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Evaluation of a Performance Assessment Methodology for Low-Level Radioactive Waste Disposal Facilities

Evaluation of Modeling Approaches

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Abstract

This report represents an update to our earlier reports on low-level waste performance assessment. This update addresses needed improvements and recommended approaches to the existing state of the art in modeling, treatment of uncertainty, and use of data. Greater attention is paid to developing an integrated approach to performance assessment than was done in earlier developments of the methodology. Furthermore, insights are being developed by participating in validation exercises, and by evaluating which validation data are needed to improve confidence in the methodology. It is emphasized that the performance assessment methodology update is a work in progress; the recommendations given here will form the general directions toward which the methodology is heading, but some of the specific approaches may continue to evolve as the research progresses.

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FOREWORD

This technical contractor report is a product of Sandia National Laboratories under project FIN L1153. The purpose of this program is to update and improve a performance assessment methodology for low-level radioactive waste disposal facilities previously developed under FIN A1764.

NUREG/CR-5927 is not a substitute for NRC regulations and compliance is not required. The approaches and/or methods described in this NUREG/CR are provided for information only. Publication of this report does not necessarily constitute NRC approval or agreement with the information contained herein.

1. Introduction

1.1 Background

A low-level radioactive waste performance assessment methodology was developed by Sandia National Laboratories (SNL) for use by the U.S. Nuclear Regulatory Commission (NRC) in evaluating license applications under Section 10 of the Code of Federal Regulations, Part 61 (10 CFR Part 61) [Kozak et al., 1990b]. The purpose of the methodology is to allow NRC to confirm a licensee's evaluation of postclosure impacts. These performance assessment analyses are the basis for providing reasonable assurance that the performance objectives in 10 CFR Part 61.41 are met.

The methodology must be flexible enough to handle a wide range of potential low-level waste disposal facilities. The performance assessment modeling may need to be either very simple, or more complex, and models are included in the methodology for both of these possibilities. Since the methodology is modular, the analyst may substitute more complicated models for only part of the analysis when appropriate.

The components needed for performance assessment modeling of a low-level waste facility are shown in Figure 1.1. Before choosing the models to be implemented in the methodology, a literature survey was performed to identify existing models and codes for each required process. In addition, general site characterization data requirements were identified, and significant sources of uncertainty were discussed [Kozak et al., 1989a]. That work formed the basis for current models in the methodology. These models are shown in Figure 1.2. The primary impetus for choosing many of these codes was their flexibility in modeling a wide variety of problems. Further justification and discussion on some modeling areas was provided in Kozak et al. [1990a]. Some areas are modeled very conservatively in the methodology; this approach was taken when no adequate model was available, or when details of the processes themselves were poorly understood.

SNL was subsequently contracted to update and improve the methodology where necessary, and to build confidence in the models in the methodology. This is a report to assess whether the current models in the methodology are adequate, and based on this assessment, to identify additional models and codes that may be useful to include in the methodology. This report represents an update to the discussions found in Kozak et al. [1989a, 1989b, 1990a]. The intent is to update the information in these reports by including discussions on new models and codes that have become available since the preparation of the original methodology. In addition, since developing the methodology, we have applied it several times to performance assessment test cases [Chu et al., 1991; Kozak;[1] Kozak and Rao;[2] Kozak and Feeney[3]], and this additional experience allows an improved assessment of the modeling needs.

Furthermore, insights are being developed by participating in validation exercises, and by evaluating which validation data are needed to improve confidence in the methodology. Volume 2 of this report covers the validation needs for the modeling areas in the methodology [Olague et al., 1993]. Priorities are set for the most important validation problems that need to be addressed.

1.2 Scope of the Report

This report is an update to our earlier work, described in Shipers [1989], Shipers and Harlan [1989], and Kozak et al. [1989a, 1989b, 1990a, 1990b], as well as other pertinent documents and papers on low-level waste performance assessment [e.g., Starmer et al., 1988; Deering and Kozak, 1990]. It must be read in that context, since it is not intended to be a stand-alone guide to performance assessment of low-level radioactive waste disposal facilities. Rather, it is a summary of needed improvements and recommended approaches to the existing state of the art in modeling, treatment of uncertainty, and data availability. In addition, the performance assessment methodology update is a work in progress; the recommendations given will form the general directions toward which the methodology is heading, but some of the specific approaches may continue to evolve as the research progresses.

The goal of the methodology is to enable the NRC to evaluate postclosure, off-site doses from a low-level radioactive waste disposal facility for comparison with the regulatory performance measures of 10 CFR Part 61. Inadvertent intruders receive adequate protection through

[1] Kozak, M.W., "Preliminary Analysis of Cases 1a and 1b," FIN A1764 letter report, submitted to F.W. Ross, NRC/NMSS, June 1991.
[2] Kozak, M.W., and R.R. Rao, "Analysis of NSARS Case 1," FIN A1764 letter report to NRC, submitted to F.W. Ross, NRC/NMSS, August 1991.
[3] Kozak, M.W., and T.A. Feeney, "Analysis of NSARS Case 2a," FIN A1764 letter report to NRC, submitted to F.W. Ross, NRC/NMSS, September 1992.

the waste classification scheme developed as part of the Environmental Impact Statement for 10 CFR Part 61 [NRC, 1981]. As a result, intruder analyses will only be required in a low-level waste license application under special circumstances, when an exemption from the waste classification system is proposed [Kozak et al., 1990a]. Consequently, little attention will be given to intruder analyses in this report; the focus of the methodology is on evaluation of off-site doses to the maximally exposed individual of the public.

1.3 Structure of the Report

This volume covers three primary topics. First, a general assessment of the methodology and of performance assessment as a whole is given. In particular, revisions are discussed for the areas of uncertainty analysis and user friendliness. A pathway assessment is presented for a variety of disposal options in Chapter 3. The intent of this pathway assessment is to ensure that the methodology contains adequate coverage of all types of modeling that it might be faced with. In essence, the evaluation in Chapter 3 is a review of the methodology for completeness. The second aspect of the review of the methodology is to reevaluate the models in the methodology, given the results of the completeness review, and the passage of time. This review of the modeling needs is given in Chapters 4 through 8 for each modeling area of the methodology. Chapter 8 contains a summary of the evaluation of the methodology.

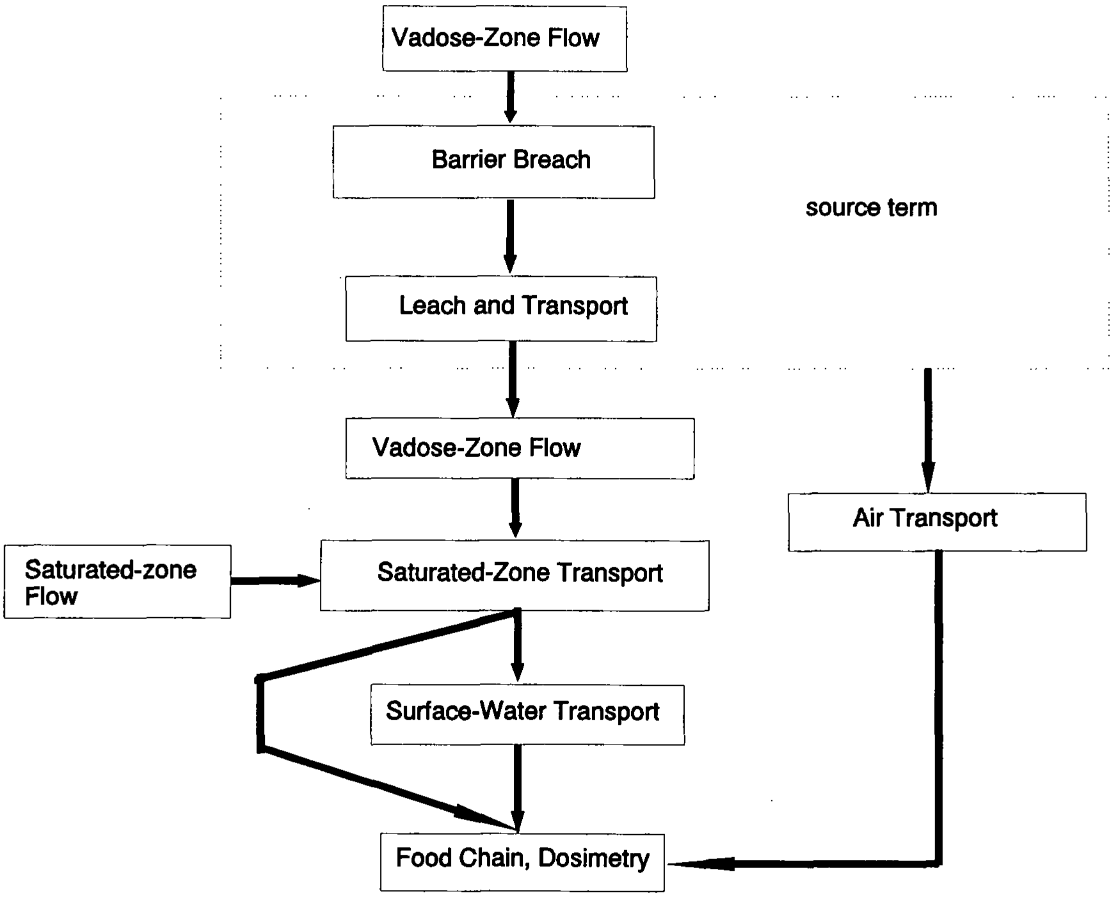

Figure 1.1 Processes included in the methodology

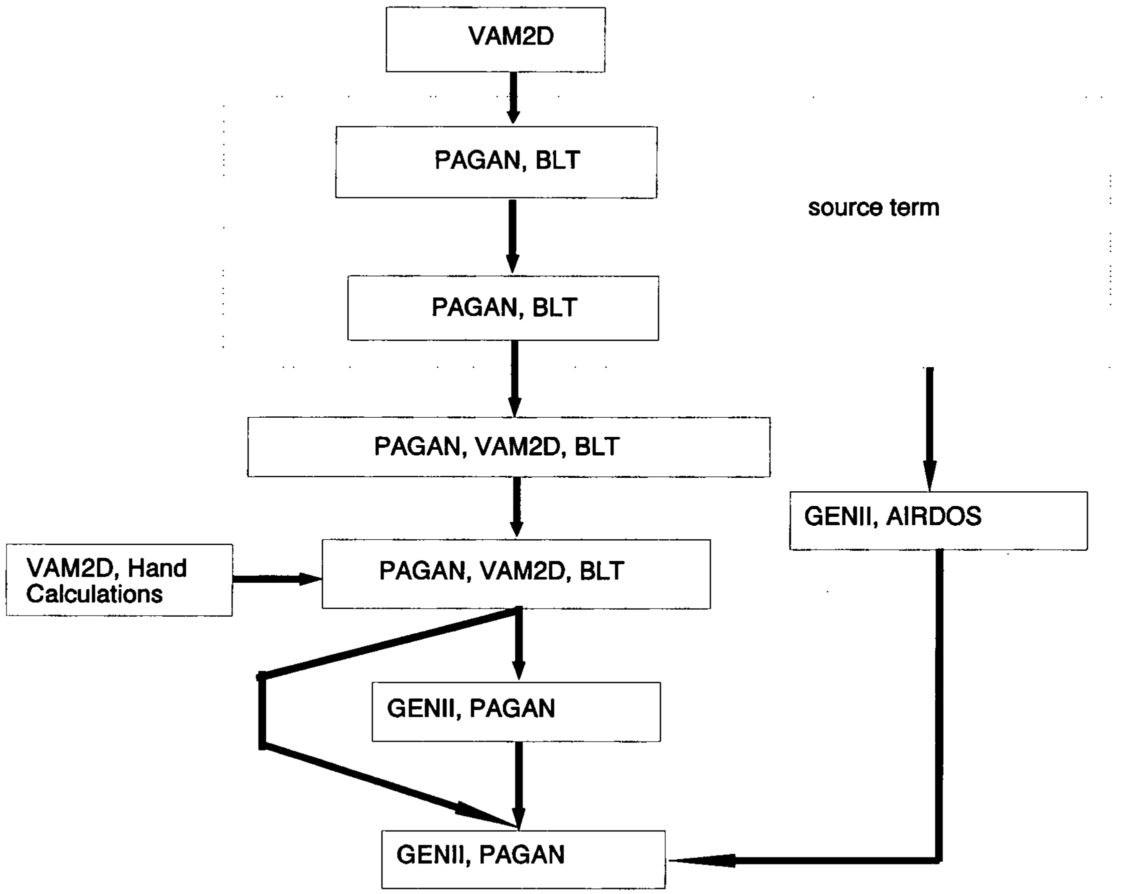

Figure 1.2 Modeling approaches in the original methodology

2. General Considerations

In our ongoing evaluation of the methodology, we are assessing aspects of performance assessment that were not stressed in the original methodology development. These areas are (1) uncertainty analysis, and (2) improved user friendliness. These areas, which are outside the strict scope of modeling updates but are of general importance, are discussed in this section.

2.1 Uncertainty Analysis

The methodology is intended to be used to allow the NRC to determine if a candidate low-level waste site will meet the performance objectives specified in 10 CFR Part 61. These performance objectives specify that an off-site person may not receive a committed annual dose equivalent of more than 25 mrem whole body, 75 mrem thyroid, or 25 mrem to any other critical organ. These are deterministic performance measures, implying that assessing the performance of each site will produce one dose that can be compared to the standard. However, uncertainty analyses invariably lead to a suite of doses, which must then be compared with the deterministic standards. The purpose of this section is to point out some of the challenges, and possible solutions, associated with comparing uncertainty analysis results with a deterministic performance objective. This section is an expansion and refinement of earlier work on this subject [Kozak et al., 1991].

The uncertainties in performance assessment have been classified as model uncertainty (which spans conceptual model uncertainty and mathematical model uncertainty), uncertainty about the future of the site, and parameter uncertainty [Davis et al., 1990a]. These inherent uncertainties are dealt with using uncertainty analysis, which is a way of formally documenting, treating, and reducing the inherent uncertainty of a system. The analysis is nothing more than an identification of how much or how little confidence the analyst has in his knowledge of the modeled system [Finkel, 1990].

If all uncertainties are addressed, the result would be alternative possible doses depending on our conceptualization of the site, the mathematical models we use to represent the conceptualization, the parameters used in the mathematical model, and the conceptualization of the evolution of the site in time. To address these uncertainties, NRC currently recommends that the licensee for a low-level waste site give dose estimates with a range of minimum and maximum values; this range should take into account all uncertainties in the calculations [Starmer et al., 1988]. However, no explicit guidance is given on how the ranges should be calculated. NRC also suggests that the doses be presented as a function of time, but does not identify a method for taking into account changes in the low-level waste site as it evolves in time, nor for how long the performance assessment should be conducted. Thus, there is currently no official position on the treatment of uncertainty for low-level waste performance assessment. Consequently, the original performance assessment methodology does not contain formal uncertainty analysis.

In the following sections, we give recommendations on how to incorporate uncertainty analysis into the methodology. First, we give background on how to address and reduce the three different types of uncertainty (model, parameter, and future) based on a literature review. From the literature review, we have determined the best methods currently available and the ones that we recommend for inclusion in the methodology. Second, and more important, we have developed a strategy for implementing our recommendations into the methodology.

2.1.1 Model Uncertainty

The process of developing a site-specific model begins with the perceived real world, as defined by site-specific data [Davis and Olague, 1991]. (It is important to note that we are never able to completely perceive reality, particularly in ground-water modeling where the system cannot even be directly observed, since this incomplete perception is a primary source of uncertainty in modeling.) Next, simplifying assumptions are made to develop a conceptual model, which is a qualitative description of the processes, geometry, and boundary conditions associated with a site. These qualitative ideas are translated into a quantitative mathematical model, which is a set of equations that represent the behavior of the conceptual model. The mathematical model can then be solved, with site-specific input parameters, analytically, or with a numerical approximation (in a computer code) for the quantity of interest (e.g., effective dose equivalent).

Model uncertainty encompasses both the uncertainty in the conceptualization of the system, the uncertainty in its mathematical representation, and the uncertainty in the solution of the mathematical representation [Bonano and Cranwell, 1988]. Conceptual model uncertainty arises from a number of different sources. There may be uncertainty associated with the characterization of the perceived "real system," such as misinterpretations of the data or inadequacy of data reduction techniques. Uncertainty will also be introduced with the simplifying assumptions that are necessary to make the problem tractable. For instance, the transient, three-dimensional real world is usually modeled as a steady-state one-dimensional process. In addition, models are most commonly developed by a single analyst or a small group of analysts using only their professional judgment to resolve available data into a model. The model is therefore limited by the abilities and imagination of the developer, in addition to limitations in the available data. Also, despite the usefulness of experience with sites other than the one being considered, one can develop a poor conceptual model for an apparently similar site based on preconceived ideas [Bonano and Cranwell, 1988]. An example of this is using a saturated porous media flow model with a few modifications for saturated fracture flow.

In many cases, conceptual model uncertainty is the dominant type of uncertainty in a performance assessment: if an inadequate conceptual model is being used, uncertainty associated with the mathematical model and the model input parameters becomes irrelevant. In some cases even the processes at work are not well understood. For example, the conceptual model cannot be identified for fracture flow in porous media. There is potentially simultaneous flow in the porous matrix and in fractures in the media, but no one understands the conditions that cause one or the other to dominate the performance of the system. The physics behind the processes are not understood well enough to allow us to write down the equations describing the exchange of material between fractures and the matrix, except in a heuristic manner that has not been substantiated (or refuted) by experiments [Updegraff et al., 1991].

Besides the uncertainties associated with the underlying conceptual model, uncertainty in mathematical models arises from the methods required to arrive at a solution to the equations of interest [Davis and Olague, 1991] and from the inability to represent a conceptual model in a suitable mathematical form [Bonano and Cranwell, 1988]. If an analytical technique is used, uncertainty can be introduced if the solution to the equations is incorrect, or from the truncation of a mathematical infinite series such as an error function used as part of the solution. If a numerical solution method is used, it will almost always be implemented in a computer program. This can introduce two sources of uncertainty: errors from the numerical approximation of the equations and coding errors in the computer program. An additional source of uncertainty for computer codes is user error.

2.1.1.1 Treatment and Reduction of Model Uncertainty

Very little work has been done to treat conceptual model uncertainty [Kozak et al., 1991]. The only available approach to treating conceptual model uncertainty is to span the range of conceptual models that are consistent with site-specific data. Some have suggested that the formal elicitation of expert opinions may be a good way of spanning this range [Kerl et al., 1991, Chhibber et al., 1991a] by creating an exhaustive list of possible alternative conceptual models that are consistent with available data. By broadening the base of expertise from which the conceptual models are developed, there is increased likelihood that a conceptual model will be included that captures some potentially adverse characteristic of the site, and to the extent possible, conceptual model uncertainty is addressed. The disadvantages of this approach include increases in cost and time and reduction in the flexibility associated with formalizing expert judgment [Bonano et al., 1990].

One way to implement this approach into performance assessment has recently been proposed [Chhibber et al., 1991a,b; Heger et al., 1991]. This method associates a probability with a given conceptual model, which is interpreted as a measure of the degree of belief that the conceptual model is appropriate for the given purpose. However, Chhibber et al. [1991a] recognizes a number of difficulties in this technique. Perhaps the most important constraint is that to apply probability theory, the models should be defined so that they are mutually exclusive, exhaustive, and independent. This difficulty seems insurmountable since all the conceptual models are based on the same site-specific data. Other difficulties arise when combining and aggregating expert opinion, and when incorporating new information into probability estimates. Given these problems, we conclude that this approach is an interesting area of research, but many significant issues need to be addressed before it can be used in performance assessment.

We suggest a simpler approach. All conceptual models consistent with data should be used for performance assessment of low-level waste sites. If the models cannot be distinguished from each other by acquiring additional site-specific data, then each of the models should be considered credible. The performance assessment must then be conducted using each model, and the results used to establish the model that is the most conservative. In general, it is not possible to establish conservatism of the model a priori. The reason for this is that most models used in licensing are mixtures of conservative and non-conservative assumptions. True "bounding" analyses are rarely used, because it is usually considered that such an approach is excessively conservative. Conservatism among models can only be established by posterior comparisons of the calculated performance objective of the different models.

Model intercomparison can be used to some extent to determine conservatism. Model intercomparison is defined as a comparison of codes that implement different conceptual models of the same processes (i.e., the models may not implement the physics or chemistry in an identical manner, or may have somewhat different assumptions). For instance, one might compare results from a one-dimensional, single layer transport analysis of radionuclide migration to results from a multidimensional, multilayer model. The intercomparison can be used to identify crucial assumptions in the two approaches to modeling radionuclide migration, and these assumptions can then be the focus of validation studies. It should be noted that intercomparison is different from benchmarking, which is a comparison of computer codes that have the same conceptual model.

Overall model uncertainty can be reduced, but not eliminated, by site-specific model validation. Site-specific data is the most defensible evidence for determining the reliability of a model, since it represents the real system to be modeled. However, as discussed by Davis et al. [1991], it is not practical to conduct validation experiments for the full range of conditions of interest in performance assessment because of time and funding constraints and because extensive testing at a site may interfere with the site's geologic integrity. Therefore, validation can be used to build confidence that the uncertainties are reduced to the extent practicable. In general, the appropriateness of any performance assessment model should always be determined based on site-specific data.

We note that "conservatism" of a model is always relative to something. Ideally, we want the model to be conservative with respect to actual (perceived) site behavior. Unfortunately, we will rarely have the luxury of establishing conservatism compared to any single aspect of site behavior, much less conservatism of the overall performance assessment. Consequently, model conservatism will usually be defined with respect to other possible models or combinations of parameters. In this sense, model intercomparison must play an important role in evaluating the conservatism (hence reliability) of the analysis results.

For performance assessment, mathematical model uncertainty is usually not propagated to the results, since it is believed to be negligible compared to the other uncertainties [Davis and Olague, 1991]. Therefore, our approach to treating mathematical model uncertainty is that it should be reduced to the extent practicable, but otherwise ignored in the propagation of uncertainty. This can be accomplished through a variety of methods. If an analytical solution is used, it can be compared to other available solutions for accuracy or checked by an available expert. For numerical or semianalytical solutions implemented in computer codes, several uncertainty reduction methods should be used. Careful quality assurance procedures should be followed during the development of the computer code to avoid the introduction of coding errors. Quality assurance activities should also include verification exercises. Verification gives assurance that the model equations are implemented correctly in the computer code. This is accomplished through careful evaluation of the program and by comparison of the program output with analytical solutions or other computer programs that implement similar physical processes (i.e., benchmarking). Once the program is in operation, a configuration management system should be followed to ensure that no haphazard modifications to the program are made. Error reduction techniques should be followed, such as using a finer discretization of the domain or using more terms to represent an infinite series. The user will know that the error has been reduced to acceptable levels once a convergent solution is achieved; i.e., the discretization is made finer without the solution changing.

The uncertainty associated with conceptual models can be treated by trying to better understand the physics behind the relevant phenomena. Alternatively, for site-specific modeling an empirical model with well-chosen parameters could be adequate for the intended purpose when compared to a more complex, physics based model. This issue can only be resolved with site-specific model validation [Davis et al., 1991]. However, it is very important to realize that empirically based models cannot be extrapolated outside of the domain in which their coefficients were determined. This makes empirical models of intrinsically limited usefulness in performance assessment, since most conditions that are important in a performance assessment are not ones that can be observed at today's conditions.

Validation is the process in which confidence is gained that a model is an accurate representation of the physical processes for which the models are intended [Davis and Olague, 1991]. It offers a chance to build confidence in the conceptual model, the mathematical model, and the site-specific parameters by seeing how well the hypothetical representation of the site compares with the performance of the actual site. Validation also tests how well the mathematical representation of the physical processes compares to site-specific data. One drawback of validation is that since it tests the whole system, it does not tell you which component of the overall model is incorrect. Thus, if a site-specific model compares unfavorably with data, it is not readily apparent if the disagreement is caused by using the wrong conceptual model or incorrect parameter ranges.

We recommend that all available procedures be used to reduce the uncertainty in low-level waste performance assessment mathematical models. Careful quality assurance procedures should be followed in the development of a solution to the mathematical model. Verification should be used to ensure that the correct equations are being solved and validation should be used to ensure that the model does represent the behavior of the site.

2.1.2 Uncertainty About the Future of the Site

Uncertainty about the future of the site is the result of our inherent lack of knowledge about how the site will evolve in time. The climatic, geologic, and population conditions that will prevail in the future of the site are not known. Consequently, we must determine some method for accommodating alternative future conditions.

At the present time it is not clear what conditions need to be evaluated to meet the low-level waste regulations. NRC has suggested that different future scenarios should be modeled, but has given no explicit guidance on the selection process [Starmer et al., 1988]. Also, there is currently no regulatory guidance concerning the performance assessment time period or how to account for future conditions at a low-level waste site. NRC has stated that the low-level waste "source term inventory should be used to justify the necessary duration of the performance assessments" [Starmer et al., 1988]. This may imply that source terms containing large amounts of long-lived radionuclides, such as ^{14}C, ^{99}Tc, etc., should be modeled for longer times than source terms containing only short-lived nuclides such as ^{60}Co and tritium.

If explicit regulatory guidance was given, it would promote consistency between analyses done by regulators, states, and compacts, which in turn would tend to make them more defensible. Uncertainty about future conditions is not primarily a technical issue, but rather must be resolved by the regulators. The purpose of the following discussion is to provide possible approaches based on relevant technical information from which regulatory decisions can be made.

2.1.2.1 Treatment and Reduction of Future Uncertainty

The generalized approach to addressing future uncertainty is to span the range of high probability, high risk future conditions of the site. Identifying all important events and processes is a creative task that may depend exclusively on expert judgment [Bonano et al., 1990]. Spanning all important possible future conditions can most readily be accomplished through formalized elicitation of expert opinion. This can be done in an analogous manner to the conceptual model elicitation. However, a significant amount of elicitation has already been done for scenarios, and comprehensive lists of possible disruptive events and processes exist [Cranwell et al., 1990].

Two issues need to be addressed regarding uncertainty about the future. First, we need to determine how long the performance assessment will need to be carried out. Second, we must decide on a way to incorporate alternative views of how the site will change in time into the performance assessment methodology. We emphasize that the intent must be to span the range of likely behavior of the site over the time frame of concern; we are not interested in addressing improbable events, but rather including extreme, but very likely events in the analysis.

One logical strategy to quantifying the time frame for low-level waste performance assessment would be to model the site until the peak dose is obtained. Because low-level waste contains long-lived radionuclides (mainly ^{14}C, ^{129}I, ^{99}Tc, and the actinides), and because of the current emphasis on long-lived engineered barrier systems, this time period can become relatively long. For long time periods, it may become important to include such events as glaciation, which would not be pertinent to short time periods. Assumptions would have to be made about the future of the site, and these assumptions would depend greatly on how long a time period needs to be modeled.

Another possible strategy for quantifying a time frame for low-level waste performance assessment would be for

NRC to mandate a specific time period that needs to be modeled, similar to what was done for high-level waste in 40 CFR Part 191. Once a time period has been designated, a method needs to be developed to account for all probable, high consequence, future states of the sites. For high-level waste performance assessment, future uncertainty is accounted for by explicitly acknowledging possible alternative future scenarios. These possible scenarios are then incorporated into the overall performance assessment by assigning likelihoods to each scenario. This approach was developed in response to the requirement in 40 CFR Part 191 that high-level waste performance assessment must include all significant events and processes over a period of 10,000 years. This requirement is not explicitly included in 10 CFR Part 61.

One way to address future uncertainty for low-level waste performance assessment is to use the scenario approach that has been developed for U.S. high-level waste regulation. Scenarios are future likely conditions of the site that affect the performance assessment results. The intent of scenarios is to provide a simpler surrogate approach for modeling the likely transient future history of the site. Scenarios are treated by modeling the system as a steady-state process, then weighting the consequences of that scenario by its likelihood. The rationale for weighting the scenario consequence is to identify the contribution of that scenario to the overall performance measure, which is integrated discharge for the U.S. high-level waste regulation. For example, consider a climate change that produces increased infiltration that lasts for 1000 years. A high-level waste scenario analysis would model the steady-state integrated discharge using the higher infiltration rate, then weight it by 0.1 to incorporate its contribution to the overall integrated discharge over 10,000 years. It is important that the likelihood associated with this scenario is not related to its probability of occurrence (the probability of a wetter time period during the next 10,000 years is essentially unity), but rather is meant to identify the amount of contribution to the performance measure. It is clear that this type of rationale cannot be applied to doses, which are not integrated over time, but rather are point estimates in time. If transport from the site were fast enough under these conditions, that waste could be flushed from the disposal facility to the receptor in this wetter period, making the receptor the maximally exposed person. In this case, there does not seem to be a reason to weight a dose produced during the wet period with a small likelihood, because the probability of that dose occurring is near unity (if all the other assumptions made in the analysis are met).

On the other hand, the scenario approach has two advantages: (1) it traces assumptions about the future in a formal manner, and (2) it addresses the uncertainty associated with the future. There are two problems associated with using this approach for low-level waste performance assessment. First, because most low-level waste sites are located near the land surface, surficial events and processes that could reasonably occur over long time periods may become important (e.g., flooding, erosion, glaciation). Considering such processes would complicate low-level waste performance assessment significantly, and it is questionable as to whether or not a near-surface facility could meet the regulations with these types of events and processes occurring. For instance, in many parts of the country, if it is reasonable to assume that glaciation may occur at a site within 10,000 years, does that mean that glaciation should be included in a low-level waste performance assessment?

To illustrate the problem with using extreme events and processes to compare with the performance objectives, let us analyze direct exposure of an individual to waste exposed at the surface by glaciation, and assume that the consequence analysis of this scenario (and only this scenario) results in exceeding a dose performance measure. The regulations specify a maximum 25 mrem/year dose, so let us assume that the consequence of this analysis significantly exceeds the standard, and the person receives 100 mrem/year. Given a regulatory philosophy that requires all consequences fall below the performance measure, this facility would fail the safety assessment. Furthermore, since no near-surface facility can reasonably be expected to survive the onslaught of a glacier, it can be expected that no facility at that site could meet the objectives, and the entire location could be eliminated from consideration. However, let us consider a counter-argument. A person living near the moraine of a glacier during an ice age would undoubtedly have much larger health problems than those posed by receiving 100 mrem/year, which poses immeasurably small health risks [Gershey et al., 1990]. We have, therefore, been trapped by our inclusion of the extreme tail of the distribution into making an arguably poor regulatory decision. This example identifies the potential problem with using a combination of a full scenario analysis together with requiring that the entire dose distribution must fall below the performance objective.

The full scenario approach can be salvaged for use in low-level waste performance assessment by choosing an intermediate confidence limit for comparison with the deterministic performance objectives. For instance, the EPA provided guidance that suggested using the larger of the mean or median value of the probability curves for assessing compliance with the Individual Protection Requirement and Ground-Water Protection Requirement in 40 CFR Part 191 [EPA, 1985a]. Alternatively, the regulator may choose to use some higher confidence limit of the dose probability curve to compare against the 25 mrem objective. This approach is self-consistent, since all events and processes of possible importance are included, but it omits the tails of the distribution, and focuses the decision maker's attention on the central tendencies of the distribution. However, it discards the tails of the distribution, which will include some conditions of the site that are likely to occur.

If NRC decides that only a relatively short-term time period needs to be modeled for performance assessment, a full scenario approach may not be warranted. The approach will then be one of including events and processes that occur for the arbitrarily defined short time period, which would probably be small perturbations around the current state of the site. Analyses conducted in this way will be focused on events and processes that are likely to occur (e.g., transport to a well), rather than events distant in the future. However, this relatively short time period may not be sufficient for characterizing the peak dose because of the longevity of some constituents in low-level waste. Also, the use of concrete vaults, which may last hundreds of years, in low-level waste disposal facilities may result in moving the peak dose past the mandated time period.

One way of implementing this approach would be to conduct the performance assessment until the peak dose is reached, but only using conditions that may be reasonably expected to occur during, say, the first 100 years of the postclosure performance time period. The modeling, therefore, includes only relatively minor perturbations about the current state of the site. The question still arises as to how large the perturbations should be. As an example, consider the rainfall at the site, which is important for assessing recharge, and hence, degradation rates of engineered barriers and release rates from the facility. There is an intrinsically probabilistic aspect in defining what rainfall will be included in the analysis, for rainfall is usually treated as being stochastically distributed in time. If the analyst decides to use the 100-year "maximum probable" precipitation year in the analysis, it should be understood that there will remain a finite probability that this value will be exceeded, since by definition "maximum probable" means there is a chance this value will be exceeded. In addition, significant vegetative progression can occur even over relatively short (100-year) time periods, so the span of conditions that may need to be included in the analysis is still quite broad.

Another short time period approach is to maintain the current conditions at the site for the entire performance assessment time period. This is similar to the approach suggested by EPA for assessing compliance with the Individual Protection Requirement and Ground-Water Protection Requirement contained in 40 CFR Part 191. They assume that current conditions will exist at the site for 1,000 years [EPA, 1985a]. Use of this approach requires a definition of what "current conditions" means. Low-level waste sites may only have a few years of detailed monitoring data at the site from which to identify current conditions; these data may fortuitously only span a range of unusual conditions, such as particularly dry or wet years. Use of this data may, therefore, lead to misinterpretations about the long-term behavior of the site.

An alternative approach would be to model the projected time-dependent future history of the site; this approach has been adopted in the United Kingdom for use in comparing with a deterministic regulatory performance objective. The rationale for this approach is that, unlike the scenario approach, extreme (but likely) conditions are not modeled as steady-state conditions for the entire time period of the analysis. Recall that in the scenario approach, the 10,000-year maximum annual rainfall was modeled as a steady-state process that lasts for 10,000 years, but it is then weighted by a 10^{-4} likelihood (actually a relative frequency), since it is presumed to affect 10^{-4} of the integrated discharge. In a time-dependent analysis, the 10^{-4} climate events would only affect the dose if significant radionuclide migration occurs during that single year of the analysis.

An obvious disadvantage to the time-dependent modeling approach is that transient modeling is much more difficult and time-consuming than steady-state modeling. In addition, the future history of the site is not known, even though it is highly likely that some extreme events will occur. This means that in spite of significant additional modeling effort, little will be done to reduce the uncertainties about the future. The danger, therefore, exists of introducing more complicated analyses without improving the confidence in the results.

To summarize, there are four approaches that might be taken for quantifying uncertainties about the future of the site in the context of low-level waste performance assessment. (1) Conduct the performance assessment until peak dose or for an NRC-mandated relatively long time period, and to include possible future events and processes through a scenario evaluation. However, to use this approach in an appropriate way for decision making, the regulators may have to allow some of the tails of the distribution of dose estimates to exceed the deterministic performance objective. The choice of the particular value of the percentile for comparison with the performance objectives is entirely a regulatory decision. Although the scenario approach provides a means for systematically treating future conditions at a site and allowing for possible alternative scenarios, there are serious issues that need to be addressed. (2) Define well-established design-basis conditions, in which only events and processes that are reasonable for a NRC-mandated short time period are included in the analysis. This approach focuses attention on the events and processes that are most likely to occur in low-level waste performance assessment, but does not address the issues associated with the long-lived radionuclides. (3) Use current conditions at the site to extrapolate to the longer time period needed to characterize peak dose or for the NRC-mandated short time period chosen for performance assessment calculations. This approach focuses attention on the most likely conditions of interest, but does not take into account that extrapolation of the design-basis conditions to longer times has progressively less physical meaning as the time period expands. (4) Model the time-dependence of the future history of the site. A choice of one of these four approaches must be made by the regulators based on regulatory philosophy; there is not a clear-cut best candidate based on purely technical considerations.

2.1.3 Parameter Uncertainty

Parameter uncertainty relates to an incomplete knowledge of the performance assessment model(s) constitutive coefficients. In part, this uncertainty is identified with uncertainty in the actual values and the statistical and spatial distributions of data used to infer the model parameters. Sources of parameter uncertainty include measurement error, insufficient data, and inconsistency between measurement scale and prediction scale. In this section, we discuss each of these issues as though they were independent of model uncertainty. However it should be recognized that the two types of uncertainty are not completely independent.

2.1.3.1 Measurement Errors

Measurement errors are generally considered to be of two types: random and systematic. In the context of the larger uncertainties in performance assessments, random errors in parameter measurements are relatively minor, and will not be discussed further here. Systematic errors, however, are unavoidable in many important measurements needed for site characterization, and this is a greater concern.

Many hydrological measurements must be taken on disturbed samples, and in situ measurements are impossible for most parameters. For example, the only extant approach for fully evaluating unsaturated-zone characteristic curves consists of laboratory evaluations of core samples. The core sample will never have the same characteristics as the same soil resting undisturbed in the field; there will always be a systematic error associated with the measurement. The issue is whether the error is significant. Other common hydrological parameters suffer from this same potential systematic error; porosity, degree of anisotropy, and dispersivity, among others, are all measured primarily in the laboratory. This source of error is consistently ignored in the literature, largely because it cannot easily be evaluated: there are not usually independent methods for deriving the same information about the site to compare to the laboratory test. When field-scale nonintrusive geophysical measurements exist, such as ground-scanning radar, they are often very difficult to interpret.

2.1.3.2 Data Insufficiency

Site characterization for performance assessment requires collecting enough data to define, to the extent necessary for performance assessment, (1) the natural hydrogeologic environment at the facility, (2) the likely future climatic conditions at the site, and (3) the long-term behavior of both the natural and engineered systems under both current and future climatological conditions. This task is difficult even for engineered systems, such as concrete vaults, which are (relatively) spatially homogeneous, and which evolve in time in a relatively well understood manner. However, there is generally a large inherent spatial and temporal variability associated with natural systems which must be accounted for in data collection, data interpretation, and modeling. In fact, one of the crucial tasks in site characterization for performance assessment is to develop confidence that the spatial variability does not include a preferential path that would result in the facility violating the performance objectives.

Data sufficiency constraints can potentially prevent the development of adequate models for performance assessment. The amount of data that is sufficient to build an "adequate" model cannot be identified in general, or even defined unambiguously, for a specific site. A large proportion of professional judgment is involved in the evaluation of data sufficiency. Furthermore, model "adequacy" and data "sufficiency" are likely to be determined to some extent by political as well as technical considerations. In addition, we note that conservative parameter values may be used (to some extent) in place of detailed site characterization data. In all cases, the regulatory decision will have to be made under a condition of uncertainty, and data sufficiency relates to confidence in the regulatory decision. Consequently, conservative parameter values can be used to reduce uncertainty in the regulatory decision even if they do not promote a fundamental understanding of the site.

2.1.3.3 Scale Dependence of Measurements

One of the most important sources of parameter uncertainty is the inconsistency between the scale of parameter measurement and the model simulation scale. It is often assumed that information collected at one spatial scale can be "scaled up" to provide information of field-scale model parameters. Based on this assumption, we try to collect enough data at the laboratory and intermediate scale, and subsequently use this data to develop field-scale flow and transport parameters. This problem affects, to some extent, our ability to model the engineered barriers in waste disposal systems, but it is even more of an issue in modeling the response of the natural environment.

For instance, the vadose-zone flow analyses are highly nonlinear; this fact, coupled with the extreme and complex spatial variability that characterizes most vadose-zone flow problems, makes identifying approaches for "averaging" local measurements to obtain field-scale parameters a formidable obstacle. Of even more importance for low-level waste performance assessment is the scale dependence of dispersion, and the uncertainty that results in trying to identify an appropriate dispersivity for use in a performance assessment. Dispersion is one of the primary factors that can be used in reducing doses from ingestion of contaminated ground water, but is difficult to quantify owing to scale dependencies. Laboratory-scale dispersivities tend to be lower than field-scale dispersivities by orders of magnitude, but there is not an adequate way to quantify field-scale dispersivities without conducting field tests. Furthermore, measured dispersivities in the field depend greatly on the technique used to measure them.

2.1.4 Treatment and Reduction of Parameter Uncertainty

Parameter uncertainty is treated by propagating the uncertainty through the model calculations to identify the effects on uncertainty in model output. Extensive reviews of methods for propagating parameter uncertainty through models are given elsewhere [Doctor et al., 1988; Doctor, 1989; Maheras and Kotecki, 1990; Zimmerman et al., 1990]; therefore, we will not provide elaborate details here. Instead, we will focus on evaluating the approaches for low-level waste performance assessment. As with other types of performance assessment uncertainties, the information produced in the parameter uncertainty analysis will be compared against a fixed, deterministic performance objective.

The result of accounting for input parameter uncertainty, with the exception of bounding analysis, is a distribution of doses; therefore, the issues discussed in the previous section in relation to comparing a probabilistic answer with a deterministic regulation become relevant. As mentioned before, without any regulatory guidance, it is assumed that the fixed regulations cannot be exceeded. Therefore, the tail of the dose distribution obtained from accounting for parameter uncertainty must meet the deterministic regulations. An alternative approach, mentioned above, is to use some intermediate statistical measure of the dose distribution. This approach is comparable to the EPA's guidance that the basis for comparison between the deterministic Individual Protection Requirement (which is dose based) and Ground-Water Protection Requirement (which is concentration based) in 40 CFR Part 191 is the greater of the mean or median of the output variable distribution [EPA, 1985a].

To represent the effect of input parameter uncertainty on modeling results, the modeler must first quantify, then propagate the parameter uncertainty through the model to obtain model results. This may be accomplished in one of several ways. One approach is to conduct "bounding" analyses, in which a conservative set of parameter values is used to produce conservative dose estimates. The most common approach for treating parameter uncertainty is Monte Carlo analysis, which consists of selecting discrete sets of input parameter values from probability distribution functions of the input variables, running each set through the model, and constructing an output probability distribution function that quantifies the uncertainty associated with the input. Another approach is perturbation analysis (also called analytical stochastic modeling). This approach is similar to Monte Carlo analysis, where distributions in input parameters are used to estimate distributions in output parameters. However, based on simplifying assumptions, the model equations and solutions are derived with the probability distribution functions for the input and output parameters explicitly included. Other approaches, such as the use of fuzzy set theory, are not well developed in the risk assessment community and, therefore, will not be discussed in this report.

2.1.4.1 Bounding Analysis

The current methodology is based on using bounding parameter values. In part, this approach was taken because of the intended use of the methodology. The purpose of the methodology was for the NRC to conduct confirmatory analyses of a license applicant's evaluation [Starmer et al., 1988]. For this use, it may not always be necessary to conduct a full parameter uncertainty analysis, since the licensee should already have quantified the parameter uncertainty and identified a conservative set of model parameters.

Although adequate for NRC's purpose in some cases, generally there are several disadvantages associated with bounding analysis. First, to have confidence that a correct set of conservative input parameters is chosen, it must be compared with other likely sets of parameter values. In treating parameter uncertainty by a bounding analysis, it is assumed that the analyst can select the conservative combination of parameters a priori. In most cases, particularly for nonlinear models, this prior identification of the bounding parameters cannot be done. Therefore, one would have to go through some analysis similar to Monte Carlo to estimate bounding parameters, and the advantage of bounding analysis (simplicity) is lost.

A second drawback to bounding analysis is that using only a single realization of parameters reduces the amount of information available to the analyst and the decision maker. This can be illustrated by considering calculated dose distributions from two hypothetical sites, as shown in Figure 2.1. A bounding analysis would suggest that the two sites are similar: the standard is violated, and the maximum doses are comparable. However, there is clearly a distinction between the two cases. For Site A, there is a much higher probability that the standard has been violated, which suggests that many sets of possible parameters produce a violation. In contrast, fewer sets of parameters produce the violation at Site B. This suggests that more site characterization may be in order to attempt to narrow the input parameter distributions. Further site characterization is less likely to produce improvement in the analysis of Site A.

The goal of performance assessment should be to provide as much necessary information to the decision maker as possible. A model prediction should be provided along with an estimate of the associated uncertainty in order to maximize the information available to the decision maker [IAEA, 1989]. Furthermore, "no method based solely on point estimates provides the decision maker with all the available information on the nature and extent of uncertainty, nor does it give decision makers or other analysts a window into the process to identify and criticize the assumptions made therein" [Finkel, 1990]. Providing enough information so that it is easy to identify the modeling assumptions and associated uncertainties is important, since the ultimate goal is public acceptance. Not only is it important to present the decision maker and the public with as much information as possible, but this information is also needed to guide data collection and validation efforts.

The above discussion suggests that while the use of bounding parameter values may be appropriate in some simple situations, in many practical cases the bounding parameter values cannot be specified a priori. Furthermore, the use of a bounding analysis limits the amount of information available to the analyst and decision maker. We, therefore, conclude that bounding analysis is not the best available method for parameter uncertainty analysis in low-level waste performance assessment.

2.1.4.2 Monte Carlo Analysis

As discussed by Zimmerman et al. [1990], of the available techniques for parameter uncertainty analysis, Monte Carlo analysis is the most versatile because (1) it facilitates consistent propagation of uncertainties, (2) it can be easily applied to a series of linked models, such as are used in low-level waste performance assessment [Kozak et al., 1990a], (3) it does not require modifications to the original models; therefore, it is generally straightforward to use, (4) it is capable of dealing with large uncertainties in the input variables, since it allows full stratification over the variable ranges, and (5) it is appropriate for use with nonlinear models [Helton et al., 1991]. The primary advantage to conducting Monte Carlo analysis is that it provides model results from a large number of likely input parameter sets. Therefore, the output uncertainty is acknowledged, and there is some means for identifying whether the output uncertainty from input

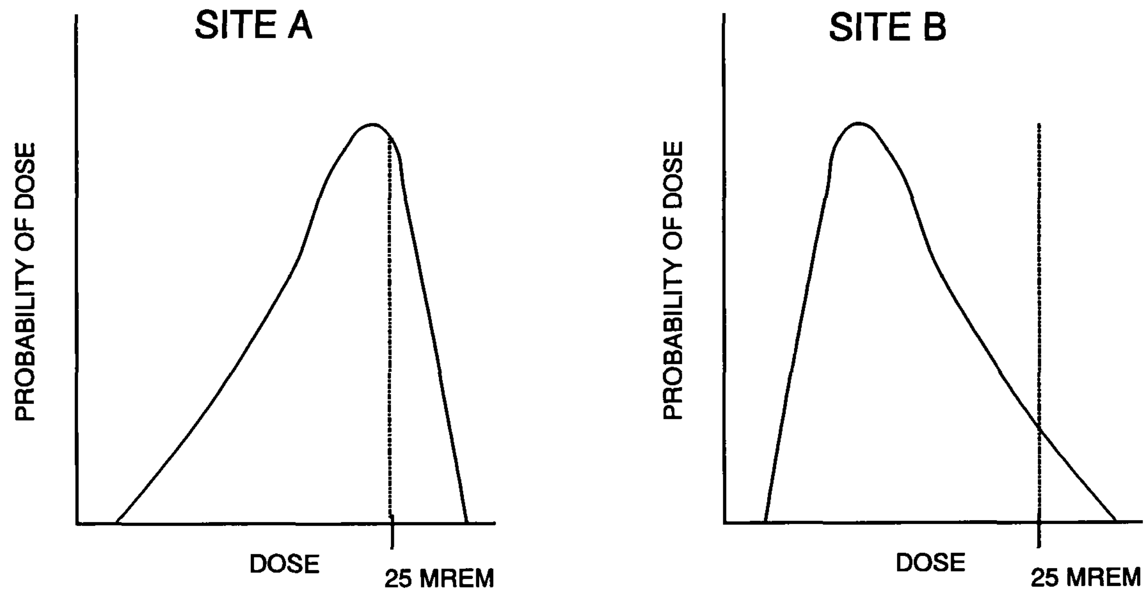

Figure 2.1 Comparison between two hypothetical dose distributions

parameter uncertainty has been bounded. Clearly, we have more confidence that the output has been bounded with increasing numbers of samples. Another advantage to this approach is that sensitivities of the model output to input parameter variations may be identified [Zimmerman et al., 1990]; this allows the analyst to identify important model parameters for future data collection efforts.

The primary disadvantages usually cited for Monte Carlo analysis are that many realizations of the data are required to span the input data range, and the parameters must be treated as uncorrelated [Harr, 1987]. However, both of these problems have been addressed to some extent. The required number of realizations can be greatly reduced by using a stratified sampling strategy, such as the Latin Hypercube Sampling method [Iman et al., 1981]. Methods are also available that allow the analyst to introduce correlations among variables [Iman and Conover, 1982], and these methods are included in the computer implementation of Latin Hypercube Sampling [Iman et al.,1981]. Even with these techniques, Monte Carlo analysis can still require extensive computer effort.

2.1.4.3 Analytical Perturbation Methods

Another approach that has been proposed involves using perturbation analyses (often called analytical stochastic methods) for ground-water flow and transport calculations [Polmann et al., 1988]. The primary reasons for using perturbation analyses are (1) to develop a computationally efficient approach, and (2) to attempt to develop improved conceptual understanding of ground-water processes by obtaining closed-form solutions to the governing equations. For instance, it is well known that the dispersivities obtained from laboratory column studies may differ substantially (by orders of magnitude) from those observed in field studies. Analytical perturbation models have been developed that can in principle, be used to predict field-scale dispersivities from local (i.e., laboratory scale) measurements [Gelhar and Axness, 1983; Dagan, 1984; Naff et al., 1988]. Other studies [Vomvoris and Gelhar, 1990] provide information on the expected variance of concentration at a given sampling point. Yet other examples relate to predictions of moisture-dependent hydraulic conductivity anisotropy in unsaturated heterogeneous media [Yeh et al., 1985].

2.1.4.4 Recommended Approach for Parameter Uncertainty

We recommend that parameter uncertainty analysis should be addressed using Monte Carlo analysis coupled with Latin Hypercube Sampling. This approach is used extensively, and has been recommended for high-level waste performance assessment [Davis et al., 1990b]. Use of this approach will provide the decision maker with considerably more information relative to bounding analysis, and more importantly, it clearly acknowledges and communicates the uncertainty associated with the model output variable due to input parameter uncertainty. It also provides some basis for assessing whether or not the model output distribution has been bounded. However, as discussed in the previous section, determining whether the complete dose history output distribution must fall below the regulatory performance measure, as opposed to some statistical measure of the distribution (e.g., mean, median, 95% confidence limit) is entirely a regulatory decision.

The only available method to reduce uncertainties associated with data insufficiency is the collection of additional data about a parameter. However, since most of the parameters of interest are potentially spatially and temporally variable, complete elimination of this source of uncertainty is impossible.

There is not a generally appropriate way to eliminate systematic errors from hydrological parameters. The best way to verify that systematic errors are small is to attempt a validation of the overall analysis. However, this approach is hindered by (usually) ambiguous validation results, by spatial variability of the parameter, and by the unavoidable convolution of potential systematic parameter errors with possible model errors. When multiple measurement approaches are available for a parameter, as many as possible should be used, and this will provide some confidence that systematic errors are insignificant. However, these types of potential errors cannot be easily quantified or identified, and the analyst should recognize this additional uncertainty in parameter values and distributions.

2.1.5 Incorporating Uncertainty Analysis into the Methodology

We have identified the need for regulatory guidance in the interpretation of the performance objectives in 10 CFR Part 61.41, particularly for incorporating future uncertainties in a comprehensive manner. Specifically, there needs to be a regulatory determination about whether the dose limits are absolute bounds that cannot be exceeded, or if some measure of the statistical distribution of doses can be used for comparison with the deterministic standard. The choice of uncertainty analysis methods to be incorporated into the methodology depends on this regulatory decision. Consequently, full implementation of uncertainty analysis methods into the low-level waste performance assessment methodology cannot commence until this regulatory guidance is available.

If NRC chooses to interpret the performance objectives as relating to a statistical measure of the dose distribution, the next step would be to identify that measure of the distribution to use. As mentioned in the previous section, one option is to use the mean or the median, whichever is higher; this option was used by EPA for assessing compliance with the Individual Protection Requirement and the Ground-Water Protection Requirement in 40 CFR Part 191 [EPA, 1985a]. However, any statistically defined measures of the dose distribution, such as confidence limits or percentiles, can be used.

The choice of what confidence limit to use is a regulatory decision, but some guidance may be available based on pragmatic grounds. Analyses of ground-water travel times performed at SNL for the Basalt Waste Isolation Program suggested that 85% confidence limits were relatively robust with respect to the number of parameter sets used to develop the output distribution, but that 90% and higher confidence limits required large numbers of parameter sets to demonstrate stability of the confidence limit value.[1] Similarly, Berthouex and Hau [1991] have shown that using high percentiles can lead to high risks of making erroneous decisions in both parametric and nonparametric analyses.

In the previous sections, we have summarized approaches for conducting uncertainty analysis for model uncertainty,

[1]Davis, P.A., personal communication, SNL, 1991.

parameter uncertainty, and uncertainty about the future of the site and recommended methods to be used for low-level waste performance assessment. These approaches are reviewed in Table 2.1.

Conceptual model uncertainty is addressed by identifying a broad range of possible conceptual models, and using each in performance assessments. Revisions of these models are made by accounting for progressive data collected, which can be used to eliminate some models from consideration. Uncertainties about the future of the site are addressed by acknowledging alternative future conditions of the site. Parameter uncertainty is addressed by using Monte Carlo analysis. Latin Hypercube Sampling of the input parameter space is recommended to reduce the computational effort associated with the analysis.

2.1.5.1 Implementation

In the following discussion, we assume that a scenario approach is adopted for treating future uncertainties. As discussed above, it is not clear that this is the best approach, nor is it clear that it will fit the regulatory philosophy that may be adopted by NRC. We use it here for simplicity of demonstration of the role of future uncertainties on the performance assessment.

The first step in implementing the treatment of uncertainty in the methodology is to formulate an initial suite of future scenarios, conceptual models, and associated parameter distributions for use in a preliminary performance assessment. These initial postulates should be consistent with available data, but should be as broad as possible within the constraints set by the data. An approach to broadening the interpretations is to get input from a large number of independent investigators. We might even invite the technical advisors of intervener organizations to participate in the preliminary data interpretation for conceptual model development. Such an approach would allow concerns of these organizations to be addressed during the performance assessment process; this approach may be very useful in developing an eventual political consensus about the results of the performance assessment. To achieve this political consensus, it is necessary to build consensus about the decision-making process beforehand. In other words, participation of intervener organizations only makes sense if they accept the technical approaches in the performance assessment process as the basis for the decision.

The overall approach to the uncertainty analysis is shown in Figure 2.2. A suite of future scenarios is identified by the analysts. For each scenario, several conceptual models may potentially be posed. To analyze the parameter uncertainty associated with each conceptual model, a number of realizations are necessary from the parameter distributions, which is depicted as the bottom tier of blocks in Figure 2.2. These realizations are the input realizations to Monte Carlo analyses of the parameter uncertainty associated with each conceptual model. Once these realizations have been propagated through the model, their results represent the total information available to make the regulatory decision.

The approach taken to use this information for comparison against the performance objectives is the root of the regulatory issue related to uncertainty analysis. For instance, NRC may choose to screen some or all alternative scenarios a priori using expert judgment and siting criteria. In this case, the analysis would be reduced to a few (or one) scenario branches with their associated conceptual model and parameter uncertainties. Alternatively, NRC may choose to evaluate all scenarios, then eliminate all or some of them a posteriori based on plausibility arguments, and make the regulatory decision on the most conservative of the remaining scenario(s). This approach was taken in the analysis of the Below-Ground Vault Prototype License Application (PLASAR) [Rogers and Associates, 1988], but not in this formal way. In the PLASAR, a "sensitivity" analysis was performed on an increased rainfall scenario, but the results were only reported in an appendix, and were not used in the main body of the report. This approach corresponds to a posterior screening of the analysis results. The advantage to this approach is that the full set of information is made available to the decision maker, who can therefore not be criticized for omitting some conditions. A third alternative is to combine all scenarios in a single probability distribution by assigning likelihoods to each scenario. The advantage to this approach is that it combines all information into a single curve from which a decision can be made. Disadvantages to this approach have been reviewed by Andersson et al. [1989], in the high-level waste context; these difficulties include problems in ensuring the scenarios are mutually exclusive and independent (which are requirements for combining them into a comprehensive distribution), and time dependence and ordering issues. In addition, as discussed above, the likelihoods of the scenarios are often related more closely to their duration than to their probability of occurrence, and this approach is not appropriate for a dose-based standard. A fourth alternative is to use the approach of Chhibber et al. [1991a], and combine scenarios and conceptual models by associating a subjective likelihood with each. This approach has the same difficulties and

Table 2.1 Summary of approaches to uncertainty treatment

Type of Uncertainty	Proposed Approach
Model uncertainty	1. Identify multiple conceptual models 2. Use all models in performance assessment 3. Use performance assessment results to identify if further data collection is necessary 4. Use the most conservative model for comparison with the performance objectives
Uncertainty about the future	Dependent on regulatory decisions
Parameter uncertainty	Monte Carlo analysis with Latin Hypercube Sampling

disadvantages as the third alternative approach discussed above, but in this case the problems are much greater in assuring that conceptual models are complete, mutually exclusive, and independent. These four alternatives are summarized in Figure 2.3.

We emphasize that, conceptually, the overall group of realizations shown in Figure 2.2 is the full amount of information that the decision maker must deal with, either explicitly or implicitly. Some scenarios, such as erosion, may be potentially eliminated a priori through siting criteria. Others, such as climate change, may be potentially incorporated into a base scenario with wider parameter value distributions (such as a broader infiltration distribution). Still others may be eliminated by the regulator, such as a decision not to include glaciation. The decision left to the NRC is whether it is appropriate to eliminate scenarios using expert judgment (and if so, which ones). If certain scenarios are to be eliminated, then regulatory guidance will be required to identify that conditions are to be retained for analysis.

2.1.5.2 Identification of Input Parameter Values

Once the initial set of conceptual models and parameter distributions has been identified, each of the models should be run through an initial performance assessment, accounting for parameter uncertainty by Monte Carlo analysis. From a mechanical standpoint, parameter uncertainty represents the bulk of the difficulty of implementing uncertainty analysis.

The analyst must identify probability distributions associated with each input variable. These input distributions can be of any form, but the form should be dictated by the information available about the variable. As discussed by Harr [1987], the amount of data available about an input distribution, together with the principle of maximum entropy, may be used to dictate the form of the distribution. Simply stated, the principle of maximum entropy identifies the least biased distribution to be the one that maximizes entropy subject to the available data. The distributions that fulfill this condition are shown in Table 2.2.

A second approach used to specify input distributions for sparse data sets involves using expert opinion. Probabilities derived by this approach are known as "subjectivist" or "personal opinion" probabilities [Berman, 1988]. It should be understood that when input parameters are specified in this manner, the output distribution also takes on the significance of a "personal opinion" probability [Vaurio, 1990]. Consequently, such an output distribution should not be interpreted as an actual frequency distribution of doses, but rather a quantification of existing expert opinion about doses. We note that all uncertainties about the site future and about conceptual models are treated by subjectivist approaches, and that the overall interpretation of the performance assessment results is (and must be) subjectivist because of the nature of the information available for the analysis [Atwood, 1988; Apostolakis, 1988]. Consequently, we should not be adverse to using subjectivist input parameter distributions, if required; however, their use should be minimized, since the amount of confidence we have in a distribution is directly related to the amount of data upon which it is based.

Once the continuous distributions have been specified, the analyst must sample from them to identify sets of parameters to be used as values in the models in the methodology. Ideally, enough realizations should be taken to completely span the range of input values. Practically, however, this ideal cannot be met. The primary disad-

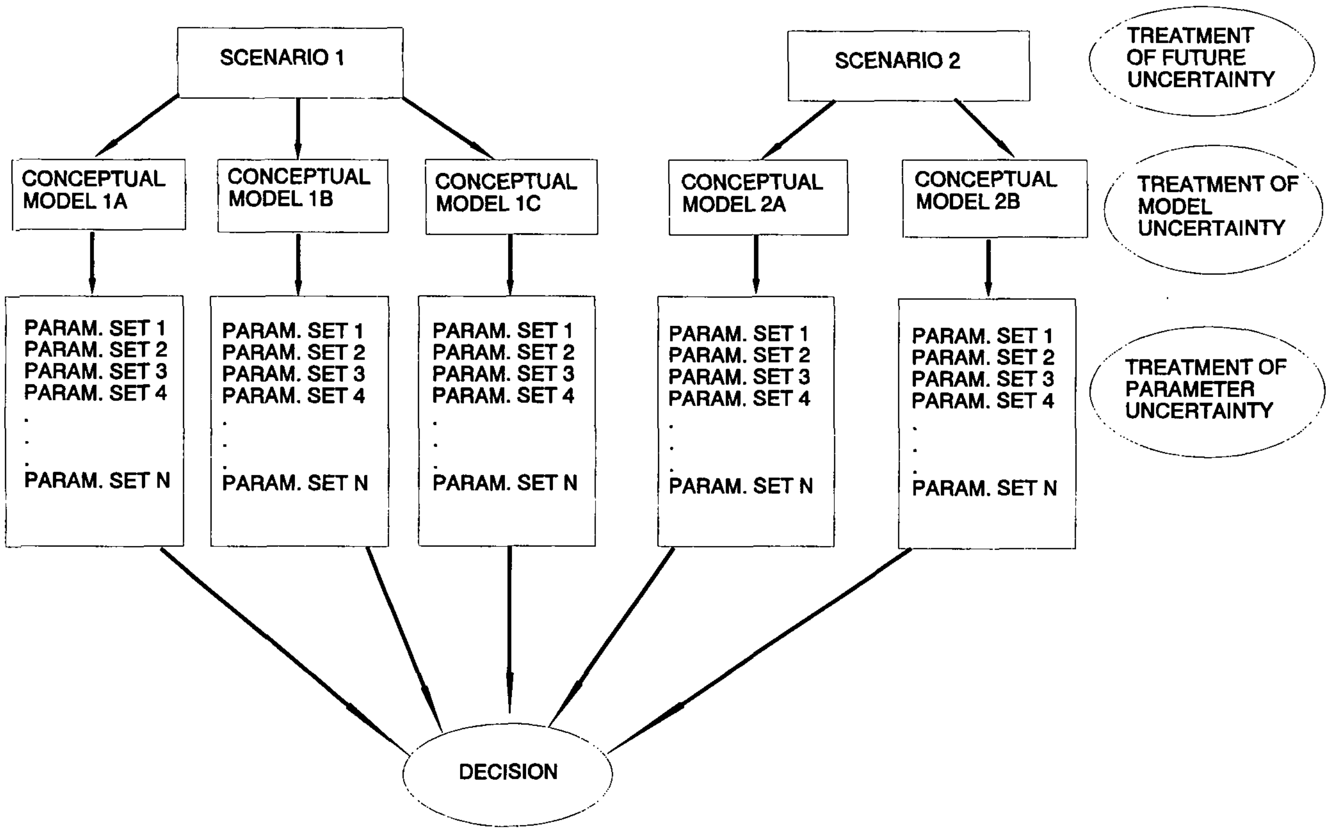

Figure 2.2 Overall approach to uncertainty analysis for low-level waste performance assessment

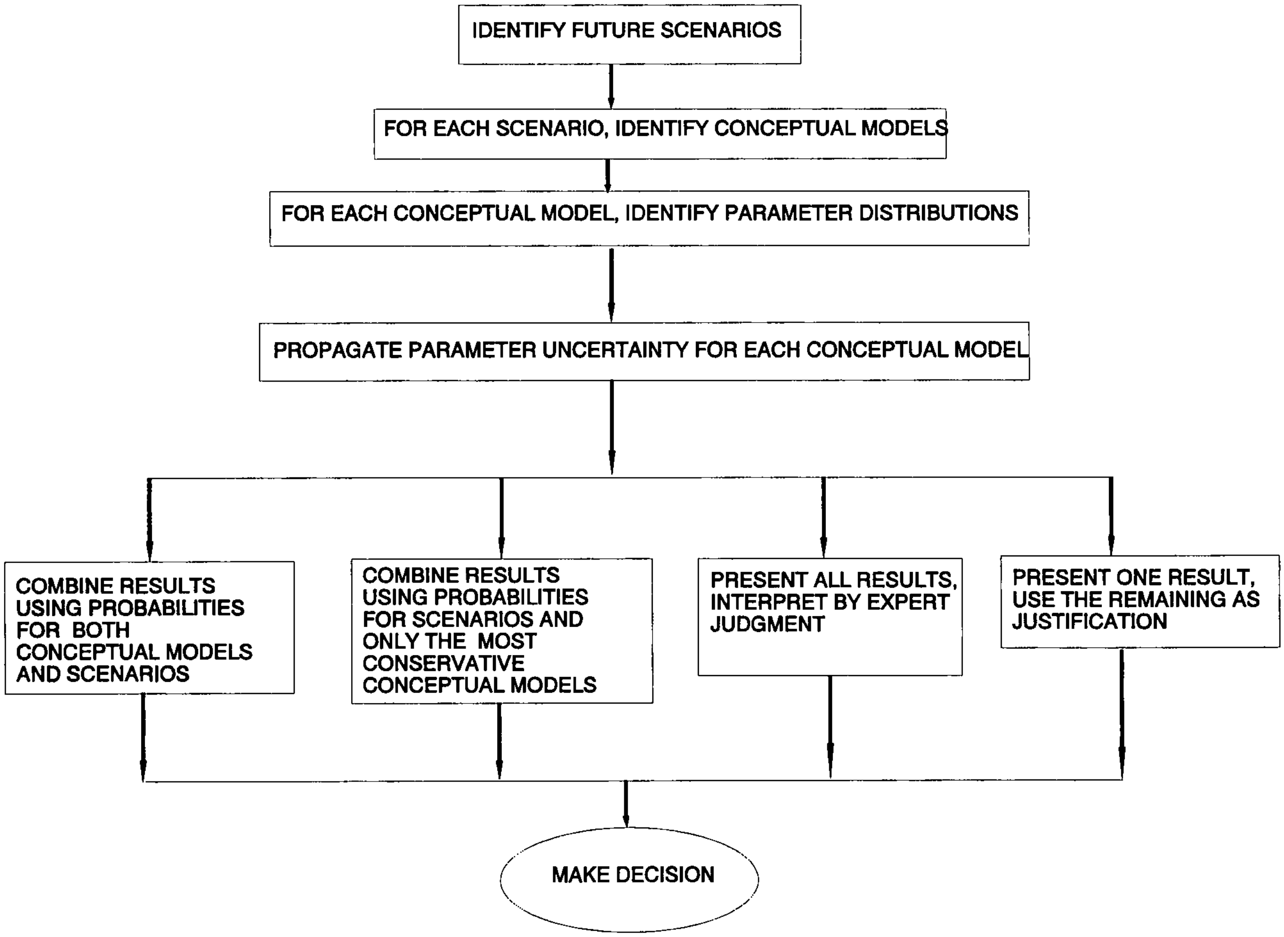

Figure 2.3 Optional approaches for combining information to make the regulatory decision

Table 2.2 Least biased distributions for varying amounts of available information [adapted from Harr, 1987]

Available Data	Assigned Distribution
Minimum and Maximum Values	Uniform
Expected Value	Exponential
Expected Value, Variance	Normal
Expected Value, Variance, Maximum and Minimum Value	Beta

vantages cited for Monte Carlo analysis are that many realizations of the data are required to span the input data range, and that the parameters must be treated as uncorrelated [Harr, 1987]. Even with Latin Hypercube Sampling, many realizations are needed to establish the tails of the output distribution with confidence.

In the ideal situation, this initial performance assessment analysis may comprise a sufficient basis from which to make a regulatory decision. The initial scenarios, conceptual models, and parameter ranges presumably include quite conservative conditions, owing to the inclusion of multiple conceptual models and broad data ranges. Consequently, if the performance measures are not exceeded, the regulator can make a relatively confident decision. However, some site-specific validation will be needed even in this ideal case to ensure that the initial concepts about the site are conservative.

A more likely situation will arise when this initial performance assessment does not meet the performance objectives for some of the more conservative conceptual models or parameter set realizations. The task of the analyst is then to gather more information about the site to support or refute the conditions of concern. An appropriate approach to complete this task is discussed in Section 2.2.1 below.

2.1.5.3 Propagating Input Parameter Uncertainty Through the Model

Conducting parameter uncertainty analyses for each conceptual model, and for multiple scenarios, can potentially consume large amounts of computer time. Of particular concern is the application Monte Carlo analysis to vadose-zone flow modeling, since the vadose-zone flow equations are strongly nonlinear. In the methodology, two-dimensional vadose-zone flow modeling is used to estimate flow into and through the disposal units. Current disposal technologies include extensive use of concrete structures, and multilayer "capillary break" covers; consequently, the flow analyses tend to be characterized by neighboring soil layers with sharply contrasting properties. These problems typically require many iterations before they are solved, and consume large amounts of time for each realization. Furthermore, it is common to be unable to achieve numerical convergence for some combinations of parameters.

These issues identify a practical obstacle to using Monte Carlo analysis for vadose-zone flow modeling; the analysis can still be done by brute force, simply grinding through many realizations of a full multidimensional flow analysis; however, this approach may require such large amounts of computer time that it becomes impractical. In such cases, a simpler flow model can be used. For instance, the multidimensional flow model may be replaced by a simple water flux through the waste that is allowed to vary over a wide enough range to represent the uncertainties. The primary disadvantage to this approach is the difficulty in setting "reasonable" limits on the uncertainty in flow rate: the distribution will generally have to be broader than if the full-flow analysis can be done. For instance, an arguably conservative flow rate through the vault might be the undisturbed infiltration rate, and this could be used as one bound on the flow rate distribution. If this is determined to be excessively conservative, the analyst might then be justified in modeling the disposal facility in more detail, and expending the extra effort to conduct the full-flow analysis.

Monte Carlo analyses are becoming increasingly more attractive, since computing costs are currently dropping dramatically, and computing speeds are increasing equally dramatically. Parallel computing is becoming available for workstations, and even for personal computers, which will further improve computing speeds. Consequently, although these problems currently exist, the concerns about Monte Carlo analysis for vadose-zone flow are diminishing.

The remaining models in the performance assessment methodology will be much easier to use in Monte Carlo analyses, since they are linear and, consequently, do not require time-consuming, iterative solutions. Groundwater transport analyses are linear, assuming that only linear sorption terms are included in the retardation term; food-chain, dosimetry, surface-water transport, and air transport models are all simple and linear. However, transport analyses of decay chains, even though they are linear, are more time consuming than single-component analyses because of the coupling of the transport equations. Nevertheless, analysis of decay chain transport is usually very fast compared to the vadose-zone flow analysis, and should not constitute a major obstacle to implementation of Monte Carlo analysis.

The Monte Carlo analysis should be integrated into the methodology in a user-friendly manner. In the current structure of the methodology, the best approach would be to implement a sampling program separately, as an addition to the suite of codes in the methodology. A stand-alone code is available to implement Latin Hypercube or random sampling [Iman et al., 1981]. This code would be run first to identify the parameter values needed in the other codes in the methodology. The performance assessment would then be run for each set of parameters. This process would have to be repeated for all conceptual models and future scenarios. Other options for implementing the uncertainty analysis are discussed in the following section.

2.2 Reduction of Uncertainty

The primary purpose of low-level waste performance assessment is to develop a confident regulatory decision about compliance of the disposal facility with the performance objectives in 10 CFR Part 61.41. Regulatory confidence should be understood to be different than scientific confidence in the context of evaluating performance assessment results. There may be significant uncertainty predicting in an absolute sense what the maximum dose will be in the future: this is scientific uncertainty. Regulatory confidence is concerned with information that may influence the regulatory decision. Scientific uncertainty may be irrelevant in making a regulatory decision, provided that there is a confident upper bound to the doses, and that regulatory requirements are met. In such a case, there is adequate confidence in the regulatory decision, even if there is large scientific uncertainty about the absolute predicted dose. Therefore, the performance assessment objective is to strive for minimal likelihood of accepting a site that will not meet the performance objectives. In this and subsequent sections, a process is described that meets this objective in an efficient and defensible manner.

The process of performance assessment has several salient features. (1) It should be iterative. During the first iteration, the analyst begins with available information and uses models that are conservative relative to that information. Subsequent iterations serve to relax conservative assumptions based on new information, such as new site characterization data, new design features, or a re-evaluation of existing information. As a side benefit, this approach minimizes the resources needed to perform site characterization. More importantly, it provides regulatory defensibility where and to the extent it is needed. The iterative nature of the process lends itself to defining endpoints. That is, performance assessment and site characterization should only be conducted to the extent needed to meet the performance objectives. Once adequate confidence is generated, the process may end. On the other hand, the process may end if the site developer finds that it is too expensive to collect enough data to produce adequate confidence. In either case, there is a finite and well-defined endpoint to the site characterization and performance assessment. (2) It provides a mechanism for using both generic and site-specific information in a defensible manner. (3) It integrates aspects of the facility development that have heretofore been considered separate from performance assessment. These aspects are site selection, facility design, site characterization, "ability to model the site," and "validation," which we take to mean development of regulatory confidence. (4) Unlike past guidance, it is a formal treatment for uncertainty analysis and is an intrinsic part of the process. Performance assessment is viewed as a method for quantifying the uncertainty associated with meeting the performance objectives, and for reducing that uncertainty to an acceptable level. The reduction of uncertainty relative to the performance objectives is an intrinsic part of the process.

The process also helps ensure that the performance assessment is substantially complete and defensible when the license application is submitted. Emphasis is placed on considering a broad range of potentially adverse conditions, then screening those conditions using subsequent site investigations. Confidence in the results is intrinsically built into the process, since each modification and the reasons for it are well documented. It is highly desirable to make the process participatory, in which case improved political acceptability may be generated as part of the process. Participatory performance assessment is also expected to lead to improved technical breadth and defensibility.

2.2.1 The Process of Performance Assessment

Performance assessment is a site characterization and modeling activity directed toward evaluating compliance of a disposal facility with the performance objectives in 10 CFR Part 61.41. In the following discussion, it should be understood that the approach to site characterization described here is directed to this end. Site characterization undertaken for monitoring or other activities is outside the scope of this discussion.

The overall process of performance assessment is shown in Figure 2.4. In this discussion, a completely participatory process has been assumed. A participatory process is defined as one in which all interested parties (stakeholders) participate in each step of the process as equal partners in developing and refuting conceptual models. In the absence of this type of approach, the process is still recommended. However, differences between participatory and nonparticipatory approaches arise at the

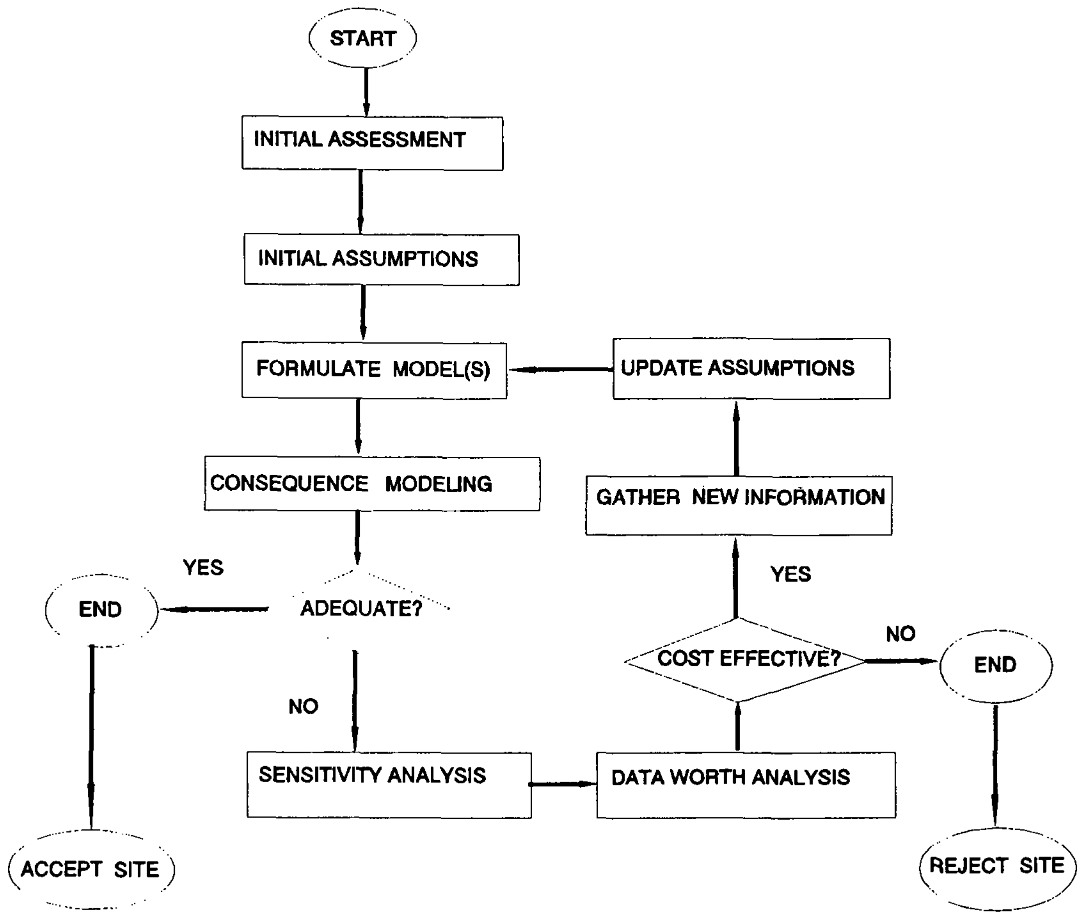

Figure 2.4 General approach to performance assessment

decision points. These will be discussed at the appropriate points in the following section.

Initial Assessment: Performance assessment begins with an initial assessment of available information about the site, and is directed toward developing an evaluation of the existing knowledge of the site and facility. It is important to note that there is at least some information available about all parts of the U.S. For instance, geological maps, regional hydrology, and weather data are generally available. Therefore, some minimal information will be available even in the worst cases. In many cases, more detailed information will also be available. This step is intended to represent a data collection activity, and an assessment of which aspects of the data are sparse. For sparse data, conservative assumptions are appropriate.

Initial Assumptions: The initially available information is next developed into assumptions used in specific conceptual models about the behavior of the system. The conceptual models should be as broad as possible within the constraints of the available information. This means that when the initial assessment shows that only sparse data are available, the initial conceptual models should include much more conservative conditions than if more information is available. The conservatism and breadth

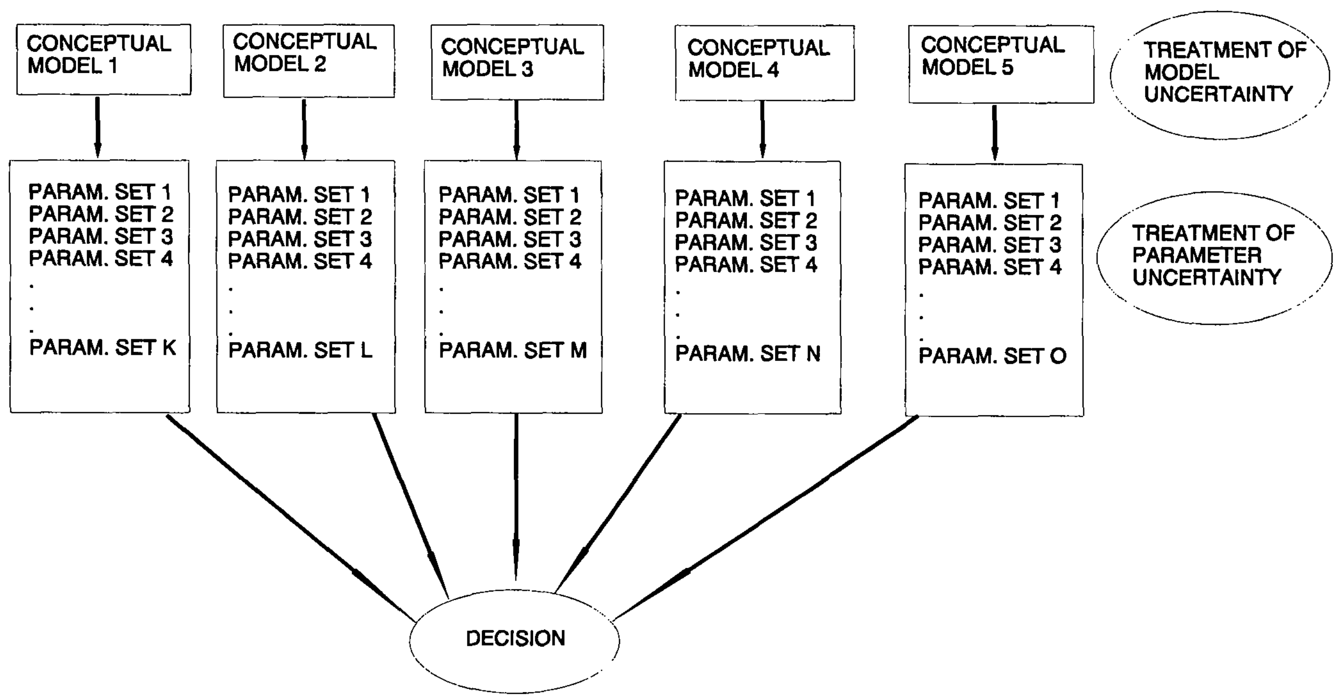

Figure 2.5 Uncertainty analysis for models and parameters

of conceptual models invoked at this stage reflects the level of initial uncertainty in the behavior of the system.

These uncertainties result in the potential for multiple conceptual models of the site, and for broad distributions in input parameters for the models. The structure of this uncertainty is depicted in Figure 2.5. In this case, five alternative conceptual models with differing parameterizations have been generated by the analysts. It should be noted that in some cases the division between different conceptual models and different parameter distributions may be artificial. The division between models and parameters is not always clear; however, the intent here is to illustrate differences between fundamentally different concepts of the behavior of the system, in contrast to different parameterizations. These conceptual models should be as conservative as possible within the constraints imposed by the understanding of the site.

Parameter distributions for each conceptual model should also be established at this point. As in the development of conceptual models, the goal at this stage is to retain the broadest ranges possible within the limits of available information. If no site-specific data are available for a particular parameter, the range can be set as the maximum physically possible, or the maximum ever measured for similar conditions. These extremes may be moderated as needed as the process progresses, and more information is collected. The important point here is that if a broad range of a parameter is used and it does not influence the regulatory decision, it does not need to be studied further. Also, as more data are collected, it may be possible to justify correlations among variables as needed. Such correlations may also depend on the models being implemented.

This stage in the process is where the participatory nature of performance assessment makes its first large impact. The breadth of conceptual models and parameters used in the analysis is directly related to the ultimate defensibility of the analysis. The breadth of analyses, in turn, is related to the number of analysts, and to the elimination of potential bias toward nonconservatism in developing the models. A license applicant, working independently at this stage, may have an inherent bias that precludes consideration of all possible adverse conditions in developing conceptual models. However, if the license applicant solicits conceptual models from regulators and other interested parties at this stage, there is much greater assurance that the range of potential conceptual models has been spanned. Also, by including analysts that tend to be more risk averse, there will be a greater tendency to capture more conservative models.

Also, it is important at this stage to emphasize the importance of retaining the breadth of possible conceptual models. It is not typically possible to identify the relative conservatism of conceptual models or parameter sets before the analysis is run. Therefore, conservatism among analyses must be identified after the analyses are performed.

A final consideration related to the eventual defensibility of the analyses is that independent quality assurance auditors should be able to trace all modeling results, thus demonstrating that they can be reproduced.

Formulate Models: At this stage of the analysis, the analysts formulate mathematical representations of their conceptual models. These mathematical expressions will usually be represented and solved in particular computer codes. However, the representation of conceptual models should never be constrained by the limitations of some computer code, simply because it is available or easy to use. Implementation of models should be done based on site-specific physical and chemical process considerations, not on code capabilities. At its worst case, this means that the analyst must develop a computer code for the express purpose of evaluating a particular conceptual model. However, it is expected that this level of effort will rarely be necessary, since a large number of computer codes exist, and these can be used to represent a broad range of potential conceptual models.

Consequence Modeling: The purpose of consequence modeling is to propagate the parameter uncertainty associated with each conceptual model through the mathematical models, in such a way that a distribution of calculated doses is produced for each conceptual model. One approach for propagating parameter uncertainty through the models is to use Monte Carlo analysis. However, this is not necessarily the only acceptable approach [Zimmerman et al., 1990].

Evaluation of Adequacy: The evaluation of adequacy is somewhat different, depending on whether a participatory or nonparticipatory performance assessment approach is used. If a participatory approach is used, the assessment of adequacy relates only to a simple comparison between the consequence analyses and the performance objectives. The reason for this simplicity is that any technical concerns by any interested parties have already been addressed as part of the performance assessment process. The decision to be made is between accepting the performance assessment if the performance objectives have been met, or driving further information collection if they have not.

By contrast, if a nonparticipatory approach is used, the performance assessment is presented to the regulator for a regulatory decision at the end of the process. The regulator must then review the initial conceptual models and parameter distributions for completeness and conservatism. The regulator must also evaluate the justifications for changing assumptions and moderating conservatism as part of the process. At any stage of the process if the justification is inadequate, the regulator may either require collection of more data, or may reject the performance assessment.

Sensitivity Analysis: Sensitivity analysis is performed on the consequence analysis results to evaluate which models and combinations of parameters were most significant in producing results in excess of the regulatory performance objectives. The primary functions of this step are to (1) identify data and assumptions that affect the regulatory decision for careful scrutiny, (2) optimize funds by specifying the most important information to be collected to reduce regulatory uncertainty, and (3) identify which assumptions and parameters *do not* influence the results. The first and third functions are important to all interested parties in the process; the second function is important to the site developer.

Sensitivity analysis is not a completely necessary step, because data could be collected randomly that would still allow the analyst to modify assumptions. However, such an approach is not conducive to optimal use of resources, nor does it build defensibility into the process.

Sensitivity analyses for identifying important parameters have been used in related high-level waste performance assessment problems [see, e.g., Bonano et al., 1989]. However, standard techniques have not yet been identified for evaluation of sensitivities relative to peak dose performance measures, nor for the evaluation of sensitivities to conceptual model (assumption) variations.

Data Worth Analysis and Evaluation of Cost Effectiveness: Data worth analysis has two functions in this process. Both functions are performed by the site developer. The first function is to identify which information can be produced at the least cost. Again, this function is related to optimal use of resources. The second function is more serious. If the data needed to eliminate a conceptual model or parameter range from consideration are very extensive, owing to site complexity or other factors, it may be more cost effective to reject the site and proceed with another site.

In a completely participatory performance assessment process, this is the only criterion that is used to reject a site. That is, the other participants can continue to recommend new iterations indefinitely, and it is only the site developer that decides to reject the site based on economic considerations. In a nonparticipatory process, this is one of two rejection points. The other point is if the license applicant, while working independently, has made unacceptable conclusions.

Gather New Information: Once sensitivity analysis has identified the critical information needed to reduce regulatory uncertainty, that information must be gathered. New information should not be gathered simply to understand the site without regard to reducing regulatory uncertainty. (The reader is reminded that this overall approach is directed toward the needs of performance assessment. The needs of monitoring or other technical areas must be addressed independently. In general, modeling to support a monitoring plan may need more extensive data. Such data may be incorporated into the performance assessment structure without difficulty, but are not necessarily required by performance assessment.) Information can be one of four types: new site characterization data, new facility design, adjunct modeling studies, or new design basis information. New site characterization data may be generated by, for instance, drilling a new well. A new facility design might influence, for example, how barrier degradation is modeled, or it might also permit geochemical modeling to be used to allow credit for reduced solubility limitations. New design basis information might consist of specifying inventory limitations to reduce calculated off-site doses in the subsequent iteration.

Any of these sources of new information can drive the subsequent consequence analysis iteration. Deciding that the previous iteration was "too conservative" without new justification is an inadequate reason for modifying assumptions.

Update Assumptions: The principles of this step are the same as the initial assumption step. In this case, however, assumptions are modified based on a larger knowledge base. Subsequent model formulation may involve elimination of a conceptual model, modification of a conceptual model, or addition of a new model introduced by new information. However, it should be noted that if the initial step included a broad range of conceptual models, the updated models should always trend toward less conservatism. They should still always be conservative relative to the information available when they are formulated.

2.2.2 Defensibility of Analyses

A common misconception about assessing low-level waste disposal facility performance is that performance assessment is conducted after the conclusion of site characterization, and when the facility design is completed. In this approach, site investigations are vaguely directed at attempting to "scientifically understand" the site, and there are no feedback loops to assess the progress of site investigations at resolving performance issues. The risks of using this approach are (1) data needed to establish defensibility in the performance assessment may not be collected, (2) too many data of some particular variety may be collected, resulting in wasted resources, (3) the investment of large amounts of resources prior to assessing safety may produce a large incentive (or at least the perception of an incentive) to demonstrate facility safety; that is, the process may become biased, and (4) data may be forced to fit the input needs on a preselected suite of models, while alternative data interpretations are not encouraged. Consequently, this approach is not conducive to building defensibility in performance assessments.

In contrast to this approach, the process described in Section 2.2.1 intrinsically builds confidence as subsequent iterations are performed. Defensibility is produced by using formal uncertainty analysis, which quantifies the effect of alternative conceptual models and parameter distributions on calculated doses. Confidence is developed since the range of potential adverse conditions has been identified, and progressively screened based on site-specific information. Those aspects of the model that lead to violation of the performance objectives can be identified, and site investigations can be performed to revise assumptions about those aspects. Confidence is produced by the visibility of this process, in that it should be made apparent how alternative assumptions have been refuted.

The process works best to build confidence if it is performed in a completely participatory manner. That is, all interested parties may propose potential adverse conditions that are evaluated as part of the performance assessment process. If those concerns are addressed as part of the process rather than in an adversarial manner after completion of performance assessment, two important issues may be addressed. (1) it is much more likely that the analysis will be technically complete and defensible, spanning a more complete range of potentially adverse conditions, and (2) if likely opponents have their concerns addressed during the process, there should be fewer adversarial hurdles to clear at the end.

If a participatory approach is not used, this process is still recommended for use by the license applicant. However, the role of the regulator must then be to evaluate in detail the span of models and parameters used in each iteration to ensure completeness and conservatism, and to evaluate the justification for modifying assumptions at each step of the process. That is, the license applicant should be prepared to identify which adverse conditions were considered, and how those conditions were addressed as part of the process. The risk of this approach, from the applicant's standpoint, is of working in a vacuum of other ideas, and possibly not considering some adverse issue, which is then introduced by regulators or interveners at the end of the process instead of at the beginning.

The performance assessment process discussed in Section 2.2.1 also builds confidence by combining site characterization activities with performance assessment modeling. Performance assessments developed in this way are assured to be site-specific modeling exercises, not generic models of dubious site-specific applicability.

The decision that the site meets the performance measures is the final step in the process, and is expected to be the result of a number of iterations undertaken to refute particular conservative conceptual models or parameter distributions. As part of that process, the initially conservative models have gone through successive screening and revision. The result is still a set of conservative models and parameters, but they are conservative relative to a new, broader set of site-specific information. In a participatory performance assessment, any decision to reject a site will be made by the license applicant, as part of a cost analysis that shows that confidence cannot be built for a particular site within reasonable cost bounds. One example of this might be a complex site in which an extensive site characterization would be required to refute a conceptual model. This is a practical definition of the requirement in 10 CFR Part 61 that the site must be "modelable."

2.3 Summary of Uncertainty Analysis Recommendations

The proposed treatment of uncertainty described in the preceding section is the most important and far-reaching recommendation in this report. Consequently, it is worthwhile to reiterate the most important points about the proposed treatment of uncertainty.

First, the methodology would benefit from a formal treatment of uncertainty. Such an analysis would (1) provide confidence that all important parameter ranges and models have been explored, (2) provide a wealth of information to decision makers about the results of the analysis, including the amount of uncertainty in the results, and (3) potentially promote a common ground upon which all interested parties can agree.

The uncertainty analysis should take the following form. Parameter uncertainty should be addressed by Monte Carlo analysis. For efficiency, Latin Hypercube Sampling should be used to reduce the number of realizations needed for the analysis. Conceptual model uncertainty should be addressed by including a broad set of conceptual models in the analysis. From the perspective of the low-level waste regulations, it is undesirable to pursue the idea of assigning probabilities to conceptual models. The decision should be made based on the most conservative of the conceptual models that cannot be eliminated using site-specific data. Conservatism here is defined after all models have been carried through the analysis. The most difficult issues in low-level waste performance assessment uncertainty analysis are related to future uncertainties. It was concluded that a scenario approach could be used in analyses for comparison with the performance objectives in 10 CFR Part 61.41, but only if a portion of the dose probability distribution is permitted to exceed the performance objectives. The scenario approach was shown to be more appropriate for an integrated discharge performance objective than for a dose-based performance objective. A possible alternative approach might be to perform time-dependent future histories of the site; this approach is expected to be both self-consistent and appropriate, but will make the analyses more complex.

Prior to full implementation of uncertainty analyses in low-level waste performance assessment, regulatory guidance is needed from NRC in areas that are more related to regulatory philosophy than to technical considerations. Guidance is needed on the following topics:

How to treat uncertainties about the future: First, there needs to be a determination about the required duration of the analyses. Second, guidance is needed about which extreme (but likely) events are to be included in the analysis. As the duration of the analysis increases, the need for the latter guidance becomes more important.

The conditions that should be compared to the performance objectives of 10 CFR Part 61.41: Specifically, guidance is needed about whether all calculated doses must fall below the performance objectives, or if some portions of the dose distribution may exceed the performance objectives.

2.4 User Friendliness

Methodology could be improved in the area of user friendliness. At the present time, a significant amount of skill in using codes is required to implement the methodology. In addition, without automation, using the methodology can be a time-consuming and tedious business. Consequently, it is recommended that automated linkages be developed between codes in the methodology. In spite of the need for automated links between codes, it is extremely desirable to maintain the flexibility of the methodology. This suggests that multiple links will be needed to retain all options available to the user. For instance, it may be necessary to develop code couplers between GENII and the ground-water transport codes: NEFTRAN, PAGAN, BLT, and VS2DT. A coupling to each of these codes will be needed to retain the current flexibility of the methodology. The necessity of flexibility dictates much more complex coupling between codes than is necessary for a single-site methodology, such as the high-level waste methodologies for basalt [Bonano et al., 1989], tuff [Gallegos, 1991], and salt [Cranwell et al., 1987].

At the present time, one typical overall performance assessment analysis is conducted as follows. VS2DT is run to evaluate the flow field in the disposal unit for intact and failed conditions. The VS2DT output file is then edited using a spreadsheet to delete all but the source region. This file contains velocity vector components and moisture contents, which can be combined to calculate the transport velocity in the waste region. This information can then be used in the PAGAN source term model. PAGAN is then used to calculate ground-water concentrations as a function of time. These concentrations are input to GENII for the exposure analysis of interest. The analyst must run GENII sequentially for a number of concentrations to produce the dose history. All of these linkages are straightforward (with practice), but it is clear that the methodology would greatly benefit by having them automated.

A potential solution to the user-friendliness problem would be to develop an integrated system for conducting low-level waste performance assessments. In the past, integrated system codes invariably restricted the analyst to a very narrow set of possible analyses. For the methodology, however, it is considered to be of overriding importance to retain flexibility in the possible approaches. In this way, a wide variety of conceptual models can be evaluated, and the methodology can continue to be used for a broad variety of conditions. To date, there are no existing codes with the requisite flexibility *and* user friendliness.

An approach that has the potential to be flexible and user friendly is currently being developed at SNL. The approach is to integrate multiple models, an uncertainty analysis package, and a geostatistical analysis package into a single package on a single computational platform. The current system has been developed to optimize the placement of monitoring wells for existing waste sites (Figure 2.6). The user interacts with the system through a graphical user interface that allows direct access to site data, which are stored in a geographical information system. The user can use the maps stored in the geographical information system to develop conceptual models of the site; the assumptions that are made in developing the conceptual model identify which of the possible process models are appropriate for the analysis. The system currently contains analytical models for vadose-zone flow and transport, MODFLOW and MODPATH, and an additional analytical solution for transport in the aquifer.

Extension of the existing monitoring well optimization package to low-level waste performance assessment is quite straightforward, and consists of adding more modeling capabilities. We are currently adapting this system to include the models in the low-level waste performance assessment methodology. In addition, we will expand the user interface to include a conceptual model manager, which will trace assumptions and ensure consistency between site data and conceptual models used in the analysis. This step will use the methodology to improve the credibility of modeling results.

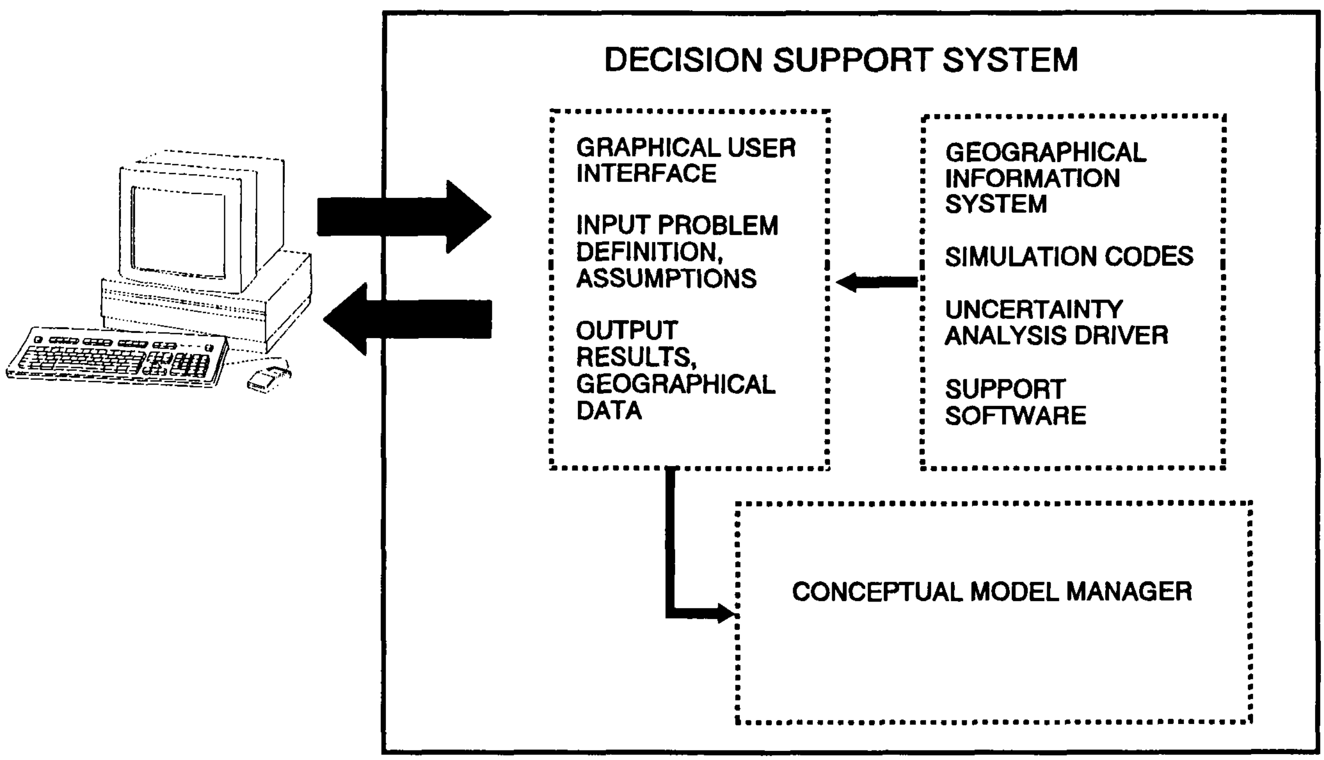

Figure 2.6 The decision support system structure

3. Pathway Assessment for Alternative Disposal Technologies

This chapter is an update of previously conducted pathway assessments [Shipers, 1989; Shipers and Harlan, 1989]. The purpose of the present pathway analysis is to determine if the current methodology can satisfactorily analyze the pathways and phenomena likely to be important for the full range of potential disposal options. If the methodology needs improvement in some modeling area, steps will be taken to update the methodology in that area. The result from this process will be assurance that there is adequate coverage of the types of modeling needed to conduct performance assessment analyses for a broad range of environmental conditions and types of disposal options.

We emphasize that the pathway assessment has been performed for a generic site. For specific sites, there may be fewer important pathways. We have attempted to be conservative in keeping pathways in the lists that will usually be of marginal importance. In this way we can build confidence that we have spanned the range of cases likely to be encountered at a real site.

Information about the various disposal options has been extracted from published reviews of alternative near-surface disposal options, including Bennett and Warriner [1986], Miller and Bennett [1986], Trevorrow and Schubert [1989], Rogers and Associates [1987], Denson et al. [1987], Denson et al. [1988], McAneny [1986], and Bennett [1985], among others.

3.1 Role of Pathway Analysis in Performance Assessment

Pathway analysis is used as a starting point to identify potential significant exposure pathways for the low-level waste disposal site under consideration, so that available resources can be concentrated on the most important aspects of the problem. Once the key pathways have been determined, models are assembled that can represent radionuclide transport from the disposal facility to the environment through these pathways. Assessing the performance of the site with respect to the relevant regulations is conducted based on the modeling results. Pathway analysis is qualitative and relies mainly on expert judgment. Efforts are made to err on the side of conservatism; pathways that may not be significant for a specific disposal site are kept for generality.

3.2 Previous Work on Pathway Analysis

Shipers [1989] presented an exhaustive list of pathways between low-level waste and humans, considering up to four intermediate transport media with no recycling between them. This analysis of pathways considered media in both environmental (non-living) media and biological media (the food chain). The result was an enormous number of combinations of the media that could physically occur (Shipers and Harlan [1989] cite the number to be over 8000). Many of these combinations result from the inclusion of the food chain in the pathway.

Shipers and Harlan [1989] determined the key pathways for shallow-land burial disposal from their list of all potential pathways. They defined the key pathways as those having both high likelihood and potentially high consequence. The performance assessment methodology [Kozak et al., 1990b] subsequently developed was a suite of mathematical models implemented in computer codes that can simulate the important processes in all the key pathways.

3.3 General Comments on Pathway Assessment

The current chapter expands the previous work by Shipers and Harlan to explicitly consider significant pathways for alternative low-level waste disposal methods, including belowground vault, tumulus, aboveground vault, shaft, and mine disposal. The following terms are defined:

Below-grade disposal: Emplacement of the waste below the natural surface grade.

Above-grade disposal: Emplacement of the waste above the natural surface grade.

Belowground disposal: Waste is covered with earthen materials at the time of closure.

Aboveground disposal: Vault remains as a freestanding structure after closure.

The primary purpose of this chapter is to evaluate whether the methodology is sufficiently general in its capability to model all pathways likely to be important in a performance assessment. In this study, we define the key pathways that the performance assessment methodology must be capable of analyzing as ones with high significance regardless of likelihood. The basis for this choice is that 10 CFR Part 61 has deterministic performance objectives. Performance assessments carried out for a comparison with these performance objectives are technical analyses of the significance or consequence of the pathway. The likelihood of the pathway is not a consideration once the modeling has begun. Shipers and Harlan defined the key pathways as those having both a high likelihood of occurring and a high significance if they do occur.

We have chosen to truncate the number of environmental media to be considered in series at three. Environmental media are defined as potential transport conduits between the source and man. Hence, we include pathways such as source → ground water → surface water → land plants, but not pathways with more environmental media. (Note that an additional implicit final link in the pathway is man.) There are two reasons for truncating the series at this point. First, in the list of important generic pathways generated by Shipers and Harlan [1989], the key pathways were found to consist of two environmental media or less. Thus, by investigating three pathways, we are being somewhat conservative. Second, various scoping analyses performed using the current methodology have corroborated the assumption of Shipers and Harlan [1989] that secondary media (such as surface water in the above example) are far less important than primary media because of dilution effects. Media beyond the secondary ones are proportionately less important. We, therefore, believe that very little information is lost by omitting pathways greater than three media. However, when assessing the pathways for a particular site, these generic guidelines may not hold, and further pathways may need to be assessed.

We begin the process of identifying key pathways with an inclusive list of all possible pathways, and then systematically screen the unimportant pathways from consideration. The rules for eliminating pathways are as follows. Pathways are screened when the release is considered unlikely or insignificant. For instance, if releases to surface water and soil are rated as unlikely for a given disposal method, then considering subsequent media along these paths would not be appropriate. Similarly, if the consequence of human inhalation of gaseous radionuclides is considered low, the effects in subsequent media are considered very low. However, for pathways that consider the passage of radionuclides from a medium in which their significance is low to another medium in which bioaccumulation can occur, this screening criterion does not apply.

The biointrusion pathways do not enter the three-media pathway lists because they all involve a transfer from the biota to an environmental medium (e.g., an animal dying or excreting into the environment) or between unlikely biotic transfers (e.g., land animals → aquatic animals can only happen by an animal dying in the water, which is clearly not a primary pathway). Such pathways will tend to be of very low significance compared to other, more direct pathways.

Keep in mind when considering the relative importance of the pathways that drinking contaminated well water overwhelmingly dominates the dose in the performance assessment. Other pathways are generally accepted to be less important. However, at a specific site, the pathways may still lead to doses that contribute significantly to the total dose. This distinction was acknowledged in ranking the significance of a pathway. The non-well pathways were ranked in importance with respect to each other.

The airborne pathway, which includes both release of radioactive gas by the waste and direct particulate entrainment of the waste, has been incorporated into all of the lists of pathways. This is contrary to the pathway lists given in Shipers and Harlan [1989], who considered the likelihood of airborne releases to be high, but the significance to be low. We concur with this assessment as it relates to gas production, since doses from such releases are generally accepted to be minor. Nevertheless, there is uncertainty about how to model gas production [Kozak et al., 1990a]; therefore, the significance of releases from low-level waste facilities may be somewhat larger than is anticipated. We have retained the air pathway for direct entrainment following exposure of the waste at the surface. The significance of this release is likely to be high, but for most disposal options it is extremely unlikely. We have also retained the air pathway since analysis of the air pathway is explicitly required in 10 CFR Part 61.

The analysis requirement is also explicit in the rule for the biotic intrusion pathways, hence these have also been retained in the lists, except for the deep disposal options, for which biointrusion will clearly not play a role. There are two aspects to the biotic intrusion pathway. The intrusion may be by wild animals, plants, or insects, or by domestic plants or animals. In the case of wild biota, the likelihood of intrusion is high (except for deep disposal) but the significance is low, since wild biota are usually not major dietary items. For domestic biota, the likelihood of intrusion is low (crops are usually shallow rooted, and domestic animals generally do not burrow), but the significance can be high. For generality, we have ranked some biotic intrusion pathways as high likelihood, high significance. However, when analyzing a specific disposal option at a particular site, we will usually be concerned about intrusion by either wild or domestic biota, but not both. Therefore, in specific cases, it is unlikely that both likelihood and significance will be high for biointrusion.

External exposures from penetrating radiation are not explicitly included in the pathway lists, but are intended to be implicitly included in the sense that the exposure route between the environment and man is not specified. Concentrations in, for example, ground water can potentially produce both external and internal exposures. Low-level waste inventories contain nuclides that emit penetrating radiation. If we choose as an example the inventory listed by Rogers and Associates [1988] and arbitrarily discard isotopes with half-lives less than 10 years, we are left with only a handful of gamma and X-ray emitters. Of these, only Ni-59 and Mo-93 have these as their dominant form of radiation. The rest are pure beta emitters, beta emitters with secondary gamma emission (e.g., Cl-36, I-129, Cs-137), and alpha emitters that have secondary gamma emissions (e.g., uranium isotopes and the transuranics). The importance of secondary emissions should not be downplayed since the ranking of emissions as primary or secondary is based on the probability of the type of emission, not on the average or peak energies of the emissions: from a radiological perspective, a secondary radiation may be of equal or greater importance than the primary radiation. For instance, I-129 produces more gamma energy than beta energy, although it is commonly described as a beta emitter. Consequently, although penetrating radiation is secondary for almost all long-lived radionuclides in low-level waste, these radiations are not necessarily negligible, and external exposures should be considered when appropriate in the pathway analysis.

3.4 Alternative Disposal Technologies

The experience with existing shallow-land burial facilities has raised questions on the effectiveness of the method. Remedial actions are underway at all three closed shallow-land burial sites [U.S. Congress, 1989]. On the other hand, there are many shallow-land burial sites around the world that are functioning satisfactorily [Kittel, 1989]. Furthermore, there have not been appreciable off-site doses received from any of the closed shallow-land burial sites in the U.S. [Matuszek, 1988]. The problems associated with the closed sites appear to be related more to siting problems than to any inherent flaw in shallow-land burial. Nevertheless, as a result of the operational problems at existing shallow-land burial sites, there has arisen a perception that the technology is flawed, which is not well established by experience. The result of this perception is that most States and Compacts are pursuing alternative disposal technologies.

The principle behind alternatives to shallow-land burial is that it should have a lower risk/hazard ratio than shallow-land burial [Trevorrow and Schubert, 1989]. That is, for a given hazard (defined in terms of the waste as disposed) the risk (which accounts for the probability and consequence of exposure) should be lower than for shallow-land burial. Proponents of these technologies generally ignore the additional uncertainties introduced when using untested disposal methods. It is conceivable, albeit unlikely, that the use of engineered enhancements could worsen off-site doses compared to shallow-land burial; for example, catastrophic failure of the roof of a vault, followed by massive subsidence, which could potentially funnel infiltrating water through the waste and produce relatively rapid release rates of the waste. The likelihood of this type of failure can be minimized through appropriate engineering designs, but this type of failure cannot be dismissed without technical justification.

Engineered barriers can influence the performance of the facility in several ways:

1. The barrier reduces the flux of water through the disposal unit for some time until the structure fails and significant releases begin. Low-level waste is made up almost entirely of short-lived (less than 30 years) and very long-lived (greater than 5,000 years) isotopes. The delay until failure of the barriers can allow sufficient time for the short-lived inventory to decay, which can greatly decrease the total impact of the eventual release. However, the barrier will have little effect on the significance of releases of long-lived species unless it inhibits their release rate in some way.

2. Engineered barriers can mitigate doses to inadvertent intruders. However, intruders receive adequate protection from the waste classification scheme in 10

CFR Part 61, and the engineered barriers do not significantly change the situation.

3. Extensive use of concrete and grout in a vault may significantly alter the chemistry of the pore water in the vault. For instance, concrete-stabilized waste has been shown to have pore water with a pH of 11 to 13, with greater uncertainty in the Eh [Dougherty and Columbo, 1985].

These conditions tend to lead to low solubilities and high sorption of some important chemical species [Atkinson and Nickerson, 1988; Atkinson and Marsh, 1988], which has a clearly beneficial influence on off-site impacts.

For the purposes of this discussion, we note that the effect of an engineered barrier may influence the choice of model for a particular phenomenon (e.g., leaching model), but will not usually change the pathways of release. The only influence of engineered containment on the pathway assessment is that the barriers may inhibit biotic intrusion. Also, keep in mind that two disposal options may have identical pathway assessments, but different models may be needed for them.

Rao et al. [1992a] considered belowground vaults, tumulus disposal, aboveground vaults, shaft disposal, and mine disposal. A *belowground vault* refers to placing waste in shallow subsurface concrete structures. Belowground vault disposal technologies essentially represent a natural evolution from shallow-land burial, and were developed to address concerns about the inherent lack of engineering control and barriers in shallow-land burial designs. Tumulus disposal refers to placing waste above grade under a mound of earth (tumulus). *Tumulus* disposal includes designs for disposal of waste entirely above grade and for combinations of above-grade and below-grade disposal. Disposal designs can include engineered enhancements (as in earth-mounded concrete bunkers) or disposal beneath a tumulus without additional enhancements (as in mill-tailing impoundments). In some cases, the specific design can influence the pathways of interest, since the presence of a concrete vault may lessen the likelihood of biointrusion or erosive direct exposure of the waste. An *aboveground vault* refers to a structure that relies on its own structural stability to provide isolation of the waste. This is the only commonly cited near-surface disposal option in which the waste is not covered by geological materials. Rogers and Associates [1987] concluded that aboveground vaults may have difficulty meeting the technical requirements for stability in 10 CFR Part 61. *Shaft disposal* facilities are holes bored into the earth, which have a large length to diameter ratio [Trevorrow and Schubert, 1989]. Shaft disposal has also been called the "augered hole" method since an auger rig is often used to construct the shafts. This method has been used by several different investigators, including the U.S. Department of Energy (DOE) [Dickman and Boland, 1983; Cook, 1984] and Atomic Energy of Canada, Ltd. (AECL) [Feraday, 1982]. The term *mine disposal* refers to underground mines, not surface (open-pit or open-cast) mines. The pathway analysis can be applied to other underground excavations as well.

Evaluations of the various disposal technologies in terms of operational, cost, and safety considerations generally rank the below-grade options (including shallow-land burial) and tumulus disposal about equally. Aboveground vaults generally are more poorly ranked [Rogers, 1989]. However, such conclusions should not be generalized, and rankings need to be performed for specific sites. Rogers and Associates [1987] concluded that earth-mounded bunkers, modular canisters, and belowground vaults provide only marginal performance improvement over shallow-land burial, at significantly larger costs and occupational exposures. They also concluded that aboveground vault technology requires additional time and analysis to ensure that all the requirements of 10 CFR Part 61 are met. Gershey et al. [1990] provide an informative summary of the positive and negative attributes of the alternative disposal technologies discussed here.

Some citizen groups and state agencies have shown a preference for above-grade placement of waste [Trevorrow and Schubert, 1989]. Several States and Compacts are actively pursuing above-grade technologies (e.g., Appalachian Compact, Central Interstate Compact). It has been suggested that above-grade facilities are preferred in regions with high infiltration rates, and that below-grade facilities are preferred for sites with low infiltration rates [U.S. Congress, 1989]. Above-grade facilities require considerably more land area than below-grade facilities; this adds an additional siting requirement. The ridge-trough topography of a closed above-grade facility may limit eventual unrestricted use leading to lower likelihood of intrusion [U.S. Congress, 1989]. On the other hand, the topography may provide incentive for an intruder to level the area, thus exposing waste.

Some have suggested that above-grade structures decrease the likelihood of inadvertent intrusion [Trevorrow and Schubert, 1989]; others suggest that above-grade facilities may attract curious potential intruders [Rogers, 1989; U.S. Congress, 1989]. In either case, the potential inadvertent intruder is assumed to receive adequate protection through the waste classification system of 10 CFR Part 61. If an exemption from the classification scheme is requested, an intruder analysis is required to demonstrate adequate protection to the intruder. In either event, the likelihood of the intrusion event is not in question, and the significance is expected to be relatively insensitive to the disposal system design. Therefore, although consideration of the probability of intrusion is important in choosing a disposal option, it is not relevant to the evaluation of potential pathways.

Concepts for disposal have been proposed in addition to the ones identified above, including modular-concrete-container disposal both aboveground and belowground [Rogers, 1989], improved shallow-land burial, and deep-trench burial [Trevorrow and Schubert, 1989]. We consider these options to be subsets of the technologies described in this report. For instance, aboveground-modular-canister disposal is equivalent (from a performance assessment perspective) to an aboveground vault with the waste grouted in place. Similarly, belowground-modular-canister disposal is similar to either shallow-land burial or belowground vault disposal with the waste grouted in place [Rogers and Associates, 1987]. The differences between the technologies are in operational and design considerations that do not affect the pathways or modeling requirements for postclosure performance assessment.

3.5 Role of Temporal Progression in Pathway Analysis

As the site evolves in time, the possibility exists for a number of initiating events to alter the characteristics of the facility enough that alternative exposure pathways become important. Since 10 CFR Part 61 has an open-ended performance period, it could be argued that low-probability initiating events should be included in the methodology. Whether such events are included or not is a regulatory issue that we will not address here. However, we anticipate that NRC will conduct performance assessments based on relatively minor perturbations about the current state of the site. Consequently, we may discard low-probability initiating events such as volcanic activity. Furthermore, we may discard processes such as landsliding and erosion of the natural grade since they are restricted by Subpart D of 10 CFR Part 61. We consider erosion of artificial grades, such as tumulus covers, to be a separate issue from erosion of the surrounding natural grade since it is not covered in Subpart D.

Remaining temporal evolution problems reduce to three primary ones: water table changes, subsidence, and vegetative progression. Water table changes may occur because of crop irrigation, other man-made disturbances to the local hydrology, or climate change. Again, we anticipate that NRC will only wish to analyze events that can be expected to occur at the site given the current conditions at the site. It, therefore, seems unreasonable to include extreme (and low-probability) changes in the hydrology. There is, of course, a somewhat arbitrary distinction between events that are anticipated to occur at the site, and events that are only somewhat likely at the site. The decision about the level of extremity to be included in the performance assessment must be made by professional judgment, since regulatory guidance on this issue is not included in 10 CFR Part 61.

Subsidence can be expected to occur to some degree at all disposal facilities and, over the short term, represents an aspect of the facility that will evolve in time. Early on, less subsidence can be expected for facilities with engineered support for the soil overburden. However, the principal effects of subsidence on the performance assessment are to change the flow field in the unit (e.g., distributed infiltration becoming more intense localized infiltration), and to alter the values and uncertainty in input parameters (infiltration rate). The presence and extent of subsidence will not affect the pathways that we need to analyze, except in the extreme case in which waste is directly exposed by massive subsidence. This extreme event can be made very unlikely by appropriate engineering design of the facility. Schultz et al. [1990] have introduced the idea of using a cover that is robust to the effects of subsidence during the period of active subsidence, and adding a multilayer cover following that time. This approach appears to be very promising for ensuring cover performance of shallow-land burial, for which the period of active subsidence corresponds to the operational and institutional control periods. However, subsidence problems associated with concrete structures may occur at much later times. For instance, the most important subsidence in the life cycle of a belowground vault may be following collapse of the roof; this can be expected to occur long after institutional control ends.

The effect of vegetative progression on low-level waste performance assessment has received relatively little attention to date. Vegetative progression can be expected to occur at all sites, to some degree, after the loss of institutional control. Progression may consist, for instance, of the displacement of grassy cover vegetation by deeper rooting trees or shrubs. A second example would be replacement of cover vegetation by crops once the site is opened to unlimited use. In either case, the progression leads to two effects. The first effect is that the likelihood of biotic intrusion may change, depending on the root depths before and after progression. The progression may be toward either deeper or shallower rooting plants, and hence to either a greater or lesser likelihood of biointrusion. The second effect is a change in the water budget. Progression toward trees, for instance, may lead to larger canopy holdup, which would alter the amount of water available for infiltration. Progression may tend to either increase or decrease infiltration.

Vegetative progression is also linked to climate change, since wetter or drier climates will tend to favor the incursion of new species into the area. The issue of climate change will not be addressed in this chapter; however, the issues associated with climate change, as well as other possible future evolutions of the site, are discussed below in the section on uncertainty analysis.

3.6 Summary of Pathway Assessment

Details of the results of the pathway assessments for specific disposal options may be found in Rao et al. [1992a]; we present only a brief summary here.

Given that a belowground vault facility is essentially a shallow-land burial facility augmented with engineered structures and barriers, the critical pathways for belowground vault disposal are similar to those identified by Shipers and Harlan [1989] for shallow-land burial. Some pathways can be eliminated for belowground vaults, since the engineered barriers tend to make certain pathways of much lower likelihood.

Two styles of tumulus disposal may be considered. The first style is a simple tumulus with neither additional concrete support nor overpacks in the design. The second style is an earth-mounded concrete bunker with a roof that supports the tumulus. Including such a roof temporarily reduces the potential for subsidence and erosion, and also reduces the likelihood of biointrusion.

Ground-water is the primary important release pathway for both belowground vaults and tumuli. Erosion is likely to be of little concern in belowground vault designs because of regulatory siting criteria; erosion of the tumulus can be minimized by appropriate engineering design. Incorporation of a thick rip-rap cobble layer in a cover would inhibit biointrusion from the surface and may discourage potential inadvertent human intruders. Lateral biointrusion would be impeded by the concrete wall in belowground vault designs. Gases generated by low-level waste decomposition could be easily transported through earthen covers, but appropriate engineering designs can be used to influence gas transport directions.

Aboveground vaults rely entirely on their concrete structure to isolate the waste from the surface environment. Given the current uncertainty about the longevity of disposal structures, it is usual in performance assessment analyses to assume the vault fails completely at some time during the performance period. For earth-mounded concrete bunkers and belowground vaults, such failure results in enhanced releases to ground water, but does not directly expose the waste at the surface. By contrast, for aboveground vaults, failure of the vault may mean direct exposure of the waste at the surface. The pathways reflect this contrast in increased importance of releases to soil, surface water, and air.

The ground-water pathway is the most significant pathway for shaft disposal. Biotic intrusion is not as important because of depth of disposal. The air pathway is likely to occur since gas will be generated by the decomposition of low-level waste, which then can be transported to the land surface by both diffusion and barometric pumping. However, the disposal depth increases the gas transport time, which can decrease the radioactivity of the gas by decay. The surface-water pathway is also not as important since direct exposure of the waste is improbable. Although an engineered cover such as concrete may not be present, the likelihood of human intrusion is small because of the small land surface area of the shaft.

According to McAneny [1986], at a mine disposal facility "ground water is the principal and perhaps only credible release path for radionuclides." This is because mines are often located in the saturated zone and ground-water intrusion is probable at some point in the life of the facility. Thus, this pathway has both a high likelihood of occurring as well as a large significance if the contaminated ground water is ingested.

In summary, disposal methods can be categorized in groups based on their depth of disposal. In deep disposal options (shaft and mine disposal), the key pathways are identical, and result entirely from releases to ground water. The shallow belowground disposal options (tumulus, shallow-land, and belowground vault disposal) also may be grouped together from a pathway analysis perspective. For these disposal options, the ground-water pathway remains the most important, but other pathways, such as biointrusion, can become more credible than for deep disposal options.

Aboveground vault disposal cannot be grouped with any of the other disposal options. In the absence of geological materials covering the disposal facility, the analyst cannot eliminate the potential for exposure of the waste at the ground surface. If the waste becomes exposed, many more pathways can become important that do not play a role in any of the below-ground disposal technologies. For instance, releases to surface water runoff should usually be considered for aboveground disposal, as should direct exposure. Other pathways, such as biointrusion, become much more significant than in below-ground disposal, since the waste can become directly accessible at the ground surface.

4. Ground-Water Flow and Transport Modeling

Ground-water hydrology is subdivided into two areas in the methodology. The first area is the evaluation of infiltration, and the second area is the analysis of the ground-water flow and transport processes. The distinction between these areas is shown in Figure 4.1, which depicts flow and transport processes in and near an intact near-surface disposal facility. Processes affecting infiltration are rainfall, evaporation, transpiration, and runoff. The resulting balance of these processes provides a boundary condition for the ground-water flow processes.

The water that percolates deeply will flow around or through the intact vault. Significant releases may occur from an intact vault if enough fractures or flaws exist in the concrete to substantially elevate its hydraulic conductivity, or if enough joints have failed to provide significant flow paths between concrete slabs.

More importantly, however, is the time in the future when the integrity of the waste containers and engineered barriers can no longer be relied upon. The primary difference between performance assessment analysis of intact and failed vaults is a change in the approach used for modeling ground-water flow through the waste. The flow processes near a failed vault are depicted in Figure 4.2; the vault may (or may not) retain structural stability, but it no longer provides an impediment to flow through the waste. The conditions usually of greatest interest in the performance assessment are those shown in Figure 4.2, since the peak dose is much more likely to be associated with failed conditions (non-design conditions) than with intact (design) conditions. Consequently, while the capability to model the intact vault is important, it is even more important to be able to model failed and partly failed conditions.

4.1 Infiltration Evaluation

Current Approaches

A specific method for evaluating infiltration was not included in the methodology; we concluded after a literature review that a sufficient general method was unavailable to evaluate the broad conditions for which the methodology must be capable of handling [Gee and Hillel, 1988; Balek, 1988]. Instead, we suggested that the licensee should support infiltration estimates with a combination of field data and site-specific modeling. A steady-state value of infiltration derived from these approaches is used in the methodology as a boundary condition for vadose-zone flow modeling. In general, use of a steady-state flux condition is much simpler than a transient condition for the long time periods of the performance assessment. However, the steady-state assumption has not been decisively shown to be conservative compared to episodic infiltration.

Status

Since the publication of the methodology, Pacific Northwest Laboratory (PNL) has developed the Infiltration Evaluation Methodology (IEM) [Smyth et al., 1990]. The basis of the IEM is that rainfall (in time) and soil properties (in space) may be treated as though they are stochastically distributed, and the IEM has been developed in an attempt to account for these variabilities. In its original form, the IEM was an "approach;" we recognized that "because of the site-specific nature of estimating drainage, the IEM does not recommend specific models to be used" [Smyth et al., 1990]. This approach is, therefore, in concurrence with the recommendations of Kozak et al. [1989a], that are incorporated in the methodology. However, work at PNL is currently in progress to recommend a group of models that will provide somewhat general infiltration evaluation capabilities.

Evaluation

The primary barrier to improved infiltration modeling is the large uncertainty in data and parameters used in validation of the models [Olague et al., 1993]. To date, there is not a consensus about the conditions for which particular models are appropriate [Balek, 1988; Knutsson, 1988]. Olague et al. [1993], recommend combining different types of experimental data to identify the conditions for which alternative approaches are appropriate.

Specific modeling approaches for infiltration are needed in the methodology. Currently, the experimental basis for choosing one approach over another is not available. Consequently, we recommend that a comprehensive comparative analysis should be conducted among the models, as is discussed by Olague et al. [1993].

4.2 Ground-Water Flow and Transport

Current Approaches

The current methodology contains the computer codes PAGAN [Kozak et al., 1990a; Chu et al., 1991] and

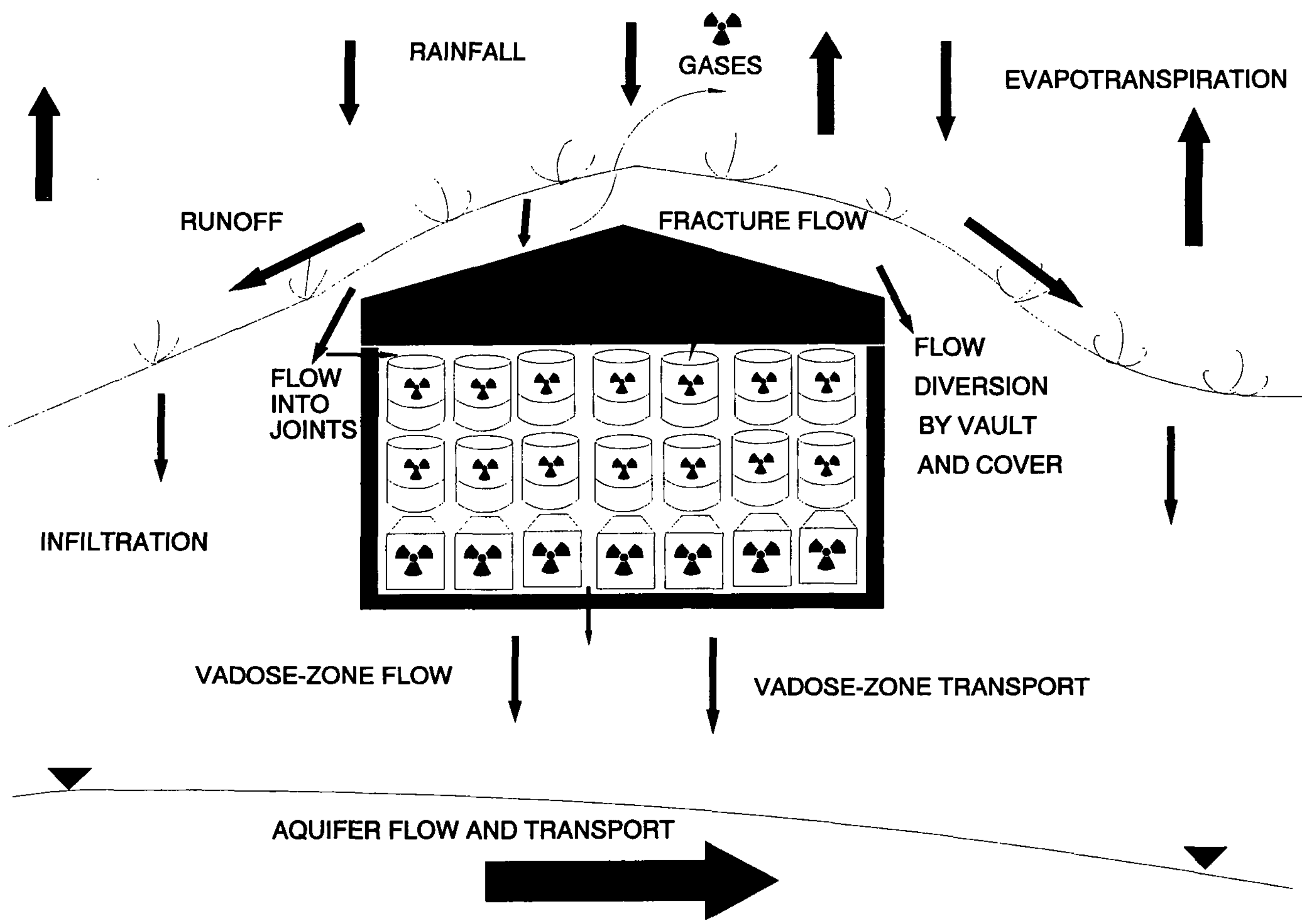

Figure 4.1 Flow processes in and around an intact disposal facility

VAM2D [Huyakorn et al., 1989]. PAGAN performs transport analyses of the radionuclides in the source, unsaturated zone, and saturated zone given a user-defined uniform velocity. The code's analysis of the source and unsaturated zone is usually used in a way that neglects dispersion in these regions. This is believed to be a conservative assumption in most cases. The saturated-zone transport model in PAGAN allows for one-dimensional convection and three-dimensional dispersion, and uses a semianalytical solution to the convective dispersion equation. VAM2D is a fully two-dimensional finite element code for the analysis of both flow and transport in either saturated media, unsaturated media, or a combination of the two. The flow model is based on Darcy's law, including the unsaturated extension of Darcy's law (Richard's equation). Discrete fracture flow cannot be evaluated using the code. The transport model solves the convective dispersion equation in two dimensions for either simply decaying radionuclides or chains (up to four member chains, either straight or branched). VAM2D is flexible in the types of boundary conditions that can be specified for both flow and transport. In addition, VAM2D has been shown to have robust numerical methods. We chose it for use in the methodology primarily because of its flexibility, robustness, and consideration of chain transport.

Kozak et al. [1990a] discussed code errors and documentation discrepancies in VAM2D Version 5.0. Some of these errors were eliminated in Version 5.1, but we have continued to find errors in Version 5.1. Several

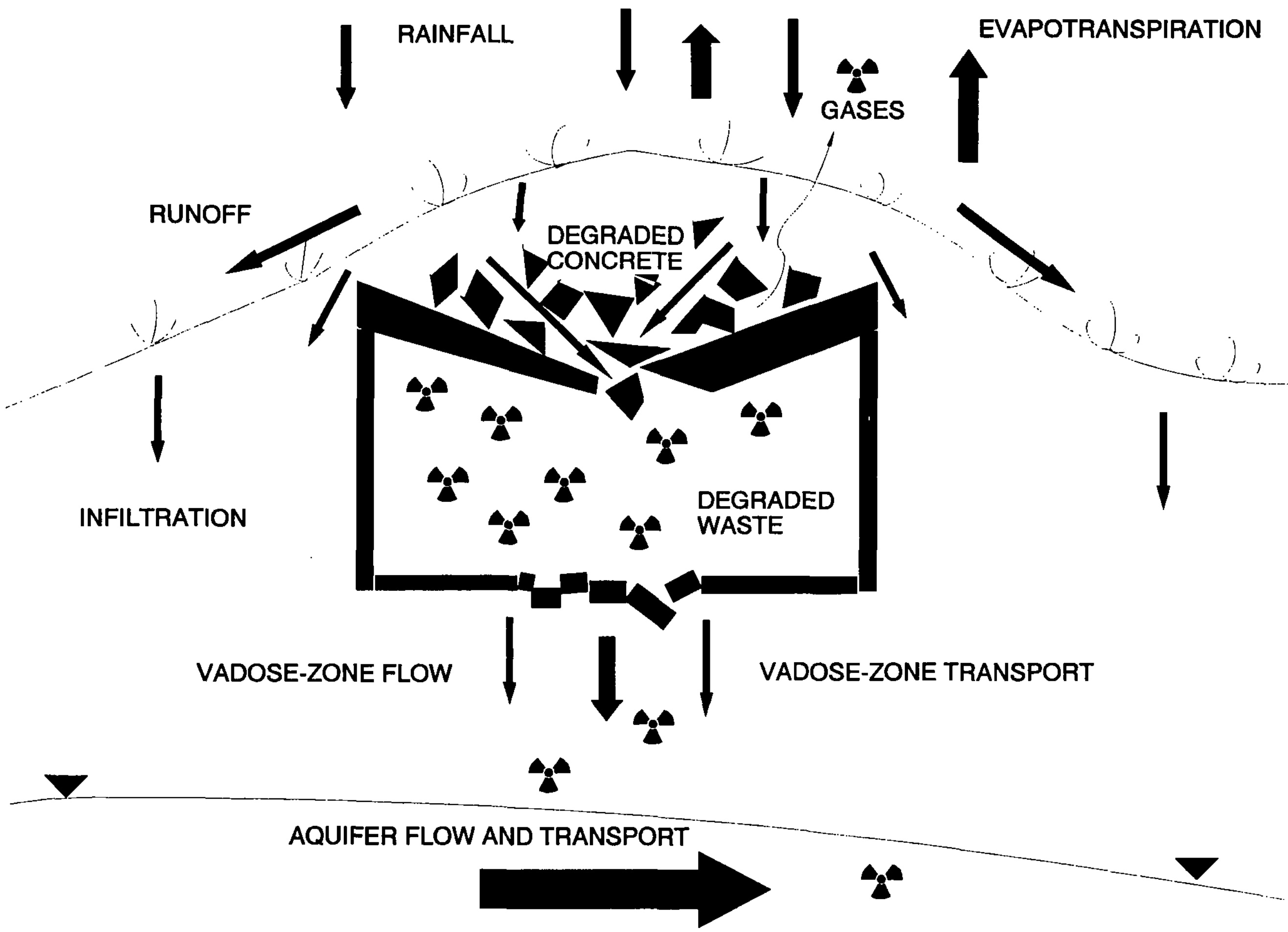

Figure 4.2 Flow processes in and near a failed vault

versions of Version 5.1 exist (we have Version 5.1d), and each of these informal sub-versions has been developed to eliminate some error; users do not receive updates as they become available, nor are they even informed of their existence. This is a poor configuration management practice that has caused us some concern. In addition, the source code is proprietary, which makes debugging a run difficult. The most recent version of the code, Version 5.2, is currently not available to us, and we have not assessed its behavior. The greatest limitations of the code are (1) it is proprietary, and the source code is not publicly available, even in hard copy form, (2) user unfriendly input structure, (3) poor configuration management practice, and (4) lingering errors in the code. These problems have not been of overriding concern to date, but they may be much more important if the code is used during the licensing process.

An updated version of PAGAN (Version 1.1) was produced in late 1990 [Kozak, 1991]. The new version uses a dispersivity defined in terms of pore velocity rather than Darcy velocity (which was used in Version 1.0); it also uses consistently conservative values for dose conversion factors for the actinides in the code. No bugs have been reported by users of Version 1.1. Some areas of possible improvement are (1) improvement of accuracy for long time periods, (2) elimination of some confusing aspects of the input shell, and (3) incorporation of Monte Carlo analysis with either random sampling or Latin Hypercube Sampling.

Status

There are many possible alternatives to using PAGAN for simple ground-water transport analyses. There are large numbers of analytical models for transport in the aquifer, and any of these may be appropriate at a particular site. However, PAGAN combines a fairly wide amount of flexibility with user friendliness, and has proved useful in several scoping analyses.

There are two possible alternatives to using VAM2D for complex ground-water modeling in the methodology:

1. Use of a different code with capabilities similar to VAM2D; and

2. Use of perturbation methods (often called analytical stochastic models) in the methodology. Both of these possibilities are discussed here.

Several possible alternatives to VAM2D have recently become available. Two of these are DCM3D [Updegraff et al., 1991] and FEMWATER3D/LEWASTE.[1] However, DCM3D only models flow at present, and FEMWATER3D/LEWASTE suffers from lack of adequate documentation and from poor quality assurance procedures. McCord[2] benchmarked a number of ground-water flow codes, and identified the code VS2DT [Lappala et al., 1987; Healy, 1990] as a potential replacement for VAM2D in the methodology. Advantages of VS2DT over VAM2D are (1) better quality assurance of the code, (2) faster convergence (in the benchmark problems), (3) nonproprietary source code, (4) a more user friendly input structure, and (5) sorption by either Freundlich or Langmuir isotherms. Disadvantages of VS2DT are (1) it cannot be used to analyze chain transport, and (2) lack of a free drainage boundary condition, which is useful in analyzing very deep unsaturated zones. This second limitation is not considered to be a severe one.

In the original development of the methodology, preference was given to codes that were able to model either saturated or unsaturated media. It has since become clear that this constraint helped reduce the number of codes needed in the methodology, but also introduced limitations. Consequently, we now consider codes that are limited in their applicability to either saturated or unsaturated media, but which introduce some additional needed flexibility to the methodology.

[1]Yeh, G.T., personal communication, ORNL, 1991.
[2]McCord, J.T., personal communication, SNL, 1991.

The first such code is MODFLOW [MacDonald and Harbaugh, 1988], developed by the U.S. Geological Survey. MODFLOW is a three-dimensional block-centered finite-difference code for evaluating saturated flow. It is extremely popular, widely used, and acknowledged to have a good quality assurance background. Use of MODFLOW allows very broad flexibility in the types of saturated-zone flow problems that can be modeled, from simple to quite complex. A particle-tracking module, called MODPATH [Pollock, 1989], has also been developed, along with a packageto generate graphical output of the pathlines calculated in MODPATH. Another package allows evaluation of the interaction between ground-water and surface-water flows [Prudic, 1989]. This interaction was treated inflexibly in the original methodology using conservative assumptions. MODFLOW is expected to be particularly useful for sites with complicated hydrology, and in reducing those complexities to simpler performance assessment conceptual models. The biggest drawback to the code is that it is not coupled with a transport code.

An alternative for evaluating transport is to incorporate NEFTRAN II [Olague et al., 1991]. NEFTRAN contains a solution to the convective-dispersion model with one-dimensional convection and one-dimensional dispersion, which are arranged in a network of stream tubes between the source and the receptor point. The primary incentives for using NEFTRAN are (1) minimization of numerical dispersion for large simulation times, (2) numerical efficiency at very long times, and (3) the ability to model multiple decay chains of any length. Ironically, the approaches used in NEFTRAN that make it efficient for long simulation times also introduce difficulties in its use for low-level waste performance assessment. Unlike the more common finite-element and finite-difference numerical solution approaches to solving the governing equations, in NEFTRAN numerical dispersion is minimized by maximizing the time step size (within certain constraints). This characteristic of the code is ideal for analyzing integrated discharge for 40 CFR Part 191, which is the original intended use of the code. However, when evaluating peak ground-water concentration for 10 CFR Part 61, it is often necessary to calculate many intermediate time steps to ensure that the analysis does not miss the time of occurrence of the peak. This issue is particularly important for time-dependent concentrations that vary rapidly in time. In this case, many intermediate time steps may need to be calculated, and this can increase numerical dispersion in NEFTRAN, thus decreasing calculated concentrations. This problem can be overcome by careful use of the code, but the analyst must be aware of the issue.

It is important to understand the limitations and uncertainties that can potentially be introduced by using a stream tube model to evaluate concentration. As pointed out by Kozak et al. [1990a], NEFTRAN is designed to calculate radionuclide flux through a cross-sectional area. Calculation of a concentration requires the introduction of a dilution volume, which must be specified by assigning an effective cross-sectional area through which transport occurs. For this reason, NEFTRAN was omitted from the original methodology. This limitation of the code can be overcome in principle by using a detailed three-dimensional flow model to identify the boundaries of pathlines that can be used in assigning the stream tube volume used in calculating concentration. This approach, which is depicted in Figure 4.3, may be useful in some cases. However, the vadose zone-aquifer interface must be evaluated with care in the flow model, since the interaction of the two zones will strongly affect the assigned area of the stream tube in the aquifer. In addition, the anisotropy of the aquifer hydraulic conductivity becomes important, since it plays an important role in identifying how broadly the stream tube spreads with distance downgradient. Other possible uses for NEFTRAN will be addressed in Chapter 5, Source Term Modeling.

Another approach that has been proposed involves the use of perturbation analyses (analytical stochastic methods) for unsaturated ground-water flow calculations [Polmann et al., 1988]. These methods reduce to an approach to conducting parameter uncertainty analysis [Freeze et al., 1990]. Parameter uncertainty can be incorporated into the methodology using Monte Carlo analysis with codes such as VAM2D or VS2DT, or using perturbation models (uncertainty analysis is discussed more fully in Section 8.1). However, to use analytical perturbation methods, the analyst is limited by a number of restricting assumptions in the models [Freeze et al., 1990]. The drawbacks to analytical perturbation methods may be summarized as follows.

- The purpose of the analytical perturbation models is to propagate input variable uncertainty through the model to identify the associated uncertainty in the output variable. Bonano et al. [1987] and Freeze et al. [1990] have noted that analytical perturbation models are only one of several approaches to account for parameter uncertainty. However, the models are currently limited to normally distributed input variable distributions. This makes the analytical perturbation models much less flexible than when Monte Carlo Sampling is used in conjunction with the Monte Carlo analysis. Furthermore, when Latin Hypercube Sampling is used in conjunction with the Monte Carlo analysis, Monte Carlo analysis may approach the analytical perturbation methods in computational efficiency [Bonano et al., 1987].

- The unsaturated-flow model is based on an exponential function for the relationship of unsaturated conductivity to pressure head [Hills and Wierenga, 1991]. This function fails at low and high saturations, which inhibits use of these models in these flow regimes. Including other characteristic functions in the model will likely eliminate the advantages of the analytical perturbation method, because the closed-form analytical structure of the solution will be lost.

- Current analytical perturbation models are based on linear perturbation theory, with high-order terms in Taylor's series in the perturbed variable truncated at the first-order terms. This limits the models to small perturbations [Freeze et al., 1990]. For the flow equation under dry conditions, the perturbations are expected to become large, and the flow model is likely to be inappropriate.

- Use of analytical perturbation analyses requires the analyst to identify a correlation length in the field data. This correlation length is usually identified by evaluating a variogram of the data. Variogram analysis has a number of limitations, among them smoothing of data perturbations and nonrobust statistical behavior [Samper and Neuman, 1989]. In addition, interpretation of variograms is often done "by eye," and this approach has been found to be quite subjective [Samper and Neuman, 1989]. There is not currently a well-established method to unambiguously identify correlation lengths from data. This means that correlation lengths cannot be specified with much confidence.

- Analytical perturbation models apply continuum equations, as do more traditional Richard's equation models, but in the analytical perturbation approach it is assumed that the variables of interest can be treated mathematically as if they are spatially random. This random field is invariably assumed to be ergodic and stationary, and these conditions are unlikely to be met in the field. Dagan [1986] has identified stationarity as a stringent constraint on using analytical perturbation ground-water models.

- The models are derived for infinite domains [Freeze et al., 1990], and the assumptions in the analytical perturbation method about correlation lengths fail

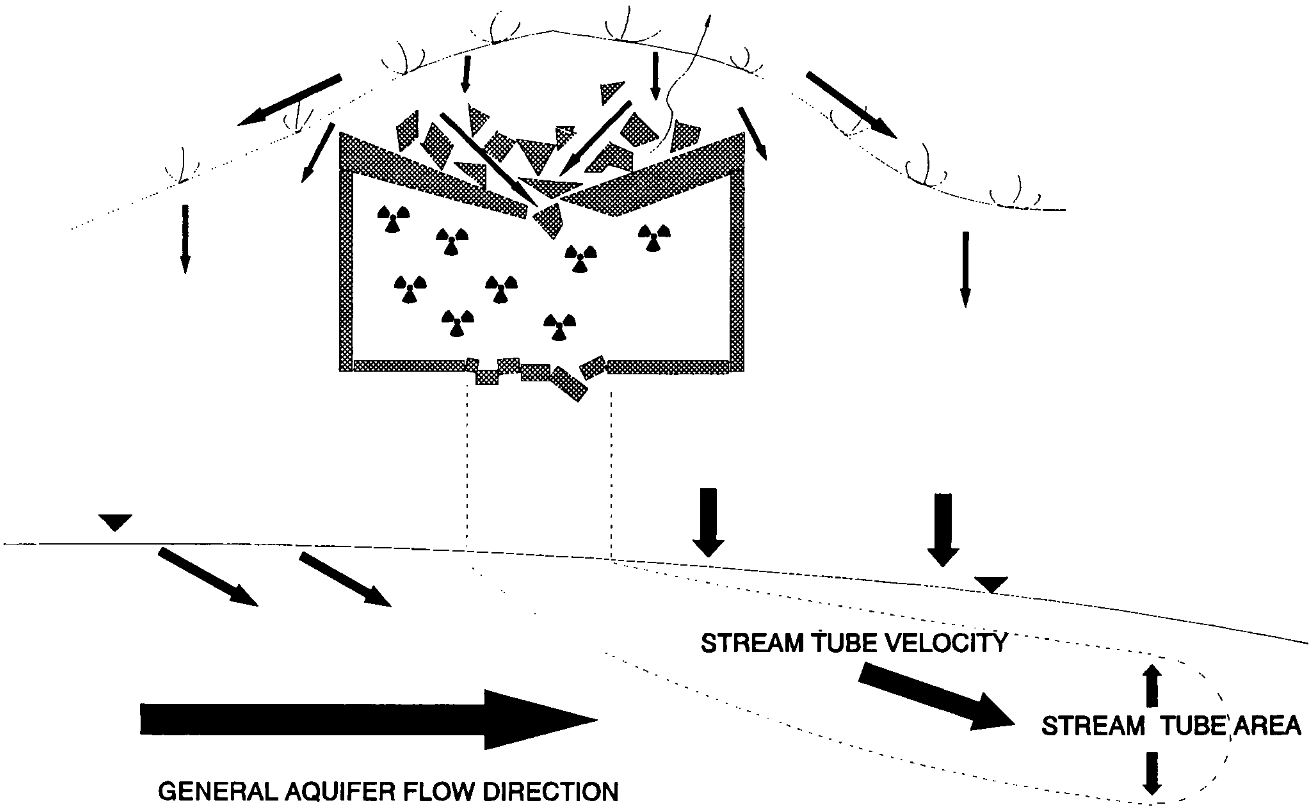

Figure 4.3 Defining an aquifer stream tube from a flow model. The cross-sectional area of the stream tube must be defined from a detailed flow model to calculate a concentration using NEFTRAN II.

near boundaries. This limitation is related to the necessity for stationarity, and identifies a limit to the usefulness of the analytical perturbation models in analyzing complex geometries.

Evaluation

In comparing PAGAN to the other alternatives available for simple analyses, we conclude that it is flexible and easy to use for scoping calculations, but the assumptions in the model may not be appropriate for many cases. Other available solutions for simple analyses may sometimes be more appropriate, but these must be chosen case by case.

VAM2D has technical capabilities that other codes do not, but its drawbacks in quality assurance and its proprietary nature are continuing sources of concern. We believe that VS2DT has some inherent advantages over VAM2D, but at present it cannot model decay chain transport, which is a major limitation. Analytical perturbation models may be useful from a purely research standpoint, but are neither flexible enough nor robust enough to be useful in performance assessment.

The ideal situation would be to develop a public code, such as VS2DT, to incorporate the capability to model decay chain transport. In this way, a code can be produced that is publicly available, that is produced under appropriate quality assurance standards, and that contains all of the important technical capabilities of VAM2D. It is recommended that VS2DT should be modified to incorporate the capability to model decay chains of arbitrary length.

MODFLOW may be useful in some cases, and introduces an additional degree of flexibility into the methodology. NEFTRAN II may also be useful in some circumstances, particularly when decay chains are important, and it is recommended that it should be incorporated into the methodology. In principle, stream tube volumes that identify the dilution volume used in calculating ground-water concentrations can be identified using MODFLOW and MODPATH. Caution must be used, however, in defining a stream tube volume for the calculation of concentration, since overestimating the stream tube volume will produce lower concentrations and doses.

5. Source-Term Modeling

Source-term models in the current methodology consist of models for the breach of engineered barriers, leaching of radionuclide chemicals into ground water, and transport of radionuclides to the boundary of the disposal unit. Current analysis methods in the methodology are the BLT code [Sullivan and Suen, 1989], and the mixing-cell cascade model [Kozak et al., 1990a], which is implemented in PAGAN [Chu et al., 1991]. These two approaches contain advantages and limitations in how they address each individual modeling area. Overall drawbacks to these methods are that neither can model chains in the source, nor can they model gas production. Furthermore, in using BLT the analyst is constrained to using FEMWATER to model the flow into and through the waste.

5.1 Engineered Barriers

We include in this discussion the time of failure and the mechanism of failure of both concrete structures and metal containers. These two aspects of the source-term analysis are the only processes that can delay the onset of releases from the disposal unit.

5.1.1 Concrete Structures

Current Approaches

Concrete structures are modeled in the current methodology as a step change in the concrete hydraulic permeability at some specified time. A flow analysis is performed using VAM2D with an unsaturated hydraulic permeability function representative of intact concrete. At the time of failure, the unsaturated permeability function is replaced with one representing a permeable soil. In essence, this approach assumes a rapid change between the conditions depicted in Figure 4.1 and those shown in Figure 4.2. There is considerable uncertainty in determining the appropriate properties for the failed vault. Less obvious, but still important, is the uncertainty in the properties of the intact concrete. The permeability of the concrete may vary considerably as a result of quality assurance practices during emplacement. The only way to ensure that the permeability of the completed concrete meets design specifications is to conduct permeability tests on the completed vault.

A step function in the flow rate through the vault is recommended at this time in the performance assessment analyses. Gradual failures of the vault can also be modeled using the current methodology models, by using a number of small step changes in the concrete flow properties. This approach would require the analyst to define a series of intermediate failed conditions between those shown in Figure 4.1 and Figure 4.2. In this approach, it is difficult to justify a gradual degradation rather than an abrupt degradation, and to specify the flow properties of the partially failed concrete. The primary reason for doing such an analysis is to take additional credit for the concrete vault. At the present time, there are such large uncertainties in modeling concrete failure that specifying gradual failure is not generally defensible.

At the time the methodology was developed, an adequate model for concrete degradation did not exist. The prime candidate for use in modeling concrete degradation, BARRIER [Shuman et al., 1988], was proprietary and could not be adequately assessed. Furthermore, use of BARRIER provided an estimate of the time of failure, but the rate of failure was assumed to be an instantaneous failure in applications of the code [Rogers and Associates, 1988]. As discussed below, the performance assessment is relatively insensitive to the time of failure, hence it was concluded that BARRIER would not significantly enhance the capabilities of the methodology.

Status and Evaluation

Since the methodology was developed, there have been developments in the area of concrete modeling. Clifton and Knab [1989] assessed the current models for the service life of concrete, and concluded that reasonable assurance of 500-year lifetimes should be possible. Walton et al. [1990] began developing models for the behavior of partially failed vaults. More recent work by Walton and Seitz [1991] has provided guidelines for the design, construction, and operation of vaults. This work included models for evaluating flow and transport under partially failed conditions. Pommersheim and Clifton [1991] identified models for evaluating the major degradation processes in concrete to be used in estimating the concrete service life. In addition, Rogers and Associates will soon produce a replacement for BARRIER, called RAESTRICT [Shuman et al., 1991]. We do not yet fully know the scope or limitations of this new code.

These improvements still suffer from the limitations of the older models. Limitations on the current models are (1) lack of long-term experience with modern concrete, and (2) lack of an adequate validation basis for models of flow through concrete. To build confidence in these models, efforts should be focused on validation experiments on old (50 to 100 years old) modern concrete, possibly experiments on ancient (1000 to 2000 years old) concretes, and comparisons with accelerated tests. It would be very useful to establish an experimental link between accelerated tests and long-term analogs to provide confidence that the accelerated tests represent long-term behavior [Olague et al., 1993]. Much of the issue of concrete longevity is related to quality assurance and quality control during construction rather than parameters that are quantifiable for use in a performance assessment [MacKenzie et al., 1986]. This introduces additional uncertainties into modeling of these structures. Olague et al. [1993] discuss a possible approach for quantifying this uncertainty.

Another drawback to these existing models is their focus on the service life of the concrete rather than the rate of change of permeability. Kozak et al. [1990a] have shown that for typical low-level waste inventories, the rate of change of vault permeability of the concrete structure is more important than the time at which failure occurs. Low-level waste is made up almost entirely of short-lived (half life < 30 years) and very long-lived radionuclides (half life greater than 5000 years). Exceptions to this rule are ^{63}Ni and the actinides. However, these radionuclides will generally be present in small concentrations; their contributions to the total dose, while not negligible, will not typically dominate the performance assessment. It is relatively easy to provide a convincing argument that a concrete structure will last longer than 100 to 200 years. After this time period, only the very long-lived constituents of low-level waste remain. It would be very difficult to demonstrate that a concrete structure will remain intact for the 2,000 to 10,000 years necessary to allow decay of some of the long-lived radionuclides. Consequently, there is very little change in the inventory during the crucial 200- to 2000-year period, and therefore, performance assessment results are relatively insensitive to the time of failure of engineered structures.

By contrast, the performance assessment is very sensitive to the rate of change of concrete hydraulic conductivity. If the change occurs slowly, releases from the disposal unit will be spread out in time, and the peak dose will be reduced. For instance, if it could be demonstrated that the vault cannot reasonably fail such that from intact conditions until complete failure could not be less than 100 years, this information would be very useful in performance assessment modeling. The step function failure model could then be replaced by a ramp failure model, and it could be expected that calculated off-site doses would decrease as a result.

Analyzing the rate of change of the hydraulic conductivity of concrete is difficult because. (1) the fundamental mechanisms are not completely understood; (2) there is no long-term experience with modern concrete from which to make our judgments; (3) there is uncertainty about the future condition of the site (see Chapter 2); and (4) the most probable failure may be gradual, but there will likely be some significant probability that the failure will be rapid.

A second issue that needs to be addressed is how multiple vaults should be modeled. We could assign a probability distribution of failure times to each vault, so that all vaults do not necessarily fail at the same time. However, the maximum dose from the possible sets of realizations of these failure times would undoubtedly result from a realization in which all vaults do fail at the same time. There is not any technical justification for formally removing this possibility from consideration. For specific sites, design considerations may suggest that the vault failure distributions may not overlap, in which case each vault may be analyzed separately, and the consequences superposed. The approach to modeling multiple vaults will be dictated by the overall approach to uncertainty analysis; optional approaches to uncertainty analysis are discussed in Section 7.1.

A third issue that needs to be addressed is the difference between the idealized planned performance of a vault and its performance as emplaced. It is unreasonable to assume that the hydraulic conductivity of a completed vault will be as low as a small laboratory sample of that same concrete. In the concrete itself, differential settlement or stress fractures may cause the concrete permeability to increase during the operational period. In addition, for most designs there will be enhanced degradation of the concrete while the vault is operational, since it will be exposed to the elements above ground. The concrete slabs in vaults must be connected by joints, which are made of materials that are not necessarily as long-lived as the concrete itself. For instance, joints may be sealed by metal that is subject to corrosion or by polymer materials whose longevity is unknown. It is probable that less confidence can be placed in the longevity of joint materials than in the longevity of concrete. Another potentially disruptive difference between idealized behavior and actual behavior might be the obstruction of vault drains by sediment or biological clogging, which is a frequent occurrence in sanitary landfills [Bass, 1984].

We, therefore, conclude that the current modeling approach in the methodology can be used to model either abrupt or gradual changes in concrete hydraulic permeability. The important issues related to this approach are (1) the hydraulic properties that are appropriate for intact, failed, and partially failed concrete are poorly understood, and (2) it may be desirable to be able to justify gradual failure behavior for the vault. Research efforts should be focused in these areas. The methodology could be only marginally improved by including a model to evaluate the service life of concrete vaults, and we do not recommend such models at this time.

5.1.2 Metal Container Degradation

Current Approaches

Degradation of metallic containers can be modeled in the methodology using either BLT or PAGAN. In PAGAN, container failures are modeled as a delay time until the onset of releases. This approach is not mechanistic, and the time of failure must be justified by some other means, such as using BLT. In BLT, container degradation is modeled using semiempirical models for corrosion. The analytical structure of the model was suggested by corrosion theory, and model parameters were obtained from field data on underground corrosion. The model and its limitations are discussed in detail in Sullivan et al. [1988], and in Kozak et al. [1989a]. The primary limitations are insufficient field data to justify some of the parameters in the model, and insufficient data on metals other than unpainted carbon steel.

Status

There have not been any significant changes to the corrosion model since the development of the methodology. However, work is in progress to make BLT more user-friendly, and to extract the Breach and Leach portions of the code to stand alone. This improvement to the code should be useful as a user-friendly approach for overall performance assessment methodology.

Evaluation

Given the increased emphasis on concrete structures in low-level waste disposal facilities, metal degradation is less important than it would be for shallow-land burial sites. We conclude that the BLT approach for metal corrosion is satisfactory for use in the methodology.

5.1.3 Degradation of Other Materials

A variety of other materials have been proposed to contain low-level wastes in disposal. In particular, polymeric materials, such as high-density polyethylene, have been proposed. There is no long-term experience with any of these materials under disposal conditions, since most of them have only existed for (at best) a few tens of years. There are no existing models, nor is there a satisfactory data base from which a model might be developed. Consequently, minimal credit should be given for the duration of such containers.

5.2 Leaching Processes and Near-Field Transport

5.2.1 Leaching Processes

Current Approaches

Leaching processes are modeled in the methodology using either of two options in the mixing-cell cascade model, or using the multiple options in BLT. The mixing-cell cascade model, depicted in Figure 5.1, was derived for surface-wash and constant-rate leaching models.

The surface-wash model was identified as being appropriate when the waste form is poorly understood, as in unconsolidated trash. The constant-rate model is considered appropriate for some releases dominated by diffusion in the waste form. In particular, if the solubility limit is reached in the waste form, a constant internal concentration will result in constant diffusional releases. This release model may, therefore, find its greatest use in modeling concrete-stabilized wastes, in which the pore fluid chemistry is dominated by the concrete. Pescatore [1991] has pointed out that a constant release model is not mathematically appropriate for diffusional releases with a zero concentration boundary on the waste container, since the mathematical solution is not convergent at time zero.

BLT is much more flexible in its treatment of the leaching process. The code contains models for surface-wash leaching, diffusional leaching (in several geometries), congruent dissolution (appropriate for activated metals), or some combination of these mechanisms [Sullivan and Suen, 1989].

Status

Since the publication of the methodology, there have been several improvements to leaching models. Sullivan and Suen [1991] improved BLT to account for concentrations surrounding containers to accumulate, which tends

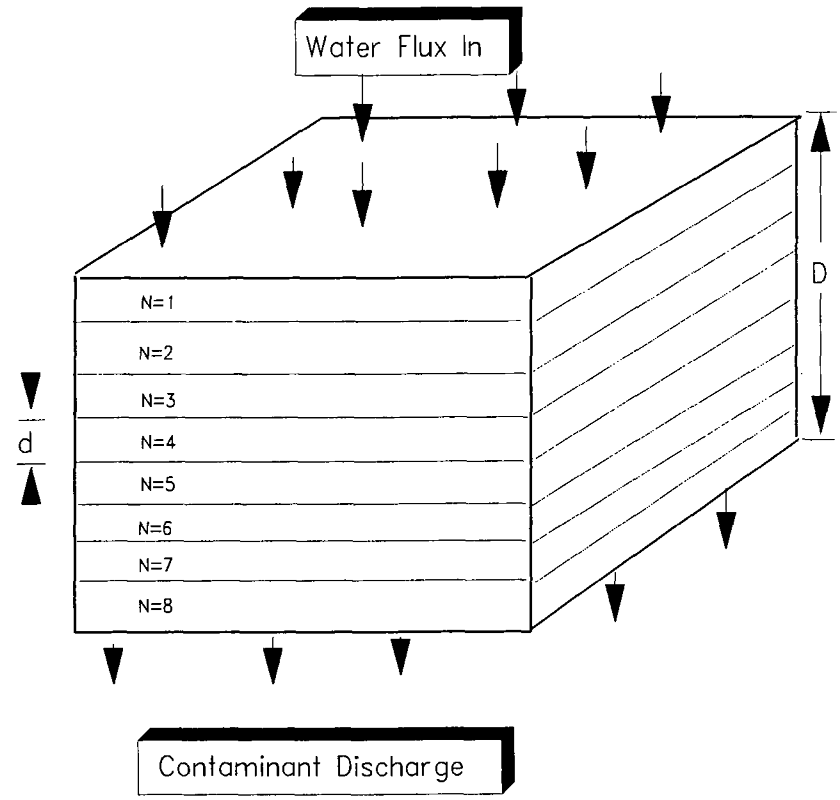

Figure 5.1 The mixing-cell cascade model

to decrease diffusional release rates. This improvement adds an additional capability to the code. In addition, a recent extension of the mixing-cell cascade model by Sullivan [1991] allows multiple simultaneous release mechanisms. This improvement also adds a new capability to a model in the methodology. Of greatest importance is the development (in progress) of a simplified model for source-term analyses, called DUST (Disposal Unit Source Term) [Sullivan, 1992]. This model is expected to be much easier to use than BLT, because the analyst will no longer be constrained to FEMWATER [Yeh and Ward, 1980] to generate a flow field to be used with BLT. Furthermore, the leaching model results will be useable by alternative transport models, and the analyst will not be constrained to FEMWASTE [Yeh and Ward, 1982].

Evaluation

The mixing-cell cascade model has typically been used with a large number of mixing cells, which corresponds to plug flow in the disposal unit. This approach generally produces larger ground-water concentrations than releases calculated from a single mixing cell model. However, when the number of mixing cells becomes large, the source-term calculation becomes quite time consuming. This numerical inefficiency can be eliminated by replacing the mixing-cell cascade model for large N by analytical solutions for plug flow releases. For instance, the surface wash model in plug flow can be represented by

$$Q = Q_0 e^{-\lambda t}, \qquad t \leq I/Q_0$$
$$Q = 0, \qquad t > I/Q_0. \qquad (1)$$

Here, Q_0 is the initial release rate, λ is the radionuclide decay factor, and I is the inventory of the radionuclide. This simplified expression for the release rate is much more numerically efficient than the full mixing-cell cascade model.

We conclude that updated versions of the current models in the methodology will be useful, and should be incorporated into the methodology as they become available. The mixing-cell cascade can be replaced in many cases by more efficient analytical expressions, and DUST is expected to be much more numerically efficient and easier to use than BLT. As in other areas of the source term, the more important issue is how a validation basis can be established that would allow the analyst to justify taking credit for phenomena that decrease release rates. The models can account more or less rigorously for solubility limitations, ion exchange phenomena, diffusion limitations in waste forms, and sorption, which can all play a significant role in reducing release rates, but which are all difficult to justify using because of the heterogeneities and uncertainties in low-level waste. This topic is discussed further in Volume 2 of this report.

5.2.2 Near-Field Transport

Current Approaches

As mentioned above, Sullivan extended the mixing-cell cascade to allow multiple simultaneous release mechanisms. The generalized mixing cell cascade will be useful in any potential future developments of the model. However, incorporating more elaborate leach-model expressions into the model (both by us and by Brookhaven National Laboratory) has resulted in expressions for the global release rate so complicated that use of this model was unjustified. It is probable that analytical mixing-cell cascade models may have been taken to their practical limit.

The alternative approach in the methodology, using a full transport analysis in VAM2D or BLT, can be cumbersome to apply, but it allows for flexibility in treating near-field transport.

Status and Evaluation

There have been no significant improvements in near-field transport modeling during the past two years. However, experience in using the methodology suggests that it may be desirable to introduce additional flexibility into the source-term analysis. Areas in which flexibility can be improved in straightforward fashion are (1) introduction of an option to use an arbitrary user-defined release rate (in tabular form), and (2) improved treatment of decay chains.

5.2.3 Decay Chains

Current Approaches

Decay chains are not explicitly considered in either the mixing-cell cascade model or in the BLT. The primary problem with modeling decay chains in the source is when the parent and daughter are assigned different retardation factors in the disposal unit. If the parent and all daughters are assigned the same retardation in the disposal unit, the analyst can evaluate only the release of the parent radionuclide, and can correct for the ingrowth of daughters as the radionuclides exit the disposal unit.

If unequal retardation factors are used, an alternative to BLT or PAGAN is necessary. In using the methodology, we assume that the analyst has calculated the inventory, including decay and daughter ingrowth, at some baseline year. The release of this inventory from particular locations in the disposal unit can be modeled using leaching models, and the resulting time-dependent release rates can be input to VAM2D to analyze transport in the disposal unit. This approach is cumbersome, because of the complexity of VAM2D. In addition, VAM2D is limited to four-member chains (either straight chain or branched), although this is not believed to be a serious limitation for the chains of importance in low-level waste inventories [Kozak et al., 1990a].

Status

One alternative approach for unequal retardation factors would be to develop analytical expressions on a case-by-case basis, assuming a mixing-cell model for the near-field transport. This analytical expression for the discharge rate can be used as input into a VAM2D analysis of the aquifer. This approach is straightforward and can easily be implemented when needed for specific cases. However, it is more desirable to have the explicit capability in the methodology to evaluate decay chains in the source.

An existing model that uses a similar approach is the NEFTRAN II source model [Olague et al., 1991]. The NEFTRAN II source model includes two options about near-field dispersion: a mixing cell model and a "flow-through" model. The flow-through model represents a convectively driven process, analogous to the releases described in Equation (1). Either source flow model can be implemented with leach limits and solubility limits. It appears that incorporating the NEFTRAN source-term model may be particularly useful for simple modeling of chain members.

Evaluation

For generality and flexibility, we recommend incorporating the NEFTRAN II source-term model into the methodology. In Chapter 4, we recommended incorporating the NEFTRAN II transport model for unsaturated-zone transport, and potentially for some saturated-zone transport problems. The NEFTRAN II source-term and transport models represent an intermediate level of complexity between very simple (PAGAN) and very complex (BLT or VAM2D), and additional flexibility over the current approaches in VAM2D and PAGAN.

5.3 Gas Production

Current Approaches and Status

Issues associated with gas generation can be separated into two discrete subjects: generation of gases by the waste, and migration of the gases to the surface. This distinction was not made in discussions of gas evolution in the original development of the methodology. Once the gases reach the surface, they form a release into the atmosphere. Depending on the exposure and pathway assumptions, this release at the soil surface may need to be used as an input into an air-transport model for the analysis of off-site doses. Air-transport modeling is discussed in Chapter 6.

Potential gaseous radionuclides in low-level waste include ^{3}H, ^{14}C, and ^{222}Rn. In general, gas generation of ^{14}C is probably most important because of its relatively long half-life; therefore, the following discussion concentrates on gas generation of ^{14}C. If the disposed waste contains naturally occurring Th-230 or depleted uranium, the potential exists for radon production and transport off-site to be a significant contributor to the maximum dose. In the case of radon, measurements are available for the disequilibria between the radon and its daughters both indoors and outdoors, which is important to its inhalation dosimetry for such a pathway [NCRP, 1988]. In addition, although radon's half-life is relatively short, it is the parent of long-lived species (particularly Pb-210), so daughters can potentially be transported in radiologically significant amounts. The short-lived daughters of Rn-222 (Po-218, Pb-214, and Bi-214) can produce significant lung doses from inhalation, since their dose-conversion factors are large. Furthermore, the possibility exists for gaseous transport of radon to plant roots, and then decay of radon to longer lived radionuclides. This could be a significant contributor of dose to man if bioaccumulation of the daughters in edible plant roots occurs. This suggests that there may be an enhanced transportation mechanism to off-site locations for daughters of radon; this transport pathway has been evaluated in the context of naturally occurring radon [NCRP, 1987], but studies related to waste disposal sites are unknown to us.

The methodology does not currently contain a way to estimate the rate, volume, or radiological component of gas production from low-level waste in a disposal facility. This is an area where there has been relatively little research, so data are unavailable to justify the use of any model at the present time. Existing approaches to modeling gas generation have been developed for different inventories than U.S. low-level waste streams (e.g., Biddle, et al., 1987; Jefferies, 1990); however, gas generation may be important for low-level waste performance assessment because

1. Releases into the gas phase may decrease the impacts of the ground-water pathway. For some low-level waste inventories, ^{14}C doses can add a substantial contribution to the ground-water dose. The doses from the air pathway and the ground-water pathway are likely to occur at very different times, so the peak dose to the maximally exposed person from the ground-water pathway may potentially be reduced by accounting for gas releases.

2. The potential exists for bioaccumulation of gaseous $^{14}CO_2$ in plants. This could concentrate otherwise unimportant releases in the gas phase; consequently, this mechanism poses a possible enhanced transport path from the disposal cells.

3. The aforementioned possibility of radon transport may provide a preferential pathway for some of its daughters.

4. Gas generation may have potentially deleterious effects on the integrity of vaults for some disposal designs [Hodgkinson et al., 1988].

Gas generation models are intimately linked to assumptions about conditions that exist inside the low-level waste disposal facility for the performance assessment time period. Conditions inside the disposal facility have large uncertainties associated with them because of the heterogeneous nature of low-level radioactive waste. These uncertainties can be reduced to some extent for stabilized waste forms, but for the most part the uncertainties are likely to remain large.

For gas production models, the mechanisms assumed to be occurring are (1) microbial biodegradation of organic materials leading to releases of $^{14}CO_2$ and $^{14}CH_4$, and (2) production of titrated H_2 gas from metal corrosion. The latter mechanism is believed to be less important, since metallic inventories are not expected to be large.

At the present time, we are unaware of any suitable models or experiments for gas generation that would be appropriate for evaluating U.S. low-level waste inventories and disposal conditions. An appropriate experiment would consist of measurements of gas generation from U.S. low-level waste in the physical and chemical conditions likely to be encountered by the waste in a disposal unit. Since most current disposal designs include massive use of concrete, the experiment should be conducted for high pH. Such an experiment may also be appropriate for many arid western sites, which are the only ones currently being considered for trench burial [Olague et al., 1993].

The second aspect of the evolution of gas from the site is its transport from the disposal vault to the ground surface. Subsurface transport of radioactive gases has received considerable attention since the mid-1970s, because of increased awareness of the potential for indoor exposure to naturally occurring Rn-222. Consequently, there is a lot of empirical and theoretical information available on subsurface transport of gases. This information will be applicable to emission of C-14, H-3, and other possible gas releases from low-level waste as well as to radon emissions.

Current thinking about gas exhalation into houses suggests that it is dominated by convective gas flow in the subsurface; measured radon concentrations in houses are too high to be explained by diffusion through the slab [Nazaroff, 1992]. The convective flow is the result of barometric pressure changes that cause transient convective transport of air into and out of the soil. However, it is not clear how the long-term average emission of gas is influenced by barometric pressure oscillations; most studies in the literature have been concerned with the temporal aspects of gas emission [NCRP, 1989; Nazaroff, 1992]. In general, at times of low pressure, gases are "pumped" from the soil, while during times of high pressure, the emission of gas is suppressed. However, the suppression of gas emission during a high-pressure period can be less than the enhancement of emission during a low-pressure period [NCRP, 1989]. This suggests that barometric pumping may provide an enhanced transport mechanism from the disposal facility to the surface, even when averaged over long times. The long-term average emission of gas may depend on the average duration of barometric pressure fluctuations, the radionuclide half life, the depth of burial, and other parameters. The influence of these parameters on long-term average gas emission from the soil surface is an aspect of subsurface gas transport that needs to be investigated from a low-level waste performance assessment standpoint.

Evaluation

Based on the above discussion, there are not currently available adequate models for gas generation in low-level waste disposal facilities. Presently, the primary limitation is an inadequate experimental database for model development. Some work is needed in this area.

Models are available for estimating gas transport through soils to the surface, which incorporate the effects of diffusion, convection, decay, daughter production, and spatial variability. These need to be evaluated to identify the scope of the phenomena needed for modeling long-term average doses for performance assessment; it is recommended that some capability is needed in the methodology for modeling subsurface transport of gases.

5.4 Geochemistry

Current Approaches

Geochemistry models are usually proposed for use in performance assessment to justify taking credit for chemical limitations to transport. These limitations generally take the form of either solubility limitations in the pore fluid or complexation with soil minerals. To model these effects, geochemical models are usually developed to identify the chemical speciation, which can then be used, together with detailed information about the chemical state of the ground water and soil, to evaluate the processes of interest.

Geochemistry models are likely to be most useful in source-term modeling, since the near-field chemical environment in vaults may be quite well conditioned compared to the surrounding natural soils. However, even in this well-established environment, geochemistry models suffer from a number of drawbacks. Olague et al. [1993] discuss the lack of an adequate validation basis for geochemical models. This lack of adequate validation, together with spatial and temporal uncertainties in geochemistry, even in the near field, led Kozak et al. [1989a] to omit all complicated geochemical models from the methodology. Geochemistry is treated in the methodology through the use of K_d sorption.

Status

The constraints on geochemistry modeling have not been improved since the methodology was developed. Consequently, there is not an impetus to change the models in the methodology to incorporate more sophisticated geochemical models. Instead, we recommend that site-specific geochemical data be collected and used to justify reasonably conservative K_d values for use in performance assessment. Detailed geochemical models may find a role in interpreting site characterization data to justify conservative values for K_d values, but will probably continue to be too complicated for performance assessment.

A popular approach to modeling geochemical phenomena has been to use "generic" data, usually derived from surrogate "conservative" data from Maxey Flats or Savannah River. This approach is embodied in generic performance assessment codes for low-level waste such as IMPACTS [Oztunali and Roles, 1986], PRESTO [EPA, 1985b], and IMPACTS-BRC for Below-Regulatory Concern waste [Oztunali and Roles, 1984; O'Neal and Lee, 1990]. This approach was also used recently by Baird et al. [1990] in a performance assessment of the Clive, UT, low-level waste site, in spite of the differences between the arid nature of Clive and the humid nature of the Savannah River site. It is often argued that K_d values derived from humid sites are conservative, *but there is no experimental or theoretical basis for this argument* [Pescatore and Sullivan, 1991; Rao et al., 1992b]. The geochemistry of any site depends on the chemical and physical form of the waste, on the chemistry of the waste, soil, and ground water, and on the hydrological flow regime of the site. Leaching results from Maxey Flats depend in a complicated way on conditions at Maxey Flats, and other sites may bear no resemblance to these conditions. The information produced for use in the IMPACTS models was only intended to be used for generic modeling in support of rulemaking; it was never intended to be used on a site-specific basis.

Evaluation

In summary, there does not appear to be a technical basis for including any complicated geochemistry in the methodology. Site-specific geochemical data should be used to identify conservative ranges for K_d to be used in performance assessment. We note that the new version of NEFTRAN allows time-dependent values for K_d [Olague et al., 1991], and these can be developed consistently from more detailed geochemistry analyses. This suggests an additional degree of flexibility introduced when using NEFTRAN. If site-specific data are unavailable, the minimum possible credit should be given for reduction of impacts as a result of geochemical effects. Surrogate geochemical data from existing low-level waste sites should *not* be used to justify K_d values when site-specific data are absent.

5.5 Source-Term Summary

The current approach to modeling concrete degradation, which is based on the use of an unsaturated-zone flow model of the concrete under intact and failed conditions, is considered adequate. However, information about the hydraulic behavior of the concrete under partly failed and completely failed conditions is needed if any credit is taken for the behavior of the system under these conditions. Furthermore, detailed models for concrete degradation are not considered to be sufficiently advanced to justify moderating the current approach, which assumes a step change in the flow properties of the concrete. Current concrete degradation models are considered to be useful primarily for design analyses. The model for metal container corrosion is considered to be the best available for performance assessment analyses. Some of the parameters required for the model have been developed from a narrow experimental base, and this can be improved. However, metal container corrosion is not important in most vault disposal systems, since the concrete is generally expected to outlast the containers by a wide margin. The greatest need in the area on concrete degradation is for experiments on permeability under several degrees of failure.

Several improvements have been made (or are currently being made) to leaching models in the methodology. These improvements generally relate to increased flexibility or efficiency in the models, and should be incorporated into the methodology as they become available. However, they do not substantially change the existing capabilities of the methodology.

A significant improvement can be made to PAGAN if a user-defined source term in tabular form is used. The user could then model arbitrary releases by entering time-dependent values of releases. This approach would allow the regulator to accept a license applicant's source-term output, and to confirm only the transport analysis independently. We recommend that this capability be developed.

Adequate data to support models for gas production and geochemistry are needed for research. Models are needed, and should be developed, but such models should not be included in the methodology until there is an adequate experimental basis for them. Treatment of subsurface transport of gaseous contaminants is better established, and such models will be needed for evaluation of gas releases from a disposal facility. We recommend that such models be investigated to determine which salient aspects of the processes are likely to be important from a performance assessment perspective.

An improved treatment of decay chains in near-field transport is needed. The mixing-cell cascade model and BLT are adequate for modeling decay chain transport *if* all retardation factors are equal. If they are not, VAM2D or analytical solutions can be used, but are not sufficiently flexible. We, therefore, recommend that the NEFTRAN II source-term model be incorporated into the methodology. This addition will significantly enhance the flexibility and capability of the methodology.

6. Surface-Water Transport, Air Transport, and Exposure Pathway Modeling

The current primary computer code for all of these areas is GENII [Napier et al., 1988]. Consequently, the status and evaluation of the code for each area has been combined into Section 5.4. Descriptions of the models used in the code are given in Sections 5.1, 5.2, and 5.3.

6.1 Surface-Water Transport

Current Approaches

The current methodology uses the surface-water transport models recommended in NRC Regulatory Guide 1.113 [NRC, 1977c]; these models are contained in the GENII computer code [Napier et al., 1988]. The GENII model can be used for a river or lake and assumes a constant flow depth, a constant convective velocity, a constant river width, a constant lateral dispersion coefficient, a straight river channel, and a continuous point discharge of contaminants [Kozak et al., 1990a].

The important pathways in low-level waste performance assessment relate only to dissolved radionuclides; consequently, radionuclide interaction with the sediments can frequently be neglected. For most cases, this is conservative since adsorption of radionuclides onto the sediments would cause liquid concentrations to be lower than those estimated neglecting sediment sorption. The possible exceptions to this would be if bottom feeding fish contributed significantly to the exposure, or if the external exposure from the contaminated surface water was important in estimating total exposure. These two situations are not typical; therefore, neglecting sediment interactions is usually acceptable, although this must be determined on a site-specific basis. The more usual effect of neglecting sediment sorption is to produce conservative estimates of exposure via the food chain [NCRP, 1984]. Nevertheless, a simple approach to sorption of radionuclides on sediments is included in GENII, and can be used if desired.

The interaction between ground water and surface water (used as an input to the surface-water models) is calculated based on conservative assumptions in the methodology [Kozak et al., 1990a]. We assume that all radionuclides distributed in the aquifer at the location of the surface-water body are discharged into the surface water. In general, we expect that much less of the plume will actually end up in surface water. Furthermore, we assume that the discharge will occur at a point at the shore; a diffuse distributed plume entering the surface water is not considered.

The methodology does not currently contain models for various kinds of ephemeral flows, such as surface runoff from above-ground vaults.

6.2 Air-Transport Modeling

Current Approaches

Although air as a transport medium was not identified by Shipers and Harlan [1989] as part of a significant pathway, Kozak et al. [1989a], included air-transport models in the methodology because of circumstances that may arise where soils become contaminated and entrained in air (e.g., dry lake bed, intermittent stream) and because air pathways may become significant in intruder scenarios. Besides simulating airborne transport of contaminated particulates, models are also needed to simulate transport of radioactive gases that may be released from a disposal facility.

For modeling airborne transport of particulates, a mass-loading factor was recommended to estimate the amount of contaminant that is entrained in the air. The mass loading model assumes that the source of 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 [Kozak et al., 1989a]. This model assumes that materials have been mixed uniformly with the soil and that soil and contaminants are suspended equally. Kozak et al. [1989a], recommend this model because conservative values can be identified, and because it is relatively simple.

Transport of the gases or particulates is calculated using a Gaussian plume model, which is recommended in NRC Regulatory Guide 1.111 [NRC, 1977b], and implemented in the GENII computer code [Napier et al., 1988]. The Gaussian plume model assumes one-dimensional convective transport, with three-dimensional dispersive transport. This model has been adopted as a standard method in regulating both radioactive [NRC, 1977a; IAEA, 1980] and other [EPA, 1978] airborne species. Gaussian plume models are derived for point sources in space. For the purposes of performance assessment, area sources can be treated using the conservative approaches described by Chu et al. [1991].

6.3 Exposure Pathway Modeling

Current Approaches

The methodology uses the pathway models found in NRC Regulatory Guide 1.109 [NRC, 1977a] to determine radionuclide intake rates for a person from concentrations in the environment. These models are contained in the computer code GENII [Napier et al., 1988], and account for bioaccumulation in plants, irrigation of various crops, inhalation, ingestion of drinking water and contaminated foods and external exposure [Kozak et al., 1991]. The models can be used to account for direct biointrusion into the waste as well as root uptake of radionuclides transported to plants located off-site.

A large degree of uncertainty is associated with these models because many assumptions have to be made about future conditions at the site (e.g., agricultural activity). Besides this, the most significant issue for food-chain models centers on radionuclide bioaccumulation, especially for ^{14}C and iodine compounds [Olague et al., 1993]. The NCRP [1984] states that few efforts have been made to validate the bioaccumulation values listed in the regulatory guide and that validation studies are needed because those in the regulatory literature are overly conservative. Another issue includes the lack of isotope-specific transfer coefficients for specific foods, although conservative assumptions should overestimate concentrations in terrestrial foods and bioaccumulation factors in aquatic food chain transport models. In addition, food consumption parameters are based on 1965 data; given the change in U.S. dietary habits since then, these values are quite suspect [NCRP, 1984]. Of course, these uncertainties are small compared to the uncertainty in U.S. dietary habits over the timescale of the performance assessment.

6.4 Status and Evaluation

Status

GENII was developed for the Hanford site, and may contain some hard wired parameters that are specific to that site. At this time, it appears likely that any issues associated with the model are in the exposure pathway models. This has been a lingering concern about the using GENII as a generic risk assessment tool.

A revised version of GENII, called GENII-S [Leigh et al., 1992], has recently become available. The primary differences between GENII and GENII-S are (1) the capability to perform probabilistic pathway analyses as well as deterministic ones, and (2) an improved user interface. The user interface implemented in GENII-S is the same one used in PAGAN [Chu et al., 1991], but has been adapted to accommodate the GENII input and output.

Evaluation

GENII-S has attractive features, and the user interface is likely to be more user friendly than the APPRENTICE shell introduced with the original GENII code. GENII-S should, therefore, be used in the methodology. The models in GENII-S are identical to the models in GENII; we briefly evaluate these models here.

The surface-water pathway is usually much less important than the ground-water well pathway, because of greater dilution and (usually) greater distances from the disposal units. Consequently, quite conservative assumptions can often be made without significantly affecting the decision resulting from the analysis. However, there may be occasions when a more elaborate method is needed for the interaction between ground water and surface water. As discussed in Chapter 4, this need can be met by including MODFLOW [MacDonald and Harbaugh, 1988] in the methodology. There may also be occasions when more elaborate models are needed to evaluate surface-water transport. Such models are available [see, e.g., Jirka et al., 1983], but are rarely used, so they are not recommended for formal inclusion in the methodology.

Surface-water runoff models may be needed to assess aboveground vaults. As discussed in Chapter 3, when an aboveground vault fails, the waste is exposed directly at the soil surface. The potential, therefore, exists for exposures to occur by exposure to surface-water runoff that has contacted the waste. Contaminated runoff might enter the food chain in several ways. First, runoff can contaminate ground water or surface water used for drinking, or it can contaminate nearby fields or irrigation ditches that are used by crops or livestock. This pathway is of concern only for aboveground vaults. We are currently unaware of plans by any State or Compact to use an aboveground vault, so to some extent the issues associated with modeling surface runoff are of secondary importance. Nevertheless, the capability to conduct such analyses may become important in the future.

We conclude that the surface-water transport models in the methodology are adequate for producing estimates of the dose that should be conservative in most cases. The models are very simple, and can be implemented either as hand calculations, or by using GENII-S. Similarly, the Regulatory Guide 1.111 air-transport models are very simple, and can be implemented easily, either in GENII-S or by using hand calculations.

The exposure pathway models are likely to be the ones containing Hanford site-specific parameters. A careful evaluation of the exposure pathway models is needed to establish confidence in GENII results. In addition, most low-level waste performance assessment analyses include (at most) exposures from drinking contaminated water, and exposures to contaminated crops and livestock. These calculations are a very small subset of the overall capabilities of GENII, so that using GENII to evaluate these simple exposure scenarios often seems excessive.

We, therefore, propose to develop a simple generic pathway analysis that includes only the pathways used most often in low-level waste performance assessments. This analysis might be implemented as a spreadsheet application or a simple, clear FORTRAN program. This simplified analysis has several advantages over GENII. First, the assumptions and limitations of the analysis can be made more transparent to the user than they are in a complicated code like GENII. Second, the concerns about possible site-specific assumptions in GENII are eliminated. GENII does need to be retained in the methodology for its generality and flexibility, but most performance assessment calculations can be done on a much simpler level.

7. Dosimetry Modeling

Current Approaches

Once the intake of radionuclides for a person have been established based on exposure pathway models (see Chapter 6), dosimetry models are needed to estimate the effect of this intake on the human body. In the methodology, internationally accepted dosimetry models [ICRP 26, 1977] based on dose-conversion factors [ICRP 30, 1982-1988] are used. The dosimetry models in the methodology are implemented in the computer code GENII [Napier et al., 1988].

Status

The ICRP dose conversion factors are based on a model of the human body. Critical organs are represented as well-mixed cells, and transfer factors are specified for the interchange of radionuclides between the organs. These transfer factors are specified by a combination of human and animal radiological studies, which are blended using professional judgment.[1] These models have not been compared against human databases of radiological exposures. Furthermore, there is no attention given to the uncertainties in the models: parameter uncertainty is acknowledged, but then neglected in the ICRP methodology. Olague et al. [1993] recommend a formal parameter uncertainty analysis to evaluate the uncertainty in ICRP 30 dose conversion factors. In addition, the conceptual model uncertainty is large for low dose rates. There is statistically significant evidence that moderately low levels of radiation may be beneficial to humans (radiation hormesis) [Luckey, 1989; Cohen, 1990]. For very low levels of radiation (25 mrem), stochastic health effects are so small that they can only be determined using epidemiological studies with populations the size of the entire Earth's population [Gershey et al., 1990]. This means that we will never be able to identify whether stochastic health effects actually exist for very low dose rates. Therefore, this represents a very large conceptual model uncertainty in the dosimetry models.

The ICRP has issued ICRP 60, containing updated recommendations since the publication of the original methodology [ICRP 60, 1990]; these recommendations supersede the recommendations of ICRP 26 [ICRP 26, 1977]. There are both minor and major differences between ICRP 26 and ICRP 60. Among the minor changes, effective dose equivalent is now merely called *effective dose*; doses to individual organs are now called *equivalent doses*, a reversion to an earlier nomenclature. Committed equivalent and committed effective doses are defined comparably to their earlier counterparts: they are the time integral of the dose following intake over 50 years for adults or 70 years for children. The term "non-stochastic" health effects has been replaced by *deterministic* health effects.

Of greater importance in low-level waste performance assessment is a change in organ and tissue weighting factors used in calculating the effective dose. Differences between tissue weighting factors between ICRP 26 and ICRP 60 are shown in Table 7.1. The weighting factors have been revised in an updated attempt to ensure that the effective dose would represent the same level of risk of stochastic health effects (denoted detriment in the ICRP documentation) regardless of the tissue or organ involved.

Evaluation

In spite of these changes, the ICRP 26/30 dose factors are still widely accepted, and are still considered to be a standard. Available published guidance [Eckerman et al., 1988] does not contain enough information to calculate effective doses according to the ICRP 60 standard; the guidance is for the ICRP 26 standard. We, therefore, conclude that these models are currently the best available from a defensible regulatory standpoint for use in the methodology. The ICRP 60 standard should be adopted once guidance is available on values for dose conversion factors.

However, we also note that to use these models, it is necessary to establish a regulatory position that maximum doses will be based on the ICRP standard man. Other assumptions about the person receiving the dose will result in different dose calculations. In keeping with the recommendations for air transport, surface-water transport, and food-chain analysis, we recommend that until the evaluation of GENII is complete, dosimetry analyses should be done using hand calculations. The dose conversion factors in GENII are not Hanford-specific, but if the other portions of the analysis are done outside of GENII, it would not make sense to use GENII just to generate dose conversion factors. The dose analyses can be implemented quite easily; we recommend that a generic simplified analysis that only includes ingestion doses should be developed for use in these analyses. The dose conversion factors can be found in Eckermann et al. [1988].

[1]O'Neal, B., personal communication, SNL, 1991.

Table 7.1 Differences in tissue weighting factors between ICRP 26 and ICRP 60

Tissue or Organ	Tissue weighting factor (ICRP 26)	Tissue weighting factor (ICRP 60)
Gonads	0.25	0.20
Red Bone Marrow	0.12	0.12
Colon	----	0.12
Lung	0.12	0.12
Stomach	----	0.12
Bladder	----	0.05
Breast	0.15	0.05
Liver	----	0.05
Oesophagus	----	0.05
Thyroid	0.03	0.05
Skin	----	0.01
Bone Surface	0.03	0.01
Remainder	0.30	0.05

8. Summary

The purpose of this report is to identify models that should be incorporated into the low-level waste performance assessment methodology. Each modeling area of the methodology has been reviewed in this report, and some additional areas of concern have been discussed.

We have introduced an additional modeling area from our earlier discussions: the area of subsurface transport of gases, reflected in Figure 8.1, which is an updated figure showing the modeling areas in the methodology.

The primary recommended modeling changes for the methodology (Table 8.1) are incorporating NEFTRAN II for the analyses of source-term and ground-water transport, incorporating MODFLOW to improve the flexibility of the methodology in treating saturated-zone flow, developing a simplified application to replace GENII for many applications, and replacing VAM2D by a code such as VS2DT after adapting the latter code to handle decay chain transport. We believe that NEFTRAN will provide additional flexibility and intermediate level of complexity that will be useful in some circumstances. The recommended code development to VS2DT is intended to solve persistent issues about using VAM2D: the code is proprietary, and it does not have ideal quality assurance. These issues are expected to become more important if the code is used in actual licensing analyses. These recommendations are shown in Figure 8.2.

Other key recommendations are as follows. A strong validation program is needed to begin building confidence in all of the models in the methodology as they apply to their regulator purpose. Formal uncertainty analysis methods are recommended for model uncertainty, parameter uncertainty, and uncertainty about the future of the site. We recommend that model uncertainty be addressed by analyzing multiple conceptual models in parallel, and differentiating between them using site-specific validation and model intercomparison. The uncertainty analysis approach recommended consists of allowing multiple conceptual models, a Monte Carlo analysis with Latin Hypercube Sampling, coupled with a full analysis of possible future states of the site. We recommend that formal approaches are needed for acknowledging all assumptions and their links to site-specific data. At its simplest level, this can be an explicit recognition of the assumptions made in the analysis. However, eventually it will be desirable to develop a formal method for using site-specific data to develop adequate (conservative) regulatory conceptual models. Finally, we recommend that effort is needed to improve the user friendliness of the methodology.

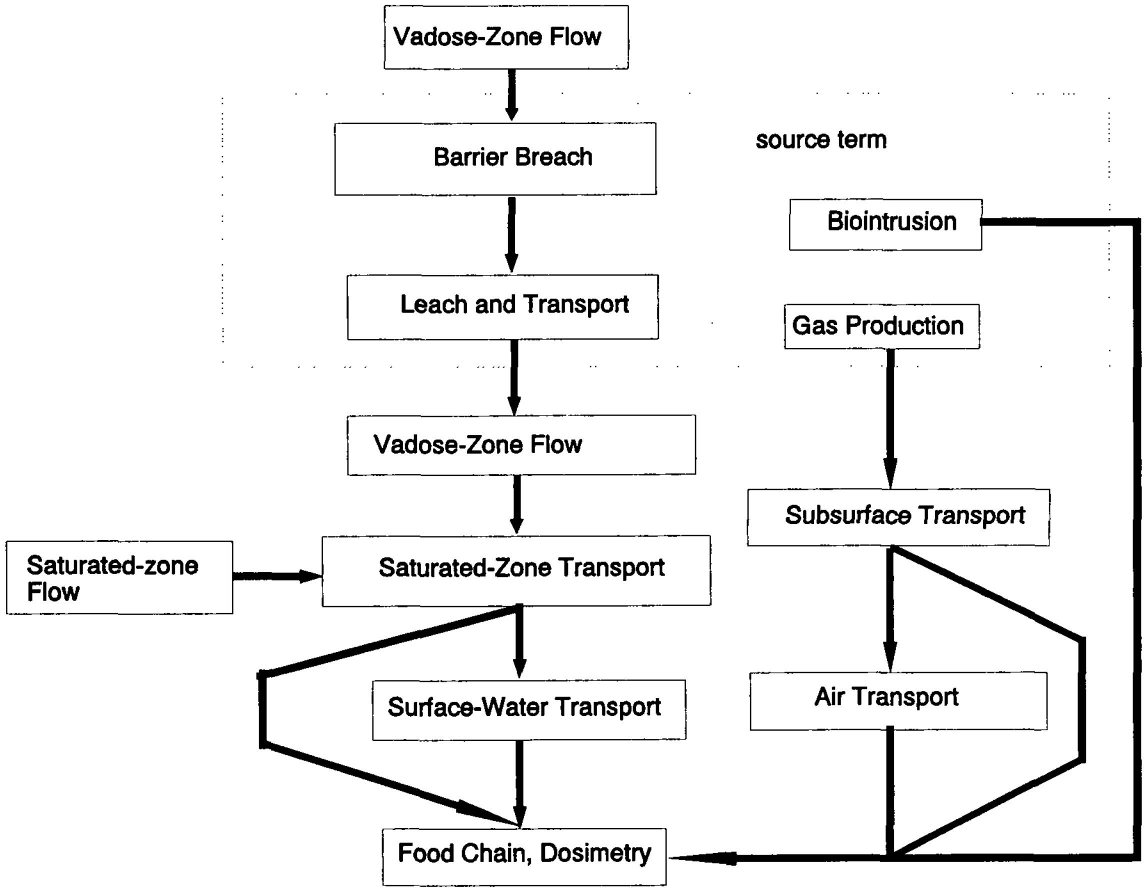

Figure 8.1 Updated processes in the methodology

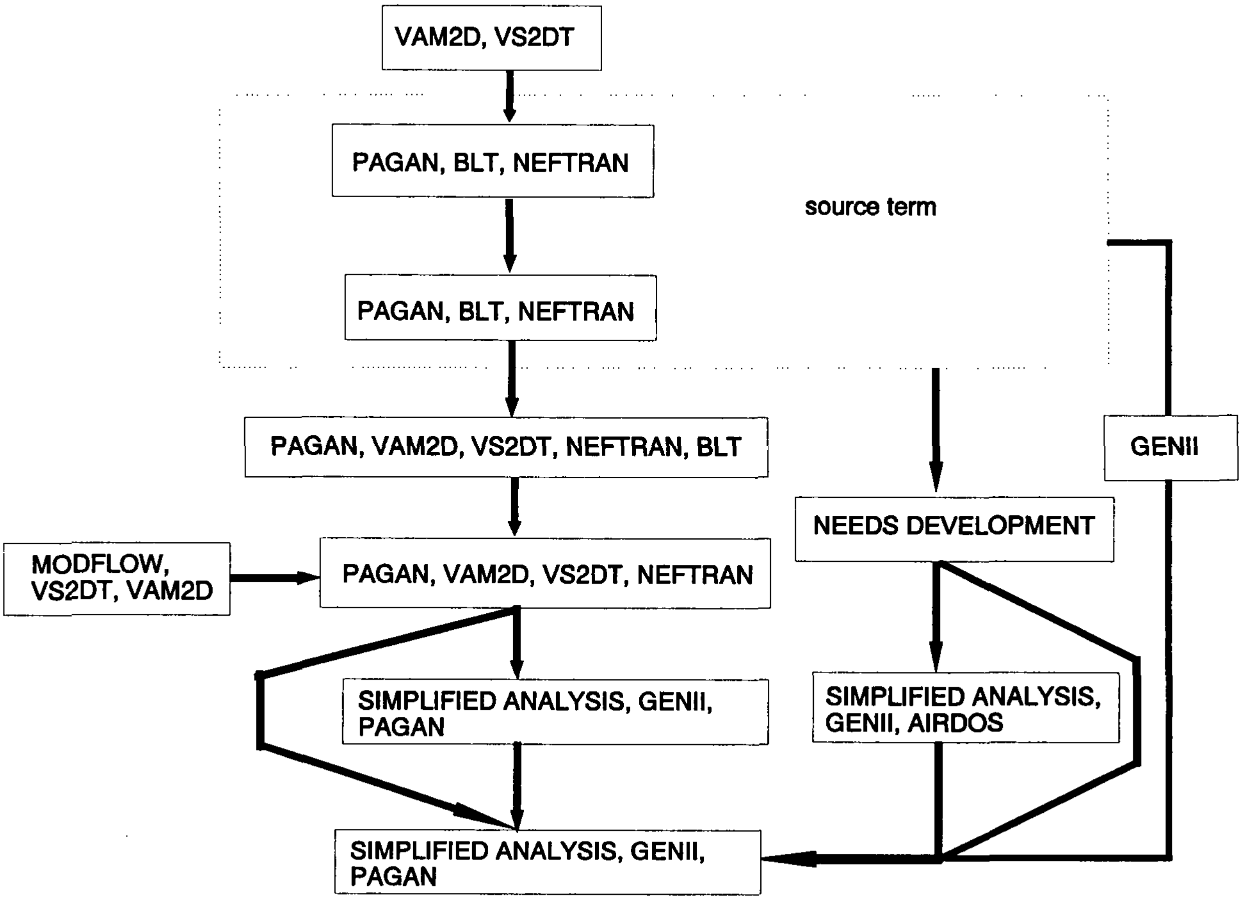

Figure 8.2 Current recommendations for the methodology

Table 8.1 Recommended changes to the models in the methodology

Modeling Area	Recommendation
Ground Water	Incorporate VS2DT, NEFTRAN II, and MODFLOW Adapt VS2DT to conduct chain decay analyses
Source Term	No change to concrete modeling Incorporate simplified Breach and Leach models Incorporate NEFTRAN II Allow tabular input Evaluate subsurface gas transport
Surface Water	Implement simplified analysis Use MODFLOW to evaluate interactions with ground water
Air	Implement simplified analysis
Food Chain	Write a simplified ingestion application
Dosimetry	Write a simplified ingestion application
Overall (All Areas)	Formal uncertainty analysis needed Improved user friendliness needed Strong validation program needed

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11. ABSTRACT *(200 words or less)*

This report represents an update to our earlier reports on low-level waste performance assessment. This update addresses needed improvements and recommended approaches to the existing state of the art in modeling, treatment of uncertainty, and use of data. Greater attention is paid to developing an integrated approach to performance assessment than was done in earlier developments of the methodology. Furthermore, insights are being developed by participating in validation exercises, and by evaluating which validation data are needed to improve confidence in the methodology. It is emphasized that the performance assessment methodology update is a work in progress; the recommendations given here will form the general directions toward which the methodology is heading, but some of the specific approaches may continue to evolve as the research progresses.

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