N | 1000000 | N | 78000 | N | |
| Phthalic anhydride | 85449 | 2.00E+00 | I | 3.43E-02 | H | | 73000 | N | 1000000 | N | 160000 | N | |
| Picloram | 1918021 | 7.00E-02 | I | | | | 500 | S | 140000 | N | 5500 | N | |
| Pirimiphos-methyl | 29232937 | 1.00E-02 | I | | | | 370 | N | 20000 | N | 780 | N | |
| Polybrominated biphenyls | 0 | 7.00E-06 | H | 8.90E+00 | H | | 0.0076 | C | 6.4 | C | 0.072 | C | |
| Polychlorinated biphenyls (PCBs) | 1336363 | | | 7.70E+00 | I | | 0.5 | S | 7.4 | C | 0.083 | C | |
| Aroclor 1016 | 12674112 | 7.00E-05 | I | | | | 2.6 | N | 140 | N | 5.5 | N | |
| Aroclor 1254 | 11097691 | 2.00E-05 | I | | | | 0.73 | N | 41 | N | 1.6 | N | |
| Polychlorinated terphenyls (PCTs) | 0 | | | 4.50E+00 | E ¹ | | 0.015 | C | 13 | C | 0.14 | C | |
| Polynuclear aromatic hydrocarbons | 0 | | | | | | | | | | | | |
| Acenaphthene | 83329 | 6.00E-02 | I | | | | 2200 | N | 120000 | N | 4700 | N | |
| Anthracene | 120127 | 3.00E-01 | I | | | | 11000 | N | 610000 | N | 23000 | N | |
| Benz[a]anthracene | 56553 | | | 7.30E-01 | E ¹ | 6.10E-01 | E ¹ | 0.092 | C | 78 | C | 0.88 | C |
| Benzo[b]fluoranthene | 205992 | | | 7.30E-01 | E ¹ | 6.10E-01 | E ¹ | 0.092 | C | 78 | C | 0.88 | C |
| Benzo[k]fluoranthene | 207089 | | | 7.30E-02 | E ¹ | 6.10E-02 | E ¹ | 0.92 | C | 780 | C | 8.8 | C |
| Benzo[a]pyrene | 50328 | | | 7.30E+00 | I | 6.10E+00 | W ¹ | 0.200 | S | 7.8 | C | 0.088 | C |
| Carbazole | 86748 | | | 2.00E-02 | H | | 3.4 | C | 2900 | C | 32 | C | |
| Chrysene | 218019 | | | 7.30E-03 | E ¹ | 6.10E-03 | E ¹ | 9.2 | C | 7800 | C | 88 | C |
| Dibenz[ah]anthracene | 53703 | | | 7.30E+00 | E ¹ | 6.10E+00 | E ¹ | 0.0092 | C | 7.8 | C | 0.088 | C |
| Fluoranthene | 206440 | 4.00E-02 | I | | | | 1500 | N | 82000 | N | 3100 | N | |
| Fluorene | 86737 | 4.00E-02 | I | | | | 1500 | N | 82000 | N | 3100 | N | |
| Indeno[1,2,3-cd]pyrene | 193395 | | | 7.30E-01 | E ¹ | 6.10E-01 | E ¹ | 0.092 | C | 78 | C | 0.88 | C |
| Naphthalene | 91203 | 4.00E-02 | W ¹ | | | | 1500 | N | 82000 | N | 3100 | N | |
| Pyrene | 129000 | 3.00E-02 | I | | | | 1100 | N | 61000 | N | 2300 | N | |
| Prochloraz | 67747095 | 9.00E-03 | I | 1.50E-01 | I | | 0.45 | C | 380 | C | 4.3 | C | |
| Profluralin | 26399360 | 6.00E-03 | H | | | | 220 | N | 12000 | N | 470 | N | |
| Prometon | 1610180 | 1.50E-02 | I | | | | 550 | N | 31000 | N | 1200 | N | |
| Prometryn | 7287196 | 4.00E-03 | I | | | | 150 | N | 8200 | N | 310 | N | |
| Pronamide | 23950585 | 7.50E-02 | I | | | | 2700 | N | 150000 | N | 5900 | N | |
| Propachlor | 1918167 | 1.30E-02 | I | | | | 470 | N | 27000 | N | 1000 | N | |
| Propanil | 709988 | 5.00E-03 | I | | | | 180 | N | 10000 | N | 390 | N | |
| Propargite | 2312358 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N | |
| Propargyl alcohol | 107197 | 2.00E-03 | I | | | | 73 | N | 4100 | N | 160 | N | |
| Propazine | 139402 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N | |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | | | | | | | |
|------------------------------------|----------|-----------------|-----------------|-----------------|-----------------|----------------|---------------------------|----------------------|----------------|---------|----------------------|-----------|-----|-------|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Soil Ingestion | | Residential mg/kg | | | | |
| Propam | 122429 | 2.00E-02 | I | | | | 730 | N | 41000 | N | 1600 | N | | | |
| Propiconazole | 60207901 | 1.30E-02 | I | | | | 470 | N | 27000 | N | 1000 | N | | | |
| Propylene glycol | 57556 | 2.00E+01 | H | | | | 730000 | N | 1000000 | N | 1000000 | N | | | |
| Propylene glycol, monoethyl ether | 52125538 | 7.00E-01 | H | | | | 26000 | N | 1000000 | N | 55000 | N | | | |
| Propylene glycol, monomethyl ether | 107982 | 7.00E-01 | H | 5.71E-01 | I | | 26000 | N | 1000000 | N | 55000 | N | | | |
| Propylene oxide | 75569 | | | 8.57E-03 | I | 2.40E-01 | 1.29E-02 | I | 0.28 | C | 240 | C | 2.7 | C | |
| Pursuit | 81335775 | 2.50E-01 | I | | | | 9100 | N | 510000 | N | 20000 | N | | | |
| Pydrin | 51630581 | 2.50E-02 | I | | | | 910 | N | 51000 | N | 2000 | N | | | |
| Pyridine | 110861 | 1.00E-03 | I | | | | 37 | N | 2000 | N | 78 | N | | | |
| Quinalphos | 13593038 | 5.00E-04 | I | | | | 18 | N | 1000 | N | 39 | N | | | |
| Quinoline | 91225 | | | 1.20E+01 | H | | 0.0056 | C | 5 | C | 0.053 | C | | | |
| Resmethrin | 10463868 | 3.00E-02 | I | | | | 1100 | N | 61000 | N | 2300 | N | | | |
| Ronnel | 299843 | 5.00E-02 | H | | | | 1800 | N | 100000 | N | 3900 | N | | | |
| Rotenone | 83794 | 4.00E-03 | I | | | | 150 | N | 8200 | N | 310 | N | | | |
| Savey | 78587050 | 2.50E-02 | I | | | | 910 | N | 51000 | N | 2000 | N | | | |
| Selenious Acid | 7783008 | 5.00E-03 | I | | | | 180 | N | 10000 | N | 390 | N | | | |
| Selenium | 7782492 | 5.00E-03 | I | | | | 50 | S | 10000 | N | 390 | N | | | |
| Selenourea | 630104 | 5.00E-03 | H | | | | 180 | N | 10000 | N | 390 | N | | | |
| Sethoxydim | 74051802 | 9.00E-02 | I | | | | 3300 | N | 180000 | N | 7000 | N | | | |
| Silver and compounds | 7440224 | 5.00E-03 | I | | | | 180 | N | 10000 | N | 390 | N | | | |
| Simazine | 122349 | 5.00E-03 | I | | 1.20E-01 | H | 4 | S | 480 | C | 5.3 | C | | | |
| Sodium azide | 26628228 | 4.00E-03 | I | | | | 150 | N | 8200 | N | 310 | N | | | |
| Sodium diethyldithiocarbamate | 148185 | 3.00E-02 | I | | 2.70E-01 | H | 0.25 | C | 210 | C | 2.4 | C | | | |
| Sodium fluoroacetate | 62748 | 2.00E-05 | I | | | | 0.73 | N | 41 | N | 1.6 | N | | | |
| Sodium metavanadate | 13718268 | 1.00E-03 | H | | | | 37 | N | 2000 | N | 78 | N | | | |
| Strontium, stable | 7440246 | 6.00E-01 | I | | | | 22000 | N | 1000000 | N | 47000 | N | | | |
| Strychnine | 57249 | 3.00E-04 | I | | | | 11 | N | 610 | N | 23 | N | | | |
| Styrene | 100425 | 2.00E-01 | I | 2.86E-01 | I | | x | 100 | S | 410000 | N | 16000 | N | | |
| Systhane | 88671890 | 2.50E-02 | I | | | | 910 | N | 51000 | N | 2000 | N | | | |
| 2,3,7,8-TCDD (dioxin) | 1746016 | | | 1.56E+05 | H | 1.16E+05 | H | 0.000005 | S | 0.00037 | C | 0.0000041 | C | | |
| Tebuthiuron | 34014181 | 7.00E-02 | I | | | | 2600 | N | 140000 | N | 5500 | N | | | |
| Temephos | 3383968 | 2.00E-02 | H | | | | 730 | N | 41000 | N | 1600 | N | | | |
| Terbacil | 5902512 | 1.30E-02 | I | | | | 470 | N | 27000 | N | 1000 | N | | | |
| Terbufos | 13071799 | 2.50E-05 | H | | | | 0.91 | N | 51 | N | 2 | N | | | |
| Terbutryn | 886500 | 1.00E-03 | I | | | | 37.00 | N | 2000 | N | 78 | N | | | |
| 1,2,4,5-Tetrachlorobenzene | 95943 | 3.00E-04 | I | | | | x | 1.8 | N | 610 | N | 23 | N | | |
| 1,1,1,2-Tetrachloroethane | 630206 | 3.00E-02 | I | | 2.60E-02 | I | 2.59E-02 | I | x | 0.41 | C | 2200 | C | 25 | C |
| 1,1,2,2-Tetrachloroethane | 79345 | | | | 2.00E-01 | I | 2.03E-01 | I | x | 0.052 | C | 290 | C | 3.2 | C |
| Tetrachloroethylene (PCE) | 127184 | 1.00E-02 | I | | 5.20E-02 | E ¹ | 2.03E-03 | E ¹ | x | 5 | S | 1100 | C | 12 | C |
| 2,3,4,6-Tetrachlorophenol | 58902 | 3.00E-02 | I | | | | | | | 1100 | N | 61000 | N | 2300 | N |
| p,a,a-Tetrachlorotoluene | 5216251 | | | | 2.00E+01 | H | | | x | 0.00053 | C | 2.9 | C | 0.032 | C |
| Tetrachlorovinphos | 961115 | 3.00E-02 | I | | 2.40E-02 | H | | | | 2.8 | C | 2400 | C | 27 | C |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Risk-Based Concentrations | | | | | | |
|--|----------|-----------------|-----------------|-----------------|-----------------|----------------|---------------------------|----------------------|--|------|--------|-------|---|
| | | | | | | | Ground Water µg/L | Industrial2 mg/kg | Soil Ingestion Residential mg/kg | | | | |
| Tetraethylthiopyrophosphate | 3689245 | 5.00E-04 | I | | | | 18 | N | 1000 | N | 39 | N | |
| Tetraethyl lead | 78002 | 1.00E-07 | I | | | | 0.0037 | N | 0.2 | N | 0.0078 | N | |
| 1,1,1,2-Tetrafluoroethane | 811972 | | I | 2.29E+01 | | x | 140000 | N | | | | | |
| Thallic oxide | 1314325 | 7.00E-05 | W ¹ | | | | 2.6 | N | 140 | N | 5.5 | N | |
| Thallium | 0 | | | | | | 2 | S | | | | | |
| Thallium acetate | 563688 | 9.00E-05 | I | | | | 3.3 | N | 180 | N | 7 | N | |
| Thallium carbonate | 6533739 | 8.00E-05 | I | | | | 2.9 | N | 160 | N | 6.3 | N | |
| Thallium chloride | 7791120 | 8.00E-05 | I | | | | 2.9 | N | 160 | N | 6.3 | N | |
| Thallium nitrate | 10102451 | 9.00E-05 | I | | | | 3.3 | N | 180 | N | 7 | N | |
| Thallium selenite | 12039520 | 9.00E-05 | W ¹ | | | | 3.3 | N | 180 | N | 7 | N | |
| Thallium sulfate | 7446186 | 8.00E-05 | I | | | | 2.9 | N | 160 | N | 6.3 | N | |
| Thiobencarb | 28249776 | 1.00E-02 | I | | | | 370 | N | 20000 | N | 780 | N | |
| 2-(Thiocyanomethylthio)-benzothiazole | 21564170 | 3.00E-02 | H | | | | 1100 | N | 61000 | N | 2300 | N | |
| Thiofanox | 39196184 | 3.00E-04 | H | | | | 11 | N | 610 | N | 23 | N | |
| Thiophanate-methyl | 23564058 | 8.00E-02 | I | | | | 2900 | N | 160000 | N | 6300 | N | |
| Thiram | 137268 | 5.00E-03 | I | | | | 180 | N | 10000 | N | 390 | N | |
| Tin and compounds | 0 | 6.00E-01 | H | | | | 22000 | N | 1000000 | N | 47000 | N | |
| Toluene | 108883 | 2.00E-01 | I | 1.14E-01 | I | x | 1000 | S | 410000 | N | 16000 | N | |
| Toluene-2,4-diamine | 95807 | | | | 3.20E+00 | H | 0.021 | C | 18 | C | 0.2 | C | |
| Toluene-2,5-diamine | 95705 | 6.00E-01 | H | | | | 22000 | N | 1000000 | N | 47000 | N | |
| Toluene-2,6-diamine | 823405 | 2.00E-01 | H | | | | 7300 | N | 410000 | N | 16000 | N | |
| p-Toluidine | 106490 | | | 1.90E-01 | H | | 0.35 | C | 300 | C | 3.4 | C | |
| Toxaphene | 8001352 | | | 1.10E+00 | I | 1.12E+00 | I | 3 | S | 52 | C | 0.58 | C |
| Tralomehrin | 66841256 | 7.50E-03 | I | | | | 270 | N | 15000 | N | 590 | N | |
| Triallate | 2303175 | 1.30E-02 | I | | | | 470 | N | 27000 | N | 1000 | N | |
| Triasulfuron | 82097505 | 1.00E-02 | I | | | | 370 | N | 20000 | N | 780 | N | |
| 1,2,4-Tribromobenzene | 615543 | 5.00E-03 | I | | | x | 30 | N | 10000 | N | 390 | N | |
| Tributyltin oxide (TBTO) | 56359 | 3.00E-05 | I | | | | 1.1 | N | 61 | N | 2 | N | |
| 2,4,6-Trichloroaniline hydrochloride | 33663502 | | | 2.90E-02 | H | | 2.3 | C | 2000 | C | 22 | C | |
| 2,4,6-Trichloroaniline | 634935 | | | 3.40E-02 | H | | 2 | C | 1700 | C | 19 | C | |
| 1,2,4-Trichlorobenzene | 120821 | 1.00E-02 | I | 5.71e-02 | H | x | 70 | S | 20000 | N | 780 | N | |
| **1,1,1-Trichloroethane | 71556 | 3.50E-02 | E ¹ | 2.86E-01 | W ¹ | x | 200 | S | 72000 | N | 2700 | N | |
| 1,1,2-Trichloroethane | 79005 | 4.00E-03 | I | | 5.70E-02 | I | 5.60E-02 | I | x | 5 | S | 1000 | C |
| Trichloroethylene (TCE) | 79016 | 6.00E-03 | E ¹ | | 1.10E-02 | W ¹ | 6.00E-03 | E ¹ | x | 5 | S | 5200 | C |
| Trichlorofluoromethane | 75694 | 3.00E-01 | I | 2.00E-01 | A | x | 1300 | N | 610000 | N | 23000 | N | |
| 2,4,5-Trichlorophenol | 95954 | 1.00E-01 | I | | | | 3700 | N | 200000 | N | 7800 | N | |
| 2,4,6-Trichlorophenol | 88062 | | | 1.10E-02 | I | 1.09E-02 | I | 6.1 | C | 5200 | C | 58 | C |
| 2,4,5-Trichlorophenoxyacetic acid | 93765 | 1.00E-02 | I | | | | 370 | N | 20000 | N | 780 | N | |
| 2-(2,4,5-Trichlorophenoxy)propionic acid | 93721 | 8.00E-03 | I | | | | 50 | S | 16000 | N | 630 | N | |
| 1,1,2-Trichloropropane | 598776 | 5.00E-03 | I | | | x | 30 | N | 10000 | N | 390 | N | |
| 1,2,3-Trichloropropane | 96184 | 6.00E-03 | I | | 7.00e+00 | I | x | 0.0015 | C | 8.2 | C | 0.091 | C |

Sources: I=IRIS H=HEAST A=HEAST alternate W=Withdrawn from IRIS or HEAST E=EPA-NECA Regional Support provisional value O=Other EPA documents
 Basis: C=Carcinogenic effects N=Noncarcinogenic effects S=West Virginia Groundwater Quality Standards

| Contaminant | CAS | RfDo mg/kg/d | RfDi mg/kg/d | CPSo kg d/mg | CPSi kg d/mg | V O C | Ground Water µg/L | Risk-Based Concentrations Soil Ingestion | | | | | | |
|--|----------|-----------------|-----------------|-----------------|-----------------|-------------|-------------------------|---|----------------------|---------|----|---------|------|---|
| | | | | | | | | Industrial2 mg/kg | Residential mg/kg | | | | | |
| 1,2,3-Trichloropropene | 96195 | 5.00E-03 | H | | | | x | 30 | N | 10000 | N | 390 | N | |
| 1,1,2-Trichloro-1,2,2- trifluoroethane | 76131 | 3.00E+01 | I | 8.57E+00 | H | | x | 59000 | N | 1000000 | N | 1000000 | N | |
| Tridiphane | 58138082 | 3.00E-03 | I | | | | | 110 | N | 6100 | N | 230 | N | |
| Triethylamine | 121448 | | | 2.00E-03 | I | | | 73 | N | | | | | |
| Trifluralin | 1582098 | 7.50E-03 | I | | 7.70E-03 | I | | 8.7 | C | 7400 | C | 83 | C | |
| 1,2,4-Trimethylbenzene | 95636 | 5.00e-02 | E ¹ | | | | x | 300 | N | 100000 | N | 3900 | N | |
| 1,3,5-Trimethylbenzene | 108678 | 5.00e-02 | E ¹ | | | | x | 300 | N | 100000 | N | 3900 | N | |
| Trimethyl phosphate | 512561 | | | 3.70E-02 | H | | | 1.8 | C | 1500 | C | 17 | C | |
| 1,3,5-Trinitrobenzene | 99354 | 5.00E-05 | I | | | | | 1.8 | N | 100 | N | 3.9 | N | |
| Trinitrophenylmethylnitramine | 479458 | 1.00E-02 | H | | | | | 370 | N | 20000 | N | 780 | N | |
| 2,4,6-Trinitrotoluene | 118967 | 5.00E-04 | I | | 3.00E-02 | I | | 2.2 | C | 1900 | C | 21 | C | |
| Uranium (soluble salts) | 7440611 | 3.00E-03 | I | | | | | 110 | N | 6100 | N | 230 | N | |
| Vanadium | 7440622 | 7.00E-03 | H | | | | | 260 | N | 14000 | N | 550 | N | |
| Vanadium pentoxide | 1314621 | 9.00E-03 | I | | | | | 330 | N | 18000 | N | 700 | N | |
| Vanadium sulfate | 36907423 | 2.00E-02 | H | | | | | 730 | N | 41000 | N | 1600 | N | |
| Vernam | 1929777 | 1.00E-03 | I | | | | | 37 | N | 2000 | N | 78 | N | |
| Vinclozolin | 50471448 | 2.50E-02 | I | | | | | 910 | N | 51000 | N | 2000 | N | |
| Vinyl acetate | 108054 | 1.00E+00 | H | 5.71E-02 | I | | | 37000 | N | 1000000 | N | 78000 | N | |
| Vinyl bromide | 593602 | | | 8.57E-04 | I | | x | 5.2 | N | | | | | |
| Vinyl chloride | 75014 | | | 1.90E+00 | H | 3.00E-01 | H | x | 2 | S | 30 | C | 0.34 | C |
| Warfarin | 81812 | 3.00E-04 | I | | | | | 11 | N | 610 | N | 23 | N | |
| m-Xylene | 1.08E+05 | 2.00E+00 | H | 2.00E-01 | W ¹ | | x | 1400 | N | 1000000 | N | 160000 | N | |
| o-Xylene | 9.55E+04 | 2.00E+00 | H | 2.00E-01 | W ¹ | | x | 1400 | N | 1000000 | N | 160000 | N | |
| p-Xylene | 1.06E+05 | | | 8.57E-02 | W ¹ | | x | 520 | N | | | | | |
| Xylene (mixed) | 1.33E+06 | 2.00E+00 | I | | | | x | 10000 | S | 1000000 | N | 160000 | N | |
| Zinc | 7.44E+06 | 3.00E-01 | I | | | | | 11000 | N | 610000 | N | 23000 | N | |
| Zinc phosphide | 1.31E+06 | 3.00E-04 | I | | | | | 11 | N | 610 | N | 23 | N | |
| Zineb | 1.21E+07 | 5.00E-02 | I | | | | | 1800 | N | 100000 | N | 3900 | N | |

Except where noted, all concentrations were obtained from USEPA Region III Risk Based Concentration Table (July 1996). The toxicity factors presented in this table may be modified for use in the development of uniform or site-specific standards.

These concentrations shall be applied where the soil ingestion pathway is the major contributor to risks identified in the site assessment. If other exposure pathways are identified, the acceptable concentrations shall be determined only in consultation with the Director, considering all exposure pathways, and all other requirements of the regulations. However, remediation of free product shall be required.

¹ Used at the discretion of the WVDEP

² Industrial risk based concentrations for carcinogens were multiplied by 10 to yield a concentration based 1×10^5 risk.

R¹ Value used in: "Interim Guidance on Screening Levels for Hazardous Substances Discovered During Site Assessments Under the Delaware Hazardous Substance Cleanup Act." Delaware DNREC, October 1995 and Connecticut "Remediation Standard Regulations" 22a-133k-3, December 1995

R² Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities" USEPA Directive 93554-12, July 1994

APPENDIX C-2: CHECKLIST TO DETERMINE THE APPLICABLE ECOLOGICAL STANDARD

This checklist is cross referenced to 60CSR3, the Voluntary Remediation and Redevelopment Rule (the Rule). This checklist is based on Section 9.5 – Ecological – De Minimis Screening Evaluation (cited as 60-3-9.5). The specific references are to subsections of this section of the Rule.

Step 1. Determine Whether a De Minimis Ecological Screening Evaluation is Appropriate for Your Site

See 60-3.9.5.a.1

Check “yes” or “no” to each of the following questions:

1.1 Has there been a release to the environment at or from the site?

yes no unknown

If the answer to 1.1 is “no”, then no further ecological evaluation is required. File this completed form with the Final Report for the site. If the answer to 1.1 is “yes” or “unknown”, proceed to Step 1.2.

1.2 Has the entire site been developed (e.g., predominantly covered by buildings, pavement, etc.)?

yes no

If “yes”, go to 1.6. If “no”, go to 1.3.

1.3 Are there any undeveloped areas on or adjacent to the site (e.g., areas that are not under intensive landscape or agricultural control)?

yes no

If the answer to 1.3 is “no” then no further ecological evaluation of terrestrial habitat is required. Continue with Step 1.4.

1.4 Are there any potential wetlands (including vernal pools) on or adjacent to the site?

yes no

If the answer to 1.4 is “no”, then no further ecological evaluation of wetland habitats is required. Continue with Step 1.5.

1.5 Are there any surface water bodies (i.e., lotic or lentic habitat) on or adjacent to the site?

___ yes ___ no

If the answer to 1.5 is “no”, then no further ecological evaluation of lotic and lentic aquatic habitat is required. Continue with Step. 1.6

1.6 Are there any terrestrial, wetland, or aquatic habitats off-site, but situated downstream, downwind, or downgradient from the site that may be affected by site-related stressors?

___ yes ___ no

1.7 Are there any project land uses for the site that would result in undeveloped areas, wetland habitat, lotic habitat, or lentic habitat?

___ yes ___ no

If the answers to 1.3 through 1.7 are “no”, then no further ecological evaluation is required. File this completed form with the Final Report of the site. If a question was answered “yes”, then go to Step 2 because a complete exposure pathway may exist for potential ecological receptors of concern.

Step 2. Identify any Readily Apparent Harm or Exceedances of Surface Water Quality Standards.

See 60-3-2-2.44 and 60-3-9.5.a.5

2.1 Have there been any incidents where harm to wildlife attributable to contaminants originating from the site has been readily apparent?

___ yes ___ no

If the answer to 2.1 is “yes”, go to 2.2; if “no”, go to Step 2.3.

2.2 Has the cause of such harm been eliminated?

___ yes ___ no

If the answer to 2.2 is “yes”, briefly describe the action taken and continue with this checklist.

If “no”, the applicant can proceed directly to the remedy evaluation or alternately proceed with a determination of a Uniform or Site Specific Ecological Standard, as described in the guidance manual prior to implementation of the remedy.

2.3 Is the site contributing to exceedances of Surface Water Quality Standards established for the protection of aquatic life (see 46 CSR1)?

yes no

If the answer to 2.3 is “yes”, the applicant can proceed directly to the remedy evaluation or, alternately, proceed with a determination of a Uniform or Site Specific Ecological Standard, as described in the guidance manual prior to implementation of the remedy.

If “no”, go to Step 3.

Step 3. Identification of Contamination Associated with Ecological Habitats

See 60-3-9.5.a.2 and 60-3-9.5.a.3

3.1 Have the environmental media (e.g., soil, surface water, sediment, biota) associated with the ecological habitat(s) identified in 1.3 through 1.6 been sampled and analyzed with regard to potential site-related contaminants of concern?

yes no

If the answer to 3.1 is “yes”, proceed to 3.2; if “no”, proceed to Step 4.

3.2 Have any site-related contaminants been detected above natural background concentrations in environmental media collected from terrestrial habitat?

yes no not applicable (no terrestrial)

3.3 Have any site-related contaminants been detected above natural background concentrations in environmental media collected from wetland or aquatic habitats (lotic or lentic habitats)?

yes no not applicable (no wetland/aquatic habitat)

If the answer to 3.3 is “yes”, go to 3.4. If the answer is “no”, go to 3.6.

3.4 Are site related contaminants presenting an ecological risk over and above “local” condition?

yes no

If the answer to 3.4 is “yes”, go to Step 4. If the answer is “no”, go to 3.5.

3.5 Have site-related releases of contaminants been stopped?

yes no

If the answer to 3.5 is “yes”, go to 3.6. If the answer is “no”, go to Step 4.

3.6 Are site-related contaminants currently migrating to aquatic habitat (e.g., lotic, lentic, or wetland habitat)?

yes no not applicable (no aquatic habitat)

If the answers to 3.2, 3.3, and 3.6 are “no” or “not applicable”, no further ecological evaluation is required. File this completed form with the Final Report for the site. If the answers to 3.2, 3.3, or 3.6 are “yes”, proceed to Step 4 because a complete exposure pathway may exist.

Step 4. Characterize the Potential Ecological Habitat

See 60-3-9.5.a.4

4.1 Describe the general land use in the immediate vicinity of the site.

| | |
|---|--|
| <input type="checkbox"/> Urban | <input type="checkbox"/> Industrial / Commercial |
| <input type="checkbox"/> Rural / Agricultural | <input type="checkbox"/> Rural / Undeveloped |
| <input type="checkbox"/> Residential | <input type="checkbox"/> Other (Describe) _____ |

4.2 For all affected areas that fulfill the descriptions in Questions 1.3 through 1.6, answer the following and provide a site map identifying the potential ecological habitat.

4.2.1 Outline the following characteristics for potential terrestrial habitats.

Location: _____

Contiguous area: _____

General topography: _____

Predominant vegetation species: _____

Primary soil type: _____

4.2.2 Outline the following characteristics for potential wetland habitats (e.g., vernal pools, marshes, etc).

Location: _____
Contiguous area: _____
General topography: _____
Predominant vegetation species: _____
Primary soil type: _____

4.2.3 Outline the following characteristics for potential lotic habitats (e.g., flowing water habitat such as rivers and streams).

Location: _____
Typical width and depth: _____
Typical flow rate: _____
Typical gradient (m/km): _____
Type of river / creek bottom: _____
Types of aquatic vegetation present: _____
Topography of the riparian zone: _____
Predominant riparian vegetation: _____
Human utilization of the river / creek and riparian zone: _____
Local conditions: _____

4.2.4 Outline the following characteristics for potential lentic habitats (e.g., standing water habitats such as lakes and ponds).

Location: _____
Is the pond / lake natural or man-made: _____
Area of the pond / lake: _____
Typical and maximum depth: _____
Brief description of sources and drainage: _____
Predominant aquatic vegetation: _____
Topography of the littoral zone: _____
Predominant vegetation in littoral zone: _____
Human utilization of the pond / lake and shoreline: _____
Local conditions: _____

4.3 Indicate if the site contains or is adjacent to any of the following types of valued terrestrial habitats:

- ___ Area designated as a National Preserve
- ___ Federal land designated for protection of natural ecosystems
- ___ National or State wildlife refuge
- ___ Designated Federal wilderness area or administratively proposed wilderness area
- ___ Federal or State land designated for wildlife or game management
- ___ National or State park
- ___ National or State forest
- ___ State designated natural area
- ___ Climax community (e.g., old growth forest)
- ___ Area utilized for breeding by large or dense aggregations of wildlife

- Area important to the maintenance of unique biotic communities (e.g., area with a high proportion of endemic species)
- Critical habitat for federally designated threatened or endangered species
- Habitat known to be used or potentially used by Federal or State designated threatened or endangered species
- Habitat needed for feeding, breeding, nesting, cover, or wintering habitat for migratory birds

4.4 Indicate if the site contains or is adjacent to any of the following types of valued wetlands:

- Area important to the maintenance of unique biotic communities (e.g., area with a high proportion of endemic species)
- Area utilized for breeding by large or dense aggregations of wildlife
- Feeding, breeding, nesting, cover, or wintering habitat for migratory waterfowl or other aquatic birds
- Spawning or nursery areas critical to the maintenance of fish / shellfish species
- Critical habitat for Federal-designated threatened or endangered species
- Habitat known to be used or potentially used by Federal or State designated threatened or endangered species.

4.5 Indicate if the site is within or adjacent to any of the following valued aquatic habitats:

- Area important to the maintenance of unique biotic communities (e.g., area with a high proportion of endemic species)
- Critical areas identified under the Clean Lakes Program
- National river reach designated as recreational
- Federal or State designated scenic or wild river
- Federal or State fish hatchery
- Trout-stocked streams or wild trout streams with verified trout production
- Habitat needed for feeding, breeding, nesting, cover, or wintering habitat for migratory waterfowl or other aquatic birds
- Spawning or nursery areas critical to the maintenance of fish / shellfish species
- Critical habitat for Federal designated threatened or endangered species
- Habitat known to be used or potentially used by Federal or State designated threatened or endangered species

4.6 Have valued terrestrial, wetland, or aquatic habitats been identified within or adjacent to the site?

yes no

(A list of agencies that can provide information that should assist in making a determination of whether the site is located within or adjacent to the areas listed in 4.3, 4.4, and 4.5 is provided at end of Section C2)

After completing 4.6, proceed to Step 5.

Step 5. Identify any Potential Ecological Receptors of Concern

See 60-3-2.2.14 and 60-3-9.5.a.4

5.1 Threatened and Endangered Species

Were any potential habitats within or adjacent to the site identified as critical habitat for Federally designated threatened or endangered species listed in 50 CFS 17.95 or 17.96, or areas known to be used by Federal or State designated threatened or endangered species?

___ yes ___ no

If “yes”, indicate which species:

Mammals:

- ___ Gray bat (*Myotis grisescens*)
- ___ Indiana bat (*Myotis sodalis*)
- ___ Virginia big-eared bat (*Corynorhinus townsendii virginianus*)
- ___ Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*)
- ___ Eastern cougar (*Felis concolor couguar*)

Birds:

- ___ Bald eagle (*Haliaeetus leucocephalus*)

Amphibians:

- ___ Cheat Mountain salamander (*Plethodon nettingi*)

Snails:

- ___ Flat-spired three-toothed land snail (*Triodopsis platysayoides*)

Clams:

- ___ Pink mucket pearlymussel (*Lampsilis abrupta*)
- ___ Tuberculed blossom pearlymussel (*Epioblasma torulosa torulosa*)
- ___ James spinymussel (*Pleurobema collina*)
- ___ Fanshell (*Cyprogenia stegaria*)
- ___ Clubshell (*Pleurobema clava*)
- ___ Northern riffleshell (*Epioblasma torulosa rangiana*)

Flowering Plants:

- ___ Shale barren rock cress (*Arabis perstellata*)
- ___ Harperella (*Ptilimnium nodosum*)
- ___ Northeastern bulrush (*Scirpus ancistrochaetus*)
- ___ Virginia spiraea (*Spiraea virginiana*)

- ___ Running buffalo clover (*Trifolium stoloniferum*)
- ___ Small whorled pogonia (*Isotria medeoloides*)

(The above list contains those federally designated threatened and endangered species that are indigenous to West Virginia. They will be revised as necessary to reflect changes to a species federal designation (e.g., addition or removal of a species from the list of federally designated species). The West Virginia Division of Natural Resources, Wildlife Resources Section should be consulted to ensure the above list is current. Note that West Virginia has not established a list of State designated threatened or endangered species. If such a list is established, the Federal designated species list will be revised to include State designated threatened and endangered species.)

5.2 Local populations that provide important natural or economic resources, functions, and values

Were any valued terrestrial, wetland or aquatic habitats listed in 4.3, 4.4, or 4.5 identified within or adjacent to the site?

- ___ yes ___ no

(The valued terrestrial, wetland, and aquatic habitats listed in 4.3, 4.4, and 4.5 may potentially contain local populations that provide important natural or economic resources, functions, and values)

If 5.1 and 5.2 are answered “no” and surface water bodies are shown to be in compliance with Appendix J, the ecological evaluation is complete and the site has passed the De Minimis Ecological Screening Evaluation. File this completed form with the Final Report for the site.

If either 5.1 or 5.2 are answered “yes”, the site does not pass the De Minimis ecological risk screening since a complete exposure pathway may exist for potential ecological receptors of concern. Further evaluation of the site is required using either the Uniform Ecological Standard or the Site-specific Ecological Standard. See Guidance Manual, Section 4.

AGENCIES

West Virginia Division of Natural Resources
Main Office
State Capitol Complex, Building 3
1900 Kanawha Boulevard
Charleston, West Virginia 25305
(304) 558-2754
<http://www.dnr.state.wv.us/default.htm>

West Virginia Division of Natural Resources
Wildlife Resources Section
PO Box 67
Elkins, West Virginia 26241
(304) 637-0245
<http://www.dnr.state.wv.us/wvwildlife/default.htm>

West Virginia Division of Forestry
1900 Kanawha Boulevard East
Charleston, West Virginia 25303
(304) 558-2788

US Fish and Wildlife Service
West Virginia Ecological Services Field Office
Elkins Shopping Plaza
PO Box 1278
Elkins, West Virginia 26241
(304) 636-6586
<http://northeast.fws.gov/wv.htm>

US Department of Agriculture
Natural Resource and Conservation Service
75 Night Street -- Room 301
Morgantown, WV 26505
(304) 291-4153

APPENDIX D: EQUATIONS FOR THE UNIFORM HUMAN HEALTH STANDARDS FOR SOIL AND DRINKING WATER

D.1 Introduction

As described in Section 3 of this guidance, if the De Minimis Human Health Standard is not appropriate for a site or the applicant does not choose to evaluate the site under the De Minimis Standard, then assessment can proceed under the Uniform Standard. Conducting a site evaluation under the Uniform Standard may be appropriate if:

- Concentrations of site-related chemicals exceed the De Minimis Human Health Standards.
- Chemicals present on the site are not appropriately evaluated under the De Minimis Human Health Standard (e.g., the potential exists for volatilization or leaching to groundwater).

If it is determined that the site is not appropriate for evaluation under the De Minimis Human Health Standard, then the assessment with the next level of complexity is the Uniform Human Health Risk-Based Standard⁷. This approach uses standard risk equations to arrive at target concentrations for chemicals in soil, groundwater, and surface water at the site. This appendix provides the equations and some default input parameters to calculate target soil and water concentrations under the Uniform Human Health Risk-Based Standard. If concentrations of chemicals at the site fall below the standards calculated using the equations presented in this appendix, then it can be reasonably assumed that concentrations of chemicals at the site present no unacceptable exposures to humans, and no further study is warranted. If the Uniform Risk-Based Standards are exceeded by site concentrations, then further study or remediation is warranted.

D.2 Exposure Pathways Considered in the Uniform Human Health Risk-Based Standard

The equations and guidance for this section are excerpted from the United States Environmental Protection Agency (USEPA) Region IX Preliminary Remediation Goals (USEPA 1996a) which are based on USEPA Risk Assessment Guidance for Superfund (RAGS: USEPA 1989) and the USEPA Soil Screening Guidance (USEPA 1996 b,c). These calculations consider human exposure to contaminants of potential concern (COPCs) in soils, air, and water and assess exposures that might occur under residential or industrial land use. Exposures from the several potential exposure pathways are taken into account and are summarized in Table D-1.

⁷ It is not required that an applicant provide an assessment under the Uniform Standard if they do not meet the requirements of the De Minimis Standard. It may be appropriate to conduct an assessment under the Site-Specific Standard rather than the Uniform Standard, as depicted in Figure 3-1 of the Guidance.

Table D-1: Typical Exposure Pathways by Medium for Residential and Industrial Land Uses

| Exposure Pathways Evaluated | | |
|-----------------------------|--|--|
| Medium | Residential Land Use | Industrial Land Use |
| Ground Water | Ingestion from drinking Inhalation of volatiles | |
| Surface Water | Ingestion from drinking Inhalation of volatiles | |
| Soil | Ingestion Inhalation of particulates Leaching to groundwater | Ingestion Inhalation of particulates Leaching to groundwater |

D.3 Input Parameters

Table D-2 provides a listing of the default input parameters for calculating residential or industrial remediation standards. The default parameters provided are consistent with the concept of evaluating a “Reasonable Maximum Exposure” (RME) and ensure that the calculated standards are health protective. Default input parameters were obtained primarily from *RAGS Supplemental Guidance: Standard Default Exposure Factors*, Office of Solid Waste and Emergency Response (OSWER Directive 9285.6-03) (USEPA, 1991a) and more recent information from USEPA’s *Soil Screening Guidance* (USEPA, 1996 2b,c), and the California Environmental Protection Agency’s (Cal EPA) *Preliminary Endangerment Assessment Guidance Manual* (Cal EPA 1994).

Table D-2: Standard Default Exposure Factors

| Symbol | Definition (units) | Default | Reference |
|------------------|---|-------------------|--|
| CSF _o | Cancer slope factor oral (mg/kg-d) ⁻¹ | Chemical-specific | IRIS (USEPA 1998), HEAST (USEPA 1995) |
| CSF _i | Cancer slope factor inhaled (mg/kg-d) ⁻¹ | Chemical-specific | IRIS, (USEPA 1998), HEAST (USEPA 1995) |
| RfD _o | Reference dose oral (mg/kg-d) | Chemical-specific | IRIS, (USEPA 1998), HEAST (USEPA 1995) |
| RfC _i | Reference dose inhaled (mg/kg-d) | Chemical-specific | IRIS, (USEPA 1998), HEAST (USEPA 1995) |
| TR _i | Target cancer risk, industrial | 10 ⁻⁵ | WV VRR Rule (WVDEP, 1997) |
| TR _r | Target cancer risk, residential | 10 ⁻⁶ | WV VRR Rule (WVDEP, 1997) |
| THQ | Target hazard quotient | 1 | WV VRR Rule (WVDEP, 1997) |
| BW _a | Body weight, adult (kg) | 70 | RAGS (Part A), USEPA 1989 (EPA/540/1-89/002) |
| BW _c | Body weight, child (kg) | 15 | Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03) |
| AT _c | Averaging time-carcinogens (days) | 25550 | RAGS (Part A), USEPA 1989 (EPA/540/1-89/002) |
| AT _n | Averaging time-noncarcinogens (days) | ED*365 | |
| IRA _a | Inhalation rate – adult (m ³ /day) | 20 | Exposure Factors, USEPA 1997 |
| IRA _c | Inhalation rate – child (m ³ /day) | 10 | RAGS (Part A), USEPA 1989 (EPA/540/1-89/002) |
| IRW _a | Drinking Water ingestion – adult (L/day) | 2 | RAGS (Part A) USEPA 1989 (EPA/540/1-89/002) |

| Symbol | Definition (units) | Default | Reference |
|---------------------|--|--------------------|--|
| IRW _c | Drinking Water ingestion – child (L/day) | 1 | PEA, Cal-EPA (DTSC 1994) |
| IRS _a | Soil ingestion – adult (mg/day) | 50 | Exposure Factors, USEPA 1997 Exposure Handbook |
| IRS _c | Soil ingestion – child (mg/day) | 100 | Exposure Factors, USEPA 1997 Exposure Handbook (OWSER No. 9285.6-03), Soil Screening |
| IRS _o | Soil ingestion – occupational (mg/day) | 50 | Exposure Factors, USEPA 1997 (OWSER No. 9285.6-03) |
| EF _r | Exposure frequency – residential (d/y) | 350 | Exposure Factors, USEPA 1997 (OWSER No. 9285.6-03) |
| EF _o | Exposure frequency – occupational (d/y) | 250 | Exposure Factors, USEPA 1997 (OWSER No. 9285.6-03) |
| ED _r | Exposure duration – residential (years) | 30 ^a | Exposure Factors, USEPA 1997 (OWSER No. 9285.6-03) |
| ED _c | Exposure duration – child (years) | 6 | Exposure Factors, USEPA 1997 (OWSER No. 9285.6-03) |
| ED _o | Exposure duration – occupational (years) | 25 | Exposure Factors, USEPA 1997 (OWSER No. 9285.6-03) |
| | Age-adjusted factors for carcinogens: | | |
| IFS _{adj} | Ingestion factor, soils ([mg·yr]/[kg·d]) | 114 | RAGS (Part B), USEPA 1997 (OSWER No. 9285.7-01B) |
| InhF _{adj} | Inhalation factor, soils ([m ³ ·yr]/[kg·d]) | 11 | By analogy to RAGS (Part B) |
| IFW _{adj} | Ingestion factor, water ([l·yr]/[kg·d]) | 1.1 | By analogy to RAGS (Part B) |
| VF _w | Volatilization factor for water (L/m ³) | 0.5 | RAGS (Part B), USEPA 1991 (OSWER No. 9285.7-01B) |
| PEF | Particulate emission factor (m ³ /kg) | See Section D, 4.3 | Soil Screening Guidance (USEPA 1996a,b) |
| VF _s | Volatilization factor for soil (m ³ /kg) | See below D, 4.3 | Soil Screening Guidance (USEPA 1996a,b) |
| Sat | Soil saturation concentration (mg/kg) | See below D, 4.4 | Soil Screening Guidance (USEPA 1996a,b) |

Footnote:

^a Exposure duration for lifetime residents is assumed to be 30 years total. For carcinogens, exposures are combined for children (6 years) and adults (24 years).

^b IRIS = Integrated Risk Information System (USEPA 1998)

^c HEAST = Health Effects Assessment Summary Tables (USEPA 1995)

WV VRR Rule = West Virginia Voluntary Remediation and Redevelopment Rule (WVDEP 1997)

Because contact rates may be different for children and adults, carcinogenic risks during the first 30 years of life were calculated using age-adjusted factors (“adj”). Use of age-adjusted factors are especially important for soil ingestion exposures, which are higher during childhood and decrease with age. However, for purposes of combining exposures across pathways, additional age-adjusted factors are used for inhalation. These factors approximate the integrated exposure from birth until age 30 combining contact rates, body weights, and exposure durations or two age groups – small children and adults. Age-adjusted factors were obtained from RAGS Part B (USEPA 1991b) or developed by analogy as described below.

For soils only, non carcinogenic contaminants are evaluated in children separately from adults. No age-adjustment factor is used in this case. The focus on children is considered protective of the higher daily intake rates of soil by children and their lower body weight. For maintaining consistency when evaluating soils, inhalation exposures are also based on childhood contact rates.

$$(1) \quad \text{ingestion (} [\text{mg}\cdot\text{yr}]/[\text{kg}\cdot\text{d}]\text{): } IFS_{adj} = \frac{ED_c \times IRS_c}{BW_c} + \frac{(ED_R - ED_c) \times IRS_A}{BW_a}$$

$$(2) \quad \text{inhalation (} [\text{m}^3\cdot\text{yr}]/[\text{kg}\cdot\text{d}]\text{): } InhF_{adj} = \frac{ED_c \times IRA_c}{BW_c} + \frac{(ED_R - ED_c) \times IRA_A}{BW_a}$$

If site-specific information suggests that the default input parameters provided are not appropriate for a site, then the site-specific inputs can be incorporated into equations. Site-specific information may be available for a variety of parameters. Types of information that may be appropriate to incorporate into a site assessment include (but are not limited to):

- Data on soil parameters from site characterization efforts, such as total organic carbon, soil density, or soil porosity
- Human activity information from land use assessments, such as exposure frequency, inhalation rates, exposure duration, or contact rates
- Miscellaneous information from interviews with individuals living in the area or involved with land use scenarios similar to those envisioned for the site.

D.4 Uniform Risk-Based Equations

The equations used to calculate remediation standards for carcinogenic and noncarcinogenic contaminants are presented in Equations D-1 through D-8. The equations update RAGS Part B equations. This methodology backcalculates a soil, air, or water concentration level from a target risk (for carcinogens) or hazard quotient (for noncarcinogens). For completeness, the soil equations combine risks from ingestion, and inhalation simultaneously.

D.4.1 Soil Equations

For soils, equations were based on two exposure routes (ingestion and inhalation).

Equation D-1: Combined Exposures to Carcinogenic Contaminants in Residential Soil

$$C(mg / kg) = \frac{TR \times AT_c}{EF_r \left[\left(\frac{IFS_{adj} \times CSF_o}{10^6 mg / kg} \right) + \left(\frac{InhF_{adj} \times CSF_i}{VF_s \text{ or } PEF} \right) \right]}$$

Equation D-2: Combined Exposures to Noncarcinogenic Contaminants in Residential Soil.

$$C(mg / kg) = \frac{THQ \times BW_c \times AT_n}{EF_r \times ED_c \left[\left(\frac{1}{RfD_o} \times \frac{IRS_c}{10^6 mg / kg} \right) + \left(\frac{1}{RfD_i} \times \frac{IRA_c}{VF_s \text{ or } PEF} \right) \right]}$$

Equation D-3: Combined Exposures to Carcinogenic Contaminants in Industrial Soil

$$C(mg / kg) = \frac{TR \times BW_a \times AT_c}{EF_o \times ED_o \left[\left(\frac{IRS_o \times CSF_o}{10^6 mg / kg} \right) + \left(\frac{IRA_a \times CSF_i}{VF_s \text{ or } PEF} \right) \right]}$$

Equation D-4: Combined Exposures to Noncarcinogenic Contaminants in Industrial Soil

$$C(mg / kg) = \frac{THQ \times BW_a \times AT_n}{EF_o \times ED_o \left[\left(\frac{1}{RfD_o} \times \frac{IRS_o}{10^6 mg / kg} \right) + \left(\frac{1}{RfD_i} \times \frac{IRA_a}{VF_s \text{ or } PEF} \right) \right]}$$

Calculation of the volatilization factor for soil (VF_s) is presented in Subsection D.4.3.1. Calculation of the particulate emission factor (PEF) is presented in Subsection D.4.3.2. Use VF for volatile chemicals (i.e., having a Henry's Law Constant greater than 10^{-5} and a molecular weight less than 200 grams/mole [gm/mole] or a PEF for non-volatile chemicals. The equation used to calculate the remediation standard should include risk due to fugitive dust and risk due to volatilization from soil, if both apply.

D.4.2 Tap Water Equations

For tap water, an upperbound volatilization constant (VF_w) may be used that is based on all uses of household water (e.g., showering, laundering, and dish washing). Certain assumptions were made in deriving this constant. For example, it was assumed that the volume of water used in a residence for a family of four is 720 liters per day (L/day), the volume of the dwelling is 150,000 L and the air exchange rate is 0.25 air changes/hour (RAGS Part B: USEPA 1991b). Furthermore, it was assumed that the average transfer efficiency weighted by water use is 50

percent (i.e. half of the concentration of each chemical in water will be transferred into air by all water uses). Note: the range of transfer efficiencies extends from 30 percent for toilets to 90 percent for dishwashers. If site-specific information is available, it may be used to develop a site-specific VF_w .

Equation D-5: Ingestion and Inhalation Exposures to Carcinogenic Contaminants in Water

$$C(\mathbf{mg} / L) = \frac{TR \times AT_c \times 1000 \mathbf{mg} / \mathbf{mg}}{EF_r [(IFW_{adj} \times CSF_o) + (VF_w \times InhF_{adj} \times CSF_i)]}$$

Equation D-6: Ingestion and Inhalation Exposures to Noncarcinogenic Contaminants in Water

$$C(\mathbf{mg} / L) = \frac{THQ \times BW_a \times AT_n \times 1000 \mathbf{mg} / \mathbf{mg}}{EF_r \times ED_r \left[\left(\frac{IRW_a}{RfD_o} \right) + \left(\frac{VF_w \times IRA_a}{RfD_i} \right) \right]}$$

D.4.3 Air Equations for Emissions from Soils

USEPA toxicity criteria indicate that risks from exposure to some chemicals in soil via inhalation after release to air may far outweigh the risk via ingestion of the soil; therefore, as presented previously in equations D-1 through D-4, calculations under the Uniform Standard have been designed to address this pathway as well. The models used to calculate standards for inhalation of volatiles/particulates are updates of risk assessment methods presented in RAGS Part B (USEPA 1991b) and are consistent with the *Soil Screening Guidance: User's Guide and Technical Background Document* (USEPA 1996a,b).

To address the soil-to-air pathways the calculations incorporate volatilization factors (VF_s) for volatile contaminants and particulate emission factors (PEF) for nonvolatile contaminants. These factors relate soil contaminant concentrations to air contaminant concentrations that may be inhaled on-site. The VF_s and PEF equations can be broken into two separate models; an emission model to estimate emissions of the contaminant from the soil and a dispersion model to simulate the dispersion of the contaminant in the atmosphere.

It should be noted that the box model in RAGS Part B has been replaced with a dispersion term (Q/C) derived from a modeling exercise using meteorological data from 29 locations across the United States because the box model may not be applicable to a broad range of site types and meteorology and does not utilize state-of-the-art techniques developed for regulatory dispersion modeling. The dispersion model for both volatiles and particulates is the AREA-ST, an updated version of the Office of Air Quality Planning and Standards, Industrial Source Complex Model (ISC2). However, different Q/C terms are used in the VF and PEF equations. Los Angeles was selected as the 90th percentile data set for volatiles and Minneapolis was selected as the 90th percentile data set for fugitive dusts (USEPA 1996b,c). A default source size of 0.5 acres was chosen for the calculations. If unusual site conditions exist such that the area source is substantially larger than the default source size assumed here, an alternative Q/C could be applied (see USEPA 1996b,c).

D.4.3.1 Soil-to-Air Volatilization Factor

Volatile chemicals, defined as those chemicals having a Henry's Law constant greater than 10^{-5} (atm·m³/mol) and a molecular weight less than 200 g/mole, should be screened for inhalation exposures using a VF_s . To calculate remediation standards for volatile chemicals in soil, a chemical-specific volatilization factor is calculated per Equation D-7. Because of its reliance on Henry's law, the VF_s model is applicable only when the contaminant concentration in soil is at or below saturation (i.e. there is no free-phase contaminant present).

The emission terms used in the VF_s are chemical-specific and may be calculated from physical-chemical information obtained from a number of sources including *Superfund Exposure Assessment Manual* (USEPA 1988), *Subsurface Contamination Reference Guide* (USEPA 1990a), *Fate and Exposure Data* (Howard 1989-1993), and *Superfund Chemical Data Matrix* (US EPA 1994c). In those cases where Diffusivity Coefficients (D_i) are not provided in existing literature, D_i 's may be calculated using Fuller's Method described in USEPA (1988). A surrogate term may be used for some chemicals that lack physio-chemical information. In those cases, a proxy chemical of similar structure may be used that may over- or under-estimate the cleanup standard for soils.

Equation D-7 forms the basis for deriving uniform soil remediation standards for the inhalation of volatiles pathway. The following parameters in the standardized equation can be replaced with site-specific data.

- Source area
- Average soil moisture content
- Average fraction organic carbon content
- Dry soil bulk density.

The basic principle of the VF_s model is applicable only if the soil contaminant concentration is at or below soil saturation ("sat", see section D.4.4). Above this level, the model cannot predict an accurate VF_s . If the Cleanup Standard calculated using VF_s is greater than the calculated "sat" (Equation D-9), the remediation standard should be set equal to "sat" in accordance with *Soil Screening Guidance* (USEPA 1996 b,c).

Equation D-7: Derivation of the Volatilization Factor

$$VF_s (m^3 / kg) = (Q / C) \times \frac{(3.14 \times D_A \times T)^{1/2}}{2 \times r_b \times D_A} \times 10^{-4} (m^2 / cm^2)$$

where:

$$D_A = \frac{[(q_a^{10/3} D_i H' + q_w^{10/3} D_w) / n^2]}{r_b K_d + q_w + q_a H'}$$

| Parameter | Definition (units) | Default |
|-----------------|--|---|
| VF _s | Volatilization factor (m ³ /kg) | -- |
| D _A | Apparent Diffusivity (cm ² /s) | -- |
| Q/C | Inverse of the mean conc. At the center of a 0.5-acre square source (g/m ² -s per kg/m ³) | 68.81 |
| T | Exposure interval (s) | 9.5×10 ⁸ |
| ρ _b | Dry soil bulk density (g/cm ³) | 1.5 |
| θ _a | Air filled soil porosity (L _{air} /L _{soil}) | 0.28 or n-θ _w |
| n | Total soil porosity (L _{pore} /L _{soil}) | 0.43 or 1 - (ρ _b /ρ _s) |
| θ _w | Water- filled soil porosity (L _{water} /L _{soil}) | 0.15 |
| ρ _s | Soil particle density (g/cm ³) | 2.65 |
| D _i | Diffusivity in air (cm ² /s) | Chemical-specific |
| H' | Dimensionless Henry's Law constant | Calculated from H by multiplying by 41 (U.S. EPA 1991a) |
| H | Henry's Law Constant (atm-m ³ /mol) | Chemical-specific |
| D _w | Diffusivity in water (cm ² /s) | Chemical-specific |
| K _d | Soil-water partition coefficient (cm ³ /g) = K _{oc} f _{oc} | Chemical-specific |
| K _{oc} | Soil organic carbon-water partition coefficient (cm ³ /g) | Chemical-specific |
| f _{oc} | Fraction organic carbon in soil (g/g) | 0.006 (0.6%) |

D.4.3.2 Soil-to-Air Particulate Emission Factor (PEF)

Inhalation of chemicals adsorbed to respirable particles (PM₁₀) were assessed using a default PEF equal to 1.316×10⁹ cubic meters per kilogram (m³/kg) that relates the contaminant concentration in soil with the concentration of respirable particles in the air due to fugitive dust emissions from contaminated soils. The generic PEF was derived using default values in Equation D-8, which corresponds to a receptor point concentration of approximately 0.76 micrograms per cubic meter (μg/m³). The relationship is derived by Cowherd (1985) for a rapid assessment procedure applicable to a typical hazardous waste site where the surface contamination provides a relatively continuous and constant potential for emission over an extended period of time (e.g., years). This represents an annual average emission rate based on wind erosion that should be compared with chronic health criteria; it is not appropriate for evaluating the potential for more acute exposures.

With the exception of specific heavy metals, the PEF does not appear to significantly affect most soil standards. Equation D8 forms the basis for deriving a generic PEF for the inhalation of particulates pathway. For more details regarding specific parameters used in the PEF model, the reader is referred to *Soil Screening Guidance: Technical Background Document* (USEPA 1996b).

Note: The generic PEF evaluates windborne emissions and does not consider dust emissions from traffic or other forms of mechanical disturbance that could lead to greater emissions than assumed here.

Equation D-8: Derivation of the Particulate Emission Factor

$$PEF(m^3 / kg) = Q / C \times \frac{3600s / h}{0.036 \times (1 - V) \times (U_m / U_t)^3 \times F(x)}$$

| Parameter | Definition (units) | Default |
|----------------|--|-------------------------|
| PEF | Particulate emission factor (m ³ /kg) | 1.316 × 10 ⁹ |
| Q/C | Inverse of the mean concentration at the center of a 0.5-acre-square source (g/m ² -s per kg/m ³) | 90.80 |
| V | Fraction of vegetative cover (unitless) | 0.5 |
| U _m | Mean annual windspeed (m/s) | 4.69 |
| U _t | Equivalent threshold value of windspeed at 7 m (m/s) | 11.32 |
| F(x) | Function dependent on U _m /U _t derived using Cowherd (1985) (unitless) | 0.194 |

D.4.4 Soil Saturation Concentration

Soil saturation ("sat") corresponds to the contaminant concentration in soil at which the adsorptive limits of the soil particles and the solubility limits of the available soil moisture have been reached. Above this point, pure liquid-phase contaminant is expected in the soil. If the Standard calculated using VF_s is greater than the calculated sat, the Standard should be set equal to sat, in accordance with *Soil Screening Guidance* (USEPA 1996 b,c). The updated equation for deriving sat is presented in Equation D-9.

Equation D-9: Derivation of the Soil Saturation Limit

$$sat = \frac{S}{r_b} (K_d r_b + q_w + H' q_a)$$

| Parameter | Definition (units) | Default |
|------------|--|---|
| Sat | Soil saturation concentration (mg/kg) | -- |
| S | Solubility in water (mg/L-water) | Chemical-specific |
| ρ_b | Dry soil bulk density (kg/L) | 1.5 |
| K_d | Soil-water partition coefficient (L/kg) | $K_{oc} \times f_{oc}$ (chemical-specific) |
| K_{oc} | Soil organic carbon/water partition coefficient (L/kg) | Chemical-specific |
| f_{oc} | Fraction organic carbon in soil (g/g) | 0.006 (0.6%) |
| θ_w | Water-filled soil porosity (L_{water}/L_{soil}) | 0.15 |
| θ_a | Air filled soil porosity (L_{air}/L_{soil}) | 0.28 or no- θ_w |
| H | Henry's Law constant ($atm \cdot m^3/mol$) | Chemical-specific |
| H' | Dimensionless Henry's Law constant | $H \times 41$, where 41 is a units conversion factor |

D.4.5 Migration to Groundwater Pathway

The methodology for calculating a soil standard for the migration to groundwater was developed to identify chemical concentrations in soil that have the potential to contaminate groundwater. Migration of contaminants from soil to groundwater can be envisioned as a two-stage process: (1) release of contaminant in soil leachate and (2) transport of the contaminant through the underlying soil and aquifer to a receptor well. The methodology considers both of these fate and transport mechanisms, and is based on the methodology presented in USEPA's *Soil Screening Guidance: Users Guide* (USEPA 1996c).

To calculate a remediation standard for soil that will protect for the migration to groundwater pathway, multiply the acceptable groundwater concentration by the dilution factor to obtain a target soil leachate concentration. A default value of 20 can be used for the dilution factor, or site information can be used to calculate a value using Equation D-10. For example, if the dilution factor is 20 and the acceptable ground water concentration is 0.05 mg/L, the target soil/water leachate concentration would be 1.0 mg/L. Next, the partition equation (Equation D-11) is used to calculate the total soil concentration corresponding to this soil leachate concentration. Alternatively, if a site-specific leach test is used, compare the target soil leachate concentration to extract concentrations from the leach tests. For further information regarding the calculations of standards based on leaching from soil to groundwater, the reader is referred to USEPA *Soil Screening Guidance: User's Guide* (USEPA 1996c).

Equation D-10: Derivation of Dilution Factor

$$dilution\ factor = 1 + \frac{Kid}{IL}$$

| Parameter/Definition (units) | Default |
|--|-----------------------|
| Dilution factor (unitless) | 20 (0.5-acre source)* |
| K/aquifer hydraulic conductivity (m/yr) | Site-Specific |
| i/hydraulic gradient (m/m) | Site-Specific |
| I/infiltration rate (m/yr) | Site-Specific |
| d/mixing zone depth (m) | Site-Specific |
| L/source length parallel to groundwater flow (m) | Site-Specific |

*Use default dilution factor of 20 or calculate a value based on site-specific parameters.

Equation D-11: Soil Screening Level Partitioning Equation for Migration to Groundwater

$$Screening\ Level\ in\ Soil\ (mg / kg) = C_w [K_d + \frac{(q_w + q_a H')}{r_b}]$$

| Parameter/Definition (units) | Default |
|--|---|
| C_w /target soil leachate concentration (mg/L) | Nonzero MCLG, MCL, or health-based level \times dilution factor |
| $K_{d,soil}$ /water partition coefficient (L/kg) | Chemical-specific |
| K_{oc} /soil organic carbon/water partition coefficient (L/kg) | $K_{oc} \times f_{oc}$ (organics) chemical-specific |
| F_{oc} /fraction organic carbon in soil (g/g) | 0.002 (0.2%) |
| θ_w /water-filled soil porosity (L_{water}/L_{soil}) | 0.3 |
| θ_a /air-filled soil porosity (L_{air}/L_{soil}) | $n - \theta_w$ |
| ρ_b /dry soil bulk density (kg/L) | 1.5 |
| n /soil porosity (L_{pore}/L_{soil}) | $1 - (\rho_b/\rho_s)$ |
| ρ_s /soil particle density (kg/L) | 2.65 |
| H' /dimensionless Henry's law constant | Chemical-specific (assume to be zero for inorganic contaminants except mercury) |

The USEPA methodology was designed for use during the early stages of a site evaluation when information about subsurface conditions may be limited. Because of this constraint, the methodology is based on conservative, simplifying assumptions about the release and transport of contaminants in the subsurface.

D.5 References

California Environmental Protection Agency (Cal EPA). 1994. Preliminary Endangerment Assessment Guidance Manual. Department of Toxic Substances Control, Sacramento, California.

Cowherd, C., G. Muleski, P. Engelhart, and D. Gillette. 1985. Rapid Assessment of Exposure to Particulate Emission from Surface Contamination. EPA/600/8-85/002. Prepared for Office of Health and Environmental Assessment, U.S. Environmental Protection Agency, Washington, DC. NTIS PB85-192219 7AS.

Howard, P.H. 1989-1993. Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Lewis Publishers, Chelsea, Michigan.

United States Environmental Protection Agency (USEPA). 1988. Superfund Exposure Assessment Manual. EPA/540/1-88/001. Office of Emergency and Remedial Response, Washington, DC.

United States Environmental Protection Agency (USEPA). 1989. Risk Assessment Guidance for Superfund: Volume I- Human Health Evaluation

Manual (Part A). Office of Emergency and Remedial Response. EPA/540/1-89/002.

United States Environmental Protection Agency (USEPA). 1990a. Subsurface Contamination Reference Guide. EPA/540/2-90/011. Office of Emergency and Remedial Response, Washington, DC.

United States Environmental Protection Agency (USEPA). 1991a. Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors. Publication 9285.6-03. Office of Emergency and Remedial Response, Washington, DC. NTIS PB91-921314.

United States Environmental Protection Agency (USEPA). 1991b. Risk Assessment Guidance for Superfund Volume 1: Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals). Publication 9285.7-01B. Office of Emergency and Remedial Response, Washington, DC. NTIS PB92-963333.

United States Environmental Protection Agency (USEPA). 1994. Superfund Chemical Data Matrix. EPA/540/R-94/009. Office of Solid Waste and Emergency Response, Washington, DC. PB94-963506.

United States Environmental Protection Agency (USEPA). 1995. Health Effects Assessment Summary Tables (HEAST): Annual Update, FY 1994. Environmental Criteria Assessment Office, Office of Health and Environmental Assessment, Office of Research and Development, Cincinnati, OH.

United States Environmental Protection Agency (USEPA). 1996a. Region 9 Preliminary Remediation Goals. San Francisco, CA. Stanford J. Smucker, Regional Toxicologist.

United States Environmental Protection Agency (USEPA). 1996b. Soil Screening Guidance: Technical Background Document. EPA/540/R-95/128. Office of Emergency and Remedial Response, Washington, DC. PB96-963502.

United States Environmental Protection Agency (USEPA). 1996c. Soil Screening Guidance: User's Guide. EPA/540/R-96/018. Office of Emergency and Remedial Response, Washington, DC. PB96-963505.

United States Environmental Protection Agency (USEPA). 1998. Integrated Risk Information System (IRIS). (www.epa.gov/ngispgm3/iris).

West Virginia Division of Environmental Protection (WVDEP). 1997. Title 60 Legislative Rule. Bureau of Environment. Division of Environmental Protection. Director's Office. Series 3 Voluntary Remediation and Redevelopment Rule. July.

United States Environmental Protection Agency (USEPA). 1997. Exposure Factors Handbook.

APPENDIX E: RELATIVE ABSORPTION FACTORS AND BIOAVAILABILITY

E.1 Introduction

This appendix provides an overview of relative absorption factors and bioavailability adjustments, the methods for measuring them, and their use in risk assessments. The two primary issues addressed are adjustment of oral toxicity values used in assessing dermal exposures, and adjustment of dermal and oral intake values to account for variations in absorption from different media. Further guidance on adjustments for absorption efficiency, including adjustments of toxicity values from administered to absorbed dose, can be found in the United States Environmental Protection Agency (USEPA) *Risk Assessment Guidance for Superfund, Appendix A* (USEPA, 1989). Terms used are defined at the end of this section.

Absorption adjustments are used in the risk characterization step to ensure that the site exposure estimate and the toxicity value for comparison are both expressed as absorbed doses, or that both are expressed as intake values. Adjustments may be necessary to match the exposure estimate with the toxicity value, if one is based on an absorbed dose and the other is based on an intake (i.e., administered dose). For the dermal route of exposure, toxicity values that are expressed as administered dose will need to be adjusted to absorbed doses for comparison. This adjustment is discussed below.

Adjustments also may be necessary to account for the different absorption efficiencies associated with different exposure media (e.g., contaminants ingested with food or soil may be less completely absorbed than contaminants ingested with water). If the medium of oral exposure in the site exposure assessment differs from the medium of exposure assumed by the toxicity value, an absorption adjustment may be appropriate to express the site exposure in terms that are comparable to the toxicity value. This adjustment is termed a relative absorption factor (RAF). For example, a substance might be more completely absorbed following exposure to the substance in drinking water than following exposure to food or soil containing the substance. A relative absorption factor would then be used to adjust the food or soil ingestion exposure estimate to match a reference dose (RfD) or cancer slope factor (CSF) based on an assumption of drinking water ingestion. This adjustment is discussed below.

E.2 Definitions

Absorbed dose. The amount of a substance that penetrates the exchange boundaries of an organism after contact. Absorbed dose is calculated from the intake and the absorption efficiency, and is usually expressed as mass of a substance absorbed into the body per unit body weight per unit time (e.g., mg/kg-day).

Administered dose. The mass of substance administered to an organism and in contact with an exchange boundary (e.g., gastrointestinal tract) per unit body weight per unit time (e.g., mg/kg-day).

Bioavailability. The bioavailability of a substance may be defined in a variety of ways, depending upon the interests of the investigator and the specific objectives

of a given study. For the purpose of this guidance, bioavailability is defined as the fraction of an administered dose that reaches the central (blood) compartment. Bioavailability defined in this manner is commonly referred to as “absolute bioavailability”.

Cancer Slope Factor. A plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime. The CSF is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a potential carcinogen.

Exposure Route. The way a chemical or physical agent comes in contact with an organism (i.e., by ingestion, inhalation, or dermal contact).

Exposure Medium. The various materials to which an organism may be exposed (e.g., water, food, or soil).

Intake. A measure of exposure expressed as the mass of substance in contact with the exchange boundary per unit body weight per unit time (e.g., mg/kg-day). Also termed the normalized exposure rate, and equivalent to administered dose.

Reference Dose. The USEPA’s preferred toxicity value for evaluating noncarcinogenic effects resulting from exposures to toxic substances.

Relative Absorption Factor. The RAF describes the absorbed fraction of a contaminant from a particular exposure medium relative to the fraction absorbed from the dosing vehicle used in the toxicity study for that compound.

Relative Bioavailability. Relative bioavailability refers to comparative bioavailabilities from different exposure media (e.g., bioavailability from soil relative to bioavailability from water), expressed in this guidance as a fractional relative absorption factor (RAF).

E.3 Bioavailability Adjustments for Assessing Dermal Exposures

E.3.1 Converting Oral Toxicity Values from Administered to Absorbed Doses

Because there are few, if any, toxicity values for dermal exposure, oral toxicity values will need to be used to assess risks from dermal exposure. The following guidance is based on Appendix A of USEPA’s *Risk Assessment Guidance for Superfund* (USEPA, 1989). Most oral toxicity values (i.e., RfDs and CSFs) are expressed as the amount of substance administered per unit time and unit body weight, whereas exposure estimates for the dermal route of exposure are eventually expressed as absorbed doses. Thus, for dermal exposure to contaminants in water or in soil, it may be necessary to adjust an oral toxicity value from an administered to an absorbed dose. An oral RfD may be converted to absorbed dose by multiplying the RfD by the fractional absorption value (ABS); i.e.,

$$\text{RfD}_O \times \text{ABS} = \text{RfD}_{\text{ABS}}$$

An oral cancer slope factor may be converted to absorbed dose by dividing the cancer slope factor by the fractional absorption value, i.e.,

$$CSF_{O/ABS} = CSF_{ABS}$$

Adjustments for an oral RfD and an oral slope factor, respectively, are shown in Examples E-1 and E-2.

If the oral toxicity value is already expressed as an absorbed dose (e.g., trichloroethylene), it is not necessary to adjust the toxicity value.

USEPA's Integrated Risk Information System (IRIS) files and Agency for Toxic Substances Disease Registry (ATSDR) toxicity profiles are good sources of oral absorption estimates. Hrudley et al (1996) also provides an extensive review of oral absorption estimates for a number of compounds. Because oral absorption may vary with the matrix administered (e.g., water vs. diet), the absorption values used should be appropriate for the matrix administered in the toxicity studies that serve as the basis for the oral toxicity values.

In the absence of any information on absorption for the substance or chemically related substances, one must assume an oral absorption efficiency. Assuming 100 percent absorption in an oral administration study that serves as the basis for an RfD or slope factor would be a very conservative approach for estimating the dermal RfD or slope factor (i.e., depending on the type of chemical, the true absorbed dose might have been much lower than 100 percent; hence, an absorbed-dose RfD should similarly be much lower, or the slope factor should be much higher). For example, some metals tend to be poorly absorbed (less than 5 percent) by the gastrointestinal tract. A relatively conservative assumption for oral absorption in the absence of appropriate information would be 5 percent.

E.3.2 Dermal Absorption Estimates for Sediment and Soil Contact

The adjusted toxicity values reflecting absorbed dose may be used to assess dermal exposure to chemicals in water, sediments, or soil. The National Center for Exposure Assessment (www.epa.gov/ncea/index.html) may be a source of oral absorption estimates for oral to dermal adjustments of toxicity values. When assessing dermal exposures to chemicals in sediments or soil, it is also necessary to estimate dermal absorption. USEPA Region III has provided guidance, with default assumptions, for a number of chemicals (USEPA, 1995). These values are listed below. The guidance is available via the internet at <http://www.epa.gov/reg3hwand/risk/solo.bsg2.htm>. Additional information is available by calling (215) 597-1309.

| Chemicals (s) | Default Dermal Absorption (%) |
|---|-------------------------------|
| Arsenic | 3 |
| Cadmium | 1 |
| Other metals | 1 |
| Volatile organic compounds (vapor pressure ≥ 95.2 mm Hg), e.g., benzene, 1,1-DCE, 1,1,1-TCE) | 0.05 |
| (vapor pressure < 95.2 , e.g., ethylbenzene, PCB, toluene, xylenes) | 3 |
| Chlorinated dioxins | 3 |
| Pentachlorophenol | 24.4 |
| Pesticides | 10 |
| PCBs | 6 |
| Other semivolatile compounds | 10 |

Although these serve as default values, applicants may wish to consult recent literature, or to conduct studies of their own. USEPA has provided comprehensive guidance regarding the assessment of dermal absorption (USEPA 1992), and has several research projects underway.

E.4 Relative Absorption Factors for Assessing Oral Exposures

E.4.1 Adjustment for Medium of Exposure

As discussed above, if the medium of oral exposure in the site exposure assessment differs from the medium of exposure assumed in the oral toxicity assessment, then an accurate assessment of site risks may require an absorption adjustment to express the exposures in the same terms. Such adjustments may be applied in assessing oral exposures to metals, pesticides, and other semivolatile organic compounds. Generally, bioavailability is expected to decrease as volatility decreases, and soil residence times increase. Frequently, toxicity values are based on have been adjusted to reflect exposures to chemicals in drinking water or diet, while the site exposure of concern is to chemicals in soil. Because the absorption of chemicals in soil is often less than their absorption from drinking water, a comparison of relative absorption efficiencies is necessary to adjust the site exposure to that on which the RfD or slope factor is based. In some cases, the absorption of a chemical from the dosing medium and the absorption from soil are both known, and an RAF can be calculated by dividing the absorption from soil by the absorption from the dosing medium. This RAF is used to adjust the chronic daily intake (CDI) value; i.e.,

$$\text{CDI} \times \text{RAF} = \text{adjusted CDI.}$$

An example calculation to adjust for medium of exposure is given in Example E-3.

In most cases, an RAF will be determined experimentally without specifically identifying absorption from the dosing medium. Methods for conducting such studies are described below. Table E-1 presents default values that may be applied for some chemicals in soil.

E.4.2 Methods of Assessing Bioavailability

Several methods are available for estimating the extent of oral absorption of compounds from environmental matrices. The method selected for a specific study will depend on the characteristics of the compound being studied and on the end use of the resulting data. Data requirements for an accurate assessment of relative bioavailability (i.e., absorption from an environmental matrix relative to absorption from the dose formulation used in the toxicity study) are substantially less rigorous than those for an accurate determination of absolute bioavailability. For this reason, and because measures of relative bioavailability are generally most useful for risk assessment, most studies are designed to determine relative bioavailability. Relative bioavailability may be determined by comparing tissue concentrations after doses are administered, or by comparing the likely extent of dissolution of different formulations in the gastrointestinal tract. Such comparisons of extent of dissolution may be conducted using *in vitro* test systems that mimic gastrointestinal tract processes. Both *in vivo* and *in vitro* methods of assessing oral bioavailability are reviewed below.

E.4.2.1 In Vivo Methods of Assessing Bioavailability

Animal models have been developed for evaluating the relative bioavailability of arsenic (swine and monkeys), cadmium (weanling rats), mercury, lead (weanling rats and weanling swine), polycyclic aromatic hydrocarbons (PAHs; mice), polychlorinated biphenyls (PCBs, rats) petroleum hydrocarbons (mice), and tetrachlorodibenzo-p-dioxin (TCDD; rats). The reader is referred to the references in Table E-2 for further information on the design and application of these animal models.

E.4.2.2 In Vitro Methods of Assessing Bioavailability

Physiologically based *in vitro* models have been developed for assessing relative lead bioavailability from soil and have been validated against results from *in vivo* studies in weanling rats (Ruby et al 1996) and weanling swine (Medlin 1997). The *in vitro* method presented in Medlin (1997) is recommended for assessing relative lead bioavailability from soil.

A physiologically available cyanide *in vitro* method has been developed by Magee et al. (1996a) in conjunction with the Massachusetts Department of Environmental Protection. This method is appropriate for evaluating the bioavailability of complexed cyanide from soil.

Additional *in vitro* methods for assessment of arsenic, cadmium, chromium, mercury, and PAH bioavailability are under development. As these methods are validated, they may also become acceptable for use in human health risk assessment.

E.4.3 Other Methods of Assessing Bioavailability

Less precise information about relative bioavailability can also be obtained using less rigorous methods, i.e., the methods described below yield qualitative information that is not appropriate for use in quantitative adjustments to risk assessments. Standard leaching tests, such as the Toxicity Characteristics Leaching Procedure (TCLP) or the Synthetic Precipitation

Leaching Procedure (SPLP) indicate whether a chemical will have limited potential to dissolve in the gastrointestinal tract. Limited ability to leach a chemical from soil may also indicate a limited ability to remove the chemical from soil during remediation.

For metals, mineralogical studies may be used to identify the specific metal compounds present in soil. If the bioavailability of the individual metal compounds – relative to the compound tested in toxicity studies relied upon by USEPA – is known, it may be possible to predict the relative bioavailability of the metal in soil. Such predictions are not likely to be as accurate as directly testing the soil, however, due to interactions of metal ions with soil constituents. Such interactions are likely to further modify the solubility and bioavailability of the metal in soil.

E.4.4 Guidance for Selecting Study Methods

This brief summary of methods for assessing oral bioavailability provides a hierarchy for evaluating bioavailability data. Animal studies are generally considered the most reliable, but are also more expensive and time consuming than *in vitro* studies. Protocols for these studies must be evaluated carefully to ensure that the study design and animal model selected are appropriate for the chemical being tested. *In vitro* methods that simulate the function of the gastrointestinal tract are generally more robust than *in vivo* studies, and are rapid and relatively inexpensive. Such studies have been validated by comparison with *in vivo* data for metals only, although studies are currently underway to adapt an *in vitro* test system for use with semivolatile organic compounds. Finally, simple leaching tests and mineralogical analyses may provide useful information for risk management and selection of remediation options, but are not expected to provide reliable quantitative bioavailability adjustments for use in deriving risk-based cleanup levels.

For a number of organic and inorganic contaminants, sufficient data are available from animal (*in vivo*) studies to provide default RAFs for these compounds in soil. Table E-1 provides a list of these default values, along with references to the studies on which they are based. If a default RAF is not provided for a specific contaminant, or a more accurate (site-specific) RAF is desired, a site-specific value may be derived using the methods discussed below.

Table E-1: Default RAFs for Oral Exposure to Contaminants in Soil

| Contaminant | RAF | Basis |
|-------------|-------------------|---|
| Arsenic | 0.40 | Freeman et al., 1993; 1995; USEPA (as cited in Medlin, 1997) |
| Cadmium | 0.50 | Schoof and Freeman, 1995 |
| Lead | 0.60 ^a | Dieter et al., 1993; Freeman et al., 1992; USEPA (as cited in Medlin, 1997) |
| Mercury | 0.30 ^b | DOE, 1995; Smucker, 1994 |
| PAHs | 0.30 | Magee et al., 1996b |
| TCDD | 0.50 | Shu et al., 1988 |

^a Numerous studies in weanling animals have indicated that RAFs for lead in soil vary widely, depending on the source and form of lead present. These results indicated that site-specific data is necessary to justify the use of a value other than the default. use of the lead RAF for risk assessment requires converting the RAF to absolute bioavailability for use in the Integrated Exposure Uptake Biokinetic Model (IEUBK, see USEPA, 1994a) or the Adult Lead Model (see USEPA, 1996).

^b Value is applicable to soils that contain predominantly elemental mercury or mervuric sulfide.

Table E-2: References for the Design of Animal Models for Oral Bioavailability Assessment

| Element | Animal Model | Reference |
|--------------|----------------|---|
| Arsenic | Monkeys | Freeman et al., 1995 |
| | Swine | Region VIII reference |
| Cadmium | Rats | Schoof and Freeman, 1995 |
| Lead | Weanling Rats | Freeman et al. 1992; Schoof et al., 1995 |
| | Weanling Swine | USEPA 1994b |
| Mercury | Various | Schoof and Nielsen, Risk Analysis, in press |
| PAHs | Rats | Goon et al., 1990, 1991 |
| | Mice | Weyand et al., 1996 |
| PCBs | Rats | [ref.] |
| Petroleum | Mice | Air Force study reference |
| Hydrocarbons | | |
| TCDD | Rats | Shu et al., 1988 |

Example E-1 -- Adjustment of an Administered to an Absorbed Dose RfD

An oral Rfd, unadjusted for absorption, equals 10 mg/kg-day.

Other information (or an assumption) indicates a 20 percent oral absorption efficiency in the species on which the RfD is based.

The adjusted RfD that would correspond to the absorbed dose would be:

$$10 \text{ mg/kg-day} \times 0.20 = 2 \text{ mg/kg-day},$$

The adjusted RfD of 2 mg/kg-day would be compared with the amount estimated to be absorbed dermally each day.

Example E-2 -- Adjustment of an Administered to an Absorbed Dose Slope Factor

An oral slope factor, unadjusted for absorption, equals (mg/kg-day)⁻¹.

Other information (or an assumption) indicated a 20 percent absorption efficiency in the species on which the slope factor is based.

The adjusted slope factor that would correspond to the absorbed dose would be:

$$1.6 \text{ (mg/kg-day)}^{-1} / 0.20 = 8 \text{ (mg/kg-day)}^{-1}.$$

The adjusted slope factor of 8 (mg/kg-day)⁻¹ would be used to estimate the cancer risk associated with the estimated absorbed dose for the dermal route of exposure.

Example E-3 -- Adjustment for Medium of Exposure

The daily oral intake of a chemical in soil is estimated to be 5 mg/kg-day.

The absorption of the chemical in drinking water is known to be 90 percent and the absorption of the chemical from soil is measured to be 45 percent.

The relative absorption of the chemical in soil is 0.5 (i.e., the FAF = 0.45 / 0.90).

The oral intake of the chemical in soil may be adjusted by the RAF, to be comparable with the oral toxicity factor (i.e., the RfD or cancer slope factor) which is based on an administered dose in drinking water.

E.5 References

Dieter, M.P., H.B. Matthews, R.A. Jeffcoat, and R.F. Moseman. 1993. Comparison Of Lead Bioavailability In F344 Rats Fed Lead Acetate, Lead Oxide, Lead Sulfide, Or Lead Ore Concentrate From Skagway, Alaska. J. Toxicol. Environ. Health 39:79-93.

DOE. 1995. Record of Decision for Lower East Fork Poplar Creek. DOE/OR/02-1370&D1. US Department of Energy, Office of Environmental Restoration and Waste Management. Prepared by Jacobs ER Team, Oak Ridge, TN.

Freeman, G.B., J.D. Johnson, J.M. Killinger, S.C. Liao, P.I. Feder, A.O. Davis, M.V. Ruby, R.L. Chaney, S.C. Lovre, and P.D. Bergstrom. 1992. Relative Bioavailability Of Lead From Mining Waste Soil In Rats. *Fund. Appl. Toxicol.* 19:388–398.

Freeman, G.B., J.D. Johnson, J.M. Killinger, S.C. Liao, A.O. Davis, M.V. Ruby, R.L. Chaney, S.C. Lovre, and P.D. Bergstrom. 1993. Bioavailability Of Arsenic In Soil Impacted By Smelter Activities Following Oral Administration In Rabbits. *Fund. Appl. Tox.* 21:83–88.

Freeman, G.B., R.A. Schoof, M.V. Ruby, A.O. Davis, J.A. Dill, S.C. Liao, C.A. Lapin, and P.D. Bergstrom. 1995. Bioavailability Of Arsenic In Soil And House Dust Impacted By Smelter Activities Following Oral Administration In Cynomolgus Monkeys. *Fund. Appl. Toxicol.* 28:215–222.

Goon, D., N.S. Hatoum, J.D. Jernigan, S.L. Schmidt, and P.J. Garvin. 1990. Pharmacokinetics And Oral Bioavailability Or Soil-Adsorbed Benzo[A]Pyrene (Bap) In Rats. *Toxicologist* 10:218.

Goon, D., N.S. Hatoum, M.J. Klan, J.D. Nerniganm, and R.G. Farmer. 1991. Oral Bioavailability Of “Aged” Soil-Adsorbed Benzo[A]Pyrene (Bap) In Rats. *Toxicologist* 11:1356.

Hrudey, S.E., W. Chen, and C.G. Rousseaux. 1996. Bioavailability in Environmental Risk Assessment. CRC Press, Inc., New York, NY.

Magee, B.H, A. Taft, W. Ratliff, J. Kelley, J. Sullivan, and O. Pancorbo. 1996a. Physiologically Available Cyanide (PAC) in Manufactured Gas Plant Wastes and Soil Samples. Abstract. 11th Annual Conference on Contaminated Soils, University of Massachusetts at Amherst. October 21–24.

Magee, B., P. Anderson, and D. Burmaster. 1996b. Absorption Adjustment Factor (AAF) Distributions For Polycyclic Aromatic Hydrocarbons (PAHs). *Human Ecol. Risk Assess.* 2(4):841–873.

Medlin, E.A. 1997. An In Vitro Method For Estimating The Relative Bioavailability Of Lead In Humans. Masters Thesis. Department of Geological Sciences, University of Colorado at Boulder.

Ruby, M.V., A. Davis, R. Schoof, S. Eberle, and C.M. Sellstone. 1996. Estimation Of Lead And Arsenic Bioavailability Using A Physiologically Based Extraction Test. *Environ. Sci. Technol.* 30(2):422–430.

Schoof, R.A., M.K. Butcher, C. Sellstone, R.W. Ball, J.R. Fricke, V. Keller, and B. Keehn. 1995. An Assessment Of Lead Absorption From Soil Affected By Smelter Emissions. *Environ. Geochem. Health* 17(4):189–199.

Schoof, R.A., and G.B. Freeman. 1995. Oral Bioavailability Of Lead And Cadmium In Soil From A Smelter Site. Poster presented at the Seventh International Congress of Toxicology, Seattle, Washington. July 3–6, 1995.

Shu, H., D. Paustenbach, F.J. Murray, L. Marple, B., Brunck, D. Rossi, and P. Teitelbaum. 1988. Bioavailability Of Soil-Bound TCDD: Oral Bioavailability In The Rat. *Fund. Appl. Toxicol.* 10:648–654.

Smucker, S. 1994. Memorandum to S. Hogan (December 1, 1994). Subject: Issue Paper For Determination Of Site-Specific Cleanup Goal For Mercury In Residential Soils. US Environmental Protection Agency, Region IX, San Francisco, CA.

USEPA. 1989. Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A). Interim Final. EPA/540/1-89/002. US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA. 1992. Dermal Exposure Assessment: Principles And Applications. EPA/600/8-91/011B. US Environmental Protection Agency, Office of Research and Development, Washington, DC.

USEPA. 1994a. Guidance Manual For The Integrated Exposure Uptake Biokinetic Model For Lead In Children. EPA/540/R-93/081. US Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA. 1994b. Protocol For Systemic Availability Of Lead To Young Swine From Subchronic Administration Of Lead-Contaminated Soil (Phase II). Ref: 8HWM-SM-TS. US Environmental Protection Agency, Region VIII, Denver, Colorado.

USEPA. 1995. Assessing Dermal Exposure From Soil. US Environmental Protection Agency, Region III, Office of Superfund Programs, Philadelphia, PA. EPA/903-K-95-003.

USEPA. 1996. Recommendations Of The Technical Review Workgroup For Lead For An Interim Approach To Assessing Risks Associated With Adult Exposures To Lead In Soil. US Environmental Protection Agency, Technical Review Workgroup for Lead.

Weyand, E.H., K. Rozett, A. Koganit, and R. Singh. 1996. Effect Of Soil On The Genotoxicity Of Manufactured Gas Plant Residue. *Fund. Appl. Toxicol.* 30(1): Part 2.

APPENDIX E: ACRONYM LIST

| | |
|-----------|---|
| ABS | fractional absorption value |
| ATSDR | Agency for Toxic Substances Disease Registry |
| CDI | chronic daily intake |
| CSF | cancer slope factor |
| IEUBK | Integrated Exposure Uptake Biokinetic Model |
| IRIS | Integrated Risk Information System |
| mg/kg-day | milligrams per kilogram per day |
| PAH | polycyclic aromatic hydrocarbons |
| PCB | polychlorinated biphenyls |
| RAF | relative absorption factor |
| RfD | reference dose |
| SPLP | Synthetic Precipitation Leaching Procedure |
| TCDD | tetrachlorodibenzo-p-dioxin |
| TCLP | Toxicity Characteristics Leaching Procedure |
| USEPA | United States Environmental Protection Agency |

APPENDIX F: RISK ASSESSMENT FOR LEAD

F.1 Introduction

Risks for lead are assessed by comparing predicted blood lead levels to target blood lead levels, rather than by calculating lifetime cancer risks (in the case of carcinogens), or comparing predicted exposure to a Reference Dose (RfD) (in the case of non-carcinogens). Blood lead levels are predicted based on environmental lead concentrations using either a childhood or an adult model. Soil lead cleanup levels are calculated by selecting a target blood lead level and doing a reverse calculation with the models to solve for soil lead concentration.

F.2 Residential Exposure Scenarios

Children are the primary population of concern in residential exposure scenarios. Children ingest more lead than do adults as a result of their frequent hand-to-mouth behavior, they absorb more of the lead they ingest than do adults, and they are more sensitive to the effects of lead than are adults. Blood lead levels for children are assessed for residential exposure scenarios since protection of the childhood population alone assures the protection of the less-susceptible adult population.

F.2.1 The USEPA Integrated Exposure Uptake Biokinetic Model for Children

The Integrated Exposure Uptake Biokinetic (IEUBK) Model (Version 0.99d) was designed by the United States Environmental Protection Agency (USEPA) to estimate blood lead levels in children up to age seven resulting from their exposure to lead in multiple environmental sources, including diet, drinking water, air, and soil and dust. The Model contains four components: exposure characterization, an absorption model, a biokinetics model, and characterization of uncertainty or variability. Using estimates of lead concentrations in the environment, the exposure component of the Model predicts the amount of lead taken into the body that is then available in the gut or lungs for absorption. The absorption component calculates the fraction of lead in the gut or lungs that is absorbed into the body's circulatory system. The biokinetic component models the distribution of lead in the body among blood, bone, liver, kidney, and other soft tissues and body fluids. Finally, the uncertainty and variability component quantifies the extent to which blood lead levels may differ among children exposed to the same environmental levels of lead. The IEUBK Model is also used to estimate acceptable soil lead levels by choosing a target blood lead level, and, in effect, running the Model "backwards" to derive a soil lead level that corresponds to that target.

F.2.2 Target Blood Lead Levels for Children

The USEPA recommends that environmental lead levels be limited to ensure that no individual child has more than a 5% probability of having a blood lead level exceeding 10 µg/dL (USEPA, 1994a). This corresponds to a population median blood lead level below the target of 10 µg/dL. The USEPA target blood lead level will be acceptable for the Voluntary Remediation

and Redevelopment Act (VRRRA) program. However, note that the Centers for Disease Control (CDC) recommends (1) rescreening of children with blood lead levels between 10 and 14 $\mu\text{g}/\text{dL}$ if a large proportion of children in a particular community have blood lead levels above 10 $\mu\text{g}/\text{dL}$, (2) nutritional and education intervention for children with blood lead levels between 15 and 19 $\mu\text{g}/\text{dL}$, with environmental investigation if blood lead levels persist in this range, and (3) environmental evaluation and remediation only when a child's blood lead level exceeds 20 $\mu\text{g}/\text{dL}$ (US CDC, 1991).

F.2.3 Exposure Parameters for Children

The USEPA Lead Guidance Manual (1994b) provides recommended values for all exposure parameters required to calculate a distribution of blood lead levels for young children. Discussed below are only those parameters relevant to the soil/dust ingestion pathway, and recommend that USEPA guidance be followed for parameter values relevant to exposures through diet or drinking water.

- *Soil lead concentrations* – Generally four or more soil samples are taken from the yard of a single residence. The required number of samples may be adjusted based on potential site use and/or degree of homogeneity of the contamination. Composite sampling is optional and may be helpful in providing adequate coverage of large properties. The average lead concentration of these samples should be used as input to the IEUBK Model for risk assessment purposes.
- *Dust lead concentrations* – When measured interior residential house dust levels are available, they can be used directly as input to the IEUBK Model. However, dust levels may not have been measured, or in the case of new construction, will not be available. In this situation, a soil-to-dust transfer coefficient can be used to estimate dust lead levels as a function of soil lead levels. USEPA recommends a soil-to-dust transfer coefficient of 0.7; that is, interior residential dust lead concentrations are assumed to be 70% that of yard soil lead concentrations. However, there are multiple potential sources of lead in interior dust, and a transfer coefficient of 0.7 likely reflects sources of interior lead other than soil, an assumption that can incorrectly inflate the apparent value of the transfer coefficient. In the absence of other sources of lead, such as lead paint, or when other sources make a small contribution to dust lead levels, a transfer coefficient of 0.3 may be more appropriate. A transfer coefficient within the range of 0.3 to 0.7 is recommended, depending on age and conditioning of housing, and the likelihood of additional sources of lead in dust. Justification should be given for the chosen value.
- *Exposure frequency* – The IEUBK Model was designed to assess uniform exposures, meaning exposures that occur every day of the year. As such, there is no explicit mechanism to deal with exposure frequency. However, the model can be used to consider reduced exposure frequencies by calculating a weighted average that reflects the fraction of each year during which a child is exposed to soil and dust with different lead concentrations. For example, if a child spends 3 months in the summer at a residence with soil lead concentrations of 200 mg/kg,

then the yearly average blood lead level arising from these exposures could be assessed by using a soil lead concentration term equal to $(1000)(3/12) + (200)(9/12) = 400$ mg/kg. This example assures both soils have the same lead bioavailability. If not, then a weighting factor adjusting for alternate bioavailabilities should also be included.

- *Soil/dust ingestion rates* – USEPA recommends a combined soil and dust ingestion rate that is age-dependent and ranges from 85 to 135 mg/day. The agency also provides several alternative sets of values (see Table 2-7) if soil/dust ingestion rates for children of a particular age are required.
- The lower set of soil/dust ingestion rates should be used for sites where there may be reason to suspect more limited soil and dust ingestion, such as at sites with full grass cover. USEPA also recommends a default split between soil and dust ingestion of 45% soil and 55% dust. The 45%-55% split may be adequate for typical suburban or rural areas, but inner-city areas with little to no yard space should more heavily weighted towards dust ingestion.
- *Soil/dust absorption* – USEPA recommends a default soil and dust absorption value for children of 0.3 or 30%. The model includes equations and parameters to effect a non-linear relationship between absorption and increasing lead intake. The default absorption value can be modified on the basis of site-specific bioavailability information, such as would result from an in vivo or in vitro test (Ruby, et al. 1996)
- *Geometric standard deviation* – The individual geometric standard deviation (GSD) is the parameter used to estimate the probability that an individual's blood lead level exceeds 10 µg/dL given the Model-predicted geometric mean blood lead level for the designated environmental exposures. USEPA currently recommends a value of 1.6. However, this value, which quantifies the spread in blood lead levels assuming exposure to uniform levels of lead in soil and dust, is quite high given that community-wide blood lead GSD values, which reflect a range of exposure conditions (e.g., a range of soil and dust lead levels) are often not much higher than 1.6. USEPA has recently estimated site-specific GSDs for the communities of Bingham Creek and Sandy Utah (USEPA, 1995a,b) of 1.43 and 1.4, respectively. It is recommended that the individual GSD be chosen from a range of 1.4 to 1.6, with justification given for the chosen value.

F.3 Commercial/Industrial Exposure Scenarios

Although evaluation of lead risks usually centers on children, there are situations in which adults may be exposed to elevated levels of lead in the environment where children are unlikely to be exposed. For example, adult blood lead levels should be assessed for commercial and industrial exposure scenarios. Of adults, the population of most concern is women of child-bearing age because of the transfer of lead from pregnant mother to fetus.

F.3.1 The Adult Model

The USEPA (1996) recommends estimation of adult blood lead levels using an approach based on an adult blood lead model developed by Bowers, et al 1994). This Model is similar to the IEUBK Model used for children in that it also contains the same four components (exposure, absorption, biokinetics, and uncertainty or variability). However, the biokinetic portion of this Model consists of a single biokinetic slope factor that relates lead uptake to blood lead. Additionally, rather than assessing lead exposures to adults from all sources, the Model focuses on exposure to soil and dust and uses a “baseline” blood lead level to represent the contributions of all other sources of lead, including past exposures. Equations for the adult model are given in Table F-1, together with an example calculation. The adult model is also used to estimate acceptable soil lead levels by running the model “backwards” to calculate a soil lead level that corresponds to a target blood lead level.

F.3.2 Target Blood Lead Levels for Adults

The USEPA recommends a target blood lead level for women of child-bearing age to ensure that the fetus has no more than a 5% probability of a blood lead level exceeding 10 $\mu\text{g}/\text{dL}$ (USEPA, 1996). The ratio of fetal to maternal blood lead levels is about 0.9. Therefore, a woman of child-bearing age should have no more than a 5% probability of having a blood lead level exceeding 11.1 $\mu\text{g}/\text{dL}$ (i.e., 10 divided by 0.9).

F.3.3 Exposure Parameters for Adults

The USEPA (1996) recommends values for all parameters in the adult model. Values recommended here are based on the best available information; however, some values differ from USEPA’s recommendations.

- *Soil lead concentrations* - Soil samples should be averaged over the exposure area and the arithmetic mean used as input to the adult model.
- *Dust lead concentrations* – Interior dust lead levels are generally not available in occupational settings, but can be estimated using a soil-to-dust transfer coefficient. If dust levels are available, they can be averaged and used directly in the model. If they are not available, a soil-to-dust transfer coefficient in the range of 0.3 to 0.7 can be used. A value in this range should be chosen based on the likelihood that there are other sources of lead besides site soils that contribute to interior dust lead levels, e.g., lead paint.
- *Exposure frequency* – Exposure frequency should be based on site-specific information. In the absence of site-specific information, a value of 250 days/year can be used, reflecting the assumption that individuals typically work 5 days a week for 50 weeks of the year.

- *Soil/dust ingestion rates* – Little information is available concerning the amount of soil and dust that adults ingest, although it is likely that adults ingest less than do children. Bowers and Cohen (1997) recommend 0.02 g/day as an average adult ingestion rate, while USEPA recommends 0.05 g/day, both of which assume an eight hour/day exposure. A value in this range may be used. The soil/dust ingestion rate should be considered the average rate for an adult during their waking hours, and thus this rate may be reduced to reflect the fraction of time that the adult spends on the site, if appropriate. Activities which involve heavy dust generation, e.g. heavy construction, may warrant a higher ingestion rate. See USEPA 1991b, Section 3.
- *Soil/dust absorption* – The fraction of lead in ingested soil and dust absorbed into the circulatory system is the product of two values: the amount of soluble lead absorbed and the ratio of this fraction for lead in soil and dust to the corresponding fraction for soluble lead. Data presented in James et al (1985) indicate that absorption of soluble lead can range from approximately 60% after a prolonged period of fasting to approximately 4% at mealtime. A time-average absorption of soluble lead that takes account of meal times and times between meals will fall close to the low end of this range. Current estimates of the time-averaged fraction of soluble lead absorbed for adults ranges from 8% (O’Flaherty (1993) to 20%). (EPA (Marjo to complete cite)). Note: Current literature should be consulted for updates of this range. This absorption value can be modified on the basis of site-specific bioavailability information, such as would result from an in vivo or in vitro test (Ruby, et al. 1996).
- *Baseline blood lead level* – USEPA recommends a baseline blood lead level for women of child-bearing age of 1.7 to 2.2 µg/dL. This blood lead level represents individuals who are not exposed to substantial sources of lead beyond background, and the range is indicative of diverse conditions across the United States (U.S.). The National Health and Nutrition Evaluation Study (NHANES) III data set (Pirkle et al 1994) indicates that the geometric mean blood lead level for women of child-bearing age (defined as age 20 to 40) in the southern region of the US is 1.54 µg/dL.
- *Biokinetic slope factor* – USEPA recommends a biokinetic slope factor of 0.4 µg/dL blood lead per µg/day lead intake.
- *Geometric standard deviation* – The individual GSD is the parameter used to estimate the probability that an individual’s blood lead level exceeds 10 µg/dL (or another target blood lead level) given the Model-predicted geometric mean blood lead for the designated environmental exposures. The NHANES III data set indicates that for women of child-bearing age (defined as age 20 to 40) in the southern region of the US, the GSD is 1.88. However, this GSD reflects blood lead variation due to differences in soil and dust lead concentrations across the broad geographical region of the south. At individual sites, heterogeneity in blood lead levels can be expected to be substantially less than across broad geographic regions because the range of exposures will not be as broad. To account for the tendency for site-specific GSDs to be smaller than GSDs for

populations living over a broad geographic region, we recommend that a value less than 1.88, such as 1.7, be used. This value can be chosen from a range of 1.6 to 1.8. A value of 1.7 is used as an example to illustrate calculation of a cleanup level in Table F-1.

Table F-1: Equation and Parameters for the Adult Blood Lead Model

$$PbB_{adult} = PbB_{baseline} + BSF \times I \times A \times \frac{EF}{365} \times (f_s C_s + f_d C_d)$$

$$PbB_{95th} = PbB_{adult} \times R_{fetal/adult} \times GSD^{1.645}$$

| Parameter | Description | Value |
|--------------------------|--|--|
| PbB _{95th} | 95 th percentile fetal blood lead level (substitute target blood lead level when calculating an acceptable soil lead level) | 95 th percentile is calculated, target for children is 10 µg/dL |
| PbB _{adult} | central estimate of adult blood lead level | calculated |
| PbB _{baseline} | baseline adult blood lead level | 1.54 (women of child-bearing age) |
| R _{fetal/adult} | ratio of fetus to maternal blood lead | 0.9 |
| GSD | geometric standard deviation of adult blood lead | 1.7 |
| BSF | biokinetic slope factor | 0.4 µg/dL per µg/day |
| I | soil/dust ingestion rate | 0.05 g/day |
| A | absorption of lead from soil/dust | 0.048 |
| EF | exposure frequency | 250 days |
| f _s | fraction of soil ingested | based on site usage |
| C _s | concentration of lead in soil | site-specific, or calculated when target blood lead level is specified |
| f _d | fraction of dust ingested, = 1-f _s | based on site usage |
| C _d | concentration of lead in dust | site-specific, or calculated from a soil-to-dust transfer coefficient |

Example Calculation of a Soil Lead Cleanup Level

Set $f_s = f_d = 0.5$, $C_d = (0.3)(C_s)$

$10 = (PbB_{adult})(0.9)(1.7^{1.645})$; $PbB_{adult} = 4.64 \text{ } \mu\text{g} / \text{dL}$

$4.64 = 1.54 + (0.4)(0.05)(0.048)\left(\frac{250}{365}\right)((0.5)(C_s) + (0.5)(0.3)(C_s))$; $C_s = 7253 \text{ mg} / \text{kg}$

F.4 References

- Bowers, T.S., B.D. Beck, and H.S. Karam. 1994. Assessing The Relationship Between Environmental Lead Concentrations And Adult Blood Lead Levels. *Risk Analysis*. 14:183-189.
- Bowers, T.S. and J.T. Cohen. 1997. Blood Lead Slope Factor Model For Adults: Comparisons Of Observations And Predictions. Proc. Vol. Of 1996 EPA Lead Model Validation Workshop, in press.
- James, H.M., M.E. Hilburn, J.A. Blair. 1985. Effects Of Meals And Meal Times On Uptake Of Lead From The Gastrointestinal Tract In Humans. *Human Toxicol.* 4:401-407.
- O'Flaherty, E.J. 1993. Physiologically Based Models For Bone-Seeking Elements. IV. Kinetics Of Lead Disposition In Humans. *Toxicol. Appl. Pharmacol.* 118:16-29.
- Pirkle, J.L., D.J. Brody, E.W. Gunter, R.A. Kramer, D.C. Paschal, K.M. Flegal, and T.D. Matte. 1994. The Decline In Blood Lead Levels In The United States: The National Health and Nutrition Examination Surveys (NHANES). *JAMA* 272:284-291.
- Ruby, M.W., A. Davis, R. Schoof, S. Eberle, and C.M. Sellstone. 1996. Estimation Of Lead And Arsenic Bioavailability Using A Physiologically Based Extraction Test. *Environ. Sci. Technol.* 30:422-430.
- US Centers for Disease Control (CDC). 1991. Preventing Lead Poisoning In Young Children. US Dept. of Health and Human Services, October.
- US Environmental Protection Agency (USEPA). 1994a. Revised Interim Soil Lead Guidance For CERCLA Sites And RCRA Corrective Action Facilities, Office of Solid Waste and Emergency Response, Directive 9355.4-1, July 14.
- US Environmental Protection Agency (USEPA). 1994b. Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children. Office of Emergency and Remedial Response. Pub. No. 9285.7-15-1, EPA/540/R-93/081, February.
- US Environmental Protection Agency (USEPA). 1995a. Site-specific Integrated Exposure Uptake Biokinetic Modeling, Kennecott Baseline Risk Assessment prepared by Life Systems, Inc. June.
- US Environmental Protection Agency (USEPA). 1995b. Evaluation Of The Risk From Lead And Arsenic, Sandy Smelter Site – Sandy, Utah – December.
- US Environmental Protection Agency (USEPA). 1996. Recommendations Of The Technical Review Workgroup For Lead For An Interim Approach To Assessing Risks Associated With Adult Exposures To Lead In Soil. December.

APPENDIX F: ACRONYM LIST

| | |
|-----------|--|
| g/dL | micrograms per deciliter |
| CDC | Center for Disease Control |
| g/day | grams per day |
| GSD | geometric standard deviation |
| IEUBK | Integrated Exposure Uptake Biokinetic Model |
| mg/day | milligrams per day |
| mg/kg/day | milligrams per kilogram per day |
| NHANES | National Health and Nutrition Evaluation Study |
| RfD | reference dose |
| USEPA | United States Environmental Protection Agency |
| VRRRA | Voluntary Remediation and Redevelopment Act |

APPENDIX G: REFERENCES TO BENCHMARK SCREENING LEVELS

Eisler, R. 1988. Lead Hazards To Fish, Wildlife, And Invertebrates: A Synoptic Review. US Fish and Wildlife Service Patuxent Wildlife Research Center, Laurel, MD. US Department of the Interior. Biological Report 85 (1.14), Contaminant Hazard review Rep. No. 14.

Grossman, G.D. 1990. Assemblage Stability In Stream Fishes: A Review. Environmental Management. 14:661-667.

Jones, D.S., R.N. Hull, and G.W. Suter II. 1996. Toxicological Benchmarks For Screening Contaminants Of Potential Concern For Effects On Sediment-Associated Biota: 1996 Revision. Oak Ridge National Laboratories, Oak Ridge, TN. 34 p., ES/ER/TM-95/R2.

Ontario Ministry of the Environment. 1993. Guidelines For The Protection And Management Of Aquatic Sediment Quality In Ontario. Water Resources Branch. ISBN 0-7729-9248-7.

Pascoe, G.A. 1994. Characterization Of Ecological Risks At The Milltown Reservoir-Clark Fork River Sediments Superfund Site, Montana. Environmental Toxicology and Chemistry. 13:1-16.

Sample, B.E., D.M. Opresko, and G.W. Suter. 1996. Toxicological Benchmarks For Wildlife: 1996 Revision. Oak Ridge National Laboratories, Oak Ridge, TN. 227 p., ES/ER/TM-86/R3.

Suter, G.W. 1995. Toxicological Benchmarks For Screening Contaminants Of Potential Concern For Effects On Freshwater Biota. Environmental Toxicology and Chemistry. 15:1232-1241.

Suter, G.W. II, and C.L. Tsao. 1996. Toxicological Benchmarks For Screening Of Potential Contaminants Of Concern For Effects On Aquatic Biota On Oak Ridge Reservation: 1996 Revision. Oak Ridge National Laboratories, Oak Ridge, TN. 104 p., ES/ER/TM-96/R2.

Stephan, C.E., et al. 1985. Guidelines For Deriving Numeric National Water Quality Criteria For The Protection Of Aquatic Organisms And Their Uses. PB85-227049. US Environmental Protection Agency, Washington, DC.

US Environmental Protection Agency. 1996. Ecotox Thresholds. Intermittent Bulletin; Volume 3, Number 2. EPA/540/F-95/038.

Will, M.E. and G.W. Suter. 1995. Toxicological Benchmarks For Screening Potential Contaminants Of Concern For Effects On Terrestrial Plants: 1995 Revision. Oak Ridge National Laboratories, Oak Ridge, TN. 123 p., ES/ER/TM-85/R2.

Will, M.E. and G.W. Suter. 1995. Toxicological Benchmarks For Potential Contaminants Of Concern For Effects On Soil And Litter Invertebrates And Heterotrophic Process. Oak Ridge National Laboratories, Oak Ridge, TN. 155 p., ES/ER/TM-126/R1.

If none of the above references fit the screening needs of a particular project, LRS can consult EPA's Ecological Effects Test Guidelines – Series 850, a guidance on toxicity testing to

develop benchmark screening levels. The most recent version with supporting documentation can be found on the internet at <http://www.hsrp.ornl.gov/ecorisk/ecorisk.html>.

APPENDIX H: SITE SPECIFIC RISK ASSESSMENT

H.1 Risk Assessment Equations and Parameters

This section briefly outlines the standard equations used to calculate intake of contaminants in various media. The equations in this section can be used for so-called “point-estimate” risk assessments, where each parameter in the equation is replaced with a single value (the “point estimate”), and the equation yields a single estimate of risk. “Intake” is expressed in terms of mg/kg-day. In the case of cancer risks, this value reflects a lifetime average. The product of the intake value and the cancer slope factor for a contaminant is the lifetime risk of cancer. In the case of noncancer risks, the intake value reflects the average over a “chronic” exposure period, typically defined to be seven years or longer. The ratio of the intake value to a contaminant’s reference dose is the hazard index. Exposures corresponding to a hazard index less than 1.0 are considered to be without appreciable risk, even among susceptible sub-populations. If the hazard index exceeds 1.0, exposure may be sufficient to cause adverse health effects.

The generic equation for intake, denoted I (as outlined by USEPA, 1989, Exhibit 6-9), is

$$I = C \times \frac{CR \times EFD}{BW} \times \frac{1}{AT}$$

Where:

- C = Chemical concentration; the average concentration contacted over the exposure period (*e.g.*, mg/L water).
- CR = Contact rate; the amount of contaminated medium contacted per unit of time or event (*e.g.*, L/day).
- EFD = Exposure frequency and duration; describes how often and for how long exposure occurs. Often calculated using two terms (EF, or “exposure frequency,” reported in days/year, and ED, or “exposure duration,” reported in years).
- BW = Body weight; the average body weight over the exposure period (kg).
- AT = Averaging time; period over which exposure is averaged (days).

The quantity C is a chemical-specific parameter. The quantities CR , EFD , and BW are parameters that describe the exposed population. The parameter AT is a fixed quantity whose value depends on whether risk being evaluated is carcinogenic or non-carcinogenic.

The remainder of this section details intake equations for the following exposure scenarios:

- Ingestion of chemicals in water
- Dermal contact with chemicals in water

- Ingestion of chemicals in soil
- Dermal contact with chemicals in soil
- Inhalation of airborne chemicals
- Ingestion of fish

Additional exposure pathways may be considered in a site-specific risk assessment, for example, see USEPA (1997).

H.1.1 Ingestion of Chemicals in Water

The United States Environmental Protection Agency (USEPA) (1989) quantifies intake of chemicals in water via ingestion in Exhibit 6-11 of the Agency's document. Specifically, intake is calculated as:

$$I = \frac{CW \times IRW \times EF \times ED}{BW \times AT}$$

Where:

| | | |
|-----|---|--|
| CW | = | Chemical concentration in water (mg/L) |
| IRW | = | Ingestion rate of water (liters/day) |
| EF | = | Exposure frequency (days/year) |
| ED | = | Exposure duration (years) |
| BW | = | Body weight (kg) |
| AT | = | Averaging time (days) |

This equation is applicable to ingestion of any source of water, including tap water, or, for example, surface water ingested while swimming.

H.1.2 Dermal Contact with Chemicals in Water

USEPA (1989) quantifies intake of chemicals via dermal contact with water in Exhibit 6-13 of the Agency's document. Specifically, intake is calculated as:

$$I = \frac{CW \times SA \times PC \times ET \times EF \times ED \times CF}{BW \times AT}$$

Where:

| | | |
|----|---|--|
| CW | = | Chemical concentration in water (mg/L) |
| SA | = | Skin surface area available for contact (cm ²) |
| PC | = | Chemical-specific dermal permeability constant (cm/hr) |
| ET | = | Exposure time (hours/day) |
| EF | = | Exposure frequency (days/year) |
| ED | = | Exposure duration (years) |
| CF | = | Volumetric conversion factor (1 L/1000 cm ³) |

BW = Body weight (kg)
AT = Averaging time (days)

This equation is applicable to dermal contact of any source of water, including tap water, or, for example, surface water in which an individual comes into contact while swimming.

H.1.3 Ingestion of Chemicals in Soil

USEPA (1989) quantifies intake of chemicals via ingestion of soil in Exhibit 6-14 of the Agency's document. Specifically, intake is calculated as:

$$I = \frac{CS \times IRS \times CF \times FI \times EF \times ED}{BW \times AT}$$

Where:

CS = Chemical concentration in soil (mg/kg)
IRS = Ingestion rate for soil (mg soil/day)
CF = Conversion factor (10^{-6} kg/mg)
FI = Fraction ingested from contaminated source (unitless)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (days)

H.1.4 Dermal Contact With Chemicals in Soil

USEPA (1989) quantifies intake of chemicals via dermal contact with soil in Exhibit 6-15 of the Agency's document. Specifically, intake is calculated as:

$$I = \frac{CS \times CF \times SA \times AF \times ABS \times ED \times EF}{BW \times AT}$$

Where:

CS = Chemical concentration in soil (mg/kg)
CF = Conversion factor (10^{-6} kg/mg)
SA = Skin surface area available for contact (cm^2 / event)
AF = Soil to skin adherence factor (mg/cm^2)
ABS = Chemical specific absorption factor (unitless)
EF = Exposure frequency (events/year)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (days)

H.1.5 Inhalation of Airborne Chemicals

USEPA (1989) quantifies intake of chemicals via inhalation in Exhibit 6-16 of the Agency's document. Specifically, intake is calculated as:

$$I = \frac{CA \times IRA \times ET \times EF \times ED}{BW \times AT}$$

Where:

| | | |
|-----|---|---|
| CA | = | Contaminant concentration in air (mg/m ³) |
| IRF | = | Average ingestion rate for fish (g/day) |
| CF | = | Conversion factor (10 ⁻³ kg/mg) |
| FI | = | Fraction ingested from contaminated source (unitless) |
| EF | = | Exposure frequency (days/year) |
| ED | = | Exposure duration (years) |
| BW | = | Body weight (kg) |
| AT | = | Averaging time (days) |

H.1.6 Ingestion of Fish

Intake of contaminants from fish is calculated as (USEPA, 1989):

$$I = \frac{CF \times IRF \times CF \times FI \times EF \times ED}{BW \times AT}$$

Where:

| | | |
|-----|---|---|
| CS | = | Chemical concentration in fish (mg/kg) |
| IRF | = | Average ingestion rate for fish (g/day) |
| CF | = | Conversion factor (10 ⁻³ kg/mg) |
| FI | = | Fraction ingested from contaminated source (unitless) |
| EF | = | Exposure frequency (days/year) |
| ED | = | Exposure duration (years) |
| BW | = | Body weight (kg) |
| AT | = | Averaging time (days) |

It should be noted that the USEPA recommended default value for fish ingestion rate (USEPA, 1997) is the daily intake averaged over a year (not the average amount of fish consumed per meal), and thus should be used in conjunction with an exposure frequency of 365 days/year.

H.1.7 Additional Pathways

Additional pathways should be considered in the risk assessment where appropriate for the site. These pathways include:

- Dermal and inhalation exposure to an adult while showering. (USEPA recommends the use of the Foster and Chrostowski Model for determining inhalation exposure in the shower.)

- Dermal exposure to a child while bathing
- Inhalation of volatiles and particulates from soil
- Dermal contact with water while swimming
- Dermal contact with surface water while wading
- Ingestion of homegrown fruits and vegetables
- Soil contact by a construction worker (ingestion, inhalation, and dermal contact)

It is recommended that the risk assessor consult the USEPA Exposure Factors Handbook (USEPA, 1997) to determine the appropriate input parameters for a given exposure pathway.

H.2 Conservative Parameter Values for a Point Estimate Risk Assessment

One approach to the selection of conservative point estimate values is to use values that USEPA has identified as corresponding to the “Reasonable Maximum Exposure,” or “RME.” USEPA defines the RME “as the highest exposure that is reasonably expected to occur at a site (USEPA, 1989, p 6-4). Although the USEPA notes that estimation of the RME may involve professional judgment, the Agency provides values for some exposure parameters that, according to USEPA, are appropriate for this calculation. USEPA also cautions that, “The specific values identified should be regarded as general recommendations, and could change based on site-specific information...” (USEPA, 1989, p 6-5). Table H-1 summarizes the parameter values recommended by USEPA.

Table H-1: Parameter Values Recommended by USEPA

| Parameter | Water Ingestion | Water: Dermal Contact | Soil Ingestion | Soil: Dermal Contact | Inhalation |
|-----------|--|--|--|---------------------------|--|
| IR | 2.32 L/day ^a 1.4 L/day ^b 1.3 L/day ^c 0.74 L/day ^d | | 200 mg/day ^{fd} 100 mg/day ^{ge} | | 30 m ³ / day ^f 20 m ³ / day ^{hg} 10 m ³ / day ⁱ 0.6 m ³ / hr ^{jh} |
| AF ET | 50 ml/hour ^{ec} | 12 minutes ^{ki} 8 minutes ^{lj} 20 minutes ^m | | 1.45 mg / cm ² | |

Parameters that do not differ by Exposure Pathway

| | |
|----|--|
| EF | 350 days/yr or pathway dependent, depending on the nature of the exposure scenario |
| ED | 70 years (lifetime exposure) 30 years (national 90 th percentile duration for living at one location) 9 years (national median duration for living at one location) |
| BW | 70 kg (adult) 16 kg (child aged 1 to 6 years) Age-specific data |
| SA | See table H-2, different combination of arms, hands, and legs may be appropriate for different exposure pathways |
| FI | Fraction of ingestion soil from contaminated site: Exposure and pathway scenario specific |

Parameters for Additional Pathways

| | |
|-------------------------------------|---|
| IR _(fish) | 8 g/day (Mean) 25 g/day (95 th Percentile) (Fish consumption by Recreational Freshwater Anglers (USEPA, 1997)) |
| ED _(swimming) | 60 mins/event (50 th percentile) (USEPA, 1997) 180 mins/event (90 th percentile) (USEPA, 1997) |
| EF _(swimming) | 1 event/month (average, adult) (USEPA, 1997) |
| ED _(construction worker) | 25 years or site-specific |
| EF _(construction worker) | 250 days/yr or site-specific |

Source: U.S. EPA, 1989, Exhibits 6-11, 6-12, 6-13, 6-14, 6-15, 6-16, except where otherwise noted.

Notes:

- a. Adult, 90th percentile (USEPA, 1997, Table 3-30)
- b. Adult, average (USEPA, 1997, Table 3-30)
- c. Child, 1-10 yr, 90th percentile (USEPA, 1997, Table 3-30)
- d. Child, 1-10 yr, average (USEPA, 1997, Table 3-30)
- b. Adult, average
- ec. Ingestion rate while swimming
- fd. Children, 1-6 years old, upper bound value
- ge. Adult upper bound value
- f. Adult upper bound value
- g. Adult, average
- h. Adult, average
- i. Child, average
- jh. Inhalation rate while showering
- ki. Showering, 90th percentile duration
- lj. Showering, average duration. (USEPA, 1997).
- m. Bath duration, recommended value for all ages (USEPA 1997 Table 15-176)

USEPA recommends that when calculating the RME value, the 90th or 95th percentile values specified in Table H-1 should be used. For the purpose of conducting a Tier I point-estimate screening assessment, it is recommended that this guidance be followed.

Body surface area values for the 50th percentile individual appear in Table H-2 (USEPA, 1989, Exhibit 6-15). Based on USEPA's guidance, it is recommended that these values be used when conducting a Tier I point-estimate screening assessment.

Table H-2: 50th Percentile Body Surface Area

| Total Body Surface Area (m²): 50th Percentile | | |
|--|-------------|---------------|
| Age (Years) | Male | Female |
| 3 < 6 | 0.728 | 0.711 |
| 6 < 9 | 0.931 | 0.919 |
| 9 < 12 | 1.16 | 1.16 |
| 12 < 15 | 1.49 | 1.48 |
| 15 < 18 | 1.75 | 1.60 |
| Adult | 1.94 | 1.69 |

| Body-Part Specific Surface Area (m²): 50th percentile | | | |
|--|-------------|--------------|-------------|
| Age (Years) | Arms | Hands | Legs |
| 3 < 4 | 0.096 | 0.040 | 0.18 |
| 6 < 7 | 0.11 | 0.041 | 0.24 |
| 9 < 10 | 0.13 | 0.057 | 0.31 |
| Adult | 0.23 | 0.082 | 0.55 |

In addition to the population-specific parameters described in the Tables H1 and H2, there are several chemical-specific values that must be quantified. In general, use of the average or median value for each of these parameters is recommended.

One parameter – the chemical concentration – is both chemical-specific and site-specific. Following USEPA’s guidance, it is recommended that the 95 percent upper confidence limit on the arithmetic mean be used as the chemical concentration. Calculation of the upper confidence limit on the mean depends on whether the observations follow a normal distribution or a lognormal distribution. USEPA (1992) has disseminated guidance outlining calculation of the 95% upper confidence limit on the mean in both of these cases.

H.3 References

US Environmental Protection Agency (USEPA). 1989. *Risk Assessment Guidance for Superfund. Volume II: Environmental Evaluation Manual*. Office of Emergency and Remedial Response (Washington, DC). EPA-540/1-89-001. March.

US Environmental Protection Agency (USEPA). 1992. Office of Solid Waste and Emergency Response (Washington, DC). *Supplemental Guidance to RAGS: Calculating the Concentration Term*. OSWER Publication 9285.7-081; NTIS PB92-963373. 8p. May 1992.

US Environmental Protection Agency (USEPA). 1997. Office of Health and Environmental Assessment, Exposure Assessment Group. *Exposure Factors Handbook – Volumes I, II and III*. EPA/600/P-97/002FA,B,C.

APPENDIX H: ACRONYM LIST

| | |
|-----------------|-------------------------------------|
| ABS | chemical specific absorption factor |
| AF | soil to skin adherence factor |
| AI | annual intake |
| AT | averaging time |
| BW | body weight |
| C | chemical concentration |
| CA | contaminant concentration in air |
| CF | volumetric conversion factor |
| cm ² | square centimeters |
| cm ³ | cubic centimeters |
| cm/hr | centimeters per hour |
| CR | contact rate |

| | |
|---------------------|--|
| CRG | cleanup remediation goal |
| CS | chemical concentration in soil |
| CSF | cancer slope factor |
| CW | chemical concentration in water |
| DIR | daily intake rate |
| ED | exposure duration |
| EF | exposure frequency |
| EFD | exposure frequency and duration |
| ET | exposure time |
| ET _{shoer} | exposure time for bathing |
| FI | fraction ingested from contaminated source |
| GM | geometric mean |
| GSD | geometric standard deviation |
| hr | hour |
| IR _{Air} | inhalation rate for air |
| IR _{Soil} | ingestion rate for soil |
| IR _{Water} | ingestion rate for water |
| kg | kilogram |
| L | liter |
| m ² | square meter |
| m ³ | cubic meter |
| mg | milligram |
| ml | milliliter |
| PC | chemical-specific dermal permeability constant |
| R | ratio |
| RfD | reference dose |
| RME | Reasonable Maximum Exposure |
| SA | skin surface area |
| SES | socio-economic status |
| SPLP | Synthetic Precipitation Leaching Procedure |
| USEPA | United States Environmental Protection Agency |

APPENDIX I: PROBABILISTIC METHODOLOGIES IN RISK ASSESSMENT

I.1 Tiered Approach to the Use of Probabilistic Risk Assessment

The risk equations provided in Appendix H can also be used for probabilistic risk assessments as well as “point-estimate” risk assessments. In the latter case, each value in the equation is replaced with a single value (the “point estimate”), and the equation yields a single estimate of risk. A probabilistic risk assessment characterizes each parameter by a probability distribution, which places positive weight on each plausible value for that parameter. The risk estimate generated by a probabilistic risk assessment is also a probability distribution, describing the range of plausible risk estimates for the population of interest, as well as the relative frequency for each risk value. This information can inform the risk manager as to the fraction of the population subject to risks exceeding a specified threshold, as well as the degree of certainty associated with these estimates. Section I.2 discusses the nature and use of the risk values generated by a probabilistic analysis in greater detail.

Although probabilistic risk assessment provides more information than a point-estimate risk assessment, it requires more information to execute. It is also far more computationally intensive, typically requiring specialized (though easily available) software, and, depending on the degree of sophistication, the development of assessment-specific computer programs. It is therefore desirable to develop point-estimate risk assessments as a first step in such a way that, in some cases, they make the execution of a probabilistic assessment unnecessary.

Recall that a probabilistic risk assessment can quantify the fraction of individuals exceeding a specified risk threshold, along with the degree of certainty associated with that estimate. Such information is unnecessary if you can determine from a point estimate risk assessment that it is likely that only a very small fraction of the members in a population will incur health risks exceeding some threshold of interest. In order to reach such a conclusion using a point estimate risk assessment, it is necessary to assign a sufficient number of parameters “conservative” (meaning health-protective) values. If the point estimate risk calculated on the basis of these assumptions is below the risk threshold of interest, then the risk manager can conclude that, with a sufficient level of confidence, members of the population, will not, for the most part, be subject to unacceptable risks. In this situation, it is unnecessary to conduct a probabilistic assessment.

However, if the conservative point estimate assessment yields an unacceptable risk value, it cannot be concluded that populations’ risks are, in reality, unacceptable. In this case, the calculated risk estimate may be too high because either the risks really are unacceptable, or because the use of conservative parameter assumptions have inflated what would have been considered an acceptable risk. Since it is impossible to tell which of these explanations is true, it is necessary to conduct a probabilistic assessment to generate more detailed information about the nature of the risks to which the population is subjected.

In summary, the tiered approach calls for the risk assessor to first conduct a point estimate assessment using conservative values for selected parameters, as outlined in Appendix H. If that assessment yields an acceptable risk, the risk manager can conclude that risks are acceptable. No further computations are necessary. However, if the point estimate is

unacceptable, the risk assessor may proceed to the second tier, which involves execution of a probabilistic assessment.

I.2 Probabilistic Risk Assessment

This section describes Monte Carlo simulation, a technique used to conduct Tier II probabilistic risk assessments. Section I.2.1 discusses the concepts of “variability” and “uncertainty.” These concepts are key to development of valid parameter distributions for a Monte Carlo simulation, and to the proper interpretation of the risk estimates from such a simulation. Section I.2.2 then describes how Monte Carlo simulation works and how to interpret the results of a Monte Carlo simulation.

I.2.1 Uncertainty and Variability

Monte Carlo risk assessment allows parameters to be assigned more than one value. However, the possibility of assigning more than one value to a parameter can reflect either or both of two phenomena, referred to as “uncertainty” and “variability.” At an USEPA workshop on the subject of Monte Carlo analysis conducted during May of 1996 (USEPA, 1996), variability and uncertainty were defined as follows:

- Variability represents the natural heterogeneity or diversity in a well-characterized population. Variability:
 - Usually not reducible through further measurement or study.
 - Is a property of the population.
 - Reflects physical, chemical, and biological phenomena
- Uncertainty represents ignorance (or lack of perfect knowledge) about poorly characterized phenomena or models. Uncertainty:
 - May sometimes be reducible through further measurement or study.
 - Is a property of the analyst.
 - Reflects limitations of our understanding of the physical world.

Body weight is an example of a quantity that varies across members of the population but for which uncertainty is not substantial. A probability distribution for body weights places weight on each value that is proportional to the frequency with which is thought to occur in the population.

On the other hand, the average concentration of a contaminant in an exposure unit does not vary (there is one true value that holds for all members of the population), but which can be uncertain due to finite sampling of that quantity. In this case, the distribution characterizing this

quantity places weight on alternative values that are proportional to each alternative's plausibility. The weight on all values between two endpoints divided by the weight placed on all values is equal to the probability that the true value lies between the specified endpoints.

Finally, there are quantities that are both uncertain and variable. This case is the most complex. For example, it is reasonable to suspect that the soil ingestion rate varies among children. It is also true that this quantity is very difficult to measure, making it uncertain. Characterization of both uncertainty and variability requires specification of multiple probability distributions, each representing one possible characterization of variability in the population. The fact that there are multiple distributions is a manifestation of the quantity's uncertainty. For example, consider the 95th percentile value for each of the alternative distributions. In general, these values differ. Together, they represent the uncertainty distribution for the 95th percentile for the parameter. Likewise, the collection of alternative arithmetic mean values represents the uncertainty distribution for the arithmetic mean for the parameter. It is important to note that a one-dimensional Monte Carlo analysis can characterize *either* uncertainty or variability. If the input probability distribution functions (PDFs) contain both uncertainty and variability, then the result can not be interpreted as either. For this reason, uncertainty and variability should not be combined in a one-dimensional Monte Carlo analysis.

I.2.2 Overview of Monte Carlo Technique

Conceptually, Monte Carlo simulation is straight-forward. Suppose one wishes to characterize a distribution of annual intake values (mg/year) for a population given that annual intake (AI) for a specific individual is the product of two quantities: EF (days/year), and the daily intake rate (DIR) when the individual is exposed to contaminant of concern (mg/day). Suppose that the distribution for EF is uniform between 225 days/year and 275 days per year, and that the DIR is lognormally distributed with a geometric mean of 30 mg/day and a geometric standard deviation of 2.0. Monte Carlo simulation characterizes the distribution for the annual intake by first selecting a value at random from the distribution characterizing EF, then selecting a value at random from the distribution characterizing DIR, and multiplying them together. This product is a single estimated value of AI. The simulation then repeats this process, generating a second value for AI. After repeating this process many times (*e.g.*, 1,000 or 10,000 times), the simulation has generated a large number of values for AI. Monte Carlo simulation has the property of producing a set of values that approximates the "true" distribution for the calculated quantity (in this case, AI), assuming that the distributions chosen for the parameters are valid, and that the relationship between the parameters and the calculated quantity is valid.

The simulation described in the preceding paragraph is referred to as a "one-stage" simulation. A one-stage simulation is sufficient when the risk assessor can assume that either the parameter distributions all represent variability, or all represent uncertainty. In the former case (all the parameter distributions represent variability), the distribution of values produced by the Monte Carlo simulation represents variability. For example, if a simulation generates 1,000 exposure values, and these values are ranked from smallest to largest, then the 950th value is an estimate of the 95th percentile of the population exposure distribution. The 500th value is an estimate of the 50th percentile of the population exposure distribution. In the latter case (all the parameter distributions represent uncertainty), the distribution of values produced by the Monte Carlo simulation represents uncertainty. For example, the 950th value is an estimate of the upper

95th percentile on the average exposure. That is, there is approximately a 95% probability that the true average exposure is less than this 95th percentile value.

If there are parameters that have important variability and uncertainty components, then a one-stage simulation may be inadequate. For example, failure to distinguish between uncertainty and variability can lead to the overstatement of variability in the population. Cohen *et al.* (1996) provides a simple example to illustrate this point. In this example, it is supposed that risk attributable to ingestion of a waterborne carcinogen is equal to the product of the chemical's cancer slope factor (CSF) and the water intake rates (IRW). The value of IRW varies from individual to individual, but is assumed to be known precisely. The CSF value, on the other hand, is assumed to be the same for all individuals, but its true value is not well-known. A simulation that fails to distinguish between the variability in IRW values and the uncertainty of the CSF value would select a random value from each, and multiply these values together to generate an estimate of risk. However, the distribution generated by repeating this procedure many times is "wider" than the distribution produced by selecting any one of the possible CSF values, holding it constant, and multiplying it by ingestion rate values drawn from the IRW distribution. Figure 1a in Cohen *et al.* (1996) illustrates several possible distributions. One such distribution is the "true" distribution (although, it is not known which is true). Moreover, any one of these distributions is narrower than the distribution generated by a one-stage simulation.

"Two-stage" simulation, which is described in detail in Cohen *et al.* (1996), properly addresses uncertainty and variability in cases where both are important. The two-stage simulation has two iterative loops, referred to as the "inner loop" and the "outer loop." The inner loop is, effectively, a one-stage simulation that is restricted to drawing values from probability distributions representing variability. The result generated by the inner loop is a single distribution of risks representing the range of risks predicted for a population given a single set of uncertain parameter values (*e.g.*, the fixed value for the geometric mean and geometric standard deviation of the soil ingestion rate for children). After execution of the inner loop, the outer loop selects a new set of uncertain parameter values from the appropriate distributions, and the inner loop is executed again. This process is repeated, yielding a large number of population risk distributions, each (in general) corresponding to a different set of uncertain parameter values. Alternatively, the differences between the variability distributions generated by each iteration of the outer loop can be viewed as the influence of uncertainty on the estimated distribution of risks in the population.

At this time, we are unaware of software that implements two-stage simulations without specialized programming⁵. Moreover, execution of a two-stage simulation can be computationally intensive since the total number of iterations is equal to the product of the number of iterations per execution of the inner loop (typically on the order of 1,000 to 10,000) and the number of executions of the outer loop (typically on the order of 1,000). Hence, the total number of iterations can range from 1,000,000 to 10,000,000.

The large amount of information produced by a two-stage simulation presents a challenge to the interpretation of the results. We recommend that after each iteration of the outer loop, critical sample statistics be saved, along with the uncertain parameter values used for that inner

⁵ Cohen *et al.* (1996) describe the implementation of a two-stage simulation for a Superfund risk assessment. The simulation was implemented using the SAS statistical programming language (SAS, 1997).

loop simulation. For example, after one execution of the inner loop, uncertain parameter values saved might include the geometric mean and geometric standard deviation of the soil ingestion distribution, while sample statistics for the calculated value that are recorded might include the population average risk, the population median risk, and the population 95th percentile risk. Table I-1 illustrates the saved information for each execution of the outer loop.

Table I-1: Illustrative Results From a Two-Stage Simulation

| Outer Loop Iteration | Uncertain Parameter Values: Subscript i,j refers to the value of the i th parameter for the j th simulation | | | | Simulation Results | | |
|----------------------|--|---------------------|---------------------|---------------------|-------------------------|------------------------|--------------------------|
| | 1 | U _{1,1} | U _{2,1} | U _{3,1} | U _{4,1} | Average ₁ | Median ₁ |
| 2 | U _{1,2} | U _{2,2} | U _{3,2} | U _{4,2} | Average ₂ | Median ₁ | 95th Pct ₁ |
| 3 | U _{1,2} | U _{2,2} | U _{3,2} | U _{4,2} | Average ₃ | Median ₁ | 95th Pct ₁ |
| ... | | | | | | | |
| 1,000 | U _{1,1000} | U _{2,1000} | U _{3,1000} | U _{4,1000} | Average ₁₀₀₀ | Median ₁₀₀₀ | 95th Pct ₁₀₀₀ |

Note that each result column characterizes a distribution for the corresponding statistic. By sorting a column of values, confidence intervals can be quantified. For example, if the column of averages are sorted, the 90 percent confidence interval for the average risk has a lower bound equal to the 50th value (out of 1,000 values), and an upper bound equal to the 950th value. It should also be noted that the distribution of values for each of the uncertain parameters will closely resemble the assumed distributions for these parameters.

The information in Table I-1 can also be used to conduct a sensitivity analysis. As outlined in Cohen *et al.* (1996), the influence of each uncertain quantity on any of the results can be estimated by regressing the calculated value of interest against all of the uncertain parameter values. Uncertain parameters that strongly influence that calculated value will have a large incremental R². Reducing the uncertainty associated with influential parameters will have the greatest impact on reducing the range of plausible calculated values generated by the simulation.

I.3 Guidelines for the Development of Input Distributions for Key Parameters

This subsection provides guidance on the development of distributions reflecting both uncertainty and variability (when appropriate) for parameters that appear in the equations outlined in Appendix H. The parameters discussed include:

- Subsection I.3.1: Ingestion rate for water (IRW)
- Subsection I.3.2: Ingestion rate for soil (IRS)
- Subsection I.3.3: The inhalation rate for air (IRA)
- Subsection I.3.4: The soil-to-skin adherence factor (AF)
- Subsection I.3.5: Exposure time for bathing (ET_{shower})

- Subsection I.3.6: Exposure frequency
- Subsection I.3.7: Exposure duration
- Subsection I.3.8: Body weight
- Subsection I.3.9: Body surface area
- Subsection I.3.10: Fraction of ingested soil from contaminated site
- Subsection I.3.11: Averaging Time
- Subsection I.3.12: Contaminant Concentration

Distributions for each of these parameters are discussed in turn.

I.3.1 Water Ingestion Rate (IRW)

Roseberry and Burmaster (1992) fit lognormal distributions to both total water intake and tap water intake data, for several age groups. For each age group, the authors report that the data closely follow a lognormal distribution. The summary statistics for these best-fit lognormal distributions are shown in Table I-2. These distributions are appropriate for describing variability in water intake. The summary statistics are based on large samples, so the uncertainty in these quantities should be small compared to the variability. Unless there is reason to believe that water consumption for a particular population of interest deviates substantially from national norms, we recommend use of these distributions.

Table I-2: Distribution Statistics: Total Water Intake

| Age group (years) | TotalWater Intake (ml/day) | | TapWater Intake (ml/day) | |
|----------------------|-------------------------------|------|-----------------------------|------|
| | GM | GSD | GM | GSD |
| 0-1 | 1,074 | 1.34 | 267 | 1.85 |
| 1-11 | 1,316 | 1.40 | 620 | 1.65 |
| 11-20 | 1,790 | 1.41 | 786 | 1.71 |
| 20-65 | 1,926 | 1.49 | 1,122 | 1.63 |
| > 65 | 1,965 | 1.43 | 1,198 | 1.61 |
| all | 1,785 | 1.50 | 963 | 1.70 |

Source: Calculated from values in Table I of Roseberry and Burmaster (1992).

I.3.2 Soil Ingestion Rate (IRS)

We characterize the IRS for children and adults separately. USEPA recommends a mean soil ingestion rate of 100 mg/day for children (USEPA Exposure Factors Handbook 1997). A typical upper bound value for soil ingestion by children is 200 mg/day. If we assume that this

upper bound value represents the 95th percentile, we can fit a lognormal distribution using these two data points. This yields a distribution with a GM of 88.4 mg/day, and a GSD of 1.64. USEPA recommends a mean soil ingestion value of 50 mg/day for adults, suggesting that the adult ingestion rate is approximately one half that of children (USEPA 1997). Based on this guidance, we recommend that the distribution used for children also be used for adults, but with a geometric mean that is half of the geometric mean assumed for children. Thus for adults we obtain a distribution with a GM of 44.2 and a GSD of 1.64.

I.3.3 Inhalation Rate (IRA)

Finley *et al.* (1994) present percentiles for the distribution of inhalation rates by age; Table I-6 lists these values. Since derivation of these values are based on assumptions regarding long term average metabolic rates, they represent average inhalation rates over extended periods. For each age group the data closely follow a lognormal distribution. The parameters of the best-fit lognormal distributions are shown in Table I-3.

The text in Section 3.3 of Finley *et al.* (1994) further describes the derivation of the percentile values listed in Table I-4. Finley *et al.* provide no indication that these statistics are particularly uncertain. However, risk assessors must keep in mind that while these long term average values may be valid for the population in general, upper end values may be most appropriate for particularly active subpopulations. Moreover, breathing rates vary substantially for each individual over the course of a day. While sleeping, inhalation rates are particularly low. On the other hand, individuals with physically intense occupations will have relatively high inhalation rates during working hours.

Table I-3: Distribution Percentiles: Inhalation Rate by Age (m³/day)

| | 5% | 10% | 25% | 50% | 75% | 90% | 95% | 99% |
|-----------|------|------|------|------|------|------|------|------|
| Age < 3 | 3.3 | 3.6 | 4.1 | 4.7 | 5.5 | 6.2 | 6.7 | 7.8 |
| Age 3-10 | 6.1 | 6.5 | 7.3 | 8.4 | 9.7 | 10.9 | 11.8 | 13.8 |
| Age 10-18 | 9.1 | 9.8 | 11.2 | 13.1 | 15.3 | 17.7 | 19.3 | 22.5 |
| Age 18-30 | 10.5 | 11.3 | 12.8 | 14.8 | 17.1 | 19.5 | 21.0 | 24.6 |
| Age 30-60 | 8.4 | 9.1 | 10.2 | 11.8 | 13.6 | 15.4 | 16.7 | 19.2 |
| Age > 60 | 8.5 | 9.2 | 10.4 | 11.9 | 13.7 | 15.6 | 16.7 | 19.6 |

Source: Table V in Finley *et al.* (1994)

Table I-4: Distribution Statistics: Inhalation Rate by Age (m³/day)

| | GM | GSD |
|-----------|------|-----|
| Age < 3 | 4.7 | 1.2 |
| Age 3-10 | 8.4 | 1.2 |
| Age 10-18 | 13.2 | 1.3 |
| Age 18-30 | 14.8 | 1.2 |
| Age 30-60 | 11.8 | 1.2 |
| Age > 60 | 12.0 | 1.2 |

I.3.4 Soil to Skin Adherence Factor

Finley *et al.* (1994) present percentiles for the distribution of soil-to-skin adherence factors, shown in Table I-5. The authors note that values reported in the literature do not vary significantly by age, so is not necessary to develop age-specific distributions for the adherence factor. They also report that “soil type, particle size, and indoor vs. outdoor exposure have minimal influence on soil adherence and therefore do not require consideration in development of a standard distribution.” As a result, a single distribution is sufficient for the adherence factor.

The data shown in Table I-5 are well characterized by a lognormal distribution with a geometric mean of 0.5 mg soil per cm² skin and a geometric standard deviation of 1.9. We recommend that this distribution be used to characterize uncertainty in the adherence factor, as Finley *et al.* (1994) state that it “is believed to predominantly reflect measurement uncertainty.”

Table I-5: Distribution Percentile: Soil to Skin Adherence Factor

| Percentile | Value (mg soil/cm ² skin) |
|------------|--------------------------------------|
| 0.05 | 0.014 |
| 0.1 | 0.03 |
| 0.25 | 0.05 |
| 0.5 | 0.25 |
| 0.75 | 0.6 |
| 0.9 | 1.2 |
| 0.95 | 1.7 |

Source: Table IX in Finley *et al.* (1994).

USEPA’S Exposure Factors Handbook (USEPA, 1997) recommends a slightly different approach for estimating soil adherence to skin, using data from a study by Kissel *et al.* (1996). The study reported soil loading on exposed skin surfaces for different activities. The analyst may use the soil loading value for the activity which best approximates the exposure scenario. (See the USEPA) Exposure Factors Handbook.) USEPA notes that insufficient data are available to develop a distribution or a probability function for soil loading.

I.3.5 Exposure Time for Bathing

USEPA’s Exposure Factors Handbook (USEPA, 1997) provides a frequency distribution for average shower duration (Table 14-18). The table reports a shower duration of 7 minutes at the 53rd percentile, and 15 minutes at the 96th percentile. If we fit a lognormal distribution to these data, we obtain a GM of 6.8 minutes and a GSD of 1.6. Since the lognormal places positive weight on shower duration values that are implausible (i.e., values very close to zero, and values that are arbitrarily large), it is recommended that the distribution be truncated at its 2.5 percentile value of 2.7 minutes, and its 97.5 percentile value of 17 minutes.

I.3.6 Exposure Frequency

The distribution for this parameter depends on factors specific to the exposure scenario. In general, the exposure frequency for a residential exposure scenario can be assumed to be approximately 350 days per year (which reflects an assumption that individuals spend two weeks per year away from home – *e.g.*, on vacation). The exposure frequency for occupational scenarios can be assumed to be approximately 250 days per year (which reflects the assumption that individuals work 5 days per week for 50 weeks per year).

It is suspected that for these standard scenarios, this parameter is not highly uncertain. On the other hand, for special exposure scenarios (*e.g.*, a trespasser visiting a site that is off-limits to the public), the exposure frequency may be highly uncertain. Very often, no empirical data for this parameter will even be available. In these cases, the risk assessor should take into account meteorological conditions that would limit the frequency of exposure (*e.g.*, it is unlikely that trespassers would visit an outdoor site on days when it rains; when the temperature is below freezing, access to contaminated soil may be limited).

I.3.7 Exposure Duration

Typically, exposure duration is assumed to reflect the amount of time an individual lives at a single location. Cohen *et al.* (1996) describe the derivation of lognormal distributions describing this parameter for both rural and urban households using data published by Israeli *et al.* (1992). In both cases, the data are well described by the lognormal. For urban households, the geometric mean is 5.32 years, with a geometric standard deviation of 6.48. For rural households, the geometric mean is 6.48 years, with a geometric standard deviation of 3.20.

For occupational exposures, exposure duration is assumed to reflect occupational tenure. Shaw and Burmaster (1996) detail values for the amount of time individuals spend between job changes. This information is presented for men and women separately by industry type.

I.3.8 Body Weight

USEPA (1997) reports that children's body weights are well described by lognormal distributions. Table I-6 lists these parameters, which are provided for each year of age through age 20.

Table I-6: Distribution Parameters: Children’s Body Weight (kg)

| Age (years) | Females | | Males | |
|----------------|---------|------|-------|------|
| | GM | GSD | GM | GSD |
| 0.5-1 | 8.67 | 1.16 | 9.30 | 1.14 |
| 1-2 | 10.80 | 1.14 | 11.70 | 1.13 |
| 2-3 | 12.94 | 1.12 | 13.46 | 1.13 |
| 3-4 | 14.73 | 1.15 | 15.64 | 1.12 |
| 4-5 | 16.95 | 1.14 | 17.64 | 1.14 |
| 5-6 | 19.69 | 1.18 | 19.89 | 1.15 |
| 6-7 | 22.20 | 1.19 | 22.87 | 1.16 |
| 7-8 | 24.29 | 1.19 | 24.78 | 1.16 |
| 8-9 | 27.39 | 1.17 | 27.94 | 1.20 |
| 9-10 | 31.82 | 1.24 | 30.88 | 1.18 |
| 10-11 | 35.52 | 1.22 | 36.23 | 1.22 |
| 11-12 | 40.85 | 1.25 | 40.04 | 1.29 |
| 12-13 | 45.60 | 1.24 | 43.82 | 1.25 |
| 13-14 | 50.40 | 1.24 | 48.42 | 1.24 |
| 14-15 | 54.05 | 1.21 | 55.70 | 1.20 |
| 15-16 | 54.60 | 1.17 | 59.74 | 1.17 |
| 16-17 | 57.97 | 1.18 | 66.69 | 1.18 |
| 17-18 | 59.15 | 1.18 | 66.02 | 1.18 |
| 18-19 | 58.56 | 1.16 | 70.11 | 1.17 |
| 19-20 | 60.34 | 1.16 | 70.81 | 1.17 |

Source: Parameters calculated from log parameters given in USEPA, 1996a..

Brainard and Burmaster (1992) report that adult body weights are also lognormally distributed. Body weights for women follow a lognormal distribution with a GM of 143 pounds and a geometric standard deviation of 1.2. Body weights for men are also lognormally distributed with GM = 169 pounds and GSD = 1.2.

Because body weights can be measured very accurately and the distribution of body weights in the population has been extensively studied and well characterized, the body weight distributions described here represent variability rather than uncertainty. That is, the magnitude of any inter-individual variability far exceeds the magnitude of the uncertainty in these distributions.

We recommend that it be assumed that body weights are correlated over time. In other words, it should be assumed that individuals who are relatively heavy at one age are also relatively heavy at later ages, while those who are light tend to remain light. Perfect correlation of body weights over time can be modeled by specifying that an individual’s weight percentile does not change. For example, an individual whose weight places him or her in the 30th percentile at age 10 is assumed to have a body weight placing him or her in the 30th percentile for weight at any other age. Perfect correlation can be implemented by randomly selecting the percentile for a simulated individual (*i.e.*, a value between 0% and 100%), and then selecting the corresponding value of the body weight distribution for each age listed in Table I-6.

Cancer toxicity factors (slope factors and unit risks) are derived assuming a body weight of 70 kg. These toxicity factors must be adjusted, as discussed in USEPA 1996a, if body weights other than 70 kg are assumed. Ordinarily, this assumption will be left unchanged, making adjustment unnecessary. The derivation of noncancer toxicity factors (reference doses [RFDs] and reference concentrations [RFCs]) does not depend on body weight, so no adjustment is necessary for these factors.

I.3.9 Body Surface Area

Finley *et al.* (1994) note that studies of total body surface area have found that surface area and body weight are strongly associated. Specifically, SA is equal to the product of BW and a ratio (R). The ratio R is lognormally distributed for children under age 2 years with an arithmetic mean of 641 cm² / kg and a standard deviation of 114 cm² / kg. These parameters correspond to a geometric mean of 631 cm² / kg, and a GSD of 1.19. For individual over the age of 2 years, R follows a normal distribution. For individuals between 2 and 18 years, the mean and standard deviation of this normal distribution is 423 cm² / kg and 76 cm² / kg, respectively; for individuals over the age of 18, the mean and standard deviation are 284 cm² / kg, and 28 cm² / kg, respectively.

Hence, once a body weight is determined probabilistically, as described in Subsection I.3.8, a value of R can be chosen randomly from the appropriate distribution described in the preceding paragraph. Total body surface area can then be computed by multiplying BW and R. If the surface area of some portion of the human body must be determined, it is recommended that the randomly generated total body surface area be multiplied by the appropriate mean percentage listed in Table I-7.

Table I-7: Average Fraction of Total Body Surface Area Corresponding to Head, Arms, Hands, Legs, and Feet

| Age (years) | Body Part – Percent of Total Body Surface Area | | | | |
|-------------|--|-------|-------|------|------|
| | Head | Arms | Hands | Legs | Feet |
| < 1 | 18.2 | 13.7 | 5.3 | 20.6 | 6.54 |
| 1-2 | 16.5 | 113.0 | 5.68 | 23.1 | 6.27 |
| 3-4 | 13.6 | 14.4 | 6.07 | 26.8 | 7.21 |
| 4-5 | 13.8 | 14.0 | 5.70 | 27.8 | 7.29 |
| 9-10 | 12.0 | 12.3 | 5.30 | 28.7 | 7.58 |
| Men > 18 | 7.8 | 14.1 | 5.2 | 31.2 | 7.0 |
| Women > 18 | 7.1 | 14.0 | 5.1 | 32.4 | 6.5 |

Source: Table IV in Finley *et al.* (1994). Note that Finley also provides minimum and maximum values for these parameters. However, it is unlikely that variability in the values of these parameters across members of the population would yield substantial differences in exposure or risk.

I.3.10 Fraction of Ingested Soil from a Contaminated Site

This parameter is site-specific and population-specific. Determining the range of plausible values for the population will depend on collection of either empirical data or on the adoption of reasonable assumptions. For example, if the risk assessment aims to quantify risks associated with soil ingestion at a non-residential site, it may be reasonable to assume that individuals spend fewer than half their waking hours at the site (on days when they visit the site at all), and hence that FI should be no larger than 0.5.

I.3.11 Averaging Time

It is not appropriate to use a distribution for the averaging time in either the cancer or noncancer risk equations. A 70 year lifetime should be used as the averaging time for cancer risks because the derivation of the cancer toxicity factors (slope factors or unit risks) reflects the assumption of a 70 year lifetime (USEPA, 1997).

The derivation of chronic noncancer risk parameters reflects the assumption that exposure is averaged over a “chronic period,” which is defined to be 7 years. Note that for exposures lasting more than 7 years, the noncancer risk is calculated using that 7 “window” during which average exposure is greatest. If the maximum exposure (on a mg/kg-day basis) is constant for 7 years or more, then noncancer risk can be calculated by comparing the RfD to the constant level of exposure during this period. In summary, the averaging time for noncancer risk assessment is always 7 years. However, the 7 year period over which exposure is averaged depends on the exposure pattern over time. Specifically,

- If exposure lasts for less than 7 years, the exposure compared to the RfD is total exposure (mg/kg) divided by 2,555 days (7 years × 365 days/year);
- If exposure exceeds 7 years, the exposure compared to the RfD is the maximum total exposure (mg/kg) over any 7-year period divided by 2,555 days;
- If exposure exceeds 7 years and is constant when expressed as mg/kg-day, then this constant level of exposure is compared directly to the RfD.

The ratio of the appropriately calculated exposure, as described above) to the RfD is the hazard index.

I.3.12 The Concentration Term

As noted in Section H.1, the concentration term corresponds to a contaminant’s average concentration at a “site.” Here, it is assumed that the site has been divided into exposure units, which are defined by USEPA (1996b) to be an area within which the probability of exposure to all locations is equal. It is also assumed that individuals come into contact with an exposure unit a large number of times. Hence, the average contaminant concentration to which they are exposed is equal to the average contaminant concentration within the exposure unit.

The advantage of this approach is that it simplifies analysis of variability. Specifically, variability is addressed by conducting a separate risk assessment for each exposure unit. The “drawback” of the approach is that it potentially requires division of a site into exposure units satisfying the definition specified above. For example, if a site is divided by a large stream, it may make sense to define the areas of the site on the two sides of the river as distinct exposure units. It should be noted that in a one-dimensional analysis of variability of exposure, the concentration term is the average concentration in an exposure area. The average concentration should be treated as a constant, not a variable.

It must also be kept in mind that uncertainty in the value of the average exposure unit concentration may have to be addressed. Typically, this lack of precision reflects the finite number of observations used to quantify contaminant concentrations. Characterizing the distribution of plausible values for the average within the exposure unit depends on assumptions regarding the distribution of concentration values within the exposure unit.

USEPA guidance (1992) states that if the concentration values follow a normal distribution, then the mean follows a Student’s “t” distribution. The value of a percentile (α) for the distribution describing the set of plausible values for the mean is described as

$$\bar{x} + t_{1-\alpha, n-1} (s / \sqrt{n})$$

where n is the sample size, \bar{x} is the sample mean, s is the sample standard deviation, and $t_{1-\alpha, n-1}$ is the $1-\alpha$ percentile of the student t distribution with $n-1$ degrees of freedom.

If the contaminant concentration is assumed to be lognormally distributed (which is often the case), then USEPA recommends assuming that plausible values for the mean fall within a range between a lower and upper percentile bound defined by the expression,

$$e^{\left(\bar{x} + \frac{s^2}{2} + \frac{sH}{\sqrt{n-1}} \right)}$$

where \bar{x} is the sample mean of the log of the observations, and s is the sample standard deviation of the log of the observations. The lower bound value for this range is determined by using a lower-end fractile value for H (e.g., the 5% value), while the upper bound value for this range makes use of an upper-end fractile value for H (e.g., the 95% value). A randomly selected value for the mean can be generated as follows. First, select a percentile from the uniform distribution between 0 and 1. Then identify the corresponding H value. Finally, plug this H value into the preceding equation. Cohen *et al.* (1996) describe the development of a distribution for plausible values of the mean contaminant concentration at a Superfund site using the preceding equation.

There are several techniques to determine whether concentration values are normally distributed (in which case the t statistic should be used to characterize plausible values of the mean), or lognormally distributed (in which case the H statistic should be used to calculate the distribution characterizing plausible values of the mean). First, there are tests of normality built into various statistics packages. For example, the W-test (Gilbert, 1987) indicates whether the hypothesis of normality can be rejected. Applying this test to the untransformed data indicates whether the data can be considered to be normally distributed. Applying this test to the log-transformed values of the data indicates whether the data can be considered to be lognormally distributed. Specifically, failure to reject the hypothesis of normality when applying the test to log-transformed data values indicates the data are lognormal.

A complication that may be encountered is the case in which the hypothesis of normality is rejected both when the test is applied to the untransformed data and when it is applied to the log-transformed data. In this case, graphical methods may provide a better idea of which (if either) distribution best describes the observed concentrations. One commonly used method is the probability plot method described by Gilbert (1987).

Finally, maximum likelihood techniques can be used to characterize the set of distributions that may plausibly describe the data. The collection of mean statistics corresponding to each of these distributions represents the set of plausible mean values consistent with the data. The following discussion briefly outlines the use of this technique when the data are assumed to be lognormally distributed.

Suppose we have observations denoted x_1, x_2, \dots, x_N . The log of these concentrations are denoted y_1, y_2, \dots, y_N . We start by computing the maximum likelihood estimates for the mean (denoted μ) and standard deviation (denoted σ) of the log-transformed data:

$$\mathbf{m} = \frac{1}{N} \sum_{i=1}^N y_i$$

and

$$\mathbf{s} = \sqrt{\frac{1}{N} \sum_{i=1}^N (y_i - \mathbf{m})^2}$$

Next, we define the log likelihood function for any set of estimates for the mean and standard deviation of the log transformed data set (y_1, y_2, \dots, y_N). Denote the alternative estimates for the mean and standard deviation as μ_j and σ_k . Then the log-likelihood function, $L(\mu_j, \sigma_k)$ is

$$L(\mathbf{m}_j, \mathbf{s}_k) = \sum_{i=1}^N \ln[f(x_i, \mathbf{m}_j, \mathbf{s}_k)]$$

where $f(x_i, \mu_j, \sigma_k)$ is the density function of the lognormal with parameters $\ln(\text{geometric mean}) = \mu_j$ and $\ln(\text{geometric standard deviation}) = \sigma_k$ evaluated at data point x_i . The density function $f(x_i, \mu, \sigma)$ is denoted

$$f(x_i, \mathbf{m}_j, \mathbf{s}_k) = \frac{1}{\sqrt{2\pi s_k} x_i} e^{\left[-\frac{1}{2} \left(\frac{\ln x_i - \mathbf{m}_j}{\mathbf{s}_k}\right)^2\right]}$$

Note that $L(\mu_j, \sigma_k)$ can now be expressed as

$$L(\mathbf{m}_j, \mathbf{s}_k) = \sum_{i=1}^N -\ln(\sqrt{2\pi}) - \ln(\mathbf{s}_k) - y_i - \frac{1}{2} \left(\frac{y_i - \mathbf{m}_j}{\mathbf{s}_k}\right)^2$$

The 95% confidence region is the set of all μ_j and σ_k that satisfy the relationship

$$2 \times L(\mathbf{m}, \mathbf{s}) - 2 \times L(\mathbf{m}_j, \mathbf{s}_k) \leq \mathbf{c}_{0.95,2}^2$$

where $\chi_{0.95,2}^2$ denotes the 95th percentile of the chi-square distribution with 2 degrees of freedom.

Each pair of parameters (μ, σ) in this region corresponds to a lognormal distribution, each of which has a corresponding mean. A Monte Carlo simulation can randomly select average contaminant concentration values by randomly selecting a parameter pair from this region and then calculating the arithmetic mean that corresponds to that pair.

I.4 Calculation of Cleanup Levels Using Probabilistic Techniques

Acceptable average soil contaminant levels can be derived from a “backwards” solution to the appropriate risk equation. In the case of cancer risk assessment, the original equation, which quantifies risk as a function of exposure terms (*e.g.*, exposure frequency, ingestion, rate, *etc.*) is rearranged so that the concentration term is expressed as a function of the other exposure terms and the maximum acceptable risk level (*e.g.*, 10^{-5}). In the case of non-cancer risk assessment, the original equation quantifying daily dose is rearranged so that concentration is expressed as a function of the other exposure terms and the maximum acceptable dose – *i.e.*, the RfD.

Although Burmaster et al, (1998) and Ferson (1996) have pointed out the difficulty of performing back-calculations for cleanup levels when probabilistic risk assessment is used, back-calculation can be used under certain limiting conditions, specifically, when the target risk is set by a single value (Burmaster and Thompson, 1995). An example of a single value target risk for cancer is that 95% of the population must have a risk below 10^{-6} .

The probabilistic calculation is performed doing back-calculation with the risk parameter specified as the single value representing the target (*e.g.*, 10^{-6}). This simulation yields a distribution of soil contaminant concentration values. To ensure that, for example, the 90th percentile risk does not exceed the maximum acceptable risk (in the case of carcinogens) or that the 90th percentile dose does not exceed the RfD (in the case of noncarcinogens), the 10th percentile of the soil contaminant concentration distribution generated by the simulation is selected as the maximum acceptable average concentration. In general, selection of the p fractile of the soil contaminant concentration distribution ensures that either the (1-p) fractile risk does not exceed the maximum acceptable risk or that the (1-p) fractile dose does not exceed the RfD.

Calculation of a site-specific “pickup level” involves further statistical analysis that makes use of the maximum acceptable average contaminant concentration, as described in the preceding paragraph, and information regarding the site-specific distribution of contaminant concentrations. The so-called “cleanup remediation goal” or “CRG” approach that identifies the pickup level requires use of specialized software. The methodology is outlined in Bowers *et al.* (1996). A revised version of this methodology will be available later in 1997. Removal of all soil with contaminant concentrations exceeding the pickup level ensures that the site-wide average concentration is less than the acceptable average concentration with some specified level of confidence.

I.5 References

- Bowers, T.S., N.S. Shifrin, and B.L. Murphy. 1996. Statistical Approach To Meeting Soil Cleanup Goals. *Environ. Sci. Technol.* 30 (5): 1437-1444.
- Brainard, J. and Burmaster, DE. 1992. Bivariate Distributions For Height And Weight Of Men And Women In The United States. *Risk Anal.* 12 (2): 267-275.
- Burmaster, D.E., Lloyd, K.J., and Thompson, K.M. 1995. The Need For New Methods To Backcalculate Soil Cleanup Targets In Interval And Probabilistic Cancer Risk Assessments. *Hum. Ecol. Risk Assess.* 1,89-100.
- Burmaster, D.E., Lloyd, K.J., and Thompson, K.M. 1995. Backcalculating Cleanup Targets In Probabilistic Risk Assessments When The Acceptability Of Cancer Risk Is Defined Under Different Risk Management Policies. *Hum. Ecol. Risk Assess.* 1, 101-120.
- Calabrese, E.J., E.S. Stanek, and C.E. Gilbert. 1991. A Preliminary Decision Framework For Deriving Soil Ingestion Rates. In: Calabrese, E.J, and P.T. Kostecki, eds. *Petroleum Contaminated Soils* 2(29). Lewis, Chelsea, MI.
- Cohen, J.T., Lampson, M.A., and Bowers, T.S. 1996. The Use Of Two-Stage Monte Carlo Simulation Techniques To Characterize Variability And Uncertainty In Risk Analysis. *Human and Ecological Risk Assessment.* 2(4): 939-971.
- Ferson, S. 1996. What Monte Carlo Methods Cannot Do. *Hum. Ecol. Risk Assess.* 2, 990-1007.
- Finley, B., Proctor, D., Scott, P., Harrington, N., Paustenbach, D., and Price, P. 1994. Recommended Distributions For Exposure Factors Frequently Used In Health Risk Assessments. *Risk Anal.* 14(4): 533-553.
- Gilbert, RO. 1987. *Statistical Methods for Environmental Pollution Monitoring.* Van Nostrand Reinhold (New York, NY).
- Kissel *et al.*, 1996. Kissel, J., K. Richter, R. Fenske. Field Measurements Of Dermal Soil Loading Attributable To Various Activities: Implication For Exposure Assessment. *Risk Anal.* 16(1):116-125.
- Israeli, M., C.B. Nelson. 1992. Distribution And Expected Time Of Residence For US Households. *Risk Anal.* 12 (1): 65-72.
- Roseberry, A.M., and Burmaster, DE. 1992. Lognormal Distributions For Water Intake By Children And Adults. *Risk Anal.* 12(1): 99-104.
- Shaw, C.D., and D.E. Burmaster. 1996. Distributions Of Job Tenure For U.S. Workers In Selected Industries And Occupations. *Human and Ecological Risk Assessment.* 2(4): 798-819.

US Environmental Protection Agency (USEPA). 1989. *Risk Assessment Guidance for Superfund. Volume II: Environmental Evaluation Manual*. Office of Emergency and Remedial Response (Washington, DC). EPA-540/1-89-001. March.

U.S. Environmental Protection Agency (USEPA). 1992. Office of Solid Waste and Emergency Response (Washington, DC). *Supplemental Guidance to RAGS: Calculating the Concentration Term*. OSWER Publication 9285.7-081;NTIS PB92-963373. 8p. May 1992.

US Environmental Protection Agency (USEPA). 1996a. *Summary Report for the Workshop on Monte Carlo Analysis*. Office of Research and Development (Washington, D.C.). EPA/630/R-96/010. September.

US Environmental Protection Agency (USEPA). 1996b. *Soil Screening Guidance: Technical Background Document*. Office of Solid Waste and Emergency Response (Washington, DC). May. EPA/540/R-95/128. PB96-963502.

US Environmental Protection Agency (USEPA). 1997. Office of Health and Environmental Assessment, Exposure Assessment Group. *Exposure Factors Handbook – Volumes I, II and III*. EPA/600/P-97/002FA,B,C.

APPENDIX J: OFFICE OF WATER RESOURCES IN-STREAM MONITORING PROCEDURES TO DETERMINE IMPACT ON THE SURFACE WATER FROM NON-POINT SOURCE SITE REMEDIATION PROJECTS

BACKGROUND:

The Water Quality Standards Rule, 46CSR1, Section 5, allows the Chief of the Office of Water Resources to determine, on a case-by-case basis, definable geometric limits for mixing zones for a discharge or a pollutant or pollutants within a discharge, upon the request of a permit applicant or permittee. These rules are tailored for point source discharges in order to further protect water quality after the imposition of technology-based treatment standards, best available treatment, on the point source discharge. Site remediation projects which constitute non-point sources are not required to obtain permits. Therefore, in order to protect water quality and achieve compliance with the rules, the Chief of the Office of Water Resources will require implementation of the following in-stream monitoring procedures to be used to determine the impact on the receiving stream, in conjunction with site remediation projects.

In-Stream Monitoring Procedures

All samples will be collected for the specific pollutants of concern and using accepted QA/QC procedures. Surface water samples will be collected as follows:

Upstream Sampling Location(s)

- A. 25' from property line
- B. Vertical mid-point of the water column
- C. A sampling location for each 50' of river width, spaced equally

Downstream Sampling Location(s)

- A. 25' from property line
- B. Vertical mid-point of the water column
- C. A sampling location for each 25' of river width, spaced equally, up to 100' of river width

Additional Sampling Location(s)

An additional sampling will be performed if the property (shoreline) exceeds 200'. For each additional 200' of the property, sampling locations will be as follows:

- A. A sampling point for each 25' of river width, spaced equally, up to 100' of river width.
- B. Vertical mid-point of the water column.
- C. Spaced equally between upstream and downstream sampling locations.



Please contact the West Virginia Secretary of State's Office at (304) 558-6000 or <http://www.state.wv.us/sos/adlaw/alreqser.htm> for copy of:

**TITLE 60
LEGISLATIVE RULE
DIVISION OF ENVIRONMENTAL PROTECTION
DIRECTOR'S OFFICE**

**SERIES 3
VOLUNTARY REMEDIATION AND REDEVELOPMENT RULE**

March 13, 2001

The changes between Version 2.0 and Version 2.1 are:

APPENDIX C-2: CHECKLIST TO DETERMINE THE APPLICABLE ECOLOGICAL STANDARD