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A Performance Assessment Methodology for Low-Level Waste Facilities

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Sandia National Laboratories
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ABSTRACT

A performance assessment methodology has been developed for use by the U.S. Nuclear Regulatory Commission in evaluating license applications for low-level waste disposal facilities. This report provides a summary of background reports on the development of the methodology and an overview of the models and codes selected for the methodology. The overview includes discussions of the philosophy and structure of the methodology and a sequential procedure for applying the methodology. Discussions are provided of models and associated assumptions that are appropriate for each phase of the methodology, the goals of each phase, data required to implement the models, significant sources of uncertainty associated with each phase, and the computer codes used to implement the appropriate models. In addition, a sample demonstration of the methodology is presented for a simple conceptual model.

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1.0 INTRODUCTION

A wide range of radioactive materials is categorized as low-level radioactive waste, and these constitute a large volume of waste. There are more than 23,000 companies, universities, laboratories, and government facilities that are licensed by the U. S. Nuclear Regulatory Commission (NRC) to use radioactive materials as part of their normal operational activities, and many of these produce low-level wastes. In 1987, a total of 1,845,000 ft^3 of commercial low-level waste was disposed of in the United States; these wastes contained a total of 269,550 Ci of activity [Tyron-Hopko and Ozaki, 1988]. These amounts may change in the future, but the changes are unlikely to be dramatic. Consequently, there exists, and will continue to exist, a need to dispose of considerable volumes of low-level waste.

Under the Low-Level Radioactive Waste Policy Act of 1980, and the Low-Level Radioactive Waste Policy Amendments Act of 1985, each state or compact of states is required to dispose of its own low-level radioactive wastes. At present all low-level waste produced in the United States is disposed at three sites: Barnwell, South Carolina, Richland, Washington, and Beatty, Nevada. This policy implies that a number of low-level waste disposal facilities must be sited, licensed, and constructed over the next few years. NRC and Agreement States have responsibility for licensing these sites using the federal requirements of 10 CFR Part 61 or similar state regulations.

10 CFR Part 61.41 establishes exposure limits to members of the general population from off-site releases of radioactivity during the lifetime of a facility. 10 CFR Part 61.13(a) requires pathways analyzed in demonstrating protection of the general population must include air, soil, groundwater, surface water, plant uptake, and exhumation by burrowing animals. These analyses are required to demonstrate that there is reasonable assurance that the exposure limits of 10 CFR Part 61.41 are not exceeded. Although the 10 CFR Part 61.41 radiological limits are applicable during the operational and post-operational periods, the analyses used by the NRC to determine compliance with the 10 CFR Part 61.41 performance objectives after permanent facility closure is commonly called a performance assessment.

The performance assessment methodology has been developed in a five-step program over a period of two years. The first two steps in the methodology development were identification of pathways of potential application in a low-level waste performance assessment [Shipers, 1989], and screening of those pathways to identify which are of primary importance [Shipers and Harlan, 1989]. Table 1-1 contains the important pathways identified for the undisturbed site [Shipers and Harlan, 1989]. This list was developed for a generic site, and is based on a qualitative ranking of both the likelihood of migration occurring along the pathway, and the expected consequence of the pathway. These are the only important pathways when it is assumed that (1) the containment structure and soil cover remain intact and perform as designed to provide adequate shielding and minimize infiltration, (2) large amounts of gaseous waste or organic waste that result in radionuclide-tagged decomposition gases are not disposed of at the facility, and (3) the disposal facility is designed with a minimum of 3 meters of soil cover, and possibly includes a concrete containment structure to minimize the likelihood of plant and animal intrusion.

The pathways presented in Table 1-2 were identified by Shipers and Harlan to be important after the occurrence of an event that disrupts the integrity of the disposal unit and exposes waste at the surface. Events that can disrupt the disposal unit can be placed into two categories: naturally occurring events and events resulting from human intrusion. Naturally occurring events include, among others, wind and water erosion, earthquakes and landslides, and disposal unit collapse or subsidence. The likelihood of these events is a site-specific consideration, and difficult to predict, but design and engineering criteria that minimize the likelihood of their occurrence may be easily incorporated into the facility design. Also, the siting criteria for low-level waste facilities specifies selecting a location where the likelihood of naturally occurring events disturbing the integrity of the facility is generally low.

Intruder-induced disruptive events include such activities as construction, drilling, and resource exploration or exploitation at the disposal facility. These activities can potentially compromise the integrity of the disposal unit and result in exhumation of the waste. Agricultural activities at the site can enhance mixing of the waste in the environment, and can promote radionuclide migration into the food chain. The activities of a future intruder are very difficult to predict, but siting criteria for low-level waste facilities are designed to minimize the likelihood of such disruptive events.

Site-specific conditions must in general be considered when selecting the important pathways to be analyzed. For a generic site, the pathways contributing the major portion of the dose to humans are from the source through ground water to a water well from which a person can be exposed to the contamination, and from the source through ground water to a surface-water body which is used by a person for various purposes [Shipers and Harlan, 1989]. The radiological component of gaseous releases from a low-level waste facility appears to be small [Biddle et al., 1987], and doses from gaseous radionuclides can most likely be neglected for the undisturbed facility [Shipers and Harlan, 1989]. For intruder-disturbed facilities, several air pathways may be important. Of these pathways, several are for off-site receptors, hence there is a need for air transport models in the methodology.

The third step in developing the methodology was to identify models that can be used to assess the pathways, and to demonstrate that those models can be integrated into a complete performance assessment methodology [Kozak et al., 1989a]. This third report contains discussions of models for source-term release, ground-water flow and transport, air transport, surface-water transport, food chain, and dosimetry. For an undisturbed facility the principal means by which radionuclides can be encountered by a human are by exposure to well water from a contaminated aquifer, and by

Table 1-1
Important Generic Pathways for Undisturbed Performance of
Low-Level Waste Disposal Facilities.

source-ground water-man
source-ground water------soil-----man
source-ground water--land plants--man
source-ground water--land animals-man
source-ground water-surface water-man
source-ground water------soil-------land plants---man
source-ground water--land plants----land animals--man
source-ground water-surface water-aquatic animals-man
source-ground water------soil-------land plants-----land animals--man
source-ground water-surface water--aquatic plants-aquatic animals-man

SOURCE: [Shipers and Harlan, 1989]

Table 1-2
Important Generic Pathways for Disturbed Performance of
Low-Level Waste Disposal Facilities

source------man (a)
source------air------man
source--ground water--man
source-surface water-man
source------air------------soil------man (b)
source------air--------land plants---man
source------soil-------land plants---man (a)
source--ground water--------soil------man (b)
source--ground water----land plants---man
source--ground water----land animals--man
source-surface water-------soil------man (b)
source-surface water---land plants---man
source-surface water---land animals--man
source-surface water-aquatic animals-man
source------air------------soil--------land plants---man (b)
source------air--------land plants-----land animals--man
source------soil-------land plants-----land animals--man (a)
source--ground water--------soil--------land plants---man (b)
source--ground water----land plants-----land animals--man
source-surface water-------soil--------land plants---man (b)
source-surface water---land plants-----land animals--man
source-surface water--aquatic plants-aquatic animals-man
source----air--------soil------land plants---land animals--man (b)
source--ground water---soil-----land plants---land animals--man (b)
source-surface water---soil----land plants---land animals--man (b)

SOURCE: [Shipers and Harlan, 1989]

notes: (a) These pathways are important only for on-site receptors.
(b) These pathways are important only for off-site receptors.

exposure to surface water that is hydraulically connected to a contaminated aquifer. These waters may be used for a variety of purposes, including crop irrigation, so that food-chain analyses must be performed.

The fourth step in the development of the methodology was to select computer codes that implement the methodology [Kozak et al., 1989b]. In that report the capability to perform both simple and detailed analyses for all parts of the methodology was retained, since for an arbitrary site any of the components of the methodology may require detailed analysis. Computer codes or analytical methods were recommended in this fourth report for both approaches.

The fifth step in the project was to acquire, implement, and assess computer codes for the methodology [Kozak et al., 1990]. Several of the early recommendations of Kozak et al. [1989b] were modified at this stage, and specific analytical techniques were suggested for source-term and ground-water transport calculations. These analytical methods are implemented in two simple computer codes named DISPERSE and SURFACE; the theoretical bases for these codes are given in detail in Kozak et al. [1990]. The recommended analytical methods and computer codes that resulted from this fifth project step are shown in Table 1-3. Kozak et al. [1990] also documented comparisons between DISPERSE and both VAM2D and FEMWATER/BLT simulations of well concentrations. In these analyses it was shown that DISPERSE provides a reasonable approximation to either VAM2D or FEMWATER/BLT results, and that DISPERSE predicts larger well concentrations.

1.1 Scope Of This Report

The purpose of this report is to summarize the information contained in background reports on the methodology development [Shipers, 1989; Shipers and Harlan, 1989; Kozak et al., 1989a; Kozak et al., 1989b; Kozak et al., 1990], and to provide an overview of the models and codes recommended in the methodology. Detailed input guides and operating procedures for the computer codes in the methodology will be documented in a Self-Teaching Curriculum report.[1]

A brief overview of the performance assessment methodology is provided in Chapter 2 of this report. The overview includes discussions of the philosophy and structure of the methodology, and a sequential procedure for applying the methodology. Subsequent chapters provide synopses of the components of the methodology for assessing undisturbed performance in the areas of ground-water hydrology (Chapter 3), source term (Chapter 4), radionuclide transport processes (Chapters 5, 6, and 7), and pathways and dosimetry (Chapter 8). Chapter 9 is devoted to a discussion the

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1. Chu, M. S. Y, M. W. Kozak, J. E. Campbell, B. K. Thompson, and P. A. Mattingly A Self-Teaching Curriculum for the NRC/SNL Low-Level Waste Performance Assessment Methodology, NUREG/CR-5539, SAND90-0585, Sandia National Laboratories, in press.

analysis of intruder scenarios. Chapters 3 to 8 contain discussions of models and associated assumptions that are appropriate for each phase of the methodology, the goals of each phase, data required to implement the models, significant sources of uncertainty associated with each phase, and the computer codes used to implement the appropriate models. A sample demonstration of the methodology is presented in Chapter 10, and a report summary and list of conclusions are given in Chapter 11.

Table 1-3
Recommended Techniques and Codes for the Methodology

• Percolation	• VAM2D
• Source Term	• Mixing-Cell Cascade Model • BLT • VAM2D
• Unsaturated Zone Transport	• Delay Time • VAM2D • BLT
• Saturated Zone Flow	• Darcy Model • VAM2D
• Saturated Zone Transport	• DISPERSE/SURFACE • VAM2D • BLT
• Surface Water	• GENII • SURFACE
• Air	• GENII • (AIRDOS-PC)*
• Food Chain and Dosimetry	• GENII

* not implemented

SOURCE: [Kozak et al., 1990]

2.0 OVERVIEW OF THE PERFORMANCE ASSESSMENT METHODOLOGY

This performance assessment methodology is designed to provide the NRC with a tool for performing confirmatory analyses in support of license reviews related to postclosure performance. The methodology allows analyses of dose to individuals from off-site releases under normal conditions as well as on-site doses to inadvertent intruders.

In previous reports on the development of this methodology, it was stated that analyses of intruder doses are required [Shipers, 1989; Shipers and Harlan, 1989; Kozak et al., 1989a; Kozak et al., 1989b]. These statements are incorrect and in most cases intruder-dose analyses need not be performed. A demonstration of intruder protection may consist of a demonstration that the waste classification and segregation requirements of 10 CFR Part 61 have been met, and that adequate barriers to inadvertent intrusion have been provided for. However, dose analyses may be required in special cases when an applicant requests an exemption from the 10 CFR Part 61 waste classification scheme.

2.1 Methodology Philosophy

The purpose of the methodology is to demonstrate whether doses are expected to be lower than regulatory performance objectives rather than to provide estimates of the actual expected dose. Hence, even though uncertainties may exist in calculated doses, there will be confidence that a site meets regulatory criteria. The approach used in this methodology is to emphasize reasonably conservative analyses.

The methodology has been designed to be modular in structure, which allows the NRC to confirm or verify parts of, or all of the assertions made by a licensee by examining intermediate output from the various models. The modular structure allows use of the simplest models possible but permits substitution of more complex models when needed [Starmer, 1988]. In addition, the modular structure permits updating of selected models as better models are developed. This prevents the methodology from becoming obsolete with passing time.

2.2 Description and Structure of the Methodology

Radionuclides released during routine performance of a low-level waste facility are most likely to reach the accessible environment by two principal pathways [Shipers and Harlan, 1989]. These pathways are (1) source to ground water, with subsequent human exposure to well water, and (2) source to ground water to surface water, with humans and foodstuffs coming in contact with the contaminated surface water. Other pathways such as releases to surface water or to the air may be of importance at particular sites, and the critical pathways must be identified for each site. To assess the effect of releases through the two principal groundwater pathways identified by Shipers and Harlan, the methodology must account for a number of physical and chemical processes that are expected to occur in and near the facility. The models for these processes, and the information exchanged between them, are shown in Fig. 2-1.

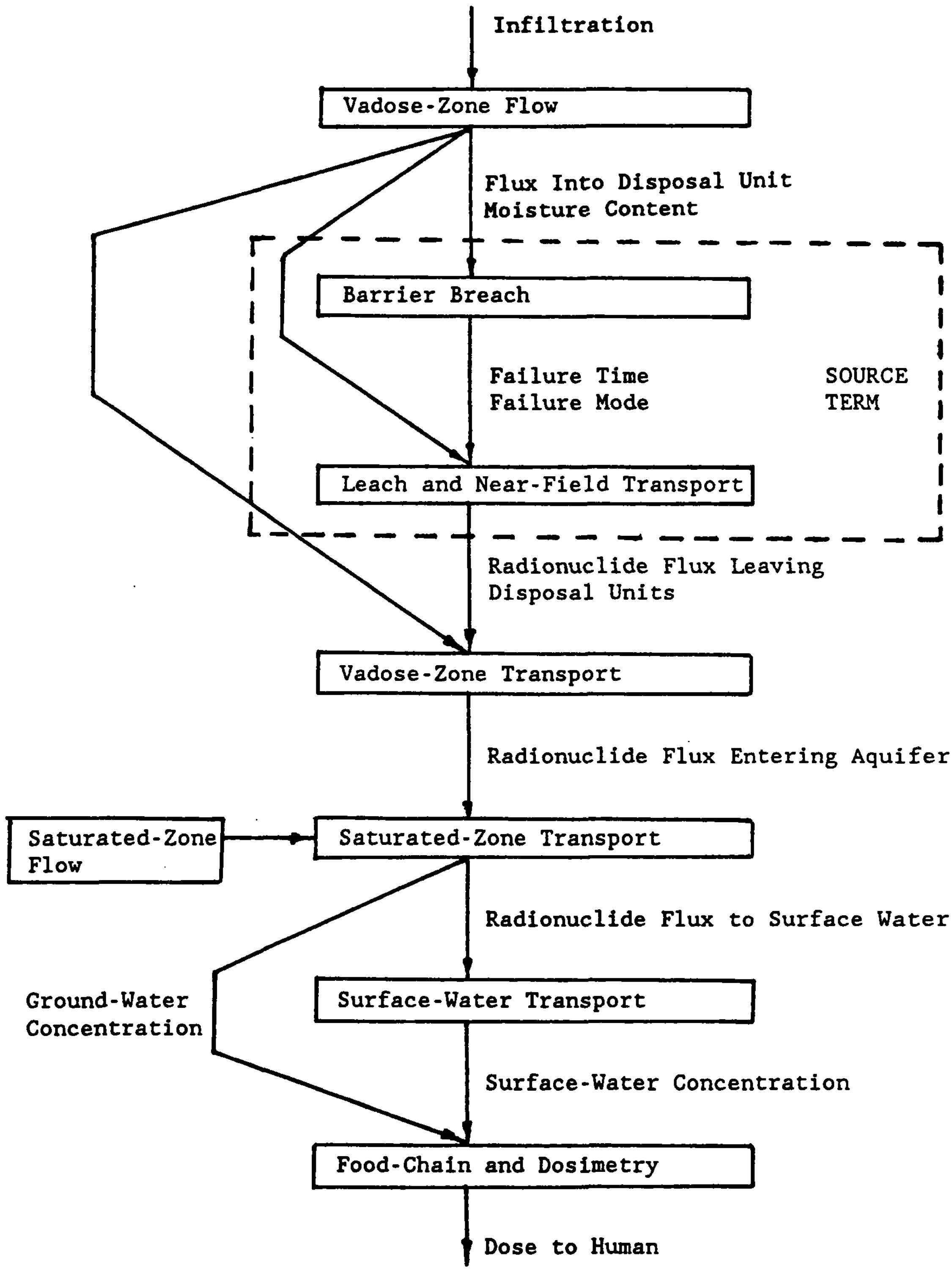

Figure 2-1: Processes in Low-Level Waste Performance Assessment for Ground-Water and Surface-Water Pathways

Percolating water impinges on the engineered cover of the disposal units. The water that passes through the engineered cover into the disposal units induces failure of concrete and steel barriers in and around the disposal units. As waste becomes accessible to the water, it can dissolve by leaching processes, and can be transported to the boundary of the disposal units (near-field transport). The overall set of processes leading to release of radionuclides from the boundaries of the disposal units is called the source term in this report. Radionuclides exiting the disposal units are convected and dispersed by water flowing in the vadose and saturated zones. Once the radionuclides enter the saturated zone, the potential exists for an individual to become exposed to them by using contaminated well water. If the aquifer is in hydraulic connection with surface waters, the potential exists for the surface waters to become contaminated, after which an individual can become exposed to the surface water.

Once either ground water or surface water becomes contaminated, the potential exists for humans to contact the contaminants in a number of ways. A person may drink contaminated water, or the water can contaminate the food chain. This contamination may occur naturally (contamination of fish in the surface water or root uptake of ground water) or through man-made intervention (consumption of well water or irrigation of crops). If these contaminated foods are consumed by an individual, they add to the total intake of radionuclides received by that person. Consumption of contaminated water and food leads to an internally received dose. Similarly, use of contaminated surface water for recreation can lead to an externally received dose that must be accounted for in the methodology. The sum of the doses from all radionuclides transported along all these pathways is the total dose to the receiving person.

These processes are modeled in this methodology by analytical methods and computer codes shown in Fig. 2-2 for each pertinent process discussed above. The capabilities of these methods and codes are summarized in Table 2-1. Vadose-zone flow processes are modeled using VAM2D [Huyakorn et al., 1989]. A multidimensional analysis of the engineered cover is recommended in most cases. From this multidimensional analysis, a one dimensional flow field in the waste disposal units can often be estimated for use in the mixing-cell cascade source-term model. More detailed source-term analyses can be performed using BLT [Sullivan and Suen, 1989] or VAM2D. Barrier breaching processes are modeled as a simple delay time to the onset of releases; as improved modeling methods are developed, they will be incorporated into the methodology. Including updated methods is made easier by the modular nature of the methodology. Radionuclide transport in the vadose zone is treated as a delay time between release from the boundary of the disposal units and entry into the aquifer. This approach corresponds to neglecting dispersion in the vadose zone. If vadose-zone dispersion is important to the site conceptual model, the analyst should use either VAM2D or BLT to analyze vadose-zone transport. Once the radionuclides enter the aquifer, transport processes are analyzed using DISPERSE for well concentrations, and SURFACE for transport through ground water to a surface-water body. If the conceptual model for the saturated zone is too complicated to be

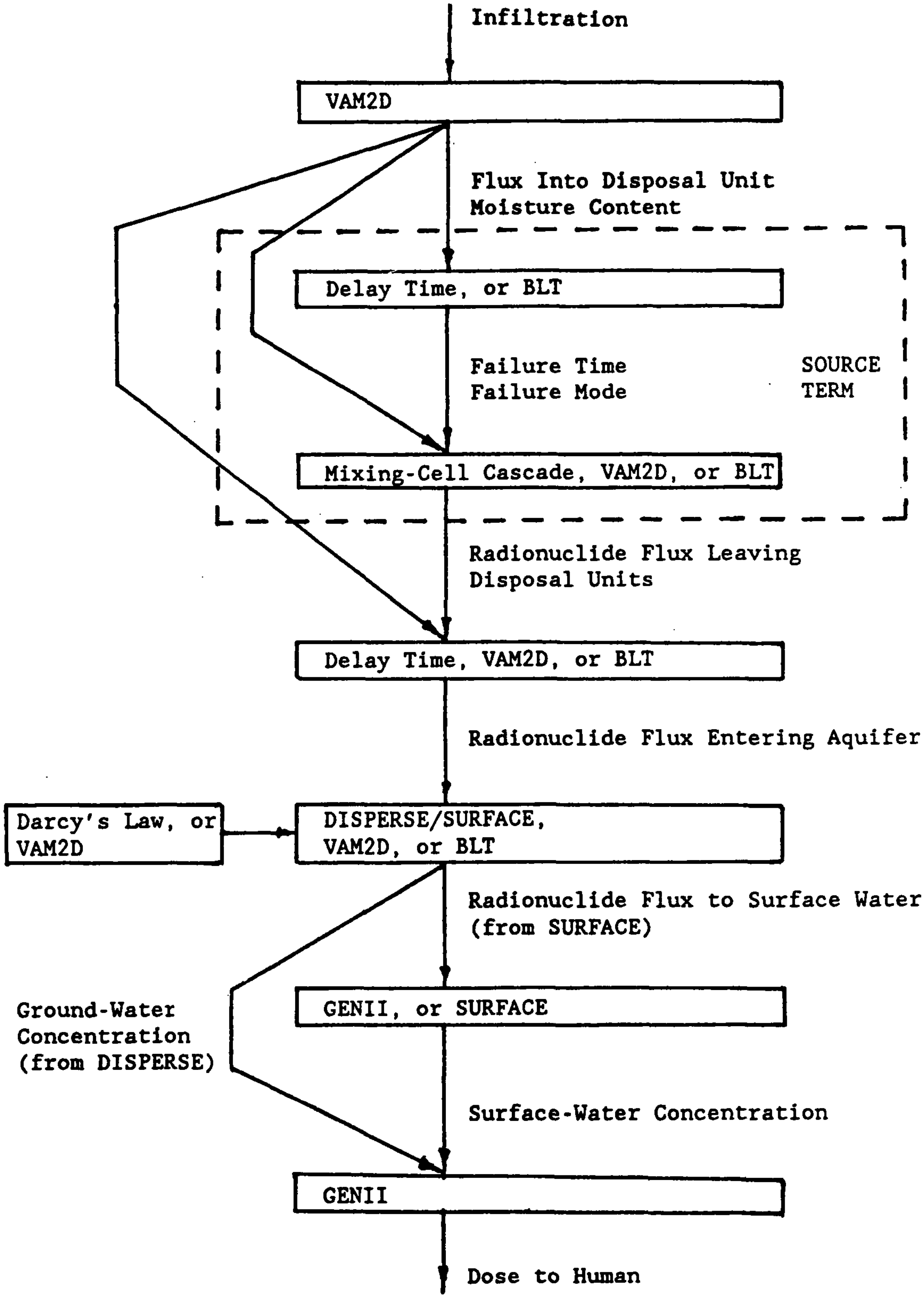

Figure 2-2: Models and Codes in the Performance Assessment Methodology for Ground-Water and Surface-Water Pathways

adequately modeled using these codes, the analyst should use either VAM2D or BLT. Saturated-zone flow processes can be modeled using either a simple one-dimensional Darcy's law approach, or by VAM2D, depending on the complexity of the conceptual model. The results of the ground-water transport analyses provide adequate input information for GENII [Napier et al., 1988]. Analyses performed for this methodology using GENII are (1) dilution in surface water, (2) air transport analyses, (3) food-chain and pathway analyses, and (4) dose calculations.

Table 2-1: Capabilities of Methods and Codes in the Methodology

VAM2D	Two-dimensional finite-element computer code for the analysis of flow and transport through porous media. Both vadose-zone and saturated-zone analyses.
BLT	Two-dimensional finite-element computer code for source-term analysis. Has ability to simulate container breach, contaminant leach, and radionuclide transport. Coupled with FEMWATER for flow analyses.
Mixing-Cell Cascade	Analytical solution for source-term analysis. Contains contaminant leach models, and accounts for arbitrary near-field dispersion.
DISPERSE	Numerical integration of analytical transport solution for the analysis of well concentrations. One-dimensional convection, three-dimensional dispersion, and time-varying source term.
SURFACE	Numerical integration of analytical transport solution for the analysis of flux to a surface-water body. One-dimensional convection, three-dimensional dispersion, and time-varying source term. Contains simple dilution model for surface-water concentrations.
GENII	Food chain, air transport, surface-water transport, and dosimetry analyses.

Both DISPERSE and SURFACE have been incorporated into a user-friendly shell, called SUNS (Sensitivity and UNcertainty analysis Shell), which facilitates data input, and which provides a variety of tabular and graphical output options. Details on the SUNS package will be described in a Self-Teaching Curriculum for the methodology.[2]

2.3 Data Requirements for the Methodology

In this section the data required in each code in the performance assessment methodology are explicitly presented. These data must, in general, be collected as part of the site characterization performed by the licensee. The data listed in this section are the data needed to analyze a relatively simple conceptual model using the computer codes listed in Section 2.2. They are not the data needed to generate a conceptual model from site characterization. Site characterization data are generally more extensive than data used in the performance assessment, and the performance assessment data set is usually a subset of the site characterization data set. The conceptual model bridges these two levels of detail in data, and in addition defines the important physical and chemical processes controlling the behavior of the site.

The models in the performance assessment methodology are for (1) ground-water flow, (2) source term, (3) ground-water transport, (4) surface-water transport, (5) air transport, and (6) pathways and dosimetry. Data requirements for each of these models are given in Tables 2-2 to 2-7. Data requirements for each of these kinds of models are given as the explicit requirements for the computer codes in the methodology, which were described in Section 2.2. The data requirements in the tables do not include information calculated from one module for use in the next module. For instance, moisture content and Darcy flow in the facility are needed for source-term analysis, but these are provided from the ground-water flow analysis, hence are not listed as source-term data requirements.

2.Chu, M. S. Y, M. W. Kozak, J. E. Campbell, B. K. Thompson, and P. A. Mattingly A Self-Teaching Curriculum for the NRC/SNL Low-Level Waste Performance Assessment Methodology, NUREG/CR-5539, SAND90-0585, Sandia National Laboratories, in press.

Table 2-2: Data Requirements for Ground-Water Flow Modeling

- An infiltration boundary condition is needed for estimation of flux into the facility. Infiltration must be determined from one of a number of analysis and measurement methods. Determination of an appropriate method for estimating infiltration is a site-specific issue. In general, infiltration represents the difference between site-specific rainfall and evaporation, transpiration, and runoff.

- Physical dimensions of the flow domain: depth to water table, dimensions of the engineered cover, thickness of aquifer, etc.

- Soil properties for each soil of interest in the natural surroundings, and in the engineered cover. These properties include porosity, effective porosity, saturated hydraulic conductivity, and soil bulk density.

- Unsaturated-zone soil properties for each soil of interest in the natural surroundings, and in the engineered cover. These properties are the characteristic curves (θ-Ψ curves) and the conductivity curves (K-Ψ or K-θ curves), which include information on residual moisture content and saturated moisture content. In VAM2D the characteristic curves and conductivity curves can be specified as parameters of the van Genuchten equation or the Brooks-Corey equation, which are both empirical equations.

- Hydraulic head distributions in the aquifer, and hydraulic gradient, generally determined from field measurements in observation wells.

Table 2-3: Data Requirements for Source-Term Modeling

- Inventory by radionuclide either at the time of closure or at the time at which releases begin. It must also be specified if the waste is stabilized or unstabilized, and if the types of waste are physically separated, as in separate trenches for Class A waste and Class B/C waste.

- It may be possible in some cases to specify data that indicate limitations on radionuclide releases. These data may include solubility limitations, sorption capabilities (K_d) in the facility, or diffusion limitations (specify diffusion coefficient in the waste form and container dimensions) for stabilized waste. It must always be emphasized that any assumption about release limitations must be strongly justified by the licensee using site-specific conditions. Conservatism should always be the guide in making assumptions about release limitations.

Table 2-4: Data Requirements for Ground-Water Transport Modeling

- Soil properties are required that have already been discussed in Table 2-3.
- Longitudinal and transverse dispersivities must be specified for the soil below the water table. Dispersion is neglected in the unsaturated zone. Dispersion data can be estimated from small-scale experiments, for such dispersivities result in conservatively small dispersivities.
- Retardation in the aquifer can be included, but conservatively small values should be used, and then only if justified on a site-specific basis. Retardation is calculated from site-specific K_d measurements.

Table 2-5: Data Requirements for Surface-Water Modeling

- The surface hydrology must be characterized using field measurements, including river flow rate, exchange flow rates between the surface water and ground water, or between surface-water bodies.
- Transport parameters required in the GENII surface-water transport model are average depth and width of the surface-water body, average water flow rate in the surface water, rate of water flow from the aquifer (effluent discharge), distance from the release point to the receptor (which must be assumed for a given scenario), transit time to irrigation withdrawal, and offshore distance to water intake.

Table 2-6: Data Requirements for Air Transport

- For the simple Gaussian-plume models in GENII the only information needed are estimates of wind speed, distance to receptor, and atmospheric stability class. In general, it should be appropriate to use this method with conservative estimates of each of these data. These are also the only data needed to implement the virtual source method for analyzing area sources [Kozak et al., 1990].
- Somewhat more complicated Gaussian-plume models are also included in GENII that can account for a variety of site specific conditions, and may require information on joint frequency data, terrain features, and atmospheric stratigraphy. These more complicated models are not expected to be needed often.

Table 2-7: Data Requirements for Pathways and Dosimetry

- A number of parameters must be specified in GENII for food-chain analyses: consumption rates and holdup times for meat, poultry, milk, eggs, leafy vegetables, other vegetables, fruit, and cereals. In addition, the irrigation rate and source of irrigation are required.
- Parameters in GENII for recreational exposures are hours of exposure from swimming, boating, and shoreline activities, and surface-water transit time from release point to recreational site. These must generally be assumed for a particular scenario.

2.4 Procedure for Applying the Methodology

The purpose of this performance assessment methodology is to allow NRC to confirm whether a licensee's analyses and assumptions are reasonable. This goal can potentially be reached in one of several ways. The NRC staff may choose to use the methodology in its entirety to perform a full performance assessment, or they may choose to use parts of the methodology to analyze subsets of the licensee's assertions. The procedure for performing the complete analysis should be similar in principal to the approach that should be used by the licensee during development of the full performance assessment.

In order to develop a full performance assessment analysis that can be meaningfully compared to the performance objectives in 10 CFR Part 61.41, the analyst should begin by identifying appropriate pathways that may be important at the specific site. These pathways should then be ranked in order of importance to the overall performance of the site. The next step in the analysis should be to develop a conceptual model of the site that appropriately accounts for transport of radionuclides through the crucial pathways previously identified. Once a conceptual model has been developed, the processes can be analyzed using appropriate mathematical models. The model results, including associated uncertainties, can then be compared to the performance objectives. In practice, this approach should be iterative; the choice and ranking of pathways, conceptual model, and choices of mathematical models are all candidates to be tested and revised on successive iterations, which should converge to a satisfactory and defensible performance assessment of the site.

The remainder of this section is devoted to a brief description of the sequential procedure for applying the methodology for analysis of releases via the ground-water pathway. The overall procedure and principals in developing a conceptual model and implementing the methodology are similar for all performance assessment analyses. However, the ground-water pathway is expected to be the crucial pathway for many low-level waste facilities [Shipers and Harlan, 1989]. The procedure for applying the methodology to the analysis of a ground-water pathway is shown pictorially in Fig. 2-3. Further discussions on the models, assumptions, and data requirements are given later in this report.

The first step in modeling a ground-water system is the development of a conceptual model. This development involves an abstraction of site characterization data into a form that is capable of being modeled. This generally involves imposing a number of simplifying assumptions, including simplification of the appropriate governing equations to reflect the physical situation. Simplifications are usually made about the geometry of the system, spatial and temporal variability of parameters, isotropy of the system, and also about the influence of surroundings on the behavior of the system.

A good conceptual model for performance assessment accounts for the important aspects of the system without involving excessive complexity. Clearly identifying the necessary level of complexity is not a trivial task, and the analyst should emphasize descriptions of the conceptual model, and discuss how it incorporates the important characteristics of the site. The complexity of the model is dependent on the purpose of the analysis [National Research Council, 1990]; for instance, a conceptual model for use in performance assessment may be simple if it provides satisfactory confidence in site performance. A conceptual model can be considered adequate if (1) it accounts for the most important physical, chemical, and geological characteristics of the system, and (2) it adequately represents the response of the system to changes in stresses. In practice, the development of a conceptual model proceeds by making an assumption, testing the assumption against available site-specific data, and modifying the result based on the comparison with data. However, there are no rigid rules in the development of conceptual models.

The next step in the performance assessment is to analyze the site hydrology. Performance assessment calculations are performed for long periods of time, and seasonal or annual variations in hydrology may not contribute greatly to the performance of the site. Consequently, one can usually assume steady-state flow. This assumption cannot be made, however, when the system is artificially stressed over a long period of time. For instance, significant use of well water by a large population may alter aquifer characteristics over tens of years. Such stresses can lead to temporal and spatial variabilities that cannot be ignored. In some cases the direction and velocity of ground-water flow may change.

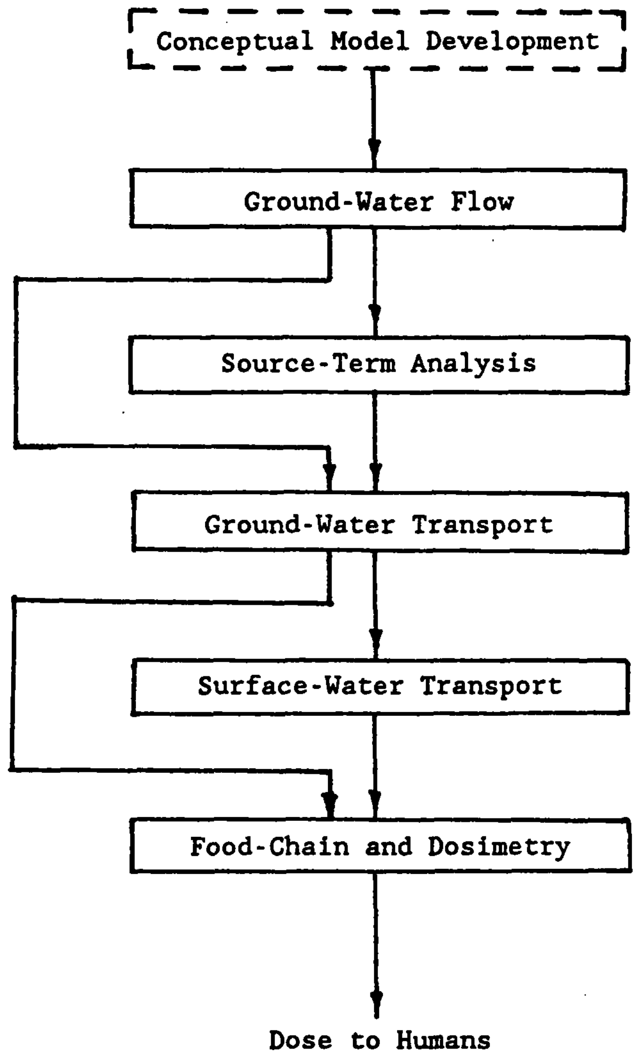

Figure 2-3: Performance Assessment Methodology Procedure for Ground-Water and Surface-Water Pathways

Analysis of the engineered cover system will usually require multidimensional simulation. Much of the water entering the cover is diverted to the edges of the cover; this is the purpose of the cover system. The partitioning of water that flows through the cover and water that is diverted is a multidimensional process, and is usually best analyzed as such. The computer code VAM2D [Huyakorn et al., 1989] has been recommended for use in this analysis based on its flexibility in handling difficult simulations and highly nonlinear soil properties [Kozak et al., 1990]. Once the flux of water into the facility has been determined, unsaturated zone flow through and below the disposal unit can usually be assumed to be one dimensional. This simplified flow field can be excerpted from the multidimensional flow field generated using VAM2D. In the event that an approximate one-dimensional flow field cannot be generated from the VAM2D simulation, VAM2D can be used for the full analysis of percolation, source term release, and ground-water transport.

Darcy velocities and moisture contents determined in the ground-water flow analysis are required inputs both for source-term calculations, and for ground-water transport analysis. The source-term is defined here as the release of radionuclides from the boundary of the disposal units. For the ground-water pathway this includes models for breach of engineered barriers, leach rates of radionuclides from waste forms, and the transport and mixing of radionuclides with water that occurs within the facility.

Failure of concrete structures is modeled in this methodology as a delay time to the onset of releases. There is no adequate existing model to analyze the details of failure of concrete structures to estimate the failure time [Clifton and Knab, 1989].

One of two methods can be used to analyze the breach rate of waste form containers in the methodology. A simple approach can be used, in which the failure of containers is modeled as a delay time to the onset of releases. Alternatively, the method of Sullivan et al. [1988] can be used to analyze the breach of carbon-steel containers. This method is incorporated into the BLT computer code [Sullivan and Suen, 1989].

In this methodology leaching processes are treated as either a simple rinse model, or as diffusion-limited leaching. The amount of dispersion in the disposal units can have a marked effect on the release rate from the facility; hence it is inappropriate to use a model that does not adequately span the range of possible dispersive behavior in the disposal units [Kozak et al., 1990]. In this methodology dispersion in the disposal units is described by a cascade of mixing cells; this approach allows the analyst to model to full range of dispersive behavior.

The source-term models used in this methodology describe the release of a single radionuclide, so the source-term analysis must be repeated for each radionuclide in the inventory. The result is a set of release-rate histories that provide the input to ground-water transport analyses. Ground-water transport analyses include transport from the facility to the aquifer, and transport in the aquifer to a nearby well, and to nearby surface water, if any. Transport in the unsaturated zone can be analyzed simply for many facilities. If the water table is shallow, one can neglect the unsaturated zone and assume source-term releases occur directly into the aquifer. For sites at which the water table is deep, the methodology treats the unsaturated zone as a delay time between release from the disposal units and entry into the aquifer. If dispersion in the unsaturated zone is of primary importance in the conceptual model, VAM2D can be used to analyze unsaturated-zone transport.

Kozak et al. [1990] have discussed analytical solutions based on a Green's function approach for transport in the saturated zone. For an arbitrary source-term release function, the Green's functions must be integrated numerically. These numerical integrations are implemented in the computer codes DISPERSE and SURFACE, which are updated versions of the computer codes described by Codell et al. [1982]. More detailed transport simulations can be run if the conceptual model merits the detail, but the simple approaches have been shown to be conservative for a simple conceptual model [Kozak et al., 1990]. Computer codes available for detailed transport analyses are VAM2D and BLT [Sullivan and Suen, 1989]. Both source-term and transport analyses are intrinsically transient, for the inventory decreases with time. Therefore the source-term and transport analyses are modeled as transient processes that occur in a steady-state flow field.

In the event that the contamination in the aquifer discharges to a surface-water body, the results of the ground-water transport analysis must include the flux of radionuclides into any nearby surface water. In this case the surface-water pathway must also be considered in the conceptual model.

Once the concentration of each radionuclide is determined as a function of time in well water and surface water (if necessary), the dose history can be generated by applying environmental pathway and dosimetry analyses for each time of interest. Pathways and dosimetry models consist of simple, linear, multiplication factors that convert an environmental concentration to an annual committed dose. Appropriate NRC pathways models [NRC, 1977a] and ICRP dosimetry models [ICRP 26, 1987] are implemented in GENII.

2.5 Applications of the Methodology in the Regulatory Process

This performance assessment methodology provides a tool for the NRC to use in compliance assessments of low-level waste site license applications. In addition, the methodology may be used to perform parametric analyses on the processes of interest. These analyses may provide insight into which parameters are most important to demonstrating compliance with regulatory performance measures, hence which data are most crucial. Furthermore, parametric analyses can be used to identify design changes or inventory changes that can potentially improve facility performance.

The more detailed hydrology codes used in the methodology may also find application in compliance demonstrations for the technical requirements of Subpart D of 10 CFR Part 61.50. For instance, transient analysis of the ground-water system may be used to demonstrate that changes in rainfall do not cause the water table to intrude into the facility. The detailed codes may also be used to analyze site characterization data in the development of a conceptual model. In this role, the detailed codes may be used to provide justification that a simple conceptual model is appropriate.

3.0 GROUND-WATER FLOW MODELING

A key to proper ground-water flow analysis is the development of an appropriate conceptual model. General approaches to developing a conceptual model from site characterization data are discussed in Section 2.4 of this report, and in Kozak et al. [1989a]. In the overall scheme of the methodology, these flow fields are analyzed prior to the source-term and ground-water transport analyses. In large measure, the complexity of the ground-water flow conceptual model will dictate the complexity needed for transport analysis.

A ground-water flow field is necessary for modeling both the source term and the rate of radionuclide transport through ground water. The ground-water flow analysis has three main aspects: (1) modeling surface processes that lead to infiltration (2) modeling flow of water into, through, and below the disposal units to the water table, and (3) modeling aquifer flow. In this chapter are discussed appropriate models, assumptions, data requirements, and uncertainties associated with developing and performing the ground-water flow analysis of a conceptual model. The flow models in this methodology are based on models of porous media; the capability to analyze fractured media is outside the scope of the methodology.

3.1 Unsaturated-Zone Flow Modeling

3.1.1 Infiltration and Percolation

Precipitation is subject to evaporation, transpiration, surface runoff, and infiltration. Infiltration is the downward entry of water into the soil. Percolation is the movement or redistribution of infiltrated water through the unsaturated zone [USGS, 1989]. In performance assessment, percolation has two components of interest. A portion of percolated water is diverted by the engineered cover around the waste-containing disposal units. The amount of percolated water that is not diverted, which we call flux into the disposal unit, is of particular importance in source-term and transport modeling. Water flux into the disposal unit can occur directly through the engineered cover system, or in some circumstances may be due to lateral flow under the cover. Operationally, in this methodology the flux into the disposal unit is determined using a flow simulation of the vadose zone using infiltration as a boundary condition at the upper edge of the computational domain. In this section a number of methods are discussed for determining infiltration for use as a boundary condition in percolation analyses.

The primary features that will influence the balance between infiltration and evaporation, transpiration, and runoff are surficial ones: the type and thickness of the topsoil, the type of vegetation, and the surficial topographical features of the site [Fenn et al., 1975]. In addition, the temporal and spatial variability of climatic processes can be important in estimating infiltration. Furthermore, to adequately estimate infiltration for the long time periods of a performance assessment, the analyst must make some assumption about the future state of the surficial characteristics and climatic conditions. One of two assumptions can be made about the future state of the site: (1) it may be reasonable to assume that the future characteristics of the site are similar to the present characteristics, or (2) the present and future characteristics may be reasonably assumed to be dissimilar. In this methodology, we do not specifically address climate change, and limit the following discussion to the influence of surficial characteristics. The primary effect of including climate change in the methodology will be to increase the uncertainty in the value of infiltration.

If the present-day and expected future characteristics of the site are similar, one can use estimates of present-day infiltration to predict future infiltration. The assumption here is that the facility will not significantly influence the surficial characteristics of the site. For instance, if the site is currently on a grassy plain, and if the closure plan includes replanting native grasses over the cover, it seems reasonable to assume that the runoff, evaporation, and transpiration characteristics of the site will be similar before and after. On the other hand, if a site is currently tree-covered, but is expected to be grass-covered following closure, one cannot directly use estimates of present-day infiltration to predict future infiltration. When present and future states are markedly dissimilar, there will be much more uncertainty in infiltration predictions than if the present and future states are expected to be similar.

Gee and Hillel [1988] concluded that tracer tests, along with lysimeter measurements, offer the best methods for estimating infiltration at arid sites. At humid sites, they concluded that the errors in water balance techniques may be acceptable, since infiltration is a significant fraction of rainfall. Knutsson [1988] has provided a comparison of the physics of infiltration in humid and arid climates. Knutsson concluded that, given the uncertainties associated with each method, several techniques based on independent input data should be applied at any specific site, but that no unique method was generally appropriate at either humid or arid sites. Balek also concluded that none of the available techniques used in estimating infiltration are generally acceptable [Balek, 1988]. For the purposes of estimating infiltration for performance assessment, it is therefore recommended that the licensee should provide the results from several separate techniques, so that the results can be compared [Foster, 1988; Johansson, 1988]. Below is a brief discussion of some of the available techniques.

Tracer Tests. Tracer tests involve correlating the movement of some tracer in the recharging water to the movement of water. Two of the most promising tracers are tritium and ^{36}Cl residual contamination from atmospheric nuclear bomb testing. The concentration of these tracers in the vadose zone, together with the history of atmospheric bomb testing, can be used to estimate the rate of downward movement of water over the past 30-40 years.

Lysimeter Measurements. Gee and Hillel [1988] note that lysimeter measurements can yield accurate data on infiltration, and that lysimeter measurements are the only direct measurements of infiltration. The drawbacks to the use of lysimeters are identified as the large expense of their construction and maintenance, disturbance of the vegetation and soil, modification of the bottom boundary condition relative to the field, and localized nature of the data collected.

Water Balance Methods. Areal water balances involve measuring all components of the water balance except infiltration, and performing a straightforward subtraction to estimate infiltration. Simulation models of the water balance elaborate on this technique, and include empirical or semi-empirical relationships for plant transpiration and the relationship between transpiration and evaporation. Simplified water balance methods are used when there is an absence of detailed plant and soil information. Each of the water balance methods requires the analyst to have measured estimates of actual evapotranspiration and runoff, both of which are difficult to measure with precision. In arid regions, infiltration is a small fraction of precipitation and evapotranspiration, and large errors in the estimated infiltration (an order of magnitude or more) can result from subtracting two similar numbers to get a small number [Gee and Hillel, 1988]. In humid regions, the expected percentage error in infiltration should decrease, but the numerical value of the error will remain about the same.

Kozak et al. [1989b] reviewed several computer codes for calculating infiltration. All of the codes were based on the water-balance method, hence each one relies heavily on the accuracy of evapotranspiration measurements. None of the computer codes had been satisfactorily compared with field data, and there were no features that clearly distinguished one code from the next. As discussed in Chapter 2, these problems, together with the large potential errors associated with the water balance method in arid climates led us to conclude that water-balance codes are not sufficiently flexible for use in this methodology. However, it should be noted that in some circumstances, particularly in humid climates, water balances can provide a useful approximation for infiltration.

Other methods are available that primarily consist of various methods of measuring evapotranspiration. These methods may be applicable to specific sites, but are not generally well established.

Given the amount of uncertainty associated with the estimation of infiltration, it is important to address this uncertainty in the performance assessment of a low-level waste site. Although preliminary evidence shows that flux into the disposal units may not be a strong function of infiltration when the disposal units have an appropriate engineered cover [Kozak et al., 1990], infiltration should be varied parametrically to assess the impact of the uncertainty of infiltration on the final results in a performance assessment analysis.

The analysis of flux into the disposal units must account for several phenomena. First, as discussed earlier, the model must be able to distinguish the effect of differing values of infiltration. Infiltration enters the analysis of percolation as a boundary condition at the upper boundary of the analysis domain. Second, since low-level waste facilities can be expected to have an engineered cover, the percolation model must be able to analyze soil layers with greatly differing hydraulic properties. Third, it may be necessary to analyze the effect of soil heterogeneity below the cover in the disposal units, and between the disposal units and the water table.

A moisture-barrier cover is usually included as part of the design of a low-level waste facility. Designs for cover systems typically include several soil layers that provide low permeability coupled with high capillarity [Herzog et al., 1982]. Flow through such barriers is intrinsically multi-dimensional, since the purpose of the engineered cover is to laterally divert a vertical flow rate. Furthermore, flow around the cover can be an important path for water to enter the disposal units [Fayer et al., 1985; Suen, 1988]. Consequently, it is usually necessary to use multi-dimensional analysis to determine the performance of the cover. If one-dimensional analyses are used in the performance assessment, it is necessary to compare these with a multi-dimensional model of the cover to demonstrate that the one-dimensional model provides a satisfactory representation of the cover behavior.

Over the long time frames of interest in performance assessment, steady-state flow modeling is often performed, and for this case percolation should be modeled as steady state. Both constant-flux and constant-head boundary conditions have been proposed as representative of infiltration [Johnson et al., 1983]. However, a flux boundary condition appears more compatible with the results of infiltration estimation methods.

3.1.2 Unsaturated-Zone Flow Beneath the Disposal Unit

Flow in the unsaturated zone between the disposal units and the water table can generally be expected to be downward, but horizontal flow can be caused by soil layering, fractures, anisotropy, land shape, capillary forces, and transient boundary conditions. These influences can be important in modeling flow in the natural soils near the facility, and also in modeling flow into the disposal units. The presence of these complicating features may also suggest that multi-dimensional analysis of percolation is appropriate.

A number of computer codes were reviewed by Kozak et al. [1989b] for use in the analysis of vadose-zone flow. Several were rejected due to code complexity and difficulties in documentation. Several others were rejected due to a lack of flexibility in handling a wide variety of site conditions. However, each of the codes evaluated implements the appropriate physical concepts, and each can potentially be used for the analysis of vadose-zone flow for specific conditions. Kozak et al. [1990] implemented FEMWATER [Yeh and Ward, 1980] and VAM2D [Huyakorn et al., 1989], and concluded that VAM2D appears to contain numerical methods that are more flexible in handling a variety of soil properties.

In this methodology it is recommended that flow in the vadose zone be analyzed using VAM2D. VAM2D is a two-dimensional finite-element code for the analysis of ground-water flow and transport in porous media. Using VAM2D allows the analyst to examine the multi-dimensional effects caused by the trench cover system, and allows transient flow analyses if such are necessary. The flow field generated using VAM2D can often be used to develop an approximate one-dimensional flow field below the cover through the disposal units. This one-dimensional flow field can be used in the source-term analyses in DISPERSE and SURFACE, the ground-water transport codes in the methodology. The flow field generated by VAM2D can also be used to estimate the ground-water travel time from the bottom of the disposal units to the aquifer. In a complicated situation, in which a simplified flow field cannot be extracted from the VAM2D analysis, both flow and transport can be analyzed using VAM2D.

The greatest sources of uncertainty in modeling the unsaturated zone are the site characterization data themselves. Three sources of uncertainty that arise from these data are

- The infiltration boundary condition, as discussed in Section 3.1, is difficult to specify accurately from imprecise field data for evaporation, transpiration, and runoff. Furthermore, uncertainty is inherent in the extrapolation to future site surficial conditions

- The soil is spatially variable, and it is difficult to quantify underground structures without using a large number of boreholes. However, using too many boreholes can excessively disrupt the site hydrology. The naturally incomplete nature of these data leads to uncertainty in the conceptual model of the site, and also in parameter values used in the model.

- It is difficult to retrieve an undisturbed soil sample to use in determining the soil properties. Dis'urbed soil can have markedly different properties than the in situ soil, and this leads to uncertainty in values of model parameters.

3.2 Saturated-Zone Flow Modeling

The goal of analyzing the saturated zone for the performance assessment methodology is to model the rate and direction of ground water flow underlying the disposal facility for use in transport analyses. Most approaches to modeling the saturated zone are based on Darcy's law, which simply says that the flux of water through the aquifer is proportional to the total-head gradient. The proportionality constant in Darcy's law is known as the hydraulic conductivity. Hydraulic conductivity is generally spatially variable, and is sometimes significantly anisotropic. Data needed for modeling the saturated zone can be expected to be collected as part of site characterization [NRC, 1988]. These data are hydraulic conductivity as a function of position, and measured head as a function of position.

In some cases it may be necessary to account for transient flow; analysis of a well pumping large volumes of water (as in a municipal supply well) is one example of a case in which unsteady flow may be important. In this case the well can cause a large and time-varying cone of depression, and can significantly alter local flow rates.

The greatest source of uncertainty in modeling aquifer flow is derived from the deduction of model parameters from field data. Spatial variability couples with limited access locations (boreholes and monitor wells) to produce uncertainty in parameter values.

Many of the codes examined for use in saturated-zone analysis in this methodology were the same as the codes for vadose-zone analysis [Kozak et al., 1989b]. The codes were rejected for use in this methodology due to difficulties in code documentation, and due to lack of flexibility in handling a variety of situations. Nevertheless, some of the codes rejected for use in the methodology may well contain attractive features for use at a specific site. Kozak et al. [1990] determined that VAM2D [Huyakorn et al., 1989] appeared to be the most user-friendly and flexible candidate for the analysis of multi-dimensional ground-water flow.

The transport codes DISPERSE and SURFACE used in this methodology (which are described more fully in Chapter 5) require a one-dimensional, steady-state Darcy velocity field. One-dimensional flow can frequently be justified for a limited spatial domain from site characterization data. In some cases a regionally multi-dimensional flow field can be subdivided into locally one-dimensional segments with differing properties [Rood et al., 1989]. The use of steady-state flow is a common approach in performance assessment analyses, in which the long-term steady behavior is considered of more importance than seasonal or yearly variability. For more complicated situations involving multi-dimensional, spatially variable, or transient aquifer flows, VAM2D [Huyakorn et al., 1989] can be used for both flow and transport analyses.

Determination of the necessary complexity of the flow field is part of the development of the conceptual model of the site, which has been discussed in Section 2.4 of this report. In summary, there are no fixed rules for determining the appropriate level of complexity in the ground-water model, and this must be determined by professional judgement and experience of the analyst [National Research Council, 1990].

4.0 SOURCE-TERM MODELING

The source term in this methodology is defined as the time-dependent rate of radionuclide release from the boundary of the facility's disposal units. This definition means that the source-term analysis must account for the quantity of disposed radionuclides, physical and chemical forms of the radionuclides, breach rate of containers and degradation of engineered barriers, leach rates of radionuclides from containers, and transport and mixing of radionuclides with water in the disposal units.

The major components of a source-term calculation for routine releases from the facility are illustrated in Figure 4-1. In this figure, models for each of the important processes are represented by the blocks, and output from one block serves as input to the next block. Percolation analysis, which was described in Chapter 3, requires hydrologic, infiltration, and facility-design information, and is used to calculate the flux of water that enters the disposal units. This flux, together with the moisture content in the disposal unit, are required inputs for the source-term analysis. Once the water contacts the waste form, a container-degradation model is needed to simulate container failure mechanisms. The output from this model is used to estimate when and how containers fail. Similarly, the inhibition of radionuclide releases by engineered barriers, such as concrete vaults, must be able to be modeled in some fashion. After the containers and barriers fail, water contacts the waste directly and releases radionuclides by leaching mechanisms. Finally, a near-field transport model is used to calculate radionuclide transport to the boundary of the disposal unit.

A low-level waste facility is a complicated system, comprised of many engineered features and many different waste species, containers, and forms. Detailed modeling of the physical and chemical features of this complex system is extremely difficult. Consequently, some approximate approach must be used that incorporates the essential processes that are expected to occur in the disposal units. Hence, it is necessary to first identify the important features that characterize low-level waste disposal practices, and to identify important physicochemical processes that influence radionuclide release rates.

Each of the processes in the source-term analysis can be strongly influenced by geochemical processes in the disposal unit. Unfortunately, there is generally a large amount of uncertainty in geochemical processes, particularly in a complex chemical system like a low-level waste disposal unit. Consequently, careful attention must be given to justification of anything that is strongly influenced by the geochemistry in the disposal unit, such as solubility limits or retardation.

Shallow-trench burial is at present the most common method for disposal of low-level waste, but there are several proposed alternative disposal methods. Some of these alternative disposal methods are listed in Table 4-1, along with the major engineered barriers associated with each [Bennett et al., 1984]. Depending on the disposal method, water flow into the disposal unit may involve degradation of, and flow through both natural and man-made materials.

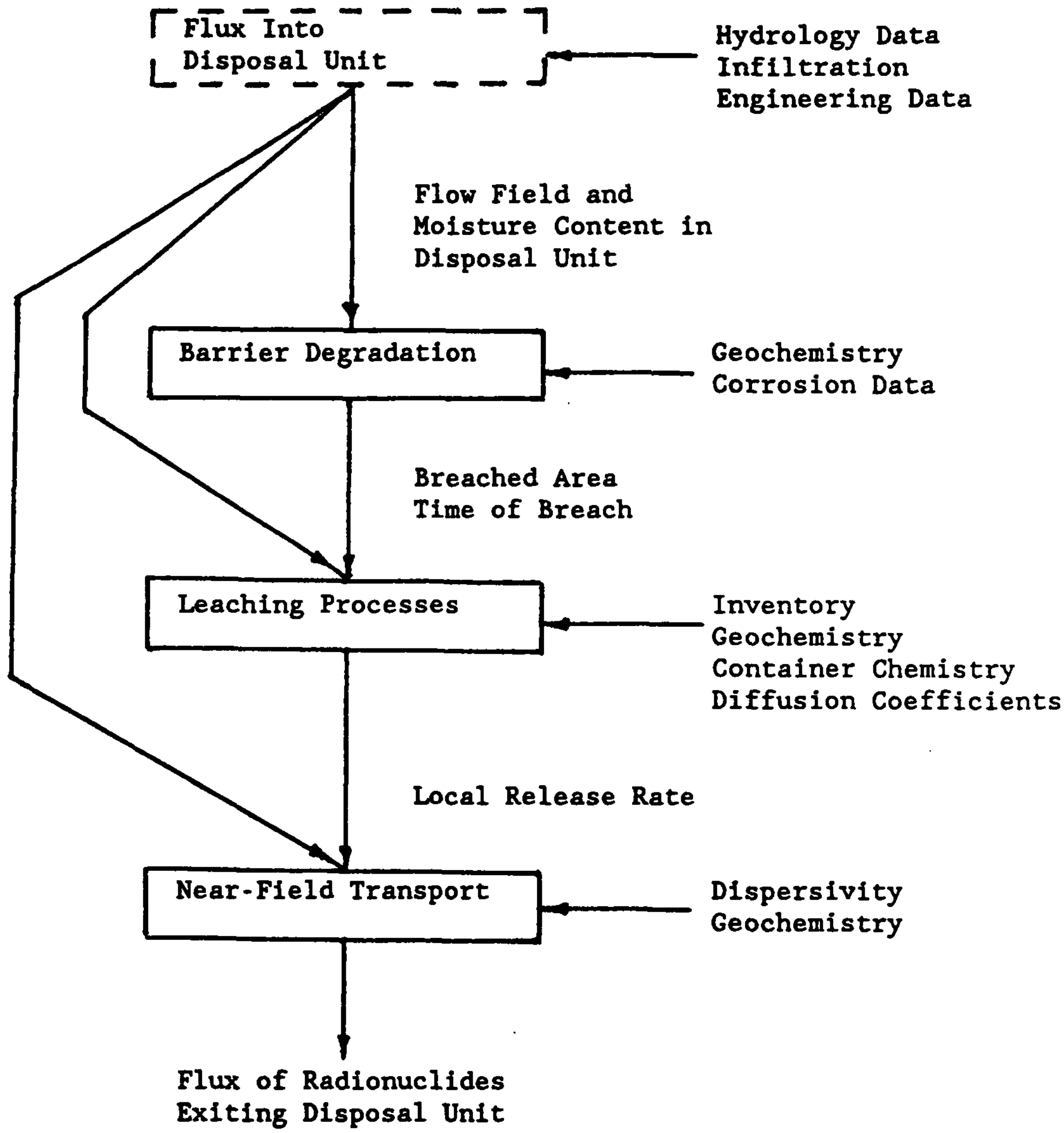

Figure 4-1 Major Components of Source-Term Models

Waste characteristics for low-level waste are given in Table 4-2. It is of interest that Class A wastes occupy most of the volume of a disposal site, but only constitute a low proportion of the total site activity [Sullivan and Suen, 1989 (Appendix 11)]. By contrast, Class C wastes constitute most of the activity in the trench, but occupy only a small volume, and are usually confined to a localized region of the facility.

Table 4-1
Some Possible Land Disposal Methods for Low-Level Waste

Method	Design Essentials
Shallow-trench burial	Trenches covered and capped with natural materials
Below-ground vault	Below-ground facility of engineered materials (e.g. concrete)
Above-ground vault	Above-ground facility of engineered materials (e.g. concrete)
Earth-mounded bunker	High activity wastes embedded in concrete below ground, and low activity wastes emplaced above ground in earthen mounds

Table 4-2
Characteristics of Waste Classifications

Class	Total Activity	Waste Form	Volume of Waste
A	Low	Steel drum	Very high
B	Medium	High-integrity container or solidified waste	Low
C	High	High-integrity container or solidified waste plus barrier against intrusion	Very low

Waste form and container type are important in determining the time to onset of radionuclide release from the waste form. Failure mechanisms may be quite different for different container types. The three container failure mechanisms germane to low-level waste are identified in Table 4-3. It should be noted that metal alloy high-integrity containers will degrade by corrosion in similar fashion as carbon-steel drums, although at a different rate. Radionuclide release mechanisms are identified in Table 4-4 for each pertinent waste form. Surface release mechanisms are those in which transport limitations in the waste form do not play a role. However, chemical limitations such as solubility limits and sorption phenomena may be important.

The modeling approach outlined in this chapter incorporates the important physical and chemical processes in source-term modeling, while retaining a level of simplicity that is appropriate given the usual uncertainty in the chemistry and morphology of the disposal units. The remainder of this chapter is devoted to more detailed discussions of the source-term modeling approach used in the performance assessment methodology.

Table 4-3
Containment Failure Mechanisms

Mechanism	Applicability
Corrosion	Steel drums, activated metals, high integrity containers with Classes B and C wastes.
Concrete degradation	Concrete caisson (Class C), Concrete vault (above or below ground)
Degradation of cover	Earthen trench cover

Table 4-4
Mechanisms of Release From Waste Form

Mechanism	Applicability
Surface Release	Trash, adsorbed liquid, surface species of stabilized waste
Diffusive Release	Solidified waste
Congruent Dissolution	Solidified waste, activated metals

4.1 Inventory

The inventory of a facility is defined here as the amount, type, and waste form of disposed radionuclides. Guidance on the level of detail necessary to identify major components of the waste for a low-level waste facility license application has been given by the NRC [1988]. For the purposes of a performance assessment, it is necessary to have projected estimates of the inventory of each radionuclide to be buried at the site. Each radionuclide is treated separately in the methodology and each radionuclide is assigned its own quantity, and physicochemical properties based on inventory information, operator experience, and generator projections. If some of the inventory is assumed to be in stabilized waste forms in which releases are limited by diffusion, data must be available for diffusion coefficients and container dimensions. In addition, data must be available to allow an estimation of the convective water velocity through the waste form. The inventory at times after closure can be determined by multiplying the inventory at closure by an appropriate decay factor.

Much low-level waste is extremely heterogeneous, both physically and chemically, and acquiring an exact knowledge of the inventory is not practical. In some cases, one may have only a rough idea of the contents of the waste. Consequently, for the purposes of performance assessment, it may be desirable to lump similar forms of a radionuclide together, and treat unknown forms conservatively. If greatly different physical or chemical forms of a particular radionuclide are present in the waste, these should be treated differently. Particular attention should be given to characteristics of waste that may enhance release rates or environmental mobilities (e.g. chelating agents).

4.2 Radionuclide Releases

Once the amount of each radionuclide and the water flux through the facility have been assessed, the next modeling step in a performance assessment is to determine the rate of radionuclide release from the disposal unit. Following Sullivan et al. [1988], three processes in the facility are considered: container or barrier breach, radionuclide leach, and near-field transport. Each of these processes is dependent on the moisture content or water flux through the facility, which are determined from the ground-water hydrology analysis.

4.2.1 Containment of the Waste

This section discusses the degradation and breach of man-made barriers to radionuclide releases including concrete structures, and steel or high-integrity containers. These features of the disposal system are considered to control access of water to the waste and egress of leachate from the waste disposal unit. The structures and to some extent the high-integrity containers also act as intruder barriers. Containers such as wooden boxes are considered to have little or no function and are generally not modeled, i.e., they are assumed to provide no waste isolation function.

Man-made barriers can perform an important function in the performance of a facility by containing the waste until the short-lived radionuclides have decayed. Many of the important isotopes in low-level waste have half lives of less than 30 years, hence the hazard of such wastes decreases rapidly [NRC, 1981; NRC, 1982; Rogers and Associates, 1988]. If a barrier serves to eliminate or significantly reduce releases for the first 100 to 500 years following closure, only long-lived isotopes may need to be accounted for in the performance assessment. On the other hand, once short-lived isotopes have decayed there is generally little change in the inventory for a long time, since the remaining isotopes remain radioactive for long times. Hence the principal effect of engineered structures on the dose from long-lived isotopes is to shift the time of arrival of the peak dose rather than to influence the magnitude of the peak dose.

The goal of modeling man-made barriers is to approximate the rate at which radionuclides become available to the surroundings. This involves identifying both the mode of failure (localized or general) of the barrier, and the rate of failure (breached area as a function of time).

4.2.1.1 Degradation of Concrete Structures

Reasonable confidence can be generated that some concretes may last 500 years [Clifton and Knab, 1989]. However, modeling methods for both the expected breach mode and the degradation rate of underground concrete as a function of time are not considered adequate at this time [MacKenzie et al., 1986; Kozak et al., 1989a]. State-of-the-art analyses, such as those found in BARRIER [Shuman et al., 1988], or discussed by Clifton and Knab [1989], can be used to model concrete degradation to provide estimates of the performance of concrete structures. However, available data for concrete degradation used in these models are derived from short-term experiments, which leads to large uncertainties in long-term predictions [Kozak et al., 1989a]. Consequently, there is no formal method for treating concrete degradation and breach in this performance assessment methodology. Instead, breaching of concrete is accounted for in a simplified manner as a delay time to the onset of releases. This approach assumes that the mode of failure is general (complete loss of structure), and that the rate of failure is a step function in time. The time at which failure occurs is assumed, not calculated. Nevertheless, this approach can be used to assess the sensitivity of the performance assessment to barrier breach time.

4.2.1.2 Container Corrosion

Kozak et al. [1989a] concluded that the container corrosion model of Sullivan et al. [1988] provides a satisfactory compromise between site-specific corrosion studies and detailed electrochemical modeling of corrosion, but that several constants used in the model need better experimental corroboration. This corrosion model provides the breached area of a single container as a function of time. The model is semi-empirical, and is based on the corrosion data of Romanoff [1957] and Gerhold et al. [1981].

If the conceptual model requires detailed modeling of corrosion in the performance assessment, the computer code BLT (Breach, Leach, and Transport) can be used [Sullivan and Suen, 1989]. This computer code contains the corrosion model of Sullivan et al. [1988], and has the following characteristics:

- Requires only pH, amount of aeration, clay fraction, soil moisture, and container characteristics to estimate breach rate.

- Breach model is integrated with leach and transport models in a comprehensive source-term code.

Uncertainties in this corrosion model arise from several sources. First, several of the corrosion phenomena are poorly understood, in particular the rate of pit growth and the areal pit density. Furthermore, the model is derived from a best fit to generic corrosion data, and the results may not always be conservative [Sullivan et al., 1988]. The data were collected for bare carbon steel, and there is uncertainty about applying the model to painted steel [Kozak et al., 1989a]. In addition, the failure rate of containers can be dependent on the method of emplacement [MacKenzie and Smalley, 1985], and this effect is not accounted for in the breach model. Perhaps the greatest uncertainty involves predicting corrosion phenomena from incompletely understood geochemical data. Soil pH and aeration can vary dramatically with both position and time, and geochemical modeling is inadequate to predict the changes in these parameters in a changing, degrading disposal unit. The model is also limited in that it cannot be used to simulate the degradation of high-integrity containers. However, there are no competing extant models for the degradation of high-integrity containers, and there are limited data on the subject [Soo, 1988].

Based on the uncertainties in data needed to implement this corrosion model, a simplified approach to modeling failure of engineered barriers may often be more appropriate. This simpler approach is to treat the container failure time as an delay time to the onset of radionuclide releases. This simple approach is not predictive about the failure time of the containers: the failure time is assumed, and can be varied in a sensitivity analysis. The simple modeling approach does, however, make an assumption about the failure mode. The containers are assumed to fail completely, and provide no inhibition to radionuclide releases after the failure time.

In summary, engineered barriers such as concrete can be modeled in detail using analyses similar to the ones embodied in BARRIER, or currently under development [Clifton and Knab, 1989]. However, given the uncertainties in the data required for the model, such a level of detail is not recommended at this time. Similarly, the corrosion model in BLT can be used for detailed analyses of container failure rates, but the geochemical data needed for the model are likely to be quite uncertain. An alternate approach, as recommended in this methodology, is to add a delay time to the onset of releases, which corresponds to an assumption that the barriers fail completely at some specific time. This approach is not predictive, but can be used in sensitivity analyses to determine the importance of the engineered barriers to the overall performance assessment methodology.

4.2.2 Radionuclide Leach Rates

Radionuclide-containing chemicals can dissolve when contacted by water. The process of dissolution, or preferential solute mass transfer from solid to liquid phase, is known as leaching [Treybal, 1980]. The most rapid leach rates are determined by assuming the contaminant resides at the solid surface, and is simply washed off by passing water. More detailed models may account for solubility limitations, sorption capabilities, or other mass-transfer limitations either inside or outside of the waste container.

The goal of leach modeling is to estimate the release rate of a radionuclide from a single waste container, or from a single point in the facility. That is, leach modeling provides a local release rate, not the release rate from the facility.

In models of leaching processes, a distinction was drawn between stabilized and unstabilized waste [Kozak et al., 1989a]. Unstabilized waste is extremely heterogeneous, and it is inappropriate to make any detailed assumptions about the spatial distribution of the waste, or the chemical form and properties of the waste. As a result, limitations to mass transfer from unstabilized waste should be neglected.

Kozak et al. [1990], based on the work of Sullivan and Suen [1989], recommended the use of a surface-washing model for unstabilized waste and trash. Data required for the unstabilized-waste model are minimal. The geochemistry in the disposal units will usually be unknown; hence mass-transfer limitations such as solubility limits and sorption capabilities cannot be quantified, and should be neglected. In this case the amount of radionuclide available to be dissolved in water is the total inventory of the radionuclide.

The leaching behavior of stabilized waste will usually be very different from unstabilized waste, for the two kinds of waste experience much different environments. Stabilized waste is contained in a specific stabilized shape, and the chemical environment is somewhat better defined than in unstabilized waste. As a result, the uncertainty in leach rates from stabilized waste may well be less than uncertainties in unstabilized waste, and a more detailed modeling approach may be appropriate.

Release rates from stabilized waste may frequently be limited by mass-transfer rates in the waste form. If the convective velocity of water in the waste form is significantly less than the diffusive velocity of the contaminant in the waste form, the release can be modeled as a diffusive process. It is conservative to assume the concentration boundary condition at the container surface to be zero, for this is equivalent to an assumption that mass-transfer limitations external to the container are negligible [Bird et al., 1960]. In order to justify the use of a diffusion-limited release rate, an estimate of the convective velocity in the waste form is necessary to demonstrate that a diffusion-limited process is an appropriate model. Such data may frequently be unavailable or inadequate, and in such cases a surface-wash model should be used for stabilized waste.

For diffusive releases it is necessary to have diffusion coefficients in the waste form for each radionuclide, and dimensions of the waste container. Kozak et al. [1990] recommended an approximate form of an expression for diffusion-limited leaching from cylindrical shapes that is numerically efficient. In this approach the diffusive release rate is assumed to be constant, and equal to the initial release rate, until the inventory is depleted, after which the release rate is zero. That is,

$$\begin{aligned} q &= \frac{4D_e(H+a)m}{a^2 H \theta}, && qt \le I \\ q &= 0, && qt > I \end{aligned} \qquad (4\text{-}1)$$

where q is the release rate from the waste form, D_e is the effective diffusion coefficient in the waste form, H is the height of the waste form, a is the radius of the waste form (assumed to be barrel shaped), m is the initial inventory in the waste form, and I is the total inventory in the container. As noted by Kozak et al. [1989a], the effective diffusion coefficient is an empirical parameter that includes the effect of equilibrium sorption on the diffusion process. Use of Eq. (4-1) assumes that all of the inventory is dissolved in the pore water; that is, there are no solubility limitations in the container. If such limitations can be justified with confidence, Eq. (4-1) takes the form

$$\begin{aligned} q &= 4\pi D_e(H+a)C_s, && qt \le I \\ q &= 0, && qt > I \end{aligned} \qquad (4\text{-}2)$$

where C_s is the solubility-limited concentration of radionuclide in pore water in the waste form. Equation (4-2) contains the assumption that the mass transfer rate from the solid into solution is fast, such that the pore water is maintained at the solubility limit, and that the release rate from the container is limited by the diffusion rate. Use of Eq. (4-2) will in general lead to slower release rates than either Eq. (4-1) or the surface-wash model, since the release rate is doubly limited (by solubility and diffusion) in Eq. (4-2).

Equations (4-1) and (4-2) provide estimates consistent with the time-dependent equation for diffusive release at times shortly after releases begin, and will overestimate release rates thereafter, until the inventory is exhausted.

Both solubility and sorption are extremely sensitive to geochemical parameters such as pH (acidity) and Eh (oxidation state). Unfortunately, no adequate comparisons between geochemical models and field data currently exist, and fundamental mechanisms in geochemistry are not well understood [National Research Council, 1990]. As a result, chemical limitations to mass transfer in the waste form or outside the waste form in the disposal unit must be justified using adequate data and modeling to provide adequate confidence in the values used. The use of solubility limit inside the waste form will probably be easier to justify than if used outside the waste form, because the chemistry of the pore water in the stabilized waste form is likely to be less heterogeneous than that in the disposal unit.

There will generally be substantial uncertainty in the chemical and physical form of low-level waste, the ground-water chemistry in the facility, and the interaction of the waste with surrounding water and soil. Consequently, there is substantial uncertainty in calculated leach rates, and as a result conservative approaches should be used. The reader may do well to recall at this point that the purpose of the methodology is to compare estimates of facility performance with regulatory criteria, not to predict the actual performance of the facility. Hence, conservative approaches are acceptable when dealing with large uncertainties.

More complicated and detailed leach models are incorporated into BLT [Sullivan and Suen, 1989]. These leach models include surface-wash, diffusion-limited, and congruent-dissolution approaches. These models are appropriate if sufficient confidence exists in the chemical behavior of the disposal units to justify the additional complexity of the models.

4.2.3 Near-Field Transport

As radionuclides leach from waste forms, they undergo dilution with surrounding waters, and are transported by those waters to the disposal unit boundary. This process of mixing and transport is defined here as near-field transport.

The primary goal of near-field transport modeling is to estimate the amount of mixing and dilution that the contaminant undergoes in the disposal unit. Mixing in the facility is governed by the dispersion coefficient in the facility. However, due to large (and unknown) spatial variabilities in the facility, it may not be possible to specify a dispersion coefficient with any confidence.

Kozak et al. [1989a] identified two methods for the analysis of near-field transport. The Brookhaven near-field transport model [Sullivan and Suen, 1989], involves solving the convective-dispersion equation in the disposal units. Kozak et al. concluded that the only disadvantage to this model is its complexity. This method incorporates the appropriate chemistry and physics of the near-field transport, but may be excessively complicated given the usual uncertainties about morphology and chemistry in the disposal unit. The second near-field transport modeling approach identified by Kozak et al. [1989a] was the use of a mixing-cell model, typified by the Robinson model [Robinson et al., 1988] and the NEFTRAN source-term model [Longsine et al., 1987]. It was noted that this model may be appropriate when details of the disposal facility morphology are unknown. However, Kozak et al. [1990] showed that use of this model is equivalent to assuming infinite dispersion in the disposal unit, which in some cases may be neither conservative nor justifiable. Consequently, Kozak et al. [1990] introduced a mixing-cell cascade model that retains the simplicity of the mixing-cell model, but that allows the analyst to vary dispersion over the full range of possible values. The mixing-cell cascade model therefore represents a compromise between complexity and flexibility in treating dispersion. A schematic representation of the mixing-cell cascade model is shown in Fig. 4-2.

The equations that are used in the mixing-cell cascade analysis for releases from unstabilized waste are

$$Q\ (t) = Q_0 e^{-\alpha N t} \sum_{n=1}^{N} \frac{(\alpha N t)^{n-1}}{(n-1)!}, \tag{4-3}$$

where N is the number of mixing cells assumed to be in the vertical direction of the disposal unit, Q_0 is the initial release rate from the disposal unit, Q is the time-dependent source-term release rate leaving the bottom of the unit, $\alpha = v/\theta DR$, where θ is the moisture content in the unit and $R = 1 + \rho(1-\varepsilon)K_d/\theta$ is the retardation factor in the unit. The model applies a spatially uniform initial concentration, which assumes that the radionuclide is approximately evenly distributed. This condition is an acknowledgement that the spatial distribution of a radionuclide in the facility will usually not be known with confidence, and hence the inventory must be averaged over the disposal unit. The initial release rate of contaminant from the disposal unit can be determined from

$$Q_0 = \frac{m\ v}{\theta\ D\ R}, \tag{4-4}$$

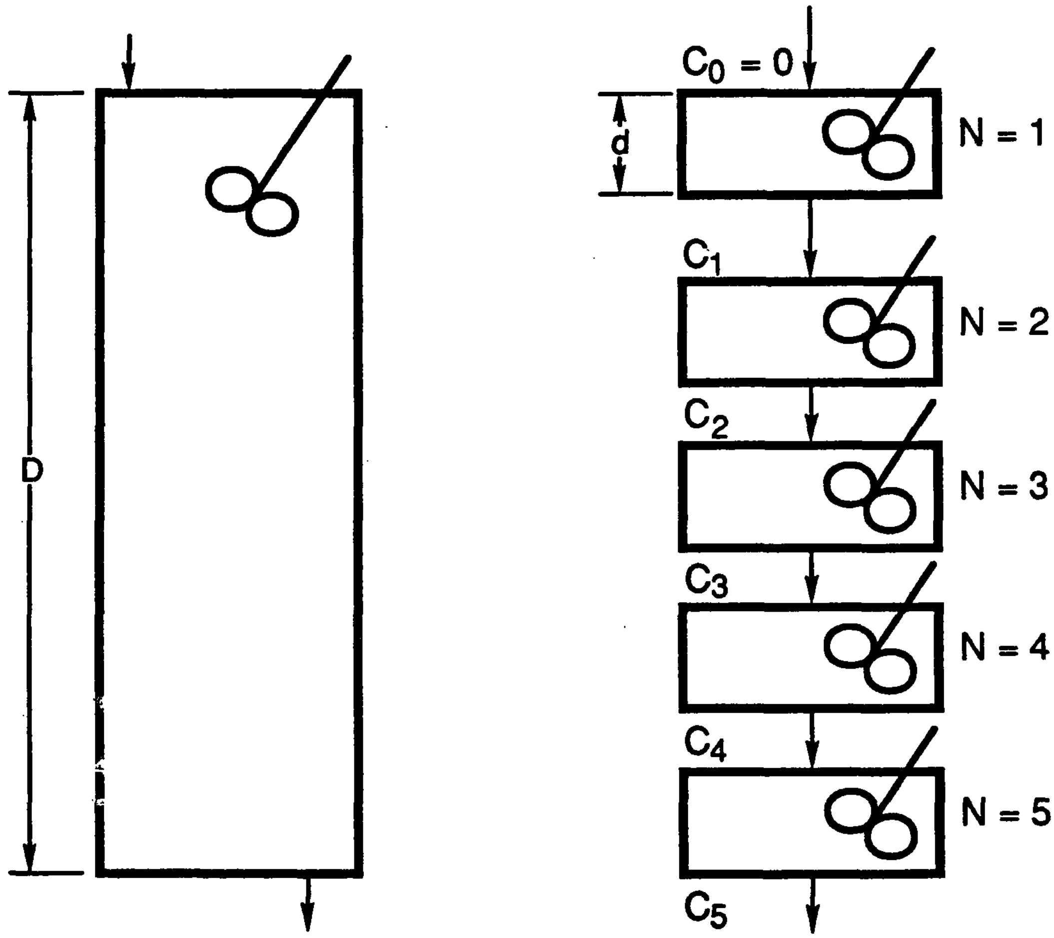

Figure 4-2: Mixing-Cell Cascade Model

where m is the total inventory of the radionuclide in the disposal unit, v is the vertical Darcy velocity through the unit, and D is the total depth of the disposal unit. As discussed in Section 4.2.2, use of Eqs. (4-3) and (4-4) assumes that mass-transfer rates between solid and liquid phases are rapid, that no solubility limits exist, and that all of the waste is immediately available to be dissolved.

Releases from a disposal unit in which diffusion-limited leaching occurs is modeled using a constant release-rate leach model. The equation for releases from a mixing-cell cascade with a constant leach rate is

$$Q(t) = qN \left[1 - e^{-\alpha Nt} \sum_{n=0}^{N} \left(1 - \frac{n}{N}\right) \frac{(\alpha Nt)^n}{n!}\right], \qquad qN \leq I$$

$$Q(t) = e^{-\alpha N(t-\tau)} \sum_{n=1}^{N} Q_{N-n}(\tau) \frac{[\alpha N(t-\tau)]^{n-1}}{(n-1)!}, \qquad qN > I \tag{4-5}$$

where q is the leach rate from each container in the disposal unit, given by Eq. (4-1), τ is the time at which the inventory is exhausted (the time when $qN = I$), and $Q_{N-n}(\tau)$ are the release rates at time τ from intermediate mixing cells. In the derivation of this equation it has been assumed that the initial concentration of radionuclide in interstitial water is zero; that is, no radionuclides exit containers prior to the start of diffusive leaching.

These analytical expressions for source-term releases incorporate dispersion in a simplified way, but retain the capability to model the full range of dispersive behavior. The mixing-cell source-term model is included in the ground-water transport codes DISPERSE and SURFACE, and has the following characteristics:

- Dispersion is accounted for in a simplified fashion that does not require specification of a dispersion coefficient. Instead, dispersion is modeled using a cascade of geometrically similar mixing cells into which radionuclides are released.

- Zero dispersion in the facility can be approximated by using a large number of mixing cells. When the number of mixing cells becomes large, the volume of each cell becomes small, and the overall dispersion tends to zero.

- Infinite dispersion can be modeled using a single mixing cell. Intermediate numbers of mixing cells can be used to model intermediate dispersive behavior.

- Both surface-wash and constant-rate leach models are incorporated into the mixing-cell cascade model. These leach models can be used to analyze unstabilized and stabilized waste, respectively.

4.3 Summary of Source-Term Modeling

Kozak et al. [1989a] reviewed the models for source-term releases for the various physical processes that are expected to occur in low-level waste disposal units. Kozak et al. [1989b] examined the use of BLT (Breach Leach and Transport) [Sullivan and Suen, 1989] and BARRIER [Shuman et al., 1988] as source-term computer codes for use in the methodology. It was concluded that BLT is a satisfactory code for detailed analyses of releases from low-level waste facilities, but that the complexity of the analysis may be unjustified for many performance assessment analyses. It was also concluded that BARRIER can be used to increase confidence in the longevity of concrete, but that its main limitation is in the lack of adequate data on concrete degradation processes, and that this limitation is important. However, as noted by Kozak et al. [1990], the value of the failure time for concrete may frequently be of minor importance to the overall goals of the performance assessment methodology. As a result, neither BARRIER nor any similar code is included in the methodology at this time.

After screening these various processes, models, and codes applicable to source-term modeling, the following source-term analysis was developed and recommended by Kozak et al. [1990]. As discussed in Chapter 3, flux of water into the disposal units is analyzed using VAM2D. Usually a multidimensional analysis of the vadose-zone flow will be appropriate. From this flow field, one can often extract an approximate one-dimensional flow through the disposal units. This flow field can be used in the mixing-cell cascade source-term model, which contains a surface-wash leach model for unstabilized waste, and an approximate approach for diffusion-limited leaching for stabilized waste. Sorption can be accounted for by specifying a retardation factor, which assumes the existence of linear (K_d) sorption isotherm. Sorption can have a dramatic effect on predicted release rates, and values for K_d must always be carefully justified using appropriate geochemical data and models. Mass-transfer limitations in stabilized waste forms are more likely to be justifiable than such limitations in unstabilized waste or outside of the stabilized waste form. It is likely that geochemical data and models will contain much uncertainty, hence sorption and solubility limits are expected to be similarly uncertain. Waste container and concrete structure failure times are incorporated as a simple delay time to the onset of releases.

In cases where this simple approach is determined to be inappropriate, the source-term analysis can be performed using BLT [Sullivan and Suen, 1989], or using the release and transport options in VAM2D [Huyakorn et al., 1989]. As in other aspects of the performance assessment, determining the appropriate level of detail is not a trivial task, and is largely the result of professional judgment.

5.0 GROUND-WATER TRANSPORT MODELING

Ground-water transport refers to the motion of water-borne radionuclides in soils, and includes both convective and dispersive transport mechanisms. Results from the ground-water flow and source-term models are used as inputs for ground-water transport modeling. As a result, the complexity of the transport analysis depends on the complexity of the flow analysis in addition to the complexity of the conceptual model. Steady-state ground-water flow may frequently be assumed, but releases from the source are intrinsically transient. As a result, it is necessary to model transient transport. The number of dimensions that should be modeled depends on the conceptual model of the site; one-dimensional transport modeling will often be appropriate. In addition to water flow and source-term release rates, the only other data needed for transport modeling are dispersivities and sorption data.

The goals of ground-water transport modeling are (1) to estimate aquifer concentrations that may become accessible to humans through a well, and (2) to estimate the rate of radionuclide flux from the aquifer into surface-water bodies. These are the two principal radionuclide pathways considered in the undisturbed performance of the facility [Shipers and Harlan, 1989]. Of these two pathways, the well pathway will usually be the more important. As a result, the solution method chosen for transport should be well suited for the analysis of ground-water concentrations.

Three of the most common analysis techniques used to analyze ground-water transport are numerical solutions of the convective-dispersion equation, analytical solutions of the convective-dispersion equation, and the stream-tube approach. These approaches were reviewed by Kozak et al. [1989a] for applicability to low-level waste performance assessment. Kozak et al. [1989a] concluded that for steady-state ground-water flow, either analytical solutions or the stream-tube approach were potentially valuable tools, but that for unsteady-state flow, or strongly spatially variable flow, numerical methods were necessary.

A consideration that was not adequately stressed by Kozak et al. [1989a] is that stream-tube models are better for analyzing radionuclide fluxes than ground-water concentrations. Calculating a concentration using a stream-tube model requires introducing an essentially arbitrary dilution volume [Simmons et al., 1986]. As a result, stream-tube models are appropriate tools for analyzing radionuclide fluxes to a surface-water body, but should be used with caution and skepticism in analyzing ground-water concentrations.

Kozak et al. [1989b] reviewed a number of computer codes that implement numerical solutions to the convective-dispersion equation, and in addition reviewed TRANSS [Simmons et al., 1986] and NEFTRAN [Longsine et al., 1987], which implement the stream-tube modeling approach. It was concluded that VAM2D [Huyakorn et al., 1989] and BLT [Sullivan and Suen, 1989] were appropriate candidates for numerical solution of the convective-dispersion equation. NEFTRAN was chosen by Kozak et al.

[1989b] as a stream-tube model over TRANSS, because of its ability to handle radioactive decay chains, but NEFTRAN was eliminated from consideration soon afterward due to the inadequacy of the stream-tube approach in determining ground-water concentrations, and due to the primacy of the well pathway [Kozak et al., 1990].

Once stream-tube models were eliminated from consideration, analytical and numerical solutions to the convective-dispersion equation remained. For steady, one-dimensional aquifer flow it is possible to use analytical solutions, of which a number are available. The solution developed by Codell et al. [1982] was chosen for use in this methodology for its ability to account for the important phenomena in transport in a simple and flexible way. This solution was recently used by Rood et al. [1989] in a comparison with data for radionuclide transport in the Snake River Plain Aquifer. The results, using estimated and calibrated transport parameters, were in reasonable agreement with field data. This analysis was a calibration exercise, not a validation of the model, but the work nevertheless shows that the Green's function solution can be used to analyze a real aquifer system.

The Green's function solution can handle arbitrary time dependence in the source term, and allows for three-dimensional dispersion and one-dimensional convection. The analytical Green's function must be integrated numerically for arbitrary source terms, and this numerical integration can be performed by simple and standardized methods. Green's functions are available for determining aquifer concentrations, and for radionuclide fluxes into a surface-water body, which can be used to determine surface-water concentrations [Codell et al., 1982; Kozak et al., 1990]. The numerical integration programs DISPERSE and SURFACE [Kozak et al., 1990] have greater numerical accuracy, and have source terms that are more flexible in their treatment of dispersion, than do GROUND and GRDFLX, the comparable computer codes of Codell et al. [1982].

Transport to a well in the aquifer is modeled using the analytical solution embodied in DISPERSE, which is

$$C = \frac{1}{nLWR} \int_0^t Q(\tau)\ X(x,\ t-\tau)Y(t-\tau)Z(t-\tau)\ d\tau, \tag{5-1}$$

where Q is the time-dependent source rate, n is the effective porosity, R is the retardation factor in the aquifer, L is the length of the disposal unit (the dimension parallel to the aquifer flow direction), W is the width of the disposal unit (perpendicular to the flow direction), t is time, and where

$$X = \frac{1}{2}[\mathrm{erf}(\frac{x + L/2 - u(t-\tau)/R}{\alpha_L}) - \mathrm{erf}(\frac{x - L/2 - u(t-\tau)/R}{\alpha_L})], \tag{5-2}$$

$$Y = \mathrm{erf}(\frac{W/2}{\alpha_T}), \qquad (5\text{-}3)$$

$$Z = \frac{1}{b}\,[1 + 2\sum_{m=1}^{\infty} \exp(\frac{-m^2\pi^2 D_T(t-\tau)}{b^2 R})], \qquad (5\text{-}4)$$

where b is the aquifer thickness, assumed constant. Here, x is the down-gradient distance, u is the pore velocity (the Darcy velocity divided by the effective porosity), and α_L and α_T are defined by

$$\alpha_L = [4D_L(t-\tau)/R]^{1/2}, \qquad (5\text{-}5)$$

and

$$\alpha_T = [4D_T(t-\tau)/R]^{1/2}. \qquad (5\text{-}6)$$

Here D_T is the transverse dispersion coefficient, and D_L is the longitudinal dispersion coefficient. Use of these equations assumes that the concentration available for consumption at the well is the maximum concentration in the plume: the concentration at the plume centerline and at the water table. This concentration can be expected to be a conservative estimate of the well concentration, since the actual concentration sampled by a well will be averaged over some depth, and can be expected to be lower than the maximum concentration.

DISPERSE performs a numerical integration to solve Eq. (5-1) using either Eq. (4-3) or Eq. (4-5) for Q. To generate the concentration history at the well, the integration is performed repeatedly for several times.

The flux of radionuclide into a surface-water body can be analyzed using a similar program called SURFACE. SURFACE calculates the flux of radionuclide into the surface water using a second Green's function, which is

$$F_i(t) = \frac{1}{2L(\pi D_L t/R)^{1/2}}\,[\frac{u}{R}\,(\pi D_L t/R)^{1/2}(\mathrm{erf}(z_1) - \mathrm{erf}(z_2)) - \frac{D_L}{R}\,(e^{-z_1^2} - e^{-z_2^2})], \qquad (5\text{-}7)$$

where

$$z_1 = \frac{x - ut/R + L/2}{(4D_L t/R)^{1/2}}, \tag{5-8}$$

and

$$z_2 = \frac{x - ut/R - L/2}{(4D_L t/R)^{1/2}}, \tag{5-9}$$

and where the parameters have been defined above. As in DISPERSE, this analytical Green's function is multiplied by the source rate Q and numerically integrated over time. DISPERSE and SURFACE share several subroutines and parameters, but have been set up to run as independent programs.

In this performance assessment methodology, DISPERSE and SURFACE are recommended for most analyses. The computer codes include the mixing-cell cascade source-term model that was discussed in Chapter 4. The following characteristics are common to both DISPERSE and SURFACE:

- The codes have been compiled on an IBM personal computer using a Microsoft FORTRAN compiler.
- Dispersion in the unsaturated zone below the facility is neglected. Transport in the vadose zone is modeled as a simple delay time between release from the disposal unit and entry into the aquifer.
- Releases from the source occur into the aquifer through a square area of arbitrary length and width on the water table.
- Leaching release can be specified by one of two mechanisms. A simple surface-release model can be specified, in which all of the inventory is available for release. Alternately, a constant release-rate can be specified, which can be used to approximate diffusion-limited release rates. Solubility limitations are not accounted for exterior to the waste forms, but the release may be limited by the solubility of the radionuclide in the waste form pore fluid. Retardation in the disposal unit can be included. Inclusion of sorption effects or solubility limits must be justified using site-specific data.
- Dispersion in the disposal units is accounted for by a cascade of equal-sized mixing cells. This model can span the full range of possible dispersivities.

- Radionuclide decay and retardation in the source, during the vadose-zone travel time, and in the aquifer are accounted for. Daughter-product production is not included in the analysis. Daughter-product effects can be accounted for by analyzing the parent radionuclide transport, then correcting the final concentration or flux profile to account for the daughter-product production. This approach can be used for the simple linear chains expected to be of importance in low-level waste [Kozak et al., 1990].

- Both codes are incorporated into a user-friendly shell, called SUNS, which provides for simplified data input, parameter variation, and a variety of tabular and graphical output.

DISPERSE is used to calculate the concentration available for consumption at a well; the code has the following capabilities:

- One-dimensional convection and three-dimensional dispersion. The convective velocity is constant, and dispersion coefficients are spatially uniform. Longitudinal and transverse dispersion coefficients can be different from each other.

- The aquifer may be finitely or infinitely thick, but the thickness may not vary between source and receptor.

SURFACE is used to calculate the radionuclide flux into a surface-water body; the code has the following characteristics:

- All radionuclides in the aquifer passing the a plane intersecting the surface water body are assumed to enter the surface water. This is the most conservative assumption possible about the flux into the surface water.

- The code includes a simple dilution-factor model for calculating surface-water concentrations. This model is appropriate for calculating concentrations in small rivers.

In circumstances in which the flow field or conceptual model are too complicated to justify the use of these simple modeling approaches, transport analyses may be performed using either VAM2D [Huyakorn et al., 1989] or BLT [Sullivan and Suen, 1989], which contains the transport code FEMWASTE [Yeh and Ward, 1982]. These codes allow the analyst to account for multidimensional, spatially variable, or transient processes.

6.0 SURFACE-WATER TRANSPORT

Radionuclides can enter surface waters when these are connected to a contaminated aquifer. This is the only pathway to surface water for the undisturbed performance assessment of a generic site [Shipers and Harlan, 1989]. The rate at which radionuclides enter the surface water is determined by analysis of radionuclide transport in the aquifer. An appropriate analysis is embodied in SURFACE [Kozak et al., 1990], which is based on the analytical solution of Codell et al. [1982]. A vertical plane is specified through the aquifer intersecting the surface water body, and it is assumed that all radionuclides passing through that plane enter the surface water. This is a very conservative assumption, for it includes even very deep radionuclides that, in reality, may not enter the surface water. However, in many cases the use of this conservative assumption will not be prohibitive, since surface-water pathways will often be of minor importance compared to ground-water well pathways. An alternative, less conservative, approach is to integrate the ground-water plume over the aquifer streamlines intersecting the surface-water body. In this approach, only those radionuclides that intersect the surface water are assumed to enter it. This approach is somewhat more complicated to implement than the very conservative approach discussed above, and must be implemented on a site-specific basis.

The goal of surface-water modeling is to estimate the amount of dilution that occurs when radionuclides enter the surface water. The primary factors that influence this dilution are dispersion or mixing in the surface water, and the manner in which the radionuclides enter the surface water. It is conservative to assume that the radionuclides enter the surface water at a single point, and not through a diffuse area. For small rivers and streams, perfect lateral mixing can be assumed across the river, which leads to a simple dilution-factor model of mixing [Kozak et al., 1990]. Similarly, small lakes can be considered well mixed, and radionuclide concentrations determined by a simple mass balance [Kozak et al., 1989a]. Mixing phenomena in larger bodies of water generally exhibit more complicated mixing behavior [Jirka et al., 1983]. Good mixing cannot generally be assumed in these larger surface-water bodies; there exist formal NRC guidelines for these more complicated models [NRC, 1977b]. Flow in estuaries is complicated, and estuarine data are difficult to acquire. Nevertheless, models are available for contaminant transport in estuaries [NRC, 1977b].

Data needed for the surface-water model result from characterization of the surface hydrology, including an estimate of dispersion in the surface waters. Steady-state surface hydrology can usually be assumed, provided conservative values of water levels and flows are used. This is the basis for many accepted NRC surface-water models [NRC, 1977b; NRC, 1978, Onishi et al., 1981]. Sediment sorption should only be included if adequate site-specific data are available.

Uncertainties in surface-water modeling arise due to uncertainties in field data. For instance, dispersion coefficients and sediment sorption characteristics may be largely unknown or uncertain for a particular site. These uncertainties are not expected to be of primary importance in the performance assessment methodology, since surface-water radionuclide concentrations can usually be expected to be much less than ground-water concentrations. This means that the surface-water pathways will usually be of secondary importance to the ground-water well pathway.

Surface-water transport models used in this methodology are contained primarily in GENII [Napier et al., 1988]. SURFACE contains a simple dilution-factor model that is appropriate for modeling small rivers, but at this time the code cannot be used to analyze more complicated surface-water bodies. Surface-water transport models in GENII are the models discussed by Codell et al. [1982], which are consistent with recommended NRC models [NRC, 1977b]. If GENII is used in the methodology for surface-water analyses, SURFACE is only used to determine the rate of influx of contaminants into the surface water. In GENII, surface-water concentrations can be estimated for a river or large lake under the following assumptions:

- constant flow depth,
- constant convective velocity,
- straight river channel,
- constant lateral dispersion coefficient,
- continuous point discharge of contaminants, and
- constant river width.

Use of the GENII code for surface-water dilution may lead to a more realistic estimate of surface-water concentrations than does the dilution-factor model included in SURFACE.

7.0 AIR-TRANSPORT MODELS

Airborne radionuclide release is not among the significant pathways for releases from an undisturbed low-level waste facility identified by Shipers and Harlan [1989]. However, there may be circumstances in which air transport may need to be considered in the conceptual model. For instance, contamination of a playa (dry lake bed) or intermittent stream may lead to exposure of contaminated soils and subsequent entrainment of contaminated particles in air. The importance to the performance assessment of such contamination is contingent upon the existence of a viable transport pathway to the dry surface-water body.

Similarly there are several airborne pathways that may be significant in the intruder scenarios [Shipers and Harlan, 1989]. Several of these pathways are important for offsite receptors, hence there is a need for models of airborne transport. However, it should be noted that the maximally exposed individual in an intruder scenario must be assumed to be located onsite, for ground-level releases produce concentrations that monotonically decrease downwind from the release site.

Matuszek and Robinson [1983] have discussed the radiological importance of gases produced at the West Valley, New York low-level waste site. At West Valley, gas production appears to be the predominant exposure pathway. However, the West Valley site suffers from considerable ponding [Sullivan and Kempf, 1987], and it is not clear how results from this site can be extrapolated to a well-situated facility [Kozak et al., 1989a]. Air transport models are unlikely to be used often in postclosure analyses. The models in this chapter are included for completeness, to ensure that doses from airborne pathways can be calculated when such calculations are appropriate.

To analyze the airborne transport of particulates, the analyst must first estimate the amount of contaminant that is entrained in the air at the source. Kozak et al. [1989a] recommended the mass-loading model based on its simplicity and conservatism. The mass-loading model is based on the assumption that airborne concentration can be expressed as the product of the amount of soil particles suspended in the air and the radionuclide concentration on the soil:

$$C_a = C_m C_p \quad , \qquad (7\text{-}1)$$

where C_a is the concentration of the radionuclide in the air, C_m is the concentration of the radionuclide on the soil, and C_p is the concentration of particulate matter in the air [NCRP 76, 1984]. Measured values of C_p for the United States range between 9 and 79 $\mu g/m^3$. A conservative value of 100 $\mu g/m^3$ is often used in calculations.

The mass-loading model provides an estimate of the concentration in air using very little data. The disadvantages to this method are that (1) the method is considered applicable to aged deposits of material that have been mixed uniformly with the soil, and (2) it is implicitly assumed that soil and contaminants are suspended equally easily, and this may be invalid in some instances. Nevertheless, the mass-loading model is recommended for conservative estimation of entrained contaminant concentrations. The mass-loading model can be used by itself to estimate local airborne concentrations (for instance, in the analysis of an onsite intruder), or as a source term for a model for airborne transport.

Gaussian-plume models are commonly used to assess the transport of airborne radionuclides. These models are derived from the convective dispersion equation, assuming one-dimensional convective transport and three-dimensional dispersive transport. The Gaussian-plume model has been adopted as a standard method in the regulation of both radioactive [NRC, 1983; IAEA, 1980] and non-radioactive [EPA, 1978] airborne contaminants. They have been extensively validated, are simple, and have a good theoretical basis [NCRP 76, 1984]. In addition, they are computationally efficient and require small amounts of easily obtained data [Hanna et al., 1982]. By contrast, other models often require complex computer codes for solution. These factors make Gaussian-plume models suitable for performance assessment analyses.

Several assumptions are commonly made to simplify the plume models for use in radiological assessment. Dispersion in the downwind direction is generally neglected, for convection dominates the transport. This assumption is reasonable only for source terms that are continuous in time. In addition, the first-order radioactive-decay terms are commonly neglected. However, based on the time and distance the plume has traveled, the strength of the source term can be adjusted to account for radioactive decay. In the absence of complicating terrain or atmospheric conditions, lateral and vertical eddy diffusion coefficients are functions only of downwind distance and atmospheric stability conditions.

Gaussian-plume models are derived for a continuous point source, but releases from a disturbed low-level waste facility will emanate from some finite area. The point-source assumption probably introduces errors in the predicted concentrations for small times and locations near the release point. For locations far from the release point these errors are small. Area sources can be accounted for by the virtual source method [Turner, 1970]. A virtual source is an imaginary point source upwind from the source of the airborne contaminant. The virtual source is chosen such that some characteristic dimension of the plume, say two times the lateral eddy diffusion coefficient, is equal to the lateral dimension of the source. The eddy diffusion coefficient is mathematically identical to the standard deviation in a normal distribution, so this definition equates the source area with the 95% bound of the dispersion plume. Since the eddy diffusion coefficient is a function only of distance and stability class the distance to the virtual source can be uniquely defined for each stability class. This distance is then added to the distance from source to receptor, and the analysis is performed using a point source located at the virtual point.

The analysis of air transport in the assessment of the undisturbed facility will generally be for long-term transport. Over such long times, wind speed and direction and atmospheric stability are all randomly variable, and the airborne transport can be expected to be randomly distributed in all directions. A simple approach to modeling this situation is to model the transport as occurring in constant wind speed directed from the source to the receptor at some average stability classification. This method will probably be quite conservative for long-term analyses, since the variability in wind direction is neglected. This level of conservatism is probably not crucial to the overall performance assessment analysis, since the air transport pathway is not often expected to be among the most important. However, if this method is determined to be excessively pessimistic, other models are available that can be expected to provide more detailed, less conservative results by accounting for the variable nature of the wind direction and atmospheric stability.

The air transport model in GENII [Napier et al., 1988] accommodates either the simplest form of the Gaussian-plume model, or more detailed atmospheric condition information. As a result, GENII implements the level of detail in air transport modeling that is likely to be needed in low-level waste performance assessment analyses.

8.0 PATHWAYS AND DOSIMETRY MODELING

This chapter summarizes modeling approaches for the analysis of transport through the food chain, and for dosimetry analyses. More detail on these methods has been provided in Kozak et al. [1989a].

Once radionuclide concentrations have been determined in each environmental medium of concern, the dose to an individual can be calculated using pathways and dosimetry models. Pathways models convert environmental concentrations to radionuclide intake rates for a person. Pathways can include ingestion of water or foodstuffs, or inhalation of airborne contaminants. Dose models estimate the effect of this radionuclide intake on human tissues. NRC guidance exists for appropriate models and assumptions for pathways models [NRC, 1977a], and these models and assumptions should be used in the performance assessment. Internationally accepted dosimetry models have been developed from a model of the human body, as described by the International Commission on Radiological Protection (ICRP) [ICRP 26, 1987]; these models are used in the methodology. The result of the ICRP models is a data base of dose-conversion factors that convert radionuclide intake to doses, and these have been published in ICRP 30 [1982-1988]. The recommended NRC pathways models and the recommended ICRP dosimetry models are implemented in GENII [Napier et al., 1988].

The goal of pathways and dosimetry modeling is to calculate the effective whole-body dose and doses to individual organs for the maximally exposed person. These are the doses that must be compared to the regulatory performance objectives of 10 CFR Part 61: 25 mrem effective whole-body, 75 mrem thyroid, and 25 mrem to any single organ. Doses should be calculated for each organ of importance in the ICRP methodology; due to the averaging procedure used in calculating effective whole-body equivalent it is possible for the whole-body dose to be below 25 mrem, but to have an organ dose greater than 25 mrem. Doses should be calculated for a one-year exposure period, with a 50-year commitment period.

Many of the data required for these models are standard. For instance, guidance on appropriate food consumption rates, inhalation rates, bioaccumulation factors, and other pathway model parameters has been given by the NRC [1977a]. Use of values other than these standards must be justified on a site-specific basis. The primary uncertainties in pathways and dosimetry modeling are generally caused by uncertainty in the state of the site in the future: for instance, what agricultural activities occur following institutional control.

Appropriate models for pathways, food-chain, and dosimetry are implemented in GENII [Napier et al., 1988]. This code contains models for the following pathway analyses:

- Atmospheric transport by Gaussian-plume models. Air transport analyses will generally not be necessary in intruder scenarios, for the maximally exposed person can be expected to be onsite.

- Surface-water transport models. These models require a steady-state radionuclide influx that is provided in the methodology by SURFACE.

- Soil-contamination models, including biotic transport and manual redistribution models. These can be expected to be useful in intruder-scenario analysis.

- Food chain models, including bioaccumulation in plants, and options to account for irrigation of various crops.

- Terrestrial-exposure pathways, which include inhalation, ingestion of drinking water and contaminated foods, and external exposure.

GENII also contains dosimetry models that are in accordance with ICRP 26 standards. Both external and internal dose conversion factors can be generated.

The primary limitation to GENII is that the models are for steady-state radionuclide concentrations. Consequently, to analyze transient effects it is necessary to use a quasi-steady-state approximation to the environmental concentrations. That is, one should assume the concentrations are constant during the exposure period, even though some variation occurs. In practice, this should be a good approximation, for the transient effects in the performance assessment will generally occur over longer time frames than a single year.

9.0 INTRUDER SCENARIOS

In previous reports on the development of this methodology, it was stated that analyses of intruder doses are required [Shipers, 1989; Shipers and Harlan, 1989; Kozak et al., 1989a; Kozak et al., 1989b]. These statements are incorrect and in most cases intruder-dose analyses need not be performed. A demonstration of intruder protection may consist of a demonstration that the waste classification and segregation requirements of 10 CFR part 61 have been met, and that adequate barriers to inadvertent intrusion have been provided for. However, dose analyses may be required in special cases when an applicant requests an exemption from the 10 CFR Part 61 waste classification scheme.

The purpose of this chapter is to discuss modeling considerations for the analysis of dose to an inadvertent intruder. This assessment differs from an analysis of releases from an undisturbed facility in several ways. First, human exposures from an undisturbed site primarily result from radionuclides entering the ground-water system; by contrast, an intruder can be directly exposed to waste, which can result in exposures by direct radiation, ingestion of contaminated water and food, and inhalation of airborne soil.

Analyses of intruder scenarios can be performed using GENII [Napier et al., 1988]. This computer code includes transport models for air transport and surface-water transport, includes pathways models for exposures to contaminated food and water, and includes ICRP 30 dosimetry models. GENII is probably appropriate for analyzing most intruder scenarios that may be proposed for a low-level waste site.

Three example intruder scenarios are discussed in the following sections. These scenarios are selected because they were being considered in the consequence analysis of a generic site for the development of 10 CFR 61 [NRC, 1981]. These intrusion scenarios are conservatively assumed to occur based upon consideration of typical human activities. Since these scenarios may not apply to a specific site, they are only representative of the kinds of analyses that may be required of a licensee who applies for an exemption from the intruder-protection criteria listed above. The burden of justification of any assumption is on the licensee [Starmer, 1988].

9.1 Intruder-Construction Scenario

In this scenario it is assumed that a building is constructed directly on the facility, and the waste is assumed to be excavated during construction [NRC, 1981]. A subset of this scenario, the intruder-discovery scenario, is bounded by this analysis, hence is not discussed here.

Exposures from this scenario are assumed to result primarily from inhalation of suspended contaminated soil, and from direct radiation from standing on the exposed waste and immersion in a contaminated dust cloud [NRC, 1981]. The duration of exposure can be assumed to be of typical duration for a construction project, and as such is a relatively short, acute exposure.

The analysis of this scenario is relatively straightforward, for the intruder is onsite, the waste is onsite, and transport offsite need not be considered. The concentration of airborne contaminants can be estimated using a conservative estimate of the mass-loading factor [Kozak et al., 1989a]. Surface concentrations used in direct exposure modeling can be determined by a reasonable assumption about how the soil is mixed during excavation.

9.2 Intruder-Agricultural Scenario

In this scenario a farmer is assumed to live in the building constructed in the intruder-construction scenario [NRC, 1981]. Exposures in this scenario may result from consumption of food grown in contaminated soil, consumption of well water from a well at or near the facility, and direct exposure to contaminated soil. The duration of exposure is greater than the exposure time for the construction intruder.

The analysis of this scenario is slightly more complicated than the analysis of the intruder-construction scenario, since it is necessary to account for radionuclides in food. The intruder is onsite, the waste is onsite, but some transport pathways may need to be analyzed to determine the concentrations available for uptake in nearby fields. Of these transport pathways, surface-water runoff may frequently be the most important. However, it will generally be conservative to assume that crops consumed by the farmer are grown within the site boundary, in which case transport models are unnecessary.

9.3 Intruder-Well Scenario

In this scenario an intruder is assumed to drill a well through the waste, and to consume water from the well. This scenario can be part of the intruder-agricultural scenario, but is not necessarily. Exposures in this scenario result from ingestion of contaminated well water, and direct exposure to the waste excavated by the drill.

The amount of waste excavated is minimal, and doses from excavated waste will generally not be important. The well borehole will not usually provide a preferential flow path for infiltrating water through the unsaturated zone; the exception to this is when water is ponded over the top of the borehole. This will rarely be the case, hence radionuclide releases to the aquifer will generally be the same as for the undisturbed scenario, and well concentrations can be determined as discussed in Chapter 5 of this report.

10.0 METHODOLOGY DEMONSTRATION

In this chapter a demonstration of the performance assessment methodology is given for analysis of releases to ground water. This chapter is intended to provide a brief overview of how the methodology can be applied, and to demonstrate the sequence of performance assessment analyses using the computer codes in the methodology. Only a single radionuclide is considered in this demonstration; in an actual performance assessment this analysis must be carried out for each radionuclide of concern, and the contributions of each radionuclide added to the total dose. Furthermore, in this demonstration only results for the total dose are given. In an actual performance assessment doses to critical organs must also be calculated. More details about the theoretical development of the models used here was given by Kozak et al. [1990]. A more detailed presentation of the methodology, including computer code input and output files and detailed operating procedures, is to be published under separate cover as A Self-Teaching Curriculum for the methodology.[3]

Several important points should be made about the analysis in this chapter. First, the iterative process of developing a conceptual model from site characterization data is not presented. For this demonstration the conceptual model is assumed, and the analysis is presented. Development of a conceptual model is a site-specific issue that does not follow any firm rules, and no attempt is made in this chapter to provide guidance. Second, analysis of single set of model parameters is presented in this chapter. In an actual performance assessment, uncertainty in the data must be accounted for. In addition, parametric analyses should be performed to identify the parameters that influence the performance of the facility the most. Third, the conceptual model analyzed here is a very simple one. More complicated conceptual models can undoubtedly be analyzed using the methods described here, and conceptual models for actual sites may well be more complicated than the one used in this chapter. Fourth, the calculations presented here are for a base-case analysis: the cover is assumed to function as designed for the duration of the analysis, and variations of site conditions with time are not considered.

10.1 Conceptual Model

The conceptual model to be analyzed in this chapter is shown in Fig. 10-1. The facility is assumed to be a shallow-land burial site containing 9.0 Ci of C-14 in a single trench. Trench dimensions are 12 meters by 200 meters in the horizontal plane, and the waste is buried from just below the cover to 10 meters below the ground surface, hence the vertical thickness of the waste burial zone is 8 meters. In other words, the 9 Ci inventory is contained in a 12 x 200 x 8 m^3 volume. A

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3. Chu, M. S. Y, M. W. Kozak, J. E. Campbell, B. K. Thompson, and P. A. Mattingly A Self-Teaching Curriculum for the NRC/SNL Low-Level Waste Performance Assessment Methodology, NUREG/CR-5539, SAND90-0585, Sandia National Laboratories, in press.

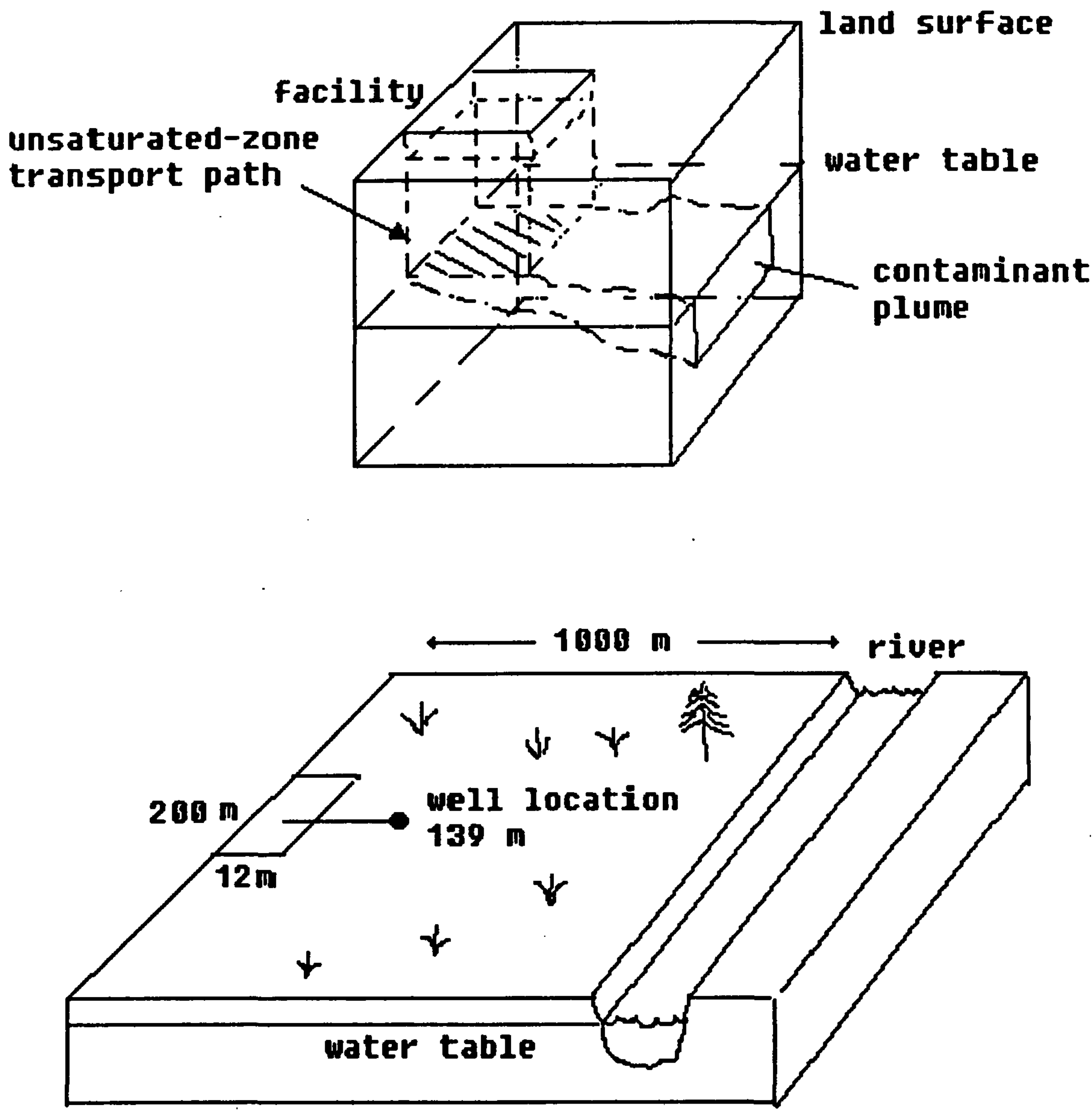

Figure 10-1: Demonstration Conceptual Model

three-layered cover over the facility acts to reduce water influx into the trench. The cover consists of a clay layer overlayed by a sand layer, which in turn is covered by the natural soil in which the facility is located. The cover extends beyond the trench dimensions in an attempt to reduce flow into the facility from around the cover; the cover extends laterally 29 meters past the waste on each side.

The characteristic curves describing the relationship between pressure head and water saturation are given by the expression derived by van Genuchten [1980], which is

$$S = S_{wr} + \frac{S_w - S_{wr}}{[1 + (\alpha|\Psi|)^{\beta}]^m}, \tag{10-1}$$

where S is the moisture content, S_{wr} is the residual moisture content, S_w is the saturated moisture content, Ψ is the capillary head, and α, β, and $m = 1-1/\beta$ are empirical parameters. The parameters that characterize the soils in the conceptual model are listed in Table 10-1; in this table K_s is saturated hydraulic conductivity. The values for these properties are from typical soil properties listed in the literature [Sullivan and Suen, 1989; Carsel and Parrish, 1988].

Unsaturated-zone hydraulic conductivity is calculated by the van Genuchten relationship between conductivity and saturation, which is

$$K_u = K_s S_e^{1/2} [1-(1-S_e^{1/m})^m]^2, \tag{10-2}$$

where $S_e = (S-S_{wr})/(1-S_{wr})$ [van Genuchten, 1980]. The characteristic curve and conductivity expressions are the standard input forms for VAM2D [Huyakorn et al., 1989].

Table 10-1
Soil Properties Used In The Conceptual Model

Soil type	S_w	S_{wr}	K_s (cm/day)	α (cm^{-1})	β
Undisturbed soil- (Material #1)	0.52	0.218	31.6	0.0115	2.03
Cover Layer-Clay (Material #2)	0.446	0.00	0.0082	0.00152	1.17
Cover Layer- Silt Loam (Material #3)	0.469	0.190	303.0	0.0050	7.09
Cover Layer- (Material #4)	0.52	0.218	31.6	0.0115	2.03

Infiltration at the site is 25 cm/yr; it is assumed that this value was estimated using an appropriate approach from those discussed in Chapter 3. In an actual performance assessment some sensitivity analysis should be performed to estimate the effect of uncertainty in infiltration on the overall performance of the facility. In this case, it has been shown that infiltration has negligible effect on the results of the performance assessment in the range 2.5 - 25 cm/yr [Kozak et al., 1990], when the cover retains its effectiveness for the duration of the analysis.

The aquifer below the facility is unconfined, and the water table is 24 meters below the ground surface. It is 49 meters from ground surface to bedrock, so the aquifer is 25 meters thick. The natural soil in which the facility resides is uniform above and below the water table, and has an effective porosity of 0.52. A river is located 1000 meters downgradient from the facility, and a water well is assumed to be drilled 139 meters downgradient from the center of the trench. The aquifer Darcy velocity is 2.3 m/yr; the pore velocity was calculated from the Darcy velocity and the effective porosity as 4.44 m/yr. Aquifer dispersivities are 2.0 m for longitudinal dispersivity, and 0.4 for transverse dispersivity. The longitudinal dispersion coefficient in the aquifer, D_L, and the transverse dispersion coefficient, D_T, are calculated from

$$D_L = a_L v, \tag{10-3}$$

and

$$D_T = a_T v, \tag{10-4}$$

where a_L and a_T are the longitudinal and transverse dispersivities, respectively, and v is the Darcy velocity. Equations (10-3) and (10-4) are generally recommended for homogeneous aquifers [Codell et al., 1982].

10.2 Methodology Calculations

In this section the conceptual model is analyzed using the performance assessment methodology. Unsaturated-zone flow is analyzed using VAM2D, which provides the water flux rate into the waste, and also provides information that allows an estimate of the travel time to the aquifer. The source-term analysis is performed using the surface-wash leach option and the mixing-cell cascade model contained in DISPERSE and SURFACE. Dispersion in the unsaturated zone is neglected, both in the trench and below the trench. Ground-water transport is modeled using DISPERSE to determine the well concentration, and using SURFACE to determine the flux into the river. GENII is used to analyze surface-water transport, food-chain pathways, and dosimetry. The emphasis in this section is on the information necessary to run the codes, and how the codes are integrated together, rather than on detailed input guides.

10.2.1 Unsaturated-Zone Flow

Water flux into the trench is determined using VAM2D with a constant 25 cm/yr flux at the upper boundary of the domain. Soil properties used in the simulation are those discussed in the previous section. Capillary-head profiles for this simulation are shown in Fig. 10-2, and moisture content profiles are shown in Fig. 10-3. Most of the region below the cover is at unit gradient, and the moisture content is approximately a constant 0.28 in the waste-containing trenches. Most of the region far from the cover is at unit gradient, but the Darcy velocity and hydraulic conductivity are larger than under the cover. Darcy velocities are calculated from total-head contours, which in this case are calculated using the water table as a datum plane. Total-head contours for this simulation are shown in Fig. 10-4. Vertical Darcy velocities in the waste-containing region are approximately spatially constant, and equal to 2.8 cm/yr, hence the cover system results in about a ten-fold decrease in flux into the disposal unit from natural conditions. The velocity is approximately uniform at 2.8 cm/yr from below the facility to just above the water table, at which point it increases rapidly.

Two estimates of the ground-water travel time can be made to bound the expected value. A lower bound is given by the travel time far from the facility, where the Darcy velocity is at its maximum. This estimate corresponds to an assumption that the cover has failed. Assuming a constant 0.32 moisture content, and Darcy velocity 25 cm/yr, the minimum travel time from the bottom of the trench to the aquifer (1400 cm) is $T = (0.32)(1400)/(25) = 18$ years. An upper bound, which is probably closer to the actual travel time, is given by the conditions in the trench: moisture content 0.28 and Darcy velocity 2.8 cm/yr. These values lead to a travel time estimate of 140 yrs. This analysis is not greatly sensitive to these travel times, since the half life of C-14 is much greater than either. Therefore either travel time can be used. Since the expected value is close to the upper bound, 140 years is used as the unsaturated zone travel time. If short-lived isotopes are of concern, a more accurate estimate of the unsaturated-zone travel time should be used.

10.2.2 Source-Term Evaluation

Leaching release rates are determined using the mixing-cell cascade model with the surface-wash leach option, which is described in Chapter 4. From the Darcy velocity and moisture content in the trench computed using VAM2D, and assuming no sorption (i.e. $R=1$), the value of $\alpha = 0.011$. Dispersion in the facility is assumed to be zero, which is reflected by a large value for N, say, $N > 30$. This assumption tends to lead to higher environmental concentrations and less dispersion at the receptor point. In addition, the pre-exponential factor Q_0 must be specified from Eq. (4-4). From the parameter values specified or calculated in this chapter, $Q_0 = 0.1$ Ci/yr.

10.2.3 Ground-Water Transport Analysis

Dispersion in the unsaturated zone is neglected, and the unsaturated zone only serves to delay the release of radionuclides into the aquifer. As

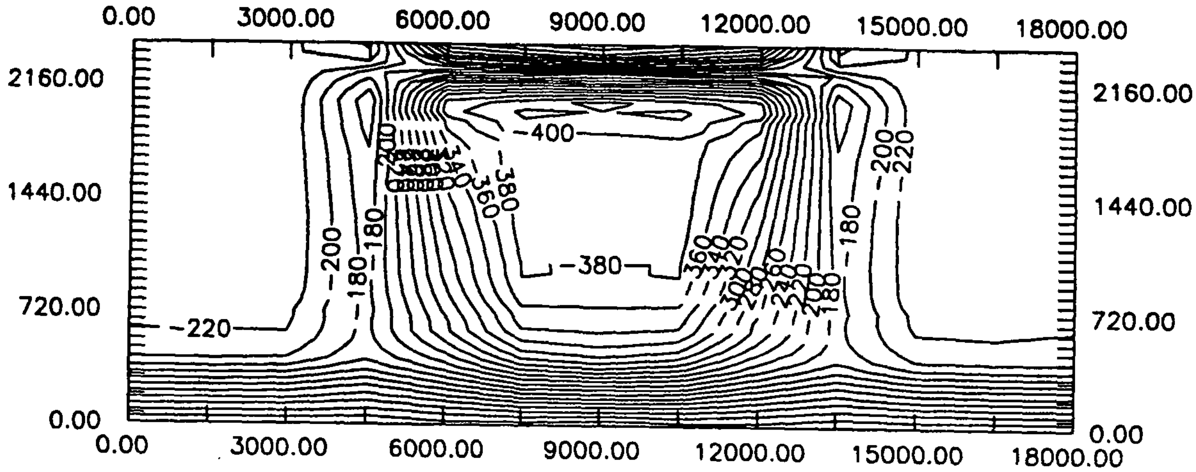

Figure 10-2: VAM2D Pressure-Head Contours

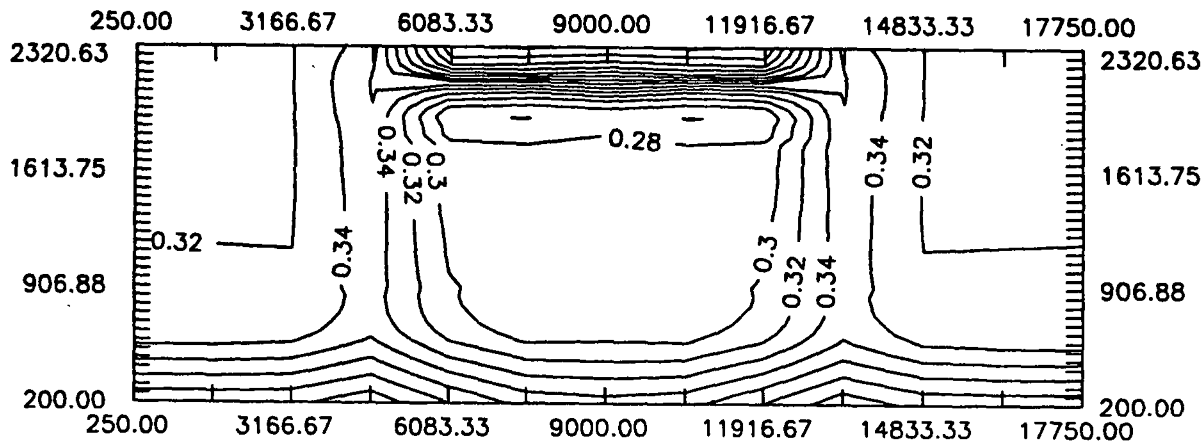

Figure 10-3: VAM2D Moisture-Content Contours

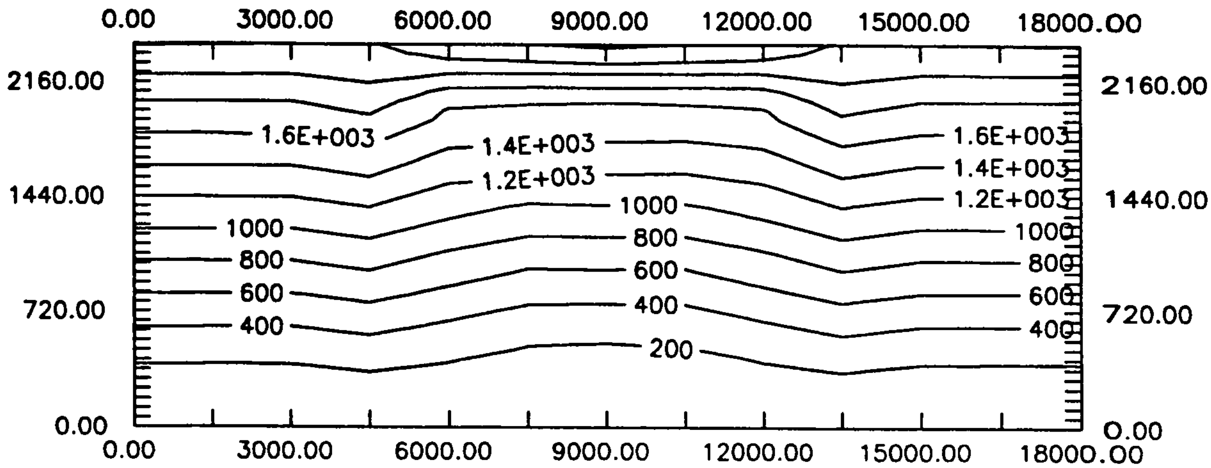

Figure 10-4: VAM2D Total-Head Contours

discussed above, the best estimate of the vadose-zone travel time is close to 140 years; since the analysis is insensitive to this value, it is unnecessary to demonstrate conservatism, or to model the travel time in greater detail.

The purpose of the ground-water transport analysis is twofold. First, the analysis must estimate the concentration history at the well located downgradient from the facility. Second, the analysis must estimate the flux of radionuclides that travel through the aquifer and discharge into the river. These two processes are treated as decoupled. That is, removal of contaminant at the water well does not reduce the flux to the river.

The history of concentrations at the water well is generated using DISPERSE, which contains the appropriate source-term analysis. The ground-water concentration is determined by a numerical integration of a Green's function in time, as described in Chapter 5. When the integration is performed for a number of times, there results a well concentration history. This concentration history is shown in Fig. 10-5. In this figure, 140 years has been added to account for the vadose-zone travel time.

The history of C-14 flux to the river is analyzed using SURFACE, which was described in Chapter 5. The flux history that results is shown in Fig. 10-6. In each of these figures, 140 years has been added to the time to account for the unsaturated-zone travel time, and radioactive decay has been accounted for during the overall travel time. In this

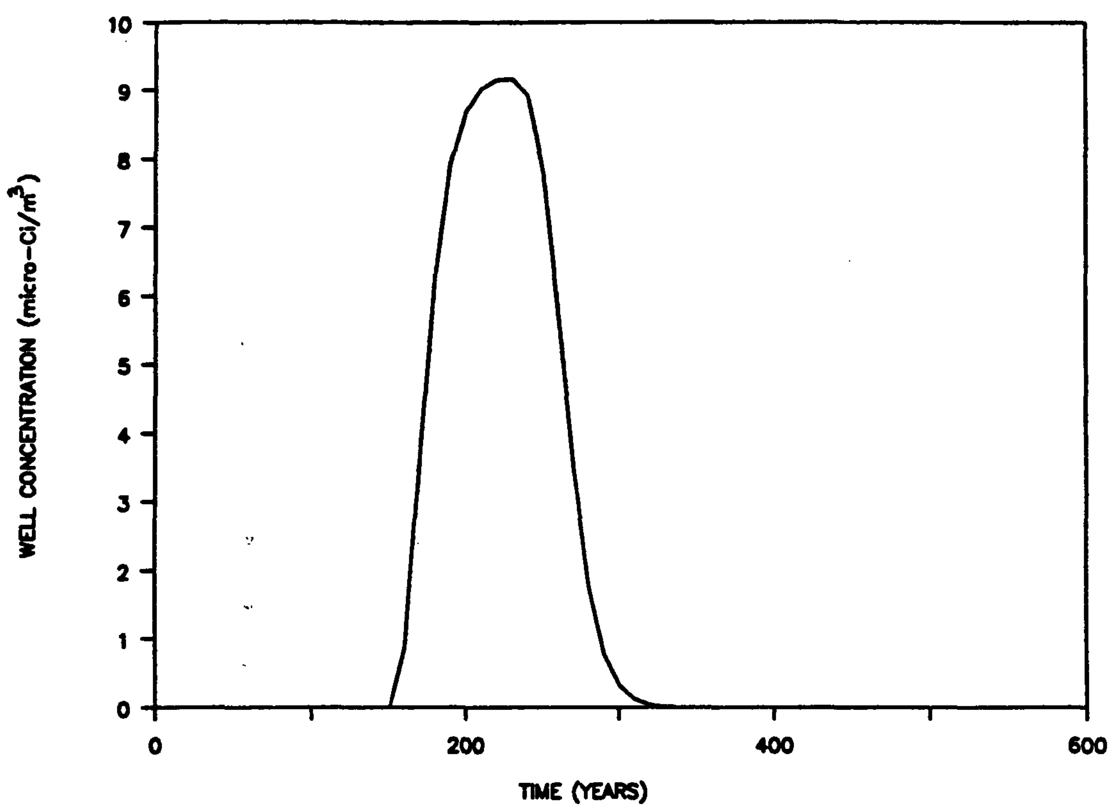

Figure 10-5: Demonstration Well-Concentration History

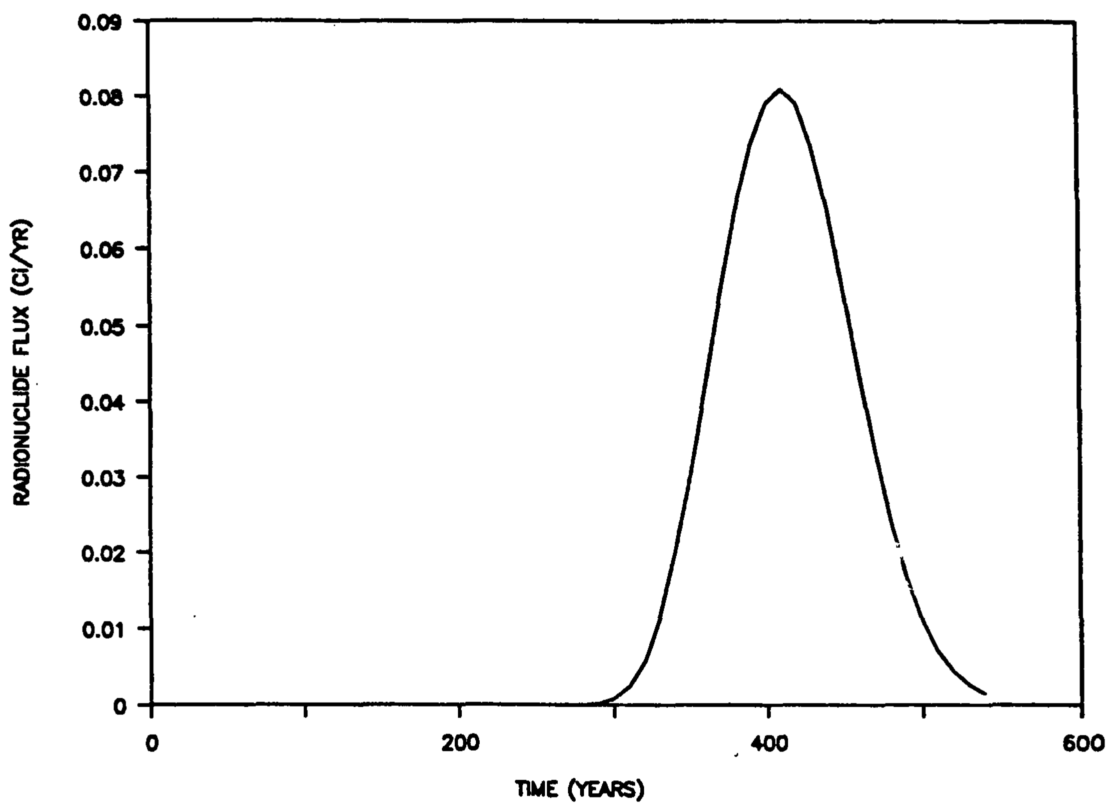

Figure 10-6: Demonstration History of Flux Into River

instance, radioactive decay is negligible, since the half-life of C-14 is significantly longer than the overall travel time.

10.2.4 Pathways and Dosimetry Analysis

GENII can be used to analyze the doses to the maximally exposed person from chronic ingestion of well water concentrations calculated using DISPERSE. In addition, exposures to surface water can be calculated using the flux calculated in SURFACE together with the surface-water transport model in GENII. The procedure for analyzing the dose is to use GENII for a number times at which ground-water concentration and flux into surface water are known. For each time, only these two values are changed; the result is a total dose history for the site.

The following parameters are used in the surface-water transport model in GENII:

- Transit time to access point for all uses: 0.5 hr.
- Rate of effluent discharge to receiving water body: 0.001 m^3/s
- Average river depth: 10 m.
- Average river width: 10 m.

These values have been assumed for the present analysis of a hypothetical site. In practice these are parameters that need to be estimated during the conceptual model development from actual site data.

Well water is assumed to be untreated, and used only for water ingestion. Surface water is used for recreation, and for irrigation of leafy vegetables, other vegetables, fruit, and cereals. Contaminated irrigation is also used on foods consumed by food animals. Doses are considered from contaminated meat, poultry, fish, cow milk, and eggs. GENII default values are used for consumption rates and exposure times, which assume all foods consumed by the person are contaminated. In short, all surface-water exposure pathways are included except water ingestion, and 100 percent of the maximally exposed person's diet is assumed to come from contaminated sources. The maximally exposed person is assumed to get 100 percent of his drinking water from the contaminated well.

Despite these conservative assumptions, and despite the conservative surface-water transport model used, doses due to surface water exposures are negligible. The dose history from this calculation is shown in Fig. 10-7. The peak dose of 13 mrem occurs at 230 years following release from the source, and is due entirely to consumption of well water. The peak contribution to the dose from surface-water pathways occurs at year 410, and is only 0.061 mrem. The dose from surface-water pathways is sensitive to the surface-water transport model parameters listed above, but in no case is the contribution significant. The dose from surface-water pathways calculated by the GENII surface-water model is substantially larger than the dose calculated from a simple dilution model. For the large river used in this demonstration, a simple dilution model is overly optimistic about the transverse dispersion coefficient in the river: in the dilution factor model the river is assumed to be transversely well mixed.

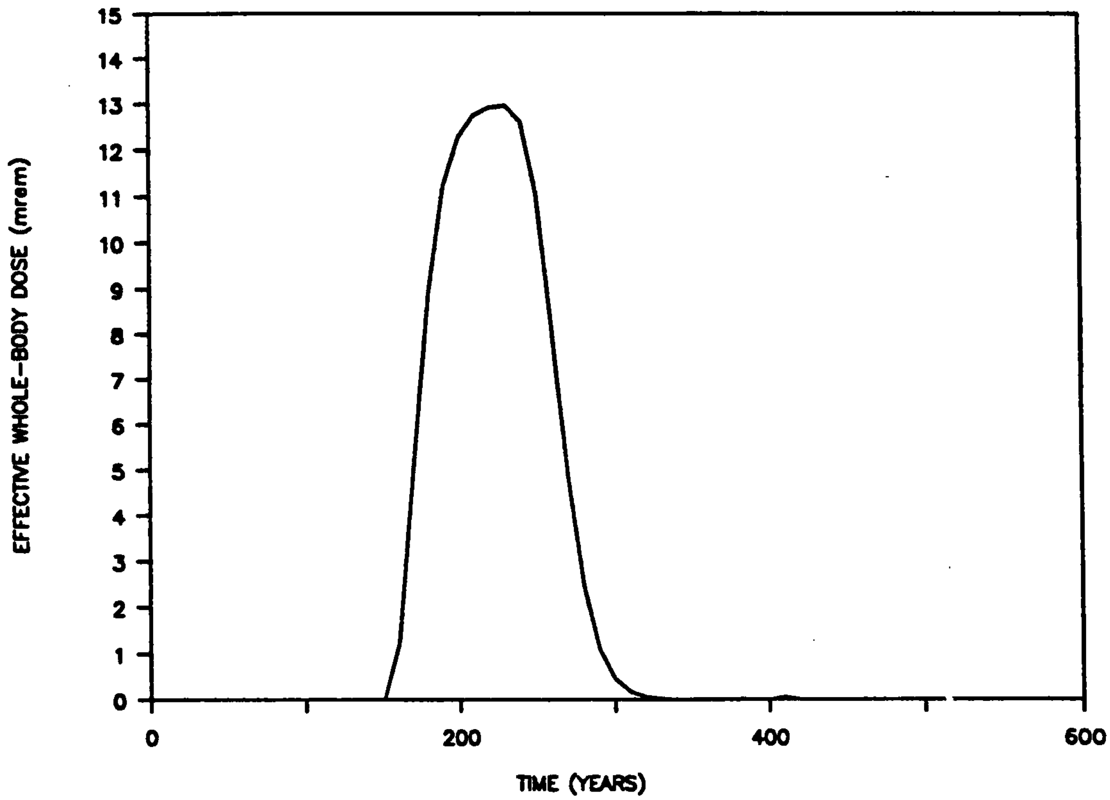

Figure 10-7: Demonstration Dose History

11.0 SUMMARY AND CONCLUSIONS

A performance assessment methodology has been developed for the NRC to use in regulatory assessment of low-level radioactive waste facility license applications. The methodology is designed to allow the NRC to compare calculated estimates of the facility performance with regulatory criteria.

The methodology has a modular structure that allows the NRC to perform confirmatory analyses on either part of, or all of the licensee's performance assessment. In addition, the modular structure allows substitution of more detailed calculations when desirable, allows comparison of the conceptual model to other reasonable conceptual models, and allows better models to be substituted into the methodology as they become available.

This report contains a summary of the background reports written for this project [Shipers, 1989; Shipers and Harlan, 1989; Kozak et al., 1989a; Kozak et al., 1989b], and a brief overview and summary of the models and computer codes incorporated into the methodology. Sources of uncertainty and data requirements are discussed for each part of the methodology, and a demonstration is given of the methodology applied to a particular conceptual model. More detailed discussion of operating procedures and input guides will be published in a Self-Teaching Curriculum.[4]

The most likely pathway for radionuclides to reach humans from a low-level waste facility is through the groundwater. The modeling components for the groundwater pathway consist of infiltration, percolation, source term, unsaturated-zone flow and transport, saturated-zone flow and transport, food-chain, and dosimetry. Infiltration is dependent on the climatic conditions and surficial features of a site. No unique method used in estimating infiltration is appropriate for a site. Therefore for performance assessment, it is recommended that the licensee should provide the results from several techniques to estimate infiltration. Flow of water into the disposal unit should usually be modeled as multi-dimensional flow through and around the engineered cover. Such analyses must generally be performed numerically using such computer codes as VAM2D [Huyakorn et al., 1989] or FEMWATER [Yeh and Ward, 1980]. Kozak et al. [1990] recommended VAM2D for use in this methodology due to its flexibility in handling a wide variety of nonlinear soil properties.

There will often be large uncertainty in modeling the source term (i.e., release of radionuclides from the disposal unit) in the performance assessment. As a result, a simple surface-wash model will often be appropriate, and such a model has been incorporated into the mixing-cell cascade model used in the computer codes DISPERSE and SURFACE [Kozak et al., 1990], which also perform ground-water transport analyses. In addition, these codes allow a constant leach rate to be specified, and

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4.Chu, M. S. Y, M. W. Kozak, J. E. Campbell, B. K. Thompson, and P. A. Mattingly A Self-Teaching Curriculum for the NRC/SNL Low-Level Waste Performance Assessment Methodology, NUREG/CR-5539, SAND90-0585, Sandia National Laboratories, in press.

this option allows solubility-limited or diffusion-limited releases to be modeled. More detailed source-term analyses are included in BLT [Sullivan and Suen, 1989], and this code can be used to model container corrosion rates and leaching processes limited by mass-transfer processes. The analyst should be aware that incorporating such processes is likely to be less conservative than using a surface-wash model, and hence these processes must be justified using site-specific data. The burden of justification of any assumption is on the licensee [Starmer, 1988].

Transport in the saturated zone can often be modeled using DISPERSE, to determine the maximum concentration at a well, and SURFACE, to determine the radionuclide flux into a surface-water body. The results from these codes, or from VAM2D or BLT, can be used as input to GENII [Napier et al., 1988], which contains surface-water transport models, air transport models, food-chain models, and dosimetry analyses. The air pathway is not generally expected to be important for the undisturbed site, but in the event that it is important the analysis can be performed using GENII.

The result of the performance assessment analysis is a series of dose histories for each radionuclide of importance. The contribution of each radionuclide to the dose must then be added together to produce the total predicted dose. This dose estimate is intended to be compared with the regulatory performance objectives in 10 CFR Part 61.41. Estimated doses are not intended to reflect actual doses that may be received by members of the general public.

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A performance assessment methodology has been developed for use by the U.S. Nuclear Regulatory Commission in evaluating license applications for low-level waste disposal facilities. This report provides a summary of background reports on the development of the methodology and an overview of the models and codes selected for the methodology. The overview includes discussions of the philosophy and structure of the methodology and a sequential procedure for applying the methodology. Discussions are provided of models and associated assumptions that are appropriate for each phase of the methodology, the goals of each phase, data required to implement the models, significant sources of uncertainty associated with each phase, and the computer codes used to implement the appropriate models. In addition, a sample demonstration of the methodology is presented for a simple conceptual model.

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