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Unit Environmental Transport Assessment of Contaminants from Hanford's Past-Practice Waste Sites

Hanford Remedial Action Environmental Impact Statement

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Abstract

The U.S. Department of Energy, Richland Operations Office (DOE-RL) contracted Pacific Northwest Laboratory (PNL)^(a) to provide support to Advanced Sciences, Incorporated (ASI) in implementing the regional no-action risk assessment in the Hanford Remedial Action Environmental Impact Statement (HRA-EIS). Researchers at PNL were charged with developing unit concentrations for soil, groundwater, surface water, and air at multiple locations within an 80-km (50-mi) radius from the center of the Hanford Installation at years 40, 100, 140, 1000, and 10,000. Using the Multimedia Environmental Pollutant Assessment System (MEPAS), PNL simulated 1) a unit release of one ci for each radionuclide and one kg for each chemical from contaminated soils and ponded sites, 2) transport (i.e., migration and fate) of the contaminants in and through various environmental media (i.e., groundwater, surface water, air, and soils), and 3) exposure/risk of four exposure scenarios, outlined by the Hanford Site Baseline Remedial Action Methodology. These four scenarios include residential, recreational, industrial, and agricultural exposures. Spatially and temporally distributed environmental concentrations based on unit releases of radionuclides and chemicals were supported to ASI in support of the HRA-EIS. Risk for the four exposure scenarios, based on unit environment concentrations in air, water, and soil, were also supplied to ASI.

This report outlines the procedure that was used to implement the unit transport portion of the HRA-EIS baseline risk assessment. Deliverables include unit groundwater, surface water, air, and soil concentrations at multiple locations within an 80-km (50-mi) radius from the center of the Hanford installation for years 40, 100, 140, 1000, and 10,000. These unit concentrations will be utilized to compute risks at any location within and outside of the Hanford installation boundaries to a radial distance of 80 km (50 mi).

(a) PNL is operated for the U.S. Department of Energy by Battelle Memorial Institute.

Summary

The U.S. Department of Energy, Richland Operations Office (DOE-RL) contracted Pacific Northwest Laboratory (PNL) to provide support to Advanced Sciences, Incorporated (ASI), in implementing the regional no-action risk assessment in the Hanford Remedial Action Environmental Impact Statement (HRA-EIS). Researchers at PNL were charged with developing unit concentrations for soil, groundwater, surface water, and air at multiple locations within an 80-km (50-mi) radius from the center of the Hanford Installation at years 40, 100, 140, 1000, and 10,000.

Researchers at PNL, in conjunction with ASI, developed the Modular Risk Analysis (MRA) methodology and implemented the MRA as a component of the HRA-EIS. The MRA modularizes the risk assessment approach into three distinct modules: source term, unit transport (i.e., Unit Transport Analysis [UTA]), and unit exposure/risk. By determining the linearities in the assessment at Hanford, each module can be structured so that its calculations are independent of the calculations in the other modules. By structuring the approach in modular form, the three assessments (i.e., source term, transport, and exposure/risk) can proceed simultaneously and independently.

The MRA uses the Multimedia Environmental Pollutant Assessment System (MEPAS) to simulate the release of contaminants (i.e., leaching, volatilization, and soil suspension) from contaminated soil or ponds, migration and fate of the contaminants in and through various environmental media (i.e., groundwater, surface water, air, and overland), exposure to receptors, and subsequent risk/hazard to the exposure. MEPAS, in conjunction with the MRA, computes unit groundwater, surface water, air, and soil concentrations at locations within an 80-km (50-mi) radius of the center of the installation at years 40, 100, 140, 1000, and 10,000. In conjunction with MEPAS, the MRA allows the analyst to connect to the Washington State Geographical Information System (GIS) and develop spatially varying two- and three-dimensional concentration and risk isopleths at different times.

This report reviews the structure associated with the MRA and discusses the structure and steps associated with the UTA. Because millions of computer modeling runs were made to generate unit soil, air, groundwater, and surface water concentrations and because the UTA output represents only an intermediate step in presenting information as part of the MRA, a summary of the output results is not presented. Instead, illustrative examples of the output are presented.

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1.0 Introduction

Human-health impacts, especially to the surrounding public, are extremely difficult to assess at installations that contain multiple waste sites and a variety of mixed-waste constituents (e.g., organic, inorganic, and radioactive). These assessments must address different constituents, multiple waste sites, multiple release patterns, different transport pathways (i.e., groundwater, surface water, air, and overland soil), different receptor types and locations, various times of interest, population distributions, land-use patterns, baseline assessments, and a variety of exposure scenarios. Although the process is complex, two of the most important difficulties to overcome are associated with 1) establishing an approach that allows for modifying the source term, transport, or exposure component as an individual module without having to re-evaluate the entire installation-wide assessment (i.e., all modules simultaneously) and 2) displaying and communicating the results in an understandable and useable manner to interested parties. An integrated, physics-based, compartmentalized approach, which is coupled to a Geographical Information System (GIS), is presented that captures the regional health impacts associated with multiple waste sites (e.g., hundreds to thousands of waste sites) at locations within and surrounding the installation. Utilizing a modular/GIS-based approach overcomes difficulties in 1) analyzing a wide variety of scenarios for multiple waste sites and 2) communicating results from a complex human-health-impact analysis by capturing the essence of the assessment in a relatively elegant manner so that the meaning of the results can be quickly conveyed to all who review them. For example, the results can be presented in two- and three-dimensional plots of concentration and risk isopleths using multimedia (i.e., intermedia) models. These isopleths consider concentrations in and exposures from soil, air, groundwater, and surface water. Two- and three-dimensional isopleths (i.e., contours) are illustrated in Figure 1.

The Modular Risk Assessment (MRA) methodology was developed by researchers at the Pacific Northwest Laboratory (PNL)^(a) in conjunction with Advanced Sciences, Incorporated (ASI), to concurrently assess multiple waste sites at multiple locations containing multiple constituents in different waste forms (e.g., soil, liquid). The MRA modularizes the risk assessment approach into three distinct modules: source term, unit-transport (i.e., Unit Transport Analysis [UTA]), and unit exposure/risk. The MRA takes advantage of the inherent linearities that exist at an installation to reduce the number of calculations required for the analysis. As opposed to calculating the environmental concentrations at one location for a single constituent, the MRA calculates the spacial and temporal distribution of multiple contaminants. In addition, the MRA accounts for the interaction between plumes emanating from different waste sites. The MRA uses the Multimedia Environmental Pollutant Assessment System (MEPAS) to simulate the release of contaminants (i.e., leaching, volatilization, and soil suspension) from contaminated soil or ponds, migration and fate of the contaminants in and through various environmental media (i.e., groundwater, surface water, air, and overland), exposure to receptors, and subsequent risk/hazard to the exposure. MEPAS, in conjunction with the MRA, computes unit groundwater, surface water, air, and soil concentrations at locations within an 80-km (50-mi) radius of the center of the installation at years 40, 100, 140, 1000, and 10,000. In conjunction with MEPAS, the MRA allows the analyst to connect to the state GIS and develop spacially varying two- and three-dimensional concentration and risk isopleths at different times.

(a) PNL is operated for the U.S. Department of Energy by Battelle Memorial Institute.

Figure 1. Schematic Illustrations of Two- and Three-Dimensional Risk Isopleths (Developed by Advanced Sciences, Incorporated, Richland, Washington)

The purpose of this effort was to implement the UTA by generating unit concentrations for soil, groundwater, surface water, and air at multiple locations within an 80-km (50-mi) radius at years 40, 100, 140, 1000, and 10,000. Because millions of computer modeling runs were made to generate unit soil, air, groundwater, and surface-water concentrations and because the UTA output represents only an intermediate step in presenting information as part of the MRA, a summary of the output results is not presented, although the output data are available on request. The scope of the effort associated with generating unit concentrations for all constituents is summarized as follows:

- define transport scenarios
- determine characteristics in the environmental settings
- collect and review physical source-term data
- develop structure for performing unit transport simulations and the database associated with those simulations for soils, ponds, rivers, and groundwater
- develop structure for analysis of unit transport results in groundwater (soils), groundwater (ponds), surface water (Columbia River), and atmosphere
- develop interface and pre- and post-processing programs
- identify distribution coefficients (Kds) per environmental setting per constituent
- modify MEPAS to run as compartments, providing spatially varying results
- determine the number of matrix simulations
- implement the unit transport analyses through groundwater (soil), groundwater (ponds), surface water (Columbia River), and atmosphere
- generate unit concentrations in soil, groundwater, surface water, and air at multiple locations within an 80-km (50-mi) radius of the center of the Hanford installation for multiple constituents in two waste forms (i.e., contaminated soils and ponded wastes) for years 40, 100, 140, 1000, and 10,000.

This report is divided into the following eight sections:

MRA Approach -- This section describes the MRA methodology.

Calculations for Linearity -- For the MRA to be effective, its structure must be modularized and applied to a mathematically linear system. This section mathematically describes the linearities in the system and presents analyses to demonstrate the linearity.

Implementation Steps of MRA at Hanford -- To meet the end points associated with the MRA, a ten-step process was implemented. Each step is briefly discussed.

Pathline Analysis Using the Coupled Fluid, Energy, and Solute Transport (CFEST) Model -- To calculate concentrations at future times at locations other than the sources, the CFEST model was used to generate pathlines to spacially identify the location of the contamination as it emanated from the sources and moved through the environment.

Unit Transport Analysis Using MEPAS -- This section presents a description of the multimedia model MEPAS.

Summary of Source-Term and Environmental Setting Assumptions -- This section describes the source-term and environmental setting data utilized in the UTA.

UTA of MRA at Hanford -- This section describes the unit transport analyses for the atmospheric and water transport exercises at Hanford. Illustrative example results are supplied, and a discussion of the impacts associated with the travel time is presented.

References -- This section documents the cited literature.

2.0 Modular Risk Analysis Approach

Within the past ten years, multimedia contaminant environmental exposure and risk assessment methodologies have been the focus of a considerable amount of research and development.^(a) These multimedia methodologies describe the release of contaminants into the environment, their transport and fate through various environmental media (i.e., groundwater, surface water, overland, and air), exposures to surrounding sensitive receptors (i.e., inhalation, ingestion, dermal contact, and external dose), and health consequences resulting from the exposures.

Practically speaking, three levels of analyses can be identified (with a variety of variations possible) (Whelan et al. 1992): screening-level (ranking), "analytical" (prioritization and preliminary assessments), and numerical (detailed). Early in the process, screening models are used to identify environmental concerns. These models, often based on a structured-value approach, are designed to be used with regional/representative information (37 FR 31219-43; 40 CFR 300; Aller et al. 1985; Wells et al. 1990). Models such as the Hazard Ranking System (HRS) (FR 1982; CFR 1988) divide site and release characteristics into predetermined categories that are assigned a point value based on answers to questions. The score from such systems is useful to determine whether a situation is a problem. The score however, does not provide a risk-based relative ranking of problems.

Detailed analyses require a highly specialized assessment of potential impacts (Onishi et al. 1981; 1982; Whelan et al. 1983; Whelan and Parkhurst 1983). Such methodologies as the Chemical Migration and Risk Assessment (CMRA) methodology (Onishi et al. 1981; Whelan and Parkhurst 1983) are compositely coupled approaches that use numerically based models that are physically unlinked and represent single-medium models that are implemented independently in series. This approach is usually reserved for the most complex models, is data intensive, and is based on the expertise of the analyst. These detailed assessment models are used to determine the levels of risk associated with relatively well-defined and complex problems. Models for detailed analyses tend to focus on special sets of problems and special types of situations. Although detailed assessment tools are appropriate for their intended application, extension beyond the site-specific application is often either difficult or cost prohibitive.

Analytically/semianalytically/empirically based multimedia models (referred to as "analytical" models) can be utilized for prioritization or preliminary assessments and fill the gap between initial-screening and highly specialized numerical models.^(b) These physics-based models may contain some

(a) Whelan and Parkhurst 1983; Whelan et al. 1983, 1986, 1987a, 1987b, 1988, 1989, 1992; Cohen and Ryan 1985; McKone 1985; Mackay and Paterson 1986; Hartz and Whelan 1988; Hoopes et al. 1988; Mayer 1988; Napier et al. 1988; Buck et al. 1989; Droppo et al. 1989a, 1989b, 1990, 1993; Droppo and Hoopes 1989; NCITR 1989a, 1989b; Streng and Peterson 1989; 1992 Draft report, U.S. Environmental Protection Agency; Yu et al. 1993.

(b) Johanson et al. 1980; Browman et al. 1982; Whelan et al. 1983, 1986, 1987a, 1987b, 1988, 1989, 1992; Whelan and Parkhurst 1983; Cohen and Ryan 1985; McKone 1985; Mackay and Paterson 1986; Hartz and Whelan 1988; Hoopes et al. 1988; Mayer 1988; Napier et al. 1988; Buck et al. 1989; Droppo et al. 1989a, 1989b, 1990, 1993; Droppo and Hoopes 1989; NCITR 1989a, 1989b; Streng and Peterson 1989; 1992 Draft report, U.S. Environmental Protection Agency; Yu et al. 1993; Tucker et al. 1994.

numerical computations (hence the semianalytical designation). The analytical models, such as the MEPAS methodology, are designed to provide environmental evaluations over a wide range of applications. These models are fully coupled approaches that use analytically, numerically, and empirically based algorithms that are combined into a single code to describe each environmental medium.

Droppo et al. (1993) illustrated the utility of analytical models in the waste-site evaluation process. The analytical-assessment models are codes with physics-based algorithms whose components can be utilized in a detailed (i.e., numerical) or an initial-screening (i.e., ranking/prioritization) assessment, where data and circumstances warrant. Figure 2 illustrates the relationships between data-input requirements, output uncertainty, and types of problems addressed by each level of assessment. As Figure 2 indicates, the computational requirements tend to be less at the earlier stages of an assessment when available data are fewer, and, correspondingly, the uncertainty associated with the output results tends to be greater. As the assessment progresses, improved site-characterization data and conceptualization of the problem increase, thereby reducing the overall uncertainty in risk estimates.

Figure 2. Relationships Between Data-Input Requirements, Output Uncertainty, and Types of Problems Addressed by Each Level of Assessment (Based on Droppo et al. 1993)

The strength of analytical multimedia models is that they integrate standard approaches into a consistent yet powerful tool. The multimedia model comprises medium-specific transport-pathway and exposure-route codes, which are based on standard, well-accepted algorithms and, hence, accepted by regulators. For example, analytical solutions to the advective-dispersive equation describe transport in the groundwater environment. When coupled, the models allow the analyst to immediately assess the entire process of contaminant release, transport, exposure, and risk at one sitting. The utility of a coupled model is exemplified by an order-of-magnitude reduction in assessment time, as compared to an uncoupled model.

Multimedia models can be effectively used to concurrently assess multiple waste sites with multiple constituents for baseline (at $t = 0$), no-action (at $t > 0$), during-remediation, and residual (post-remediation) assessments, including land-use patterns (e.g., agricultural, residential, recreational, and industrial). The multimedia models can be used to describe the environmental concentrations within each medium at all locations surrounding the waste sites to a radius of 80 km (50 mi). Spatially distributed, three-dimensional concentration isopleths can be constructed detailing the level of contamination within each environment. By coupling land-use patterns with the environmental concentrations, three-dimensional risk isopleths (as a function of land-use pattern and location) can be developed.

2.1 MRA Structure

Most assessments are conducted on a single waste site, although at large installations (e.g., U.S. Department of Energy [DOE] and U.S. Department of Defense [DoD] installations) multiple waste sites represent the norm. Under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) process, SUPERFUND sites have been aggregated into a group of centrally located sites (e.g., operable units); therefore, aggregation represents a cost-effective alternative for regional assessments. When multiple waste sites with different waste forms at different locations require assessment, a more flexible approach is required to ensure that important aspects of the assessment are not lost through gross aggregation. The MRA approach allows for evaluation of regional risk by accounting for the effects of widely scattered waste sites. Because the system is automated, the approach becomes powerful and relatively easy to apply.

Three key end points were identified for the Hanford Remedial Action Environmental Impact Statement (HRA-EIS) that dictated the type of assessment that was required. First, the system should be structured so that on-site and regional results could be presented as two- and three-dimensional concentrations and risk isopleths for four different land-use options: recreational, residential, agricultural, and industrial. Second, an automated modular approach to the assessment had to be developed, so the user could change the source-term, transport, or exposure scenarios without having to repeat the entire source-to-receptor analysis.

The assessment was divided into three independent components. Source-term data were collected simultaneously with unit transport and exposure/risk analyses. When the linearity of the system has been determined, each module can be assessed independently. The final risk is calculated by multiplying numbers from each of the source-term, transport, and exposure modules together. The modular structure of the MRA is presented in Figure 3. Finally, the approach had to address the release and transport of multiple constituents through multiple environments in different waste forms. The key to this type of an assessment is linearity and modularization. For example, if the risk is linear relative to

Figure 3. Structure of the Modular Risk Analysis (Demonstrating the Relationships Between the Source-Term, Unit-Transport, and Unit-Exposure Components)

concentration, and the concentration is doubled, then the risk is doubled also; therefore, only one calculation is required.

To meet the end points, the assessment of installations with multiple waste sites is a ten-step process:

1. connect installation to state GIS
2. subdivide installation and surrounding area within the 80-km (50-mi) radius into a grid-cell system
3. aggregate waste sites in each cell location according to waste type (e.g., contaminated soils) within the 80-km (50-mi) radius
4. define initial conditions for each grid cell within the 80-km (50-mi) radius
5. define land-use patterns and exposure scenarios for each cell within the 80-km (50-mi) radius
6. determine flow paths from the aggregated waste sites to all surrounding grid cells within the 80 km (50-mi) radius
7. implement unit transport and exposure analyses for each aggregated waste site at each cell location within the 80-km (50-mi) radius
8. collate cell concentrations by combining concentrations from multiple waste sites for each cell within the 80-km (50-mi) radius

9. calculate risk for each land-use pattern within each cell within the 80-km (50-mi) radius
10. plot two- and three-dimensional concentration and risk isopleths.

A unit-analysis approach represents the cornerstone for efficiently implementing the MRA assessment. A unit-analysis approach is based on the conditions of linearity. Linearity implies that output results are linear to the input. For example, when the input doubles (e.g., contaminant inventory) the output doubles (e.g., concentration or risk). By determining the conditions of linearity, the assessment can be divided into compartments that can be implemented independently and concurrently. The compartments are 1) source-term determination (i.e., waste site strength and degree of contamination), 2) transport modeling to redistribute the waste for future conditions, 3) exposure assessment to determine the dose to surrounding populations, and 4) risk or health consequences determination from the exposure.

Under conditions of linearity, environmental concentrations are linearly proportional to the strength of the waste at the source. Likewise, exposure, dose, and risk calculations can be performed under unit environmental concentrations. The risk at any location surrounding the installation is based on multiplicative mathematics (i.e., source strength times the unit concentration times the unit risk):

$$\text{Risk} = \frac{S_s}{S_u} \cdot \frac{C_{us}}{C_{uc}} \cdot R_u \quad (1)$$

where S_s is the actual strength of the source term, S_u is the strength of an assumed unit source term, C_{us} is the unit concentration at the receptor based on the assumed S_u and unit-transport modeling, C_{uc} is the unit concentration assumed at the receptor (which is not based on S_u or transport modeling) in the unit-exposure analysis, and R_u is the unit risk based on the unit-exposure analysis (i.e., C_{uc}).

For example, under temporally far-field conditions, the concentration at a receptor is directly proportional to the waste inventory at the source; that is, as the source inventory doubles, the environmental concentration at the receptor doubles. An example of the temporally far-field case is illustrated in Figure 4. Figure 4 shows that as the initial release at the source moves downgradient, the shape of its distribution becomes strictly Gaussian (e.g., at distance 1000 in Figure 4). In other words, the results occurring at distance 1000 based on the shape of the release illustrated in Figure 4a are the same as those that would be produced if the release was instantaneous at the source. Therefore, the distribution of the release at the source is independent of the source, except for mass or inventory that is released.

Under the MRA, the analyst can assume a unit inventory at the source (e.g., $S_u = 1$ Ci) and estimate through modeling the receptor concentration (e.g., C_{us}). When the source-term analysis is completed or if the source-term data are updated (e.g., S_s), the new receptor concentration is simply the new inventory times the predicted receptor concentration [i.e., $S_s \cdot C_{us} \equiv (S_s / S_u) \cdot C_{us}$]. Because of this linearity, the unit-transport and unit-risk calculations can be performed independently of the source-term evaluation; therefore, the source-term, transport, and exposure analyses can be performed independently and concurrently.

Figure 4. Illustrative Example of a Far-Field Problem: a) Mass Flux from the Source, b) Concentration at a Near-Field Distance, Which is Closest to the Source, c) Concentration at a Near-Field Distance, Which is at a Middle-Distance from the Source, and d) Concentration at a Far-Field Distance from the Source

2.2 Source-Term Analysis

Prior to the collection of data that specifically describe the waste at each site, environmental settings and a grid system must be determined. The installation is divided into regions where the geology, hydrology, hydraulics, and meteorology are considered to be representative of the region. Data describing each environmental setting are collected and verified by installation engineers and scientists as being representative of the areas. Figure 5 presents an illustrative example of seven environmental settings (excluding meteorology) that have been utilized to describe DOE's Hanford Installation near Richland, Washington, as part of DOE's complex-wide Programmatic Environmental Impact Statement (PEIS): 100-N Region, North of the Columbia River Region, 100-H Region, 200 East Region, 200 West Region, 300 Area Region, and Southwest Region. Two meteorological settings have also been identified for Hanford. Similar maps have been developed under the PEIS for other DOE installations (e.g., Oak Ridge National Laboratory, Los Alamos National Laboratory).

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Figure 5. Illustrative Example of Seven Environmental Settings at DOE's Hanford Installation near Richland, Washington: 100-N Region, North of the Columbia River Region, 100-H Region, 200-East Region, 200-West Region, 300-Area Region, and Southwest Region

Once the installation has been divided into environmental settings, data are collected on each known waste site and area of contamination. Typical information includes constituent and waste site-specific history; inventory; soil, groundwater, surface-water, and air concentrations; and areal extent of contamination. The solid, black squares in Figure 6 illustrate the major areas of activity and contamination at the Hanford Installation. The waste sites are further divided into waste-form types. For example, waste sites are designated as dry waste (i.e., solid waste or contaminated soils), liquid-impounded waste (e.g., holding ponds), facilities (e.g., buildings), contaminated surface water (e.g., rivers), groundwater plumes, and other. These waste forms are further divided into "active" and "inactive" categories. "Active" refers to sites that are currently and actively discharging wastes into the environment as a result of human activity (e.g., disposal to an operating landfill, stack or vent release, or liquid discharge due to a metal-plating process). "Inactive" refers to sites that discharge to the environment as a result of natural processes (e.g., leaching from a closed landfill or resuspension of contaminated top soil). Only inactive waste sites are addressed by the HRA-EIS.

Once site data have been collected, a regional GIS grid is established over an 80-km (50-mi) radius surrounding the installation. Each grid-cell size is a function of the amount and quality of data present at the waste sites and size of the installation. For example, because data are limited for the approximately 1800 waste sites at Hanford and because Hanford is a very large installation (1450-km² [560-mi²]), a regional risk and impact assessment would represent the most appropriate analysis. One-kilometer by 1-km grid cells are placed over the Hanford Installation, as illustrated in Figure 6. Waste sites and their characteristics are correlated with each grid cell. Multiple waste sites in each cell are aggregated by constituent, active and inactive categories, and waste-form type. Therefore, at time zero, current levels of contamination in the air, groundwater, surface water, and soil are known for each grid cell. If a more detailed assessment is required, the grid-cell size can be reduced to the size of the waste sites themselves, assuming appropriate data are available on this scale.

2.3 Unit-Transport Analysis

Although the current conditions at the installation have been determined under the source-term analysis, future conditions may have to be assessed. For example, the no-action alternative assesses future impacts to the surrounding population and environment if the installation were to maintain existing conditions. A remediation alternative would assess future impacts to workers, the surrounding population, and the environment due to remediation activities. A residual (i.e., post-remedial) alternative would assess impacts associated with the installation after remedial activities; that is, has remediating the sites reduced contaminant levels to within acceptable limits for planned land-use activities (e.g., agricultural, residential, industrial, or recreational)?

To assess future impacts to surrounding populations, the environment, or workers, a transport analysis must be performed to project where the contaminants might go. An analysis is performed to describe the transport, migration, and fate of contaminants in the environment at locations away from each grid-cell source. The transport analysis temporally and spatially redistributes the contaminants within the 80-km (50-mi) radius for each environmental medium (i.e., soil, groundwater, surface water, and air). The analyst can assess the impacts associated the different alternatives at any user-designated time in the future (e.g., 100 years). Therefore, contaminants are transported from their source in each grid cell to new locations within the grid system. Following redistribution of the

Figure 6. Major Areas of Contamination at the Hanford Installation, and an Illustrative 1-km by 1-km Grid Overlaid on the Hanford Installation (Developed by Advanced Sciences Incorporated, Richland, Washington)

contaminants from the transport analysis, new contaminant levels in the soil, groundwater, surface water, and air can be determined in each grid cell. Figure 7 presents an illustrative example of redistributed contaminant levels from multiple sources within the installation in the form of concentration isopleths (lines of equal concentration) within a groundwater aquifer.

Prior to the transport assessment, a sensitivity analysis is performed to determine conditions of linearity. For example, if the relationship is linear, doubling the emission rate from the source will double the contaminant concentration at the receptor. Under conditions of linearity, it is not necessary to know the source strength prior to performing the transport analysis. If, for example, the assessment is linear with respect to waste-site inventory (e.g., temporally far-field case), a unit inventory (e.g., 1 Ci or 1 kg) could be assumed for the waste site, resulting in a unit environmental concentration at the receptor. When the inventory is determined at the source, the predicted concentration at the receptor is the unit concentration times the inventory. In effect, a UTA can be performed at the installation. This unit-transport approach could also be used to determine the degree of cleanup that is required to meet a particular clean-up standard. For example, once the contaminant clean level is determined at the source, the UTA could identify the degree to which the inventory must be reduced at the site to meet a maximum contaminant level (e.g., drinking-water standard).

The UTA represents a powerful, cost-effective tool in assessing installations. First, once the environmental settings have been identified, waste-site-data gathering can be implemented concurrently with the UTA (and unit-exposure). Second, if the waste-site information is updated in the future, the UTA need NOT be performed again. Because of linearity, the new inventory (assuming linearity with inventory) is multiplied by the unit concentration at the receptor to provide the new predicted concentration at the receptor. Each time the source is changed, the predicted concentration at the receptor is determined by simple multiplication. In effect, the transport computer codes have to be run only once, which results in tremendous time and cost savings.

2.4 Unit-Exposure Analysis

An exposure analysis 1) identifies the routes through which the contaminants make their way through the food chain to expose surrounding sensitive receptors and 2) determines the intake rate associated with the exposure scenario. Table 1 illustrates several of the main routes of exposure from ingestion, inhalation, dermal contact, and external dose. Because dose or risk is linear with respect to environmental concentration, a unit-exposure analysis can be compartmentalized in a manner that is very similar to the unit-transport analysis; in effect, the exposure analysis can be implemented independently of the transport analysis. For example, if a receptor is assumed to ingest 2 L/d (5 gal/d) of water over a lifetime, the contaminant concentration in the water has no effect on the ingestion rate, although it does affect the dose/risk associated with the exposure. When in the linear range, the dose/risk is proportional to the intake rate; consequently, a unit-exposure analysis can be performed independently of source-term and unit-transport assessments.

All of the various exposure options, even those based on land-use patterns, can be performed for each grid cell within the 80-km (50-mi) radius. The analyst would then identify for each grid cell the exposure options associated with that cell. The analyst will, in effect, determine the land-use pattern for the grid cell by activating or deactivating various exposure options. For example, by activating the agricultural-scenario option, groundwater would be used to irrigate crops that are consumed 1) directly

Figure 7. Illustrative Concentration Isopleths (Developed by Advanced Sciences Incorporated, Richland, Washington)

Table 1. Illustrated Exposure Routes and Scenarios (Droppo et al. 1989a)

1. Air/Surface soil → Inhalation, External exposure,^(a) and Soil ingestion
2. Air/Surface soil → Crops → Ingestion
3. Air/Surface soil → Crops → Animals → Ingestion
4. Groundwater → Ingestion
5. Groundwater → Bathing → Ingestion/Inhalation
6. Groundwater → Irrigation → Crops → Ingestion
7. Groundwater → Irrigation → Crops → Animals → Ingestion
8. Groundwater → Animals → Ingestion
9. Surface water → Ingestion
10. Surface water → Fish/shellfish → Ingestion
11. Surface water → Irrigation → Crops → Ingestion
12. Surface water → Irrigation → Crops → Animals → Ingestion
13. Surface water → Animals → Ingestion
14. Surface water → Bathing → Ingestion
15. Surface water → Recreation → External exposure
16. Surface water → Recreation → Ingestion
17. Direct ingestion of surface soil
18. Measured food concentrations
19. Measured radiation doses^(a)

(a) For radionuclides only.

by the surrounding population and/or 2) by milk- and meat-producing cattle that are then consumed by the surrounding population; the water would not be used for personal consumption. By activating the residential scenario instead, a family would utilize the water for personal use (e.g., drinking, bathing) but not to irrigate crops. By providing the analyst with this flexibility, options and land-use patterns can be easily compared and assessed for any location within the 80-km (50-mi) radius.

2.5 Dose/Risk Calculation

Once the actual source strength and contaminant concentrations have been identified within each grid cell, based on either monitored information (e.g., current conditions) or a predicted transport assessment, the risk or dose can be calculated for each grid cell. Equation (1) illustrated the simple multiplicative calculation that is utilized in determining the risk/dose for each grid cell and the utility of compartmentalizing the assessment. Even if the transport requires re-evaluation, the source-term and exposure assessments do not have to be re-evaluated. Likewise, if the source-term or exposure component is re-evaluated and updated, the transport component does not require re-evaluation.

Although compartmentalizing the assessment utilizes limited resources in the most efficient manner, the strength of the system is its flexibility in presenting the results of the analyses. As in the case in Figure 7, risk/dose isopleths (i.e., contours of equal risk/dose) for each option can be determined for the entire area surrounding the installation. Figure 1 illustrated the two- and three-dimensional risk isopleths based on the concentration isopleths presented in Figure 7. By presenting the results in three-dimensional plots, interested parties can instantantly see regional areas of concern. By compartmentalizing the assessment, the analyst can quickly determine the waste sites, transport pathways, and exposure routes that dominate the risk. In addition, the results will allow the analyst to quickly and visually inspect the short- and long-term ramifications of different land-use patterns and remedial options. The results for different options can be overlaid to determine the strengths and weaknesses associated with each option as it relates to environmental concentration, dose, or risk.

3.0 Calculations for Linearity

Source-term area, mass, and concentration represent the important parameters driving the linearity associated with the MRA at Hanford. The importance of these parameters stems from the following assumptions of the assessment:

- Contaminant movement through the subsurface environment for all non-standing-water waste sites (e.g., contaminated soils) is mainly driven by precipitation, resulting in deep-drainage percolation.
- For those sites that do not contain concentration information, the source-term concentration (i.e., total concentration) will be determined from the mass reported for the waste site divided by the total volume identified for the waste site:

$$C_T = \frac{M}{V_T} = \frac{M}{A T} \quad (2)$$

where C_T is the total concentration (mass per total volume), M is the mass or inventory of the constituent, V_T is the total volume of contamination at the site, A is the area of contamination, and T is the thickness of the contamination. C_T is used to calculate the dissolved concentration as follows:

$$C_o = \frac{C_T}{\theta + \beta_d K_d} \quad (3)$$

where C_o is the dissolved concentration, θ is the moisture content, β_d is the bulk density, and K_d is the distribution (partition or equilibrium) coefficient for the chemical of concern.

- The contaminant mass flux from the waste site is assumed to be equal to the water mass flux times the dissolved contaminant concentration:

$$Q_c = C_o Q \quad (4)$$

where Q_c is the contaminant mass flux and Q is the water volumetric flux rate, defined as

$$Q = i A \quad (5)$$

where i is the deep drainage-percolation rate (i.e., infiltration rate, Darcy velocity, or volumetric water discharge per unit area).

Based on Equations (2) through (4), the contaminant mass flux from the waste site can be calculated as follows:

$$Q_c = \frac{M i}{T (\theta + \beta_d K_d)} \quad (6)$$

Based on inspection of Equation (6), the contaminant mass flux and, hence, the receptor concentration and risk for a given contaminant appear to be proportional and solely based on the mass at the waste site, assuming the same geologic setting. The mass flux will only have an impact on the receptor concentration as long as the concentration is below some limiting amount (e.g., C_o).

Although the area appears to have no impact on the resulting receptor concentrations, its effects appear in the solution to the advective-dispersive equation:

$$\frac{2C}{2t} + u_p^* \frac{\partial C}{\partial x} = D_x^* \frac{\partial^2 C}{\partial x^2} + D_y^* \frac{\partial^2 C}{\partial y^2} + D_z^* \frac{\partial^2 C}{\partial z^2} - \lambda C. \quad (7)$$

The instantaneous solution of advective-dispersive equation for an area source in a saturated zone is as follows:

$$C_i = \delta X Y Z \quad \text{where } X \equiv \text{flow direction} \quad (8)$$

in which

$$\delta = M / (R_f \eta_e) \quad (9)$$

$$X = \left[\frac{1}{2L} \right] \exp(-\lambda t) \left[\operatorname{erf} \left[\frac{x + \frac{L}{2} - u_p^* t}{(4 D_x^* t)^{0.5}} \right] - \operatorname{erf} \left[\frac{x - \frac{L}{2} - u_p^* t}{(4 D_x^* t)^{0.5}} \right] \right] \quad (10)$$

$$Y = \left[\frac{1}{2W} \right] \left[\operatorname{erf} \left[\frac{y + \frac{W}{2}}{(4 D_y^* t)^{0.5}} \right] - \operatorname{erf} \left[\frac{y - \frac{W}{2}}{(4 D_y^* t)^{0.5}} \right] \right] \quad (10)$$

$$Z = \left[\frac{1}{H} \right] \left\{ 1 + 2 \sum_{n=1}^{\infty} \left[\exp \left[- \frac{n^2 \pi^2 D_z^* t}{H^2} \right] \cos \left[\frac{n \pi z}{H} \right] \right] \right\} \quad (12)$$

$$R_f = 1 + \frac{\beta_d K_d}{\eta_e} \quad (13)$$

$$D^* = \frac{D + 190 \text{ cm}^2/\text{yr}}{R_f} \quad (14)$$

$$D = \alpha u_p \quad (15)$$

$$u^* = u_p / R_f \quad (16)$$

$$v_d = u_p \eta_e \quad (17)$$

$$A = L W \quad (18)$$

where C_i is the instantaneous concentration solution to the advective-dispersive equation; δ is a mass-related constant; X , Y , and Z are Green's functions in the x , y , and z directions, respectively; R_f is the retardation factor; η_e is the effective porosity; L is the length of the waste site in the groundwater flow direction; λ is the first-order degradation/decay constant; t is time; x is the travel distance; u_p^* is the pore-water contaminant velocity; D is the dispersion coefficient; W is the width of the waste site perpendicular to the groundwater flow direction; h is the aquifer thickness; v_d is the Darcy velocity; n is the index on the summation; $190 \text{ cm}^2/\text{yr}$ equals molecular diffusion; α is the dispersivity in the x , y , or z direction; A is the area of the waste site.

The contaminant mass flux from the waste site and the instantaneous concentration are combined through the convolution integral to provide the concentration at the receptor:

$$C(\tau) = \int_0^\tau Q_c(t) C_i(\tau - t) dt \quad (19)$$

where $C(\tau)$ is the concentration at the receptor at time τ , and τ is the limit of the integral representing the current time associated with the concentration.

By inspection, Equations (10) and (11) illustrate that length and width of the waste site play an integral part in the determining the concentration at the receptor; as a result, area may become an important variable for near-field problems. Spatially, far-field problems are those for which an area source looks like a point source. Temporally, the convolution integral of far-field problems simplifies to the instantaneous-concentration solution as presented in Equation (8).

The MRA analysis of the aggregated waste sites assumed that the areas were all 929 m^2 ($10,000 \text{ ft}^2$; 100 ft by 100 ft). To demonstrate the linearity of the system and to show that the assessment at Hanford is spatially far-field, an analysis was performed on an illustrative waste site in the 200-E Area. A unit release rate was assumed for the waste site. An analysis was performed that assessed how the size of the waste site affected the downgradient concentration at four distances. The

Figure 8. Concentration Versus Distance Versus Area, Demonstrating Spacially Far-Field Effects at Hanford's 200-W Area

results of this analysis are presented in Figure 8. Figure 8 presents peak concentration as a function waste-site area and downgradient distance. Four areas were assessed: point source, 10 m by 10 m (33 ft by 33 ft), 100 m by 100 m (328 ft by 328 ft), and 1000 m by 1000 m (3280 ft by 3280 ft). Seven distances were assessed: 10 m, 75 m, 100 m, 750 m, 1000 m, 7500 m, and 10,000 m (32 ft, 246 ft, 328 ft, 2460 ft, 3280 ft, 24,606 ft, and 32,808 ft). As Figure 8 illustrates, all of the areas produce results similar to that of the point-source solution at a distance of 1 km or more from the source, clearly indicating spatially far-field conditions at several kilometers from the source.

Because the aggregated waste sites are less than $(.6 \text{ mi}^2) 1 \text{ km}^2$ in size and because spatially far-field conditions have been demonstrated for grid cells at least 1 km from the cells containing the aggregated waste sites, the mass (inventory) or activity at the source represents the only source-term parameter that directly affects the downgradient receptor concentrations (as illustrated by Equation (6)). Because mass or activity is linear to receptor concentration, the unit transport analysis is based on a unit mass of 1 kg (2.2 lb) for chemicals and 1 Ci for radionuclides.

4.0 Implementation Steps of MRA At Hanford

To meet the end points associated with the MRA, a ten-step process was implemented. Each of these steps is briefly discussed below.

1. **Connect Installation to State GIS** -- MRA approach was connected to the state of Washington GIS.
2. **Subdivide Installation** -- The Hanford Installation was divided into a grid system within an 80-km (50-mi) radius of the center of the installation. The Hanford Installation was then divided into 1-km² cells, as illustrated by Figure 6. The dark boxes in this figure illustrate the operational areas at Hanford. Because the MRA approach is coupled to the state GIS, each of the 1-km² cells can be evaluated at an even higher resolution, as illustrated in Figure 9. Figure 9 presents an illustrative example of a cell from the 100-BC Area. Within this figure of higher resolution, the circular dots represent the locations of the actual waste sites within this cell. The waste sites in each 1-km² (.6-mi²) cell were aggregated according to waste-form type (e.g., contaminated soil versus ponded waste), excluding contaminated plumes that currently exist in the aquifer beneath Hanford.
3. **Aggregate Waste Sites** -- Because approximately 1800 waste sites have been identified at Hanford, it would not be productive to analyze each site individually to determine the regional impacts. From a regional perspective, multiple waste sites at the same geographical location will have an appearance similar to that of a large, single waste site. For example, a source with an areal distribution will begin to appear as a point source as the viewer moves farther away from the source. As a result, the impacts associated with the areal distribution should approach the predicted impacts that would occur from a point source as the distance from the source increases. This phenomenon is known as being spatially far-field. By using this analogy, which is clearly demonstrated to be true at Hanford (see the section titled "Calculations for Linearity"), the waste sites in any 1-km² (.6-mi²) cell can be combined and be represented as an aggregated waste site. Similar waste types would be combined with similar waste types (e.g., contaminated soils with contaminated soils and ponded liquids with ponded liquids). Figure 10 illustrates the cells in the GIS for Hanford that contain the aggregated waste sites, designating them with a square dot.
4. **Define Initial Conditions** -- Source-term characteristics for each aggregated waste site were determined from various databases maintained at Hanford and reports written on the Installation. Source-term, geologic, hydrologic, hydraulic, meteorologic, and population data were collected or calculated for each grid cell within an 80-km, (50-mi) radius of the center of the installation, where appropriate. Environmental contaminant concentrations and inventories were also determined within this 80-km (50-mi) radius of the center of the installation. These contaminant levels, which were determined through monitoring for current conditions, will be used for transport modeling for future conditions. From the source-term characteristics, four concentrations were defined for each cell at the designated time zero (i.e., 1989): groundwater, surface-water, air, and soil concentrations.
5. **Define Land-Use Patterns and Exposure Scenarios** -- Land-use patterns and exposure scenarios are identified for the air, groundwater, surface water, and soil at all grid-cell locations within an

Figure 9. Illustrative Example of the High Resolution of Waste Sites in the 100-BC Area Using the State GIS

Figure 10. Aggregated Waste Sites at Hanford that Are Mapped onto the State GIS 1-km² Grid System

80-km (50-mi) radius. The land-use patterns and corresponding exposure scenarios were those defined by the Hanford Site Baseline Risk Assessment Methodology (HSBRAM) as agreed to in the Tri-Party Agreement between DOE, the Washington State Department of Ecology, and the U.S. Environmental Protection Agency (EPA). The HSBRAM approach is documented and demonstrated in Strenge and Chamberlain (1994). Land-use scenarios are defined for agricultural, industrial, residential, and recreational activities. For information on the land-use patterns and their corresponding exposure scenarios, refer to Strenge and Chamberlain (1994).

6. **Determine Flow Paths** -- To calculate concentrations at future times at locations other than the sources, pathlines were determined from the cells containing the aggregated waste sites to surrounding locations. For example, Figure 11 presents the pathlines through the groundwater to the Columbia River from the cells containing the aggregated waste sites. These pathlines were determined using the CFEST computer model (Cole et al. 1988) for December 1992 flow conditions at Hanford. To perform the transport analysis, these pathlines were simplified and represented by a sequence of straight lines. The pathline representation that was utilized in the assessment is illustrated in Figure 12. A similar analysis is performed for the atmospheric pathway, which uses a sector-average-Gaussian approach to distribute contaminants that are released into the environment via volatilization or resuspension of contaminated particles. A computer program was written to translate the atmospheric polar coordinate system and overlay it into the state GIS cartesian coordinate system.
7. **Implement Transport and Exposure Analyses** -- When the pathline analysis was completed, the transport analysis could be performed. Because the receptor concentrations were determined to be linear with respect to inventory or activity at the source, unit source-term conditions were assumed at each of the waste sites. An activity of 1 Ci was assumed for radionuclides, and an inventory of 1 kg was assumed for nonradioactive chemicals. The release to the environment (i.e., to air, subsurface, and surface water) was calculated for contaminated soil sites. Volatile and nonvolatile contaminants, and contamination associated with soil and surface water were addressed as part of the unit transport analysis. The multimedia model MEPAS was utilized to simulate the release of the various contaminants and the contaminants migration through the air, vadose zone, saturated zone, and surface water. The transport of contaminants from ponded sites were modeled separately, as they were not linear with respect to either mass or activity alone. Four concentrations were calculated for each grid cell within the 80-km (50-mi) radius: groundwater, surface water, air, and soil.

Exposure analyses for the four land-use patterns for the HSBRAM exposure scenarios were performed. Because the exposure/risk calculations are independent of location, only four exposure scenarios need be analyzed: recreational, residential, industrial, and agricultural. Following analysis, the analyst then defined a given land-use pattern for each grid-cell location; they may be the same or different from cell to cell; the user has the option for distributing the appropriate land-use pattern throughout the 80-km (50-mi) radius. By coupling the demographic characteristics and unit concentrations for each of the four environmental media (i.e., groundwater, surface water, air, and soil), a unit risk is determined in each grid cell.

8. **Collate Cell Concentrations** -- Because plumes from different waste sites can overlap, the concentrations from multiple waste sites at downgradient cells must be addressed. When multiple waste sites transport contamination to the same downgradient cell, the concentrations are added

Figure 11. Pathlines, as Developed by CFEST Modeling, from Aggregated Waste Sites to the Columbia River

Figure 12. Pathlines that Are Approximated by a Series of Straight Lines

by constituent. The concentrations can be added because of the linearity of the system. Following the transport, migration, and fate simulation, only four concentrations were defined for each cell: groundwater, surface water, soil, and air.

9. **Calculate Risk for Each Land-Use Pattern** -- The risk is calculated by simply multiplying the source times the unit transport analysis times the unit exposure/risk analysis for each constituent for each cell. Dose/risk was calculated as it is spatially distributed at all grid-cell locations within an 80-km (50-mi) radius for times of 0, 40, 100, 140, 1000, and 10,000 years.
10. **Plot Two- and Three-Dimensional Concentration and Risk Isopleths** -- Based on air, groundwater, surface water, and soil concentrations, two- and three-dimensional isopleths were easily plotted to visualize the results.

5.0 Pathline Analysis Using the Coupled Fluid, Energy, and Solute Transport (CFEST) Model

To generate pathlines for input to MEPAS, the unconfined aquifer at the Hanford Installation was simulated with the two-dimensional version of the Hanford Installation groundwater model (Wurstner and Devary 1993). This model consists of the CFEST (Gupta et al. 1987) groundwater modeling library integrated with the commercially available geographic information system ARC/INFO. The model is used to support work under the Groundwater Surveillance Project at the Hanford Installation and has been applied to other sites as well.

5.1 Description of the CFEST Code

The CFEST code was originally designed to support the radioactive waste repository investigations sponsored by DOE's Civilian Radioactive Waste Management Program. It has also been effectively used by the chemical waste management community for conducting exposure assessments, evaluating remediation alternatives, and designing extraction and control systems for aquifers.

5.2 Physical Processes Modeled by CFEST

The CFEST code solves partial differential equations for fluid pressure, temperature, and solute concentration for multilayered, confined hydrologic systems using the finite-element method. Options exist to solve the equations for pressure, temperature, and solute concentration in either an uncoupled or a coupled form. Fluid properties of density and viscosity are used to couple the equations for simulations requiring variable density solutions. Solution of the system of coupled equations is based on linearization, with the latest iteration of pressure, temperature, and solute concentration used to compute fluid and aquifer properties for the next iteration.

Phreatic solutions can be computed for the uncoupled equations through an iterative technique that adjusts the saturated thickness so that the calculated head is the top of the system. The user has the option to solve for any or all of the dependent variables. The code is designed to simulate transient or steady-state fluid flow coupled with energy and/or solute transport. Because Hanford simulations currently do not consider differences in fluid density or viscosity, only the uncoupled equation option is used for Hanford Installation simulations.

5.3 Aquifer Geometry

In the Cartesian coordinate system, the code can be used for simulation in a horizontal plane, a vertical plane, or a fully three-dimensional regime. An option also exists for the axisymmetric analysis of a vertical cross section.

The CFEST finite-element formulation has the capability to model discontinuities, major breaks in slope or thickness, and fault zones in individual hydrogeologic units. Surface-water bodies (lake, river, seashore), recharge or pumping wells, and variations in major land uses may be modeled using the appropriate grid (node locations).

5.4 Heterogeneity

The code models heterogeneity in aquifer permeability and porosity. Anisotropy (co-linear with the Cartesian coordinates) is also accommodated. The variation in the hydraulic properties may be described homogeneously within layers or heterogeneously on an elemental basis for aquifers exhibiting a certain degree of geologic complexity.

5.5 Boundary Conditions

The code includes options for both constant and time-variant Dirichlet and Neumann boundary conditions. The Dirichlet and Neumann boundary conditions can be specified individually for each dependent variable. For example, a given node may have a specified concentration (Dirichlet) as well as a specified fluid flux (Neumann).

5.6 Initial Conditions

The user can specify the following initial conditions:

1. Hydraulic Head or Pressure -- Constant values for hydraulic head or pressure are specified throughout the region for cases of constant and variable density.
2. Temperature -- Constant temperature, temperature as a function of depth, or independent nodal values of temperature may be specified at each node.
3. Concentration -- Constant or independent nodal values of concentration may be specified at each node.

The CFEST software library was extensively tested and brought under strict software Quality Assurance/Quality Control procedures by the Office of Nuclear Waste Isolation (ONWI). A super-computer version (CFEST-SC) was developed to run on all major Unix work stations (Cole et al. 1988). The CFEST output is now graphically displayed using the ARC/INFO GIS.

5.7 Description of the CFEST/GIS Link

A GIS consists of tabular databases and geographic databases (e.g., well locations or facilities maps) linked together with relational database software. This enables the user to easily perform sophisticated spatial data analyses that might take hours or days using conventional methods. The display and analysis capabilities of a GIS are well suited to the size and diversity of databases being

generated at the Hanford Installation. The ability to visually inspect large databases through a graphical analysis tool provides a stable foundation for site assessments and groundwater modeling studies. The GIS capability of color coding data greatly aids in the visual recognition of data trends, particularly in the inspection of large numbers of chemical and radiological measurements.

The commercially available GIS ARC/INFO has been integrated with the CFEST groundwater modeling library. This GIS/modeling capability arose from the need for effective database management and graphical tools to support ongoing environmental activities coupled with the need for more effective graphical interfaces for the groundwater models. A series of ARC/INFO macro routines and FORTRAN utility programs have been developed to create an ARC/INFO - CFEST interface. For example, an ARC/INFO macro may be used to select elements that represent starting points for particle travel analyses. A FORTRAN utility program will then generate a command file used to execute the CFEST travel path module. Another ARC/INFO macro has been written to create a triangular irregular network surface from CFEST output, from which contour maps can be generated. Additional ARC/INFO macros for grid generation and parameter assignment are being used in support of the three-dimensional model development under the Groundwater Surveillance Project.

5.8 Groundwater Model Description

The finite-element grid used by the Hanford Installation groundwater model contains 997 nodes and 899 elements. Boundary conditions for the model consist primarily of prescribed head along the Columbia and Yakima Rivers, no-flow boundaries at the bottom of the aquifer and along basalt outcrops and subcrops, and constant flux along a portion of Rattlesnake Mountain to reflect spring discharges. A prescribed head boundary was also used at the entrance of Cold Creek Valley onto the Hanford Installation. Figure 13 shows the finite-element grid with boundary conditions. The transmissivity distribution (Figure 14) and initial conditions for the groundwater model were derived from the inverse calibration performed for 1979 steady-state conditions (Jacobson and Freshley 1990). A transient simulation was run for a 12-year period from 1980 to 1992 using a time step of 1 month. The monthly effluent discharges used for this simulation were based on Westinghouse Hanford Company (WHC) effluent reports. The discharge estimates were revised based on input from staff members responsible for effluent monitoring (the Groundwater Surveillance Project maintains a database of these effluent discharges). There are relatively few measurements of storativity for the unconfined aquifer, so a constant value of 0.1 was assumed. No natural recharge was included in this simulation, although it is recognized to occur on the Hanford Installation (Gee et al. 1992). Natural recharge is being included in future work with the two- and three-dimensional models being developed by the Groundwater Surveillance Project. Interconnection between the unconfined aquifer and the deeper basalt aquifers is also not accounted for in the model.

5.9 Pathline Analysis

Flow paths were calculated based on December 1992 flow conditions (the most current represented by the model; Figure 15). It was assumed that those flow conditions will remain constant for the duration of the particle tracking. Particle paths were started at elements which contain cells representing the waste sites (Figure 16) and tracked until they reached a model boundary. Figures 10 and 17 show

Figure 13. Finite-Element Grid System and Boundary Conditions for the CFEST Modeling to Determine Flow Pathlines from Aggregated Waste Sites to the Columbia River

Figure 14. Spatial Distribution of Transmissivity in the Aquifer Beneath the Hanford Installation, Overlaid on the Finite-Element Grid Used for CFEST Modeling

Figure 15. Spacial Distribution of Hydraulic Head Contours in the Aquifer Beneath the Hanford Installation During December 1992

Figure 16. CFEST Finite-Element Cells that Correspond to the Location of Aggregated Waste Sites

Figure 17. Locations of Aggregated Waste Sites and the CFEST Finite-Element Cells

the location of the cells representing waste sites in relation to the 1-km (.6-mi) grid and the finite-element grid, respectively. The pathlines generated by the model are shown in Figure 11. These pathlines were then approximated to straight-line segments using an ARC/INFO function that identifies coordinates in a line by finding the trend of the line and eliminating vertices that fall within the specified tolerable distance of the trend line. The weed tolerance used was 1 km (.6 mi), which is equal to the cell size. The resulting pathlines consist of one to five straight-line segments (Figure 12). The points representing the vertices of the pathlines were then used in MEPAS to describe the travel paths from waste sites.

5.10 Model Assumptions and Limitations

There is a certain degree of uncertainty inherent in all models that comes from the assumptions made when developing and applying a model. Many assumptions are associated with the current Hanford Installation regional groundwater model that may no longer be appropriate. Since development of the conceptual model, new hydrogeologic data have been collected, new interpretations have been made, and the hydrogeologic conditions and driving forces (e.g., boundary conditions) at the Hanford Installation have changed. The current model was developed based on the assumption that artificial recharge from Installation operations was much greater than any natural recharge from precipitation or the basalt aquifers below the unconfined aquifer system at Hanford. In the past, it was reasonable to ignore natural recharge since the flow system was dominated by effects from artificial recharge. However, artificial recharge is decreasing at the present time and is projected to decrease even more. As the flow system approaches pre-Hanford conditions, this assumption becomes invalid and it becomes crucial to include natural recharge in the model to accurately represent the water balance and the driving forces that control water movement.

Additional sources for uncertainty include the use of a constant value for specific yield and a constant average head value assigned to the Columbia and Yakima rivers. In addition, the hydraulic conductivity distribution used in the model is based on an averaging of properties across Hanford and Ringold formation layers into a transmissivity value for each element. The flow system response to water being released from storage in the Hanford formation, for example, will be quite different from the response observed in the Ringold sediments because of the difference in hydraulic properties. This assumption may greatly influence the results of a transient, two-dimensional simulation. In addition, the size of the finite elements is large in some areas, resulting in poor resolution for the solution of groundwater flow.

6.0 Unit Transport Analysis Using MEPAS

MEPAS is a fully integrated, physics-based, PC-platform, multimedia transport- and risk-computation code that is used to assess health impacts from actual and potential releases of both hazardous chemicals and radioactive materials. MEPAS is designed for site-specific assessments using readily available information. It follows DOE/EPA/U.S. Nuclear Regulatory Commission (NRC) risk-assessment guidance, as it evaluates 1) the release of contaminants into the environment; 2) contaminant movement through and transfer between various environmental media (i.e., subsurface [vadose and saturated zones], surface water, overland [surface soil], and atmospheric); 3) exposure to surrounding sensitive receptors via inhalation, ingestion, dermal contact, and external dose; and 4) risk for carcinogens and hazard for noncarcinogens. For radionuclides, MEPAS follows International Commission on Radiological Protection/National Council on Radiation Protection and Measurements (ICRP/NCRP) and EPA guidelines, allowing the user to choose the appropriate guidelines. MEPAS is designed as a comprehensive hazard assessment tool that provides risk assessments that meet or exceed requirements for the remedial investigation/feasibility studies required by CERCLA.

The transport codes comprised by MEPAS are based on analytical or semianalytical solutions to the advective-dispersive equations (subsurface, surface water, and atmospheric) or empirically based solutions (overland transport) using well-accepted approaches. The waterborne components in MEPAS address transient short- or long-term contaminant releases into the environment. The atmospheric component in MEPAS addresses long-term releases. The exposure and health consequence components in MEPAS address chronic-lifetime risk to individuals and/or populations. MEPAS includes a contaminant database of over 400 chemicals and radionuclides.

MEPAS has been through a number of technical reviews, including the following: 1) independent, external, technical peer review (1986), 2) EPA tests MEPAS as prioritization methodology (1986-1987), 3) private corporations select MEPAS in response to California's Proposition 65, 4) DOE selects MEPAS for Environmental Survey (1986), 5) Idaho National Engineering Laboratory (INEL) reviews and selects MEPAS for risk screening applications (1989), 6) National Academy of Sciences (NAS) reviews MEPAS as part of Single-Shell Tank Analysis (1989), 7) EPA reviews MEPAS as part of DOE Priority System (1991), 8) Health and Welfare, Canada selects MEPAS as a result of their independent review of 22 multimedia models (1992), 9) NAS reviews MEPAS as part of DoD Prioritization System (1992), 10) Oak Ridge National Laboratory (ORNL) reviews models and selects MEPAS for the DOE complex-wide PEIS (1992), 11) EPA-Region 10 selects MEPAS for assessments at selected Alaskan CERCLA sites (1992-1993), and 12) U.S. Department of Energy Richland, Operations Office (DOE-RL) and Advanced Sciences, Incorporated (ASI), select MEPAS for the HRA-EIS (1994), and 13) independent, external, technical peer review (1994) as part of the PEIS. MEPAS is currently undergoing benchmarking with Residual Radioactive Material (RESRAD) and MMSOILS.

The more noted applications of MEPAS include the following: 1) 20 applications on CERCLA sites (1986-1987), 2) private clients use MEPAS to assess potential risks (1986-1990), 3) DOE Preliminary Environmental Survey uses MEPAS for 16 installations (1987-1988), 4) Hanford Single-Shell Tanks projects assess relative constituent rankings (1989), 5) MEPAS sensitivity study analyzes important input variables (1990), 6) DOE Risk Information System (RIS) report ranking issues including uncertainty (1990), 7) Hanford grout studies used for hazardous waste risks (1990), 8) DOE Environmental Survey uses MEPAS for 36 installations (1988-1990), 9) State of Washington Risk-

Based standards (1990), 10) University of Indiana, University of Washington, Washington State University, and Mesa State College (Colorado) using MEPAS as a teaching tool (1991-1993), 11) graduate theses/dissertations utilize MEPAS (1991-1993), 12) linkage with Remedial Action Assessment System (RAAS) (1991-1993), 13) Office of Management and Budget (OMB) Pilot Study by ORNL at selected DOE sites (1991), 14) Hanford 100-Area Study groundwater plume risk maps (1992), 15) Hanford Mission Plan (HMP) -- Site Integrated Risk (1992-1993), 16) DOE PEIS using MEPAS (1992-1994), 17) Eielson Air Force Base -- Fate and Transport Modeling (1992-1993), 18) Spent-Nuclear-Fuels EIS at Hanford (1993); and 19) 324 Building Emergency assessment at Hanford (1994).

MEPAS was chosen because it is the only multimedia computer model that includes all of the following elements: MEPAS 1) addresses radioactive, organic, and inorganic wastes; 2) provides user flexibility in describing the geology, hydrology, hydraulics, and meteorology at an installation by allowing the use of site-specific data; 3) performs on- and off-site calculations; 4) is largely based on the solutions to the advective-dispersive equations for solute transport; 5) includes atmospheric complex terrain, channeling of winds, fugitive dust emissions, wet and dry deposition, and gaseous and particulate releases; 6) addresses both active and inactive sites and releases; 7) allows for user- or code-specified time-varying (i.e., transient), source-term emission rates; 8) addresses both active and inactive sites; and 9) addresses contaminated soils, ponded sites, liquid discharges, injection wells, and point, line, and area sources.

7.0 Summary of Source-Term and Environmental Setting Assumptions

The HRA-EIS unit transport analysis is based on data by Schramke et al. (1994) and the DOE complex-wide PEIS (Short and Smith 1995). As noted earlier, the Hanford Installation has been divided into one climatic setting, two atmospheric dispersion settings, five major hydrogeologic environmental settings, and one river setting (i.e., Columbia River). The five major geohydrologic settings are illustrated in Figure 5. Each hydrogeologic environmental setting comprises an overburden soil layer (i.e., top-soil layer), a vadose zone, and a saturated zone. Each of the environmental settings has been divided into several vadose zones. The 300, 200-W, 200-E, 100-H, and 100-N Areas contain one, five, three, one, and two vadose zone layers, respectively. This section discusses the following environmental settings: climatologic and atmospheric, geohydrologic, and surface water. Because contaminant mobility is a function of the geohydrologic environmental setting and the distribution coefficient assigned to each contaminant, a discussion is presented on the distribution coefficients that were assigned to contaminants of concern.

7.1 Climatologic and Atmospheric Settings

Climatologic and Atmospheric settings are from Schramke et al. (1994) and Stone et al. (1983). The Hanford Installation has a mid-latitude ($\approx 46^\circ\text{N}$), semiarid climate. The area's climate is strongly influenced by the Cascade Range, which moderates the influence of the Pacific Ocean, resulting in a more continental-type climate. The mountains limit much of the maritime influence of the Pacific Ocean, resulting in a much lower rate of precipitation than would normally occur if the mountains were not present. The Rocky Mountains (far to the north and east) also influence the climate of the region by protecting the region from the more severe winter storms and extreme temperature swings associated with modified arctic air masses that move southward from Canada.

Continuous hourly observations have been conducted at the Hanford Meteorology Station since 1944. Currently, 28 automated meteorological monitoring stations operate within and near the Hanford Installation. The meteorological stations indicate that average-annual precipitation at Hanford is on the order of 16 cm (6.3 in.) with more than 40% of the annual precipitation occurring between November and January. The average-annual snowfall is 33 cm (13 in.). The average-annual temperature at Hanford is 12°C (53°F) with daily-average high temperatures of 3°C (37°F) in January and 33°C (92°F) in July. Daily-average low temperatures are -6°C (22°F) in January and 16°C (61°F) in July.

Winds at Hanford are strongly influenced by their proximity to local terrain features; consequently, two atmospheric settings have been defined: 200 Areas and 300/400 Areas. Due to the Rattlesnake Hills, the prevailing wind direction in the 200 Area is from the northwest with an average velocity of 3.4 m/s (7.6 mi/hr). In the 300 and 400 Areas, the prevailing wind direction is from the southwest. For more information on climatologic and atmospheric conditions at Hanford (e.g., wind rose, joint frequency distributions), refer to Schramke et al. (1994) and Stone et al. (1983).

7.2 Geohydrologic Environmental Settings

The Hanford Installation is located within the Pasco Basin, a structural depression on the Columbia Plateau. The geology of the area is described in detail by Schramke et al. (1994), Thorne and Chamness (1992), and Tallman et al. (1979). The principal surface water features at Hanford are the Columbia and Yakima Rivers, north and east, and south of the installation, respectively. Small amounts of local surface recharge to the basin occur from precipitation and through small ephemeral streams to the west of the area. Although infiltration amounts have ranged up to 10 cm/yr (4 in./yr) (Whelan et al. 1989), 1 to 2 cm/yr (0.4 to 0.8 in./yr) is not unrealistic (depending on weather conditions), and most precipitation is returned to the atmosphere through evapotranspiration (Bjornstad 1990).

Local recharge makes its way through the vadose zone to the water table. The unconfined aquifer at Hanford occurs primarily within the Hanford formation and Ringold Formation. Prior to operational disposal of waste waters, the unconfined aquifer was almost entirely in the less transmissive Ringold Formation; however, waste-water disposal in the 200 Areas has created a groundwater mound, and the water is now moving through the more transmissive Hanford formation (Thorne and Newcomer 1992). Because the waste-water disposal facilities have shutdown, the water table is declining, and estimates of groundwater travel times and directions based on 1992 water surface elevations are expected to be conservative. Figure 15 presents the spacial distribution of hydraulic head contours in the aquifer beneath the Hanford Installation during December 1992. Figure 11 presents the pathlines for the aquifer conditions during December 1992.

Based on the available hydrogeologic data and consultation with scientists and engineers involved in site characterization at Hanford, (Schramke et al. 1994) divided the Hanford Installation into seven hydrogeologic settings (i.e., 200-E, 200-W, 100-N, 100-H, 300, Southwest, and North Areas). Figure 18 presents the hydrostatigraphy of the seven Hanford Installation environmental settings. Of the seven settings, only five contain significant past-practice waste sites (i.e., 200-E, 200-W, 100-N, 100-H, and 300 Areas); these five are associated with the UTA. Criteria used to establish the settings included the hydrogeologic characteristics of the Hanford formation, location of the water table, and number of distinct hydrogeologic units that have been described for various locations beneath the Hanford Installation. For each of these settings, a single set of hydrogeologic input parameters was selected. The hydrogeologic input parameters are representative of the specific areas where the waste sites occur within the settings. Schramke et al. (1994) present a description of hydrogeology of each environmental setting.

A summation of the characteristics associated with each environmental setting is presented in Tables 2 through 16. Tables 2 through 4 present a summary of the hydrogeologic characteristics for the 100-N Area environmental setting. Table 2 presents the infiltration rate, Table 3 presents the vadose zone parameters, and Table 4 presents the saturated zone parameters. Tables 5 through 7 present a summary of the hydrogeologic characteristics for the 100-H Area environmental setting. Table 5 presents the infiltration rate, Table 6 presents the vadose zone parameters, and Table 7 presents the saturated zone parameters. Tables 8 through 10 present a summary of the hydrogeologic characteristics for the 200-E Area environmental setting. Table 8 presents the infiltration rate, Table 9 presents the vadose zone parameters, and Table 10 presents the saturated zone parameters. Tables 11 through 13 present a summary of the hydrogeologic characteristics for the 200-W Area environmental setting. Table 11 presents the infiltration rate, Table 12 presents the vadose zone parameters, and

Figure 18. Hydrostratigraphy of the Seven Hanford Installation Environmental Settings (Schramke et al. 1994)

Table 2. Source-Term Parameter Values Used for 100-N Area

MEPAS Parameter	Parameter	Value	Reference
WS-LEACHV	Waste liquid infiltration rate (ft/d)	1.6E-4	MEPAS ^a

(a) Calculated from MEPAS analysis.

Table 3. Partially Saturated Zone Parameter Values Used for 100-N Area

MEPAS Parameter	Parameter	Value		Reference
		PSZ 1 of 2 layers	PSZ 2 of 2 layers	
WP-CLASS	Soil classification in PSZ ^(a)	Sand	Loamy sand	Short and Smith (1995)
WP-SAND	% sand in PSZ	90	87	Short and Smith (1995)
WP-SILT	% silt in PSZ	7	11	Short and Smith (1995)
WP-CLAY	% clay in PSZ	3	2	Short and Smith (1995)
WP-OMC	% organic matter content in PSZ	0	0	Short and Smith (1995)
WP-IRON	% iron and aluminum in PSZ	0	0	Short and Smith (1995)
WP-PH	pH of the pore water in PSZ	8.5	7.0	Short and Smith (1995)
WP-THICK	Thickness of PSZ (ft)	60.0	20.0	Short and Smith (1995)
WP-BULKD	Bulk density of PSZ (g/cm ³)	1.99	1.90	Short and Smith (1995)
WP-TOTPOR	Total porosity of PSZ (%)	30.0	30.0	Short and Smith (1995)
WP-FIELD	Field capacity of PSZ (%)	9.0	12.0	Short and Smith (1995)
WP-LDISP	Longitudinal dispersivity of PSZ (ft)	0.6	0.2	Short and Smith (1995)
WP-CONDUCT	Saturated hydraulic conductivity (ft/d)	40.0	5.95	Short and Smith (1995)

(a) PSZ refers to partially saturated zone.

Table 4. Saturated Zone Values Used for 100-N Area

MEPAS Parameter	Parameter	Value	Reference
WZ-CLASS	Soil classification in SZ ^(a)	Loamy sand	Short and Smith (1995)
WZ-SAND	% sand in SZ	87	Short and Smith (1995)
WZ-SILT	% silt in SZ	11	Short and Smith (1995)
WZ-CLAY	% clay in SZ	2	Short and Smith (1995)
WZ-OMC	% organic matter content in SZ	0	Short and Smith (1995)
WZ-IRON	% Iron and aluminum in SZ	0	Short and Smith (1995)
WZ-PH	pH of the pore water in SZ	7.7	Short and Smith (1995)
WZ-THICK	Thickness of SZ (ft)	35.0	Short and Smith (1995)
WZ-BULKD	Bulk density of SZ (g/cm ³)	1.90	Short and Smith (1995)
WZ-TOTPOR	Total porosity of SZ (%)	30.0	Short and Smith (1995)
WZ-EFFPOR	Effective porosity of SZ (%)	11.0	Short and Smith (1995)
WZ-PVELOC	Pore-water velocity of SZ (ft/d)	10.8	Schramke et al. (1994)

(a) SZ refers to saturated zone.

Table 5. Source-Term Parameter Values Used for 100-H Area

MEPAS Parameter	Parameter	Value	Reference
WS-LEACHV	Waste liquid infiltration rate (ft/d)	1.6E-4	MEPAS ^(a)

(a) Calculated from MEPAS analysis.

Table 6. Partially Saturated Zone Parameter Values Used for 100-H Area

MEPAS Parameter	Parameter	Value	Reference
		PSZ 1 of 1 layers	
WP-CLASS	Soil classification in PSZ ^(a)	Sand	Short and Smith (1995)
WP-SAND	% sand in PSZ	91	Short and Smith (1995)
WP-SILT	% silt in PSZ	6	Short and Smith (1995)
WP-CLAY	% clay in PSZ	3	Short and Smith (1995)
WP-OMC	% organic matter content in PSZ	0	Short and Smith (1995)
WP-IRON	% Iron and aluminum in PSZ	0	Short and Smith (1995)
WP-PH	pH of the pore water in PSZ	8.5	Short and Smith (1995)
WP-THICK	Thickness of PSZ (ft)	50.0	Short and Smith (1995)
WP-BULKD	Bulk density of PSZ (g/cm ³)	1.64	Short and Smith (1995)
WP-TOTPOR	Total porosity of PSZ (%)	38.0	Short and Smith (1995)
WP-FIELD	Field capacity of PSZ (%)	7.5	Short and Smith (1995)
WP-LDISP	Longitudinal dispersivity of PSZ (ft)	0.5	Short and Smith (1995)
WP-CONDUCT	Saturated hydraulic conductivity (ft/d)	40.0	Short and Smith (1995)

(a) PSZ refers to partially saturated zone.

Table 7. Saturated Zone Values Used for 100-H Area

MEPAS Parameter	Parameter	Value	Reference
WZ-CLASS	Soil classification in SZ ^(a)	Sand	Short and Smith (1995)
WZ-SAND	% sand in SZ	91	Short and Smith (1995)
WZ-SILT	% silt in SZ	6	Short and Smith (1995)
WZ-CLAY	% clay in SZ	3	Short and Smith (1995)
WZ-OMC	% organic matter content in SZ	0	Short and Smith (1995)
WZ-IRON	% Iron and aluminum in SZ	0	Short and Smith (1995)
WZ-PH	pH of the pore water in SZ	7.5	Short and Smith (1995)
WZ-THICK	Thickness of SZ (ft)	10.0	Short and Smith (1995)
WZ-BULKD	Bulk density of SZ (g/cm ³)	1.64	Short and Smith (1995)
WZ-TOTPOR	Total porosity of SZ (%)	38.0	Short and Smith (1995)
WZ-EFFPOR	Effective porosity of SZ (%)	26.0	Short and Smith (1995)
WZ-PVELOC	Pore-water velocity of SZ (ft/d)	11.5	Schramke et al. (1994)

(a) SZ refers to saturated zone.

Table 8. Source-Term Parameter Values Used for 200-E Area

MEPAS Parameter	Parameter	Value	Reference
WS-LEACHV	Waste liquid infiltration rate (ft/d)	1.6E-4	MEPAS ^(a)

(a) Calculated from MEPAS analysis.

Table 9. Partially Saturated Zone Parameter Values Used for 200-E Area

MEPAS Parameter	Parameter	Value			Reference
		PSZ ^(a) 1 of 3 layers	PSZ 2 of 3 layers	PSZ 3 of 3 layers	
WP-CLASS	Soil classification in PSZ	Sand	Sand	Sand	Short and Smith (1995)
WP-SAND	% sand in PSZ	90	95	90	Short and Smith (1995)
WP-SILT	% silt in PSZ	9	3	9	Short and Smith (1995)
WP-CLAY	% clay in PSZ	1	2	1	Short and Smith (1995)
WP-OMC	% organic matter content in PSZ	0	0	0	Short and Smith (1995)
WP-IRON	% Iron and aluminum in PSZ	2	2	2	Short and Smith (1995)
WP-PH	pH of the pore water in PSZ	8.5	8.5	8.5	Short and Smith (1995)
WP-THICK	Thickness of PSZ (ft)	33.0	164.0	62.0	Short and Smith (1995)
WP-BULKD	Bulk density of PSZ (g/cm ³)	1.76	1.60	1.76	Short and Smith (1995)
WP-TOTPOR	Total porosity of PSZ (%)	36.0	42.0	36.0	Short and Smith (1995)
WP-FIELD	Field capacity of PSZ (%)	19.0	8.5	19.0	Short and Smith (1995)
WP-LDISP	Longitudinal dispersivity of PSZ (ft)	0.33	1.64	0.62	Short and Smith (1995)
WP-CONDUCT	Saturated hydraulic conductivity (ft/d)	4.34	14.2	4.34	Short and Smith (1995)

(a) PSZ refers to partially saturated zone.

Table 10. Saturated Zone Parameter Values Used for 200-E Area

MEPAS Parameter	Parameter	Value	Reference
WZ-CLASS	Soil classification in SZ ^(a)	Loamy Sand	
WZ-SAND	% sand in SZ	86	Short and Smith (1995)
WZ-SILT	% silt in SZ	12	Short and Smith (1995)
WZ-CLAY	% clay in SZ	2	Short and Smith (1995)
WZ-OMC	% organic matter content in SZ	0	Short and Smith (1995)
WZ-IRON	% Iron and aluminum in SZ	0	Short and Smith (1995)
WZ-PH	pH of the pore water in SZ	7.7	Short and Smith (1995)
WZ-THICK	Thickness of SZ (ft)	30.0	Short and Smith (1995)
WZ-BULKD	Bulk density of SZ (g/cm ³)	1.64	Short and Smith (1995)
WZ-TOTPOR	Total porosity of SZ (%)	18.0	Short and Smith (1995)
WZ-EFFPOR	Effective porosity of SZ (%)	10.0	Short and Smith (1995)
WZ-PVELOC	Pore-Water velocity of SZ (ft/d)	10.8	Schramke et al. (1994)

(a) SZ refers to saturated zone.

Table 11. Source-Term Parameter Values Used for 200-W Area

MEPAS Parameter	Parameter	Value	Reference
WS-LEACHV	Waste liquid infiltration rate (ft/d)	1.6E-4	MEPAS ^a

(a) Calculated from MEPAS analysis.

Table 12. Partially Saturated Zone Parameter Values Used for 200-W Area

MEPAS Parameter	Parameter	Value					Reference
		PSZ ^(a)	PSZ	PSZ	PSZ ^(a)	PSZ	
		1 of 5 layers	2 of 5 layers	3 of 5 layers	4 of 5 layers	5 of 5 layers	
WP-CLASS	Soil classification in PSZ	Sand	Loam	Loamy Sand	Loamy Sand	Loamy Sand	Short and Smith (1995)
WP-SAND	% sand in PSZ	95	45	86	73	87	Short and Smith (1995)
WP-SILT	% silt in PSZ	3	55	12	25	11	Short and Smith (1995)
WP-CLAY	% clay in PSZ	2	0	2	2	2	Short and Smith (1995)
WP-OMC	% organic matter content in PSZ	0	0	0	0	0	Short and Smith (1995)
WP-IRON	% Iron and aluminum in PSZ	0	0	0	0	0	Short and Smith (1995)
WP-PH	pH of the pore water in PSZ	8.5	7.0	7.0	7.0	7.0	Short and Smith (1995)
WP-THICK	Thickness of PSZ (ft)	82.0	49.2	15.0	20	70.0	Short and Smith (1995)
WP-BULKD	Bulk density of PSZ (g/cm ³)	1.75	1.65	1.65	1.90	1.76	Short and Smith (1995)
WP-TOTPOR	Total porosity of PSZ (%)	36	40	40	31.0	36	Short and Smith (1995)
WP-FIELD	Field capacity of PSZ (%)	11.0	13.0	18.0	7.2	12.0	Short and Smith (1995)
WP-LDISP	Longitudinal dispersivity of PSZ (ft)	0.82	0.492	0.15	0.20	0.7	Short and Smith (1995)
WP-CONDUCT	Saturated hydraulic conductivity (ft/d)	34.0	6.46	193.0	5.67	5.95	Short and Smith (1995)

(a) PSZ refers to partially saturated zone.

Table 13. Saturated Zone Parameter Values Used for 200-W Area

MEPAS Parameter	Parameter	Value	Reference
WZ-CLASS	Soil classification in SZ ^(a)	Loamy Sand	
WZ-SAND	% sand in SZ	87	Short and Smith (1995)
WZ-SILT	% silt in SZ	11	Short and Smith (1995)
WZ-CLAY	% clay in SZ	2	Short and Smith (1995)
WZ-OMC	% organic matter content in SZ	0	Short and Smith (1995)
WZ-IRON	% Iron and aluminum in SZ	0	Short and Smith (1995)
WZ-PH	pH of the pore water in SZ	7.9	Short and Smith (1995)
WZ-THICK	Thickness of SZ (ft)	30.0	Short and Smith (1995)
WZ-BULKD	Bulk density of SZ (g/cm ³)	1.76	Short and Smith (1995)
WZ-TOTPOR	Total porosity of SZ (%)	36.0	Short and Smith (1995)
WZ-EFFPOR	Effective porosity of SZ (%)	11.0	Short and Smith (1995)
WZ-PVELOC	Pore-Water velocity of SZ (ft/d)	0.6	Schramke et al. (1994)

(a) SZ refers to saturated zone.

Table 14. Source-Term Parameter Values Used for 300 Area

MEPAS Parameter	Parameter	Value	Reference
WS-LEACHV	Waste liquid infiltration rate (ft/d)	1.6E-4	MEPAS ^(a)

(a) Calculated from MEPAS analysis.

Table 15. Partially Saturated Zone Parameter Values Used for 300 Area

MEPAS Parameter	Parameter	Value	Reference
		PSZ ^(a) 1 of 1 layers	
WP-CLASS	Soil classification in PSZ	Sand	Short and Smith (1995)
WP-SAND	% sand in PSZ	92	Short and Smith (1995)
WP-SILT	% silt in PSZ	6	Short and Smith (1995)
WP-CLAY	% clay in PSZ	2	Short and Smith (1995)
WP-OMC	% organic matter content in PSZ	0	Short and Smith (1995)
WP-IRON	% Iron and aluminum in PSZ	0	Short and Smith (1995)
WP-PH	pH of the pore water in PSZ	8.5	Short and Smith (1995)
WP-THICK	Thickness of PSZ (ft)	40.0	Short and Smith (1995)
WP-BULKD	Bulk density of PSZ (g/cm ³)	2.06	Short and Smith (1995)
WP-TOTPOR	Total porosity of PSZ (%)	26.0	Short and Smith (1995)
WP-FIELD	Field capacity of PSZ (%)	7.5	Short and Smith (1995)
WP-LDISP	Longitudinal dispersivity of PSZ (ft)	0.4	Short and Smith (1995)
WP-CONDUCT	Saturated hydraulic conductivity (ft/d)	40.0	Short and Smith (1995)

(a) PSZ refers to partially saturated zone.

Table 16. Saturated Zone Values Used for 300 Area

MEPAS Parameter	Parameter	Value	Reference
WZ-CLASS	Soil classification in SZ ^(a)	Sand	Short and Smith (1995)
WZ-SAND	% sand in SZ	92	Short and Smith (1995)
WZ-SILT	% silt in SZ	6	Short and Smith (1995)
WZ-CLAY	% clay in SZ	2	Short and Smith (1995)
WZ-OMC	% organic matter content in SZ	0	Short and Smith (1995)
WZ-IRON	% Iron and aluminum in SZ	0	Short and Smith (1995)
WZ-PH	pH of the pore water in SZ	7.1	Short and Smith (1995)
WZ-THICK	Thickness of SZ (ft)	34.0	Short and Smith (1995)
WZ-BULKD	Bulk density of SZ (g/cm ³)	2.06	Short and Smith (1995)
WZ-TOTPOR	Total porosity of SZ (%)	26.0	Short and Smith (1995)
WZ-EFFPOR	Effective porosity of SZ (%)	23.2	Short and Smith (1995)
WZ-PVELOC	Pore-water velocity of SZ (ft/d)	35.0	Schramke et al. (1994)

(a) SZ refers to saturated zone.

Table 13 presents the saturated zone parameters. Tables 14 through 16 present a summary of the hydrogeologic characteristics for the 300 Area environmental setting. Table 14 presents the infiltration rate, Table 15 presents the vadose zone parameters, and Table 16 presents the saturated zone parameters.

7.3 Surface-Water Environmental Setting

The Columbia River has also been divided into three reaches within the 80-km (50-mi) radius from the center of the installation. Each reach corresponds to a location where a major tributary enters the river. The first reach corresponds to the free-flowing Hanford reach at median flow; its median discharge is 2905 m³/s (102,600 ft³/s), based on 27 years of records (Whelan et al. 1987c). The second reach corresponds to the confluence of the Yakima and Columbia Rivers. The mean discharge of the Yakima River is 104 m³/s (3673 ft³/s), based on 57 years of records (DOE 1994), while the combined discharge is 3009 m³/s (106,270 ft³/s) (Whelan et al. 1987c; DOE 1994; EP 1991). The third reach corresponds to the confluence of the Snake and Columbia Rivers. The mean discharge of

the Snake River is 1530 m³/s (54,100 ft³/s), based on 35 years of records,^(a) while the combined discharge is 4541 m³/s (160,400 ft³/s) (Whelan et al. 1987c; EP 1991; Ruppert). The Columbia River discharges, widths, depths, and flow velocities for each of the three reaches are summarized in Table 17.

7.4 Distribution Coefficients at Hanford

Environmental settings were developed in conjunction with local experts in the fields of meteorology, geology, hydrology, hydraulics, and geochemistry. Site experts participated in the initial development, fine-tuning, and final review of Hanford's environmental settings (Schramke et al. 1994). The degree of mobility of contaminants through the subsurface region is strongly influenced by the value of its distribution coefficient (i.e., K_d, ratio of distribution of the contaminant between the soil and water phases). Based on the physical and chemical/radiological attributes of the constituents, the Hanford soils fall into three basic categories with respect to the distribution coefficients: Type I (sand with pH > 7.5), Type II (sand with pH ≤ 7.5), and Type III (loam/sandy loam with pH = 7). Table 18 presents the three Hanford soil types and their pH ranges. Distribution coefficients were developed for various aqueous species present in a natural groundwater contacting various Hanford sediments. Table 19 presents a consolidated list of the Hanford sediments composing the five Hanford geographic areas (300, 200-W, 200-E, 100-H, and 100-N) (Schramke et al. 1994) where the waste sites are located, presenting the soil distribution of the 22 soil categories associated with the overburden, vadose zones, and saturated zones at Hanford. Fifteen soils fall in Category I, five soils fall in Category II, and two soils fall in Category III.

Table 17. Summary of Hydraulic and Geometric Conditions in the Columbia River

River Reach	Discharge (ft ³ /s)	Width (ft)	Depth (ft)	Velocity (ft/s)	Reference
Hanford Reach	102,600	1480	21	3.3	Whelan et al. (1987c) Walters et al. (1983)
Downstream of Yakima R. Confluence	106,270	2170	33	1.5	DOE (1994) EP (1991) Whelan et al. (1987c) Walters et al. (1983)
Downstream of Snake R. Confluence	160,370	4335	55	0.7	Ruppert ^(a) EP (1991) Whelan et al. (1987c) Walters et al (1983)

(a) Ruppert, G. 1994. Telephone conversation with Mr. Greg Ruppert on April 27, 1995, based on U.S. Geological Survey water-discharge records.

(a) Telephone conversation with Mr. Greg Ruppert on April 27, 1994, based on U.S. Geological Survey water-discharge records.

Table 18. Regional Soil Characteristics as a Function of pH

Soil-Type Index	Soil Type	pH
I	Sand	8.5
II	Sand	7
III	Loam	7

Table 19. Soil Characteristics

Soil Type Index	Soil Zone	Soil Type	% Sand	% Silt	% Clay	% Omc	% Iron	pH
300 Area								
I	Water Top Soil	Sand	92%	6%	2%	0%	0%	8.5
I	Water PSZ ^(a)	Sand	92%	6%	2%	0%	0%	8.5
II	Water SZ ^(b)	Sand	92%	6%	2%	0%	0%	7.1
200-W Area								
I	Water Top Soil	Sand	95%	3%	2%	0%	0%	8.5
I	Water PSZ (1)	Sand	95%	3%	2%	0%	0%	8.5
III	Water PSZ (2)	Loam	45%	55%	0%	0%	0%	7
II	Water PSZ (3)	Loamy Sand	86%	12%	2%	0%	0%	7
III	Water PSZ (4)	Sandy Loam	73%	25%	2%	0%	0%	7
II	Water PSZ (5)	Loamy Sand	87%	11%	2%	0%	0%	7
I	Water SZ	Loamy Sand	87%	11%	2%	0%	0%	7.9
200-E Area								
I	Water Top Soil	Sand	90%	9%	1%	0%	2%	8.5
I	Water PSZ (1)	Sand	90%	9%	1%	0%	2%	8.5
I	Water PSZ (2)	Sand	95%	3%	2%	0%	2%	8.5
I	Water PSZ (3)	Sand	90%	9%	1%	0%	2%	8.5
I	Water SZ	Loamy Sand	86%	12%	2%	0%	0%	7.7
100-H Area								
I	Water Top Soil	Sand	91%	6%	3%	0%	0%	8.5
I	Water Psz	Sand	91%	6%	3%	0%	0%	8.5
II	Water SZ	Sand	91%	6%	3%	0%	0%	7.5
100-N Area								
I	Water Top Soil	Sand	90%	7%	3%	0%	0%	8.5
I	Water PSZ (1)	Sand	90%	7%	3%	0%	0%	8.5
II	Water PSZ (2)	Loamy Sand	87%	11%	2%	0%	0%	7
I	Water SZ	Loamy Sand	87%	11%	2%	0%	0%	7.7

(a) PSZ refers to partially saturated zone.

(b) SZ refers to saturated zone.

Table 20 contains a list of the aqueous species that would form from many of the chemicals that have been identified in the waste sites at Hanford. These chemicals represent the species that would be present if the inorganic chemicals were dissolved in water. Kd information for salts or compounds that speciate (cation or metal portion and anion or non-metal portion) is presented for the individual species that make up the compound. For compounds that undergo dissolution, Kds are identified for the individual species that are formed. There is no chemical difference and, thus, no Kd difference between various isotopes (stable or radioactive) for a given element; thus, separate Kd values have not been listed for isotopes of the same element. Due to the difficulty of determining Kd values for all organics, a zero value is assigned to them, except for the list of organics identified in Table 21, on which research has been performed to determine site-specific Kd values (Ames and Serne 1991). For more information on Hanford distribution coefficients, refer to Serne et al. (1993), Serne and Wood (1990), and Ames and Serne (1991).

Table 20. Kd Values (mL/g) for Inorganic Species

Element	Species	Soil I	Soil II	Soil III	Reference
H	$^3\text{H}_2\text{O}$	0	0	0	water $\equiv$ Kd $\equiv$ 0
Li	Li^+	0.2	0.2	0.8	MEPAS ^(a)
Be	Be^{2+}	30	70	100	Serne ^(b)
B	BO_3^{3-}	.19	.19	1.3	MEPAS
C	CO_3^{2-}	0	0	0	anion = 0
	CN^-	0	0	0	anion = 0
N	NO_2^-	0	0	0	observed ^(c)
	NO_3^-	0	0	0	observed
	NH_4^+	0.2	0.2	0.8	assume Na^+
O	OH^-	0	0	0	anion
F	F^-	0	0	0	anion
Na	Na^+	0.2	0.2	0.8	MEPAS
Mg	Mg^{2+}	7	7	14	1/10 MEPAS (Ca^{2+})
Al	Al^{3+}	300	300	350	$\approx$ 1/100 MEPAS
	AlO_2^-	0	0	1	anion
Si	SiO_4^{4-}	0	0	0	anion
P	PO_4^{3-}	50	50	50	MEPAS
S	SO_4^{2-}	0	0	0	MEPAS
Cl	Cl^-	0	0	0	MEPAS
K	K^+	0.2	0.2	0.8	MEPAS (Na^+)
Ca	Ca^{2+}	7	7	14	1/10 MEPAS (Ca^{2+})
Sc	Sc^{3+}	50	100	250	use Cr^{3+}
V	VO_4^{3-}	5	10	20	guess ^(b)
	VO_3^-	5	10	20	guess
Cr	Cr^{3+}	50	100	250	Serne
	CrO_4^{2-}	0	0	1	observed
Mn	Mn^{2+}	30	70	100	use Be^{2+}
	MnO_4^-	0	0	0	anion

Table 20. (contd)

Element	Species	Soil I	Soil II	Soil III	Reference
Fe	Fe ²⁺	15	15	30	known to adsorb better than Mg
	Fe ³⁺	50	50	50	known to adsorb better than Fe ²⁺
Co	Co ²⁺	20	12	50	observed
Ni	Ni ²⁺	20	12	50	observed
Cu	Cu ²⁺	20	12	50	same as Ni
Zn	Zn ²⁺	20	12	50	Same as Ni
Ge	Ge ⁴⁺	200	300	350	use Al ³⁺
As	AsO ₄ ³⁻	30	50	300	Serne
	AsO ₃ ³⁻	3	5	10	Serne
Se	SeO ₄ ²⁻	0	2	5	Serne
	SeO ₃ ²⁻	0	1	3	Serne
Br	Br ⁻	0	0	0	anion
Kr	Kr (gas)	0	0	0	inert gas
Rb	Rb ⁺	0.2	0.2	0.8	same as Na ⁺
Sr	Sr ²⁺	15	10	25	observed
Y	Y ³⁺	100	100	200	guess
Zr	Zr ⁴⁺	50	50	100	like Lanthanides
Nb	Nb ⁵⁺	100	100	200	observed
Mo	Mo ³⁺	50	50	100	assume like La
	MoO ₄ ⁻	0	0	1	similar TcO ₄ ⁻
Tc	TcO ₄ ⁻	0	0	0	observed
Ru	Ru ³⁺	50	50	100	assume like La
	RuO ₄ ⁻	0	1	2	anion guess ^(b)
Ag	Ag ⁺	1	1	5	monovalent sorbs better than Na ⁺
Cd	Cd ²⁺	20	12	50	like Ni
In	In ³⁺	100	100	200	guess
Sn	Sn ²⁺	200	300	350	assume Al ³⁺
	Sn ⁴⁺	100	150	175	1/2 of Sn ²⁺
Sb	SbO ₂ ⁻	0	1	2	anion guess
	SbO ₃ ⁻	0	0	1	anion guess
I	I ⁻	0	0	0	observed
	IO ₃ ⁻	0	0	0	guess
Cs	Cs ⁺	50	30	100	observed
Ba	Ba ²⁺	15	10	25	assume Sr ²⁺
La-Lu	Ce ³⁺	50	50	100	MEPAS
	Pm ³⁺	50	50	100	MEPAS
	Sm ³⁺	50	50	100	MEPAS
	Eu ³⁺	50	50	100	MEPAS
	Tm ³⁺	50	50	100	MEPAS
Ta	Ta ⁵⁺	100	100	200	guess
Re	ReO ₄ ⁻	0	0	0	observe
Au	Au ³⁺	100	100	200	guess
Hg	Hg ²⁺	100	100	200	MEPAS

Table 20. (contd)

Element	Species	Soil I	Soil II	Soil III	Reference
Tl	Tl ⁻	0.2	0.2	0.8	like Na ⁺
Pb	Pb ²⁺	100	100	200	observe
Bi	Bi ³⁺	100	100	200	assume like Pb
Po	Po ²⁺	100	100	200	assume like Pb
Ra	Ra ²⁺	30	20	50	≈ 2 x Sr
Th	Th ⁴⁺	50	50	100	like La
Pa	Pa ⁴⁺	50	50	100	like La
U	U ⁴⁺	100	100	250	Serne
	UO ₂ ²⁺	1	2	3	Serne
Np	NpO ₂ ⁺	1	0	2	Serne
	Np ⁴⁺	200	200	500	Serne
Pu	Pu ³⁺	200	200	500	Serne
	Pu ⁴⁺				
	PuO ₂ ⁺	1	1	5	Serne
	PuO ₂ ²⁺				
Am	Am ³⁺	50	50	100	like La
Cm	Cm ³⁺	50	50	100	like La
Bk	Bk ⁴⁺	50	50	100	
					like La
Cf	Cf ³⁺	50	50	100	like La
Es	Es ³⁺	50	50	100	like La

(a) Documented in the MEPAS database (Strenge et al. 1989).

(b) Recommended by R. J. Serne, Staff Scientist, Pacific Northwest Laboratory, Richland, Washington.

(c) Observed: Work R. J. Serne has performed that was germane to Hanford, and represents data discussed and found in published and current monthly reports (will be published in FY-95).

Table 21. Kd Values (mL/g) for Organics

Chemical	Soil I	Soil II	Soil III	Reference/Comment
butyl phosphates	4	4	8	Ames and Serne (1991)
tributyl phosphates	4	4	8	Ames and Serne (1991)
dibutyl phosphonate	4	4	8	assume like TBP
tributyl phosphonate	4	4	8	assume like TBP
carbon tetrachloride	0	0	0	moves in Hanford soils
trichlorethylene	0	0	0	moves in most soils
normal paraffin hydrocarbons	100	100	200	solubility in water is low
sulfamate	0	0	0	soluble; guess on Kd
oxalate	5	5	10	forms insoluble calcium salt, so guess
methyl isobutyl ketone	0	0	0.02	Ames and Serne (1991)

8.0 Unit Transport Application of MRA at Hanford

To implement the MRA at Hanford, the MEPAS model was modularized to calculate environmental concentrations as they varied spatially in the environment. The model was connected to the state GIS to ensure that groundwater, surface-water, soil, and air concentrations were supplied for every grid cell in the 80-km (50-mi) radius. The output from the application runs resulted in Unit Transport Factors (UTFs) (i.e., C_{us} in Equation (1)), based on an assumed unit source strength (i.e., S_u in Equation (1)) of 1 Ci for radionuclides and 1 kg for chemicals. When multiple aggregated waste sites contributed to the same downgradient receptor cell, the concentrations were summed, as the system is linear.

The first step in computing the UTFs is to create a data set that describes the unit release from the source, physical/chemical characteristics of the contaminants of interest, and environmental conditions for each medium that contaminants migrate through. Parameter files were created in MEPAS for each contaminant, aggregated waste site, and environmental setting; these parameter files are a record of all data utilized for model runs. MEPAS parameter files are automatically created when MEPAS (version 3.0g) is utilized. These data sets represent the overall Base Data Set for the Hanford Installation. Figure 19 illustrates the process of building the Base Data Set by providing a flow diagram demonstrating how the environmental-setting, contaminant-library, and source-term data feed into the MEPAS model and how the data sets are created for each of the five environmental settings.

Once the Base Data Set is created, two computational options are available, depending on the release mechanism and the environmental medium to which the contaminant is being released: 1) atmospheric releases due to volatilization or resuspension of contaminated soils, or 2) subsurface (vadose or saturated zone) or soil releases due to infiltration or leaching. Figure 20 illustrates a flow diagram depicting the two computational options associated with the UTA.

Figure 19. Flow Diagram Illustrating the Development of the Base Data Set

Figure 20. Flow Diagram Illustrating the Utilization of the Base Data Set and Implementation of the Unit Transport Analysis

- Atmospheric Option** -- When the atmospheric option is selected, air and soil concentrations are calculated for locations within an 80-km (50-mi) radius for 16 directional sectors (i.e., N, NNE, NE, ENE, E, ESE, SE, SSE, S, SSW, SW, WSW, W, WNW, NW, and NNW) and 12 radii (i.e., 0.100, 0.102, 0.108, 0.124, 0.167, 0.284, 0.600, 1.47, 3.83, 10.24, 27.65, and 75.00 km) from the center of the aggregated waste site; therefore, 192 sectors are populated with air and soil concentrations from the atmospheric-modeling exercise. Seven wind-stability types are also accounted for: three particulates, a depositing gas with surface residue, a nondepositing gas, a gas emission/particle deposition, and a depositing gas with no surface residue. The polar-coordinate system describing the radial distribution of atmospheric contamination is then overlaid on the state GIS Cartesian system, and the sector concentrations are automatically mapped onto the corresponding and appropriate grid-cell locations. As a final result, unit concentrations are identified in each cell for air and surface soil. The mass at the source is depleted over time, and this depletion is accounted for at the source. The different locations associated with the aggregated waste sites are also automatically accounted for when mapping the polar-coordinate results of each waste site onto the state GIS cartesian-coordinate system.

Because of the linearity of the UTA in the MRA system (e.g., same joint frequency and wind-rose data for each waste site), the number of atmospheric runs can be reduced from 5,271,552 to 506,880 for the Hanford Installation. A model run is defined as the computation of a contaminant concentration at a given point at a given time. The computational savings occur because there are only five environmental settings, even though there are 52 grid cells containing aggregated waste sites.

For each atmospheric run, environmental setting, and contaminant, the following data were utilized to calculate volatilization and resuspension, starting at time zero and continuing until year 10,000 or until the contaminant inventory was zero: contaminant name, contaminant Chemical Abstract Services (CAS) identification, contaminant half-life, decay products for radionuclides, inventories/activities, toxicity type (where appropriate), stability class, volatilization rate, and suspension-emission rate. For year 0 to 100, the time step was 1 year; for year 100 to 1000, the time step was 10 years; for year 1000 to 10,000, the time step was 100 years. For radionuclides, inventory decay was accounted for using Bateman's equation (Droppo et al. 1989b, page A.2).

Figures 21, 22, and 23 illustrate the type of output that results from the atmospheric assessment prior to converting the polar-coordinate results to the state GIS Cartesian coordinate system. Each figure presents an illustrative example of the spatial distribution of the air concentration for 4-methyl-2-pentanone at various resolutions for time 140 years. The resolution increases from Figure 21 to 23. The outer circle in Figure 21 is approximately 75 km (46 mi) in radius, the outer circle in Figure 22 is approximately 10 km (6 mi) in radius, and the outer circle in Figure 23 is approximately 0.6 km (0.4 mi) in radius. Each figure is shade coded to denote concentration levels as a function of the highest concentration presented. A shading bar is presented in the upper left-hand corner of each figure. The coding in each figure is presented in the form of a degree of shading. The uppermost and darkest shade in the coding bar represents the highest unit concentration. Each successive shade decreases by a factor of two, as related to the shade above it.

- **Subsurface/Soil Option** — When the subsurface/soil option is selected, groundwater, soil, and surface-water concentrations are calculated for every 1-km² (.6 mi²) grid-cell location within the Hanford Installation and at locations along the Columbia River within an 80-km (50-mi) radius. Only contaminated soils and ponded waste sites (i.e., West Lake and U-Pond) were modeled as part of this UTA exercise.^(a) A unit mass or activity was assumed for each source term. Because of linearity, 1 Ci was assumed for radionuclides, and 1 kg was assumed for chemicals. To convert the calculated UTA water concentrations to concentrations that reflex the actual inventory or activity at the waste site, the UTA unit concentrations must be multiplied by the actual inventory or activity present at the source. See Equation (1) for a more detailed explanation.

Figure 20 presents a flow diagram illustrating the implementation of the subsurface/soil option. Each aggregated waste site is mapped to a grid-cell location. The leaching of the contaminants from contaminated soils through the vadose zone and groundwater is then modeled to various locations surrounding the waste site. Transport to the Columbia River (i.e., surface water) is also modeled. To conserve mass, the depletion of mass/activity from the source is accounted for over time in the contaminated soil from which the constituents originated.

Unlike the atmospheric modeling exercise, where the shapes of the flow paths were identical for each aggregated waste site per environmental setting per constituent, the subsurface modeling exercise (and in particular the groundwater modeling exercise) had different flow paths for each aggregated waste site. Figure 11 illustrates the flow paths from the 52 grid cells that contain aggregated waste sites. Unlike other semianalytical groundwater models that address only unidirectional flow, MEPAS

(a) Pacific Northwest Laboratory (PNL) modeled the contaminated soil and ponded sites at Hanford. ASI modeled the groundwater plume contamination that is currently beneath the Hanford Installation.

Figure 21. Atmospheric Results for an 80-km Radius from an Aggregated Waste Site in the 200-E Area Illustrating the Spatial Distribution of 4-Methyl-2-Pentanone (Radius of Largest Circle is 75 km)

version 3.0g approximates curvilinear flow, as illustrated by the pathlines in Figure 12. As a result, each cell containing an aggregated waste site had to be modeled separately for each contaminant to each downgradient grid cell identified in Figure 10. Each modeled plume was then automatically overlaid on the state GIS grid.

There are 82 contaminants, 52 grid-cell locations containing aggregated waste sites, and 5 modeling time periods (i.e., years 40, 100, 140, 1000, and 10,000). These numbers represent the minimum number of runs to produce groundwater or surface-water concentrations in one downgradient cell. Figure 24 illustrates the spatial distribution of one chemical (nitrite) from one aggregated waste site (200-E Area) for one time (year 140). As with Figures 21, 22, and 23, Figure 24 is coded to denote concentration levels as a function of the highest concentration presented. A coding bar is presented in the upper left-hand corner of this figure. The uppermost and darkest code in the bar represents the highest unit concentration. Each successive code decreases by a factor of 10, as related to the code above it. Similar plumes would be generated for every 1) other contaminant, 2) aggregated

Figure 22. Atmospheric Results for a 12-km Radius from an Aggregated Waste Site in the 200-E Area Illustrating the Spatial Distribution of 4-Methyl-2-Pentanone (Radius of Largest Circle is 10.2 km)

waste site, 3) time period, and 4) source-term type (i.e., ponded liquid waste and contaminated soils). When multiple plumes from different sources intersect the same downgradient grid-cell location, the unit concentrations are summed to produce one (i.e., combined) groundwater concentration for that grid cell.

Due to the complexity of this analysis, the subsurface assessment resulted in the largest number of modeling runs: 8,264,256. In comparison, 1,784,640 runs were required to generate the surface water concentrations, and only 27,456 runs were required to account for the depletion of the constituents from the source.

Figure 23. Atmospheric Results for a 1.2-km Radius from an Aggregated Waste Site in the 200-E Area Illustrating the Spatial Distribution of 4-Methyl-2-Pentanone (Radius of Largest Circle is 0.6 km)

8.1 Impact of Contaminant Distribution Coefficient and Half-Life on Contaminant Arrival Time at the Columbia River

As part of the HRA-EIS, ASI will use the unit concentrations and risks to develop three-dimensional, spatially distributed, risk isopleths will be presented for the no-action, baseline risks to all areas within an 80-km (50-mi) radius of the center of the Hanford Installation for years 0, 40, 100, 140, 1000, and 10,000 years. The ASI developed risks presented for the following four HSB RAM exposure scenarios: residential, recreational, industrial, and agricultural.

Although spatially distributed risks associated with six time periods are to be presented, the isopleths do not provide the analyst with information on whether the bulk of the contaminant mass has already passed through the environmental system (i.e., passed through the 80-km [50-mi] radius) or whether the main body of contamination is still in the system. Two contaminant characteristics greatly influence the mobility and availability of the constituents in the environment: distribution coefficient

Figure 24. Two-Dimensional Spatial Distribution of Nitrite from One Aggregated Waste Site in the 200-E Area for Year 140

(Kd) and contaminant half-life. The Kd significantly influences the mobility of the contaminant in the subsurface system. The effects of Kd are mathematically described by Equations (13), (14), and (16): the higher the Kd, the longer the travel times. The half-life is also important. If the half-life is small, then the contaminant can decay away prior to reaching the receptor. Thus there is an important relationship between travel time (i.e., Kd) and half-life. Therefore, it would be very valuable to know when the peak concentration reaches the Columbia River. This travel time is a function of the contaminant's Kd and half-life, characteristics of the environmental setting, and location of the environmental setting as it relates to the Columbia River. Once the contaminant enters the Columbia River, it will move quickly out of the 80-km (50-mi) radius.

An analysis was performed to determine when the peak concentration reaches the Columbia River. Various values for Kd and half-life were assigned to contaminants that leach through each environmental setting. The ranges in these values reflect the characteristics of the contaminants associated with that environmental setting. The analysis was performed to 10,000 years. All five environmental settings were analyzed (i.e., 200-W, 200-E, 100-N, 100-H, and 300 Areas); the results of the assessment are presented in Tables 22 through 26. These tables present the travel time of the peak concentration from the source to the Columbia River as a function of Kd and half-life. The average flow distances from the 200-W, 200-E, 100-N, 100-H, and 300 Areas to the Columbia River were assumed as 21 km (13 mi), 18 km (11 mi), 1.6 km (1.0 mi), 1.1 km (0.7 mi), and 2.2 km (1.4 mi), respectively.

Table 22. Distribution Coefficient Versus Half-Life Versus Arrival Time for the 200-W Area

Kd per Layer (in mL/g)	Contaminant Half-Life (yr)		
	0.1	100	10,000
0-0-0-0-0-0 ^(a)	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 776.	The peak arrives at year 825.
0-1-0-1-0-0	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 3216.	The peak arrives at year 2851.
2-2-2-2-2-2	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 9022.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.
2-4-2-4-2-2	The contaminants never reach the Columbia River due to a short half-life.	The contaminants never reach the Columbia River due to a short half- life.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.
5-10-5-10-5-5	The contaminants never reach the Columbia River due to a short half-life.	The contaminants never reach the Columbia River due to a short half- life.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.
15-30-15-30-15-15	The contaminants never reach the Columbia River due to a short half-life.	The contaminants never reach the Columbia River due to a short half- life.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.

(a) The 200-W Area environmental setting contains five vadose zones and one saturated zone. The first five numbers in the series represents the assigned Kd values for each successive vadose zone layer; the last value represents the Kd for the saturated zone.

Table 23. Distribution Coefficient Versus Half-Life Versus Arrival Time for the 200-E Area

Kd per Layer (in mL/g)	Contaminant Half-Life (yr)		
	0.1	100	10,000
0-0-0-0 ^(a)	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 682.	The peak arrives at year 728.
1-1-1-1	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 5308.	The peak arrives at year 7292.
5-5-5-5	The contaminants never reach the Columbia River due to a short half-life.	The contaminants never reach the Columbia River due to a short half-life.	The contaminants are still moving toward the Co- lumbia River and peak will reach the river after 10,000 years.
15-15-15-15	The contaminants never reach the Columbia River due to a short half-life.	The contaminants never reach the Columbia River due to a short half-life.	The contaminants are still moving toward the Co- lumbia River and peak will reach the river after 10,000 years.

(a) The 200-E Area environmental setting contains three vadose zones and one saturated zone. The first three numbers in the series represents the assigned Kd values for each successive vadose zone layer; the last value represents the Kd for the saturated zone.

These tables can be very useful in determining when a contaminant of concern, which has leached from contaminated soils, will enter the Columbia River. For example, if the contaminant of concern 1) is leaching from contaminated soils in the 200-W Area, 2) has a Kd of zero, and 3) has a half-life of 1000 years, the peak concentration will arrive at the river between years 776 and 825 (refer to Table 23). The bulk of the contamination for this contaminant will remain in the Hanford aquifer until at least year 776.

Table 24. Distribution Coefficient Versus Half-Life Versus Arrival Time for the 100-N Area

Kd per Layer (in mL/g)	Contaminant Half-Life (yr)		
	0.1	100	10,000
0-0-0 ^(a)	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 92.	The peak arrives at year 95.
2.5-5-2.5	The contaminants never reach the Columbia River due to a short half-life.	This case was not modeled.	The peak arrives at year 6000.
5-10-5	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 4356.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.
10-20-10	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 6894.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.
15-30-15	The contaminants never reach the Columbia River due to a short half-life.	The contaminant s never reach the Columbia River due to a short half-life.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.

- (a) The 100-N Area environmental setting contains two vadose zones and one saturated zone. The first two numbers in the series represents the assigned Kd values for each successive vadose zone layer; the last value represents the Kd for the saturated zone.

Table 25. Distribution Coefficient Versus Half-Life Versus Arrival Time for the 100-H Area

Kd per Layer (in mL/g)	Contaminant Half-Life (yr)		
	0.1	100	10,000
0-0 ^(a)	The contaminants never reach the Columbia River due to a short half-life.	The contaminants have already reached the Columbia River.	The contaminants have already reached the Columbia River.
5-5	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 1550.	The peak arrives at year 3250.
15-15	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 2827.	The peak arrives at year 9000.

(a) The 100-N Area environmental setting contains one vadose zone and one saturated zone. The first number in the series represents the assigned Kd values for the vadose zone layer; the last value represents the Kd for the saturated zone.

Table 26. Distribution Coefficient Versus Half-Life Versus Arrival Time for the 300 Area

Kd per Layer (in mL/g)	Contaminant Half-Life (yr)		
	0.1	100	10,000
0-0 ^(a)	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 43.	The peak arrives at year 43.
5-5	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 2412.	The peak arrives at year 5251.
10-10	The contaminants never reach the Columbia River due to a short half-life.	This case was not modeled.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.
15-15	The contaminants never reach the Columbia River due to a short half-life.	The peak arrives at year 4470.	The contaminants are still moving toward the Columbia River and peak will reach the river after 10,000 years.

(a) The 300 Area environmental setting contains one vadose zone and one saturated zone. The first number in the series represents the assigned Kd value for the vadose zone layer; the last value represents the Kd for the saturated zone.

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