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Proceedings of the Symposium on Uncertainties Associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste

Gatlinburg, Tennessee
March 9–13, 1981

Sponsored by

U.S. Nuclear Regulatory Commission
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UNCERTAINTIES ASSOCIATED WITH THE REGULATION OF THE GEOLOGIC DISPOSAL OF HIGH-LEVEL RADIOACTIVE WASTE

Gatlinburg, Tennessee
March 9-13, 1981

Edited by
David C. Kocher

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PREFACE

The primary purpose of this symposium was to provide a forum for wide-ranging discussions on (1) technical aspects related to the development of standards for regulating geologic disposal of high-level radioactive waste, with particular emphasis on the sources and magnitudes of uncertainties associated with current methods for predicting post-closure repository performance and potential health risks to future generations, (2) important licensing and regulatory issues involved in geologic waste disposal, and (3) the current social and political climate in which issues of high-level waste management are being debated. Significant contributions to these discussions were provided by representatives from the U.S. Nuclear Regulatory Commission (NRC), U.S. Department of Energy, U.S. Environmental Protection Agency (EPA), various contractors of these three agencies, and other interested parties not affiliated with the Federal Government or its contractors. The symposium was timed to coincide with the development and publication by the NRC of the proposed technical criteria for regulating the disposal of high-level radioactive wastes in geologic repositories. An additional subject of considerable interest at the symposium was the development of environmental radiation protection standards for high-level radioactive waste by the EPA and the relationship of these standards to the NRC's proposed technical criteria. Financial support for the symposium was provided by the NRC through the Office of Standards Development, which has since been consolidated with the Office of Nuclear Regulatory Research.

The success of this symposium was the direct result of the cooperative efforts of many individuals. The organizing committee consisted of myself, Sherri Cotter, and Rowena Chester from Oak Ridge National Laboratory (ORNL), and Pat Comella and Nick Costanzi from the NRC. I would like to recognize the invited speakers (Jack Martin, Craig Roberts, Mike Cullingford, and Pat Comella of the NRC, Dan Egan of the EPA, Geoff Eichholz of the Georgia Institute of Technology, Georgia Yuan of the Natural Resources Defense Council, Karin Sheldon of the Sierra Club Legal Defense Fund, Jocelyn Olson of the State of Minnesota, Joe Lieberman of Nuclear Safety Associates, and John Stucker of the State Planning Council on Radioactive Waste Management), the workshop co-moderators (Jim Campbell of INTERA Environmental Consultants, Ron Iman of Sandia National Laboratories, and Elly Triegel and Keith Eckerman of ORNL), and the session chairmen (Nick Costanzi of the NRC, Paul Rohwer and Rowena Chester of ORNL, Nester Ortiz and Bob Cranwell of Sandia, and Mike Foley of Pacific Northwest Laboratory) for their invaluable efforts in defining the framework of the symposium and seeing that the sessions ran smoothly. The enthusiastic response of these individuals to my pleas for help is gratefully acknowledged.

All symposium attendees are indebted to Bonnie Reesor and her staff in the Conference Office at ORNL for their tireless attentions to organizing this event. We also owe a special thanks to Wilma Minor, Robin Smith, and Malinda Hutchinson of the Health and Safety Research Division, Alice Richardson and Bob Eldridge of the Information Division, and John

Goan, Ron Harris, and Dave White of the Instrumentation and Controls Division at ORNL for their essential contributions to this symposium.

It is with regret that I acknowledge that some of the papers presented orally at the symposium were not submitted by the authors for publication in these proceedings. These oral presentations have been acknowledged by publishing the titles and authors, and, for one of the contributed papers, the abstract which had previously been submitted to the organizing committee has been published with consent of the authors. The invited paper given by Dan Egan of the EPA included a handout provided for all symposium attendees describing selected provisions of EPA's disposal standards for high-level and transuranic radioactive wastes, but we have not published this draft document here. The invited paper by John Stucker of the State Planning Council on Radioactive Waste Management also included a handout of the Interim Report of the Council which was submitted to President Reagan on February 24, 1981. We have not published this report here, because the final report of the State Planning Council has since been released along with a series of technical papers supporting their recommendations.

The papers in these proceedings are presented in the same order as at the symposium. The manuscript was prepared from camera-ready photomasters supplied by the authors, and no editing or altering has been done except as requested by the authors. All papers have been reviewed and cleared as necessary by the institution with which the authors are affiliated.

The symposium emphasized the many and diverse uncertainties associated with high-level waste disposal. As most attendees are aware, however, it is certain that there are no elevators in Hamburg, Arkansas!

David C. Kocher
Chairman
Organizing Committee

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Chairman

Frank A. Costanzi

U. S. Nuclear Regulatory Commission

UNCERTAINTIES IN THE REGULATION AND LICENSING OF A HIGH-LEVEL WASTE REPOSITORY

J. B. Martin
M. J. Bell
M. R. Knapp

ABSTRACT

This report summarizes the rationale for the draft of the technical part of the Nuclear Regulatory Commission's regulation for geologic disposal of high level waste. This regulation will reduce uncertainty in predicting repository performance through the multibarrier approach, with performance objectives for three barriers which are the waste package, the engineered system, and the geologic setting. The bases for the numerical criteria assigned to each objective, as well as a fourth criterion, that of design of the facility to permit retrieval of the waste, are discussed.

INTRODUCTION

The objective of waste disposal is to isolate the waste from the environment for as long as is necessary to protect public health and safety. The EPA is preparing a safety standard which will form the overall safety standard and which will set limits on ambient levels of radioactivity resulting from any disposal system. While the EPA standard has not yet been published in proposed form, we expect that it will explicitly limit radioactive materials released to the environment to very small amounts for a long time. The licensing proceeding will essentially involve demonstration of reasonable assurance of compliance with this standard. The only available way to deal with this requirement is by using mathematical models to project repository performance far into the future. This presents the challenge of rigorously predicting future performance while limited to methods and data which are fraught with limitations and uncertainties.

The NRC staff has given this problem much consideration in selecting both a regulatory approach and the repository performance objectives contained in 10 CFR Part 60. We have emphasized those choices which we consider appropriate to facilitate expeditious closure of a licensing proceeding. Those choices which may work out in the long run but which are subjective or hard to prove rigorously and which could involve months or years of delays in demonstrating compliance in the context of a formal licensing proceeding have been de-emphasized.

SOURCES AND SIGNIFICANCE OF UNCERTAINTY

Unlike nearly all other regulatory processes in this country, the regulation and licensing of a high-level waste repository will occur in the absence of any prior experience with comparable systems. While there may be extensive experience with some system components, or certain physical phenomena, the overall repository system will be evaluated by means of predictions, which will be subject to considerable uncertainty.

One contribution to the uncertainty of such evaluations arises from the limitations that exist in our understanding of the basic physical processes. For example, Bredehoeft, England, Stewart, Trask, and Winograd [1] discuss the complex geologic and hydrologic impacts of the presence of waste in a repository, and DOE [2] recognizes that the uncertainties associated with waste-rock interactions are a major area of concern.

A second contribution to overall uncertainty consists of limitations in being able to characterize a site. It is literally not possible to accurately assess all the relevent geologic conditions and physical parameters in the volume of rock that influences the path and rate of radionuclide migration.

There is also the potential for unanticipated interactions in complex systems. Unanticipated interactions have occurred in many engineered systems whose components were presumably well characterized.

The above sources of uncertainty limit our ability to understand and model the physical and chemical phenomena of interest. However, even if these were known perfectly, those aspects of repository evaluation which rely on numerical modeling are subject to additional uncertainties. For example, the National Academy of Sciences [3] and the Interagency Review Group on Nuclear Waste Management [4] have noted uncertainties associated with the use of numerical modeling methods, which may introduce errors and uncertainties through the use of approximation techniques, undiscovered errors in algorithims, and undiscovered logic errors in complex computer codes.

TECHNICAL APPROACH OF 10CFR PART 60

Compensation for uncertainty that would otherwise confound demonstration of compliance with the EPA standard is an essential part of the NRC staff's regulatory approach.

Three alternative approaches to regulating geologic disposal of HLW were considered in the development of the technical criteria of 10 CFR Part 60. Each was examined in light of its ability to compensate for the major uncertainties in the quantitative prediction of the performance of a geologic disposal system. The alternatives considered were:

1. Regulation of repository systems by setting a single overall performance standard that must be met by the system. The performance standard in this case would be the EPA standard;
2. Regulation of repository systems by setting minimum performance standards for each of the major system elements as well as requiring the overall system to conform to the EPA standard; and
3. Regulation of repository systems by setting numerical criteria on critical engineering attributes of the system.

The NRC staff has examined each of these alternatives from the standpoint of its ability to compensate for uncertainty in evaluating compliance with the EPA standard in a licensing proceeding. The NRC staff further examined each alternative with two objectives in mind: (1) providing as much guidance and detail as may be warranted by generic considerations and (2) avoiding undue constraints upon system design.

The alternative of setting a system performance standard is often referred to as the "systems approach." It has as its principal advantage the fact that regulation would be through a single figure of merit which is the overall system performance. This alternative leaves maximum flexibility for the designer to make tradeoffs among components of the system.

Unfortunately, the systems approach as interpreted above is not practical from a regulatory point of view. As noted earlier, a quantitative assessment of the expected performance of a geologic repository is a complex and difficult task. The results of such an assessment contain the uncertainties described above. Compensation for such uncertainties in the systems approach would require imposing ancillary requirements on the system to provide necessary conservatism. Finally, a single standard does not require the designer to take advantage of the conservatisms that are inherent in independent barriers even when, as here, no one of the barriers is required to be capable of meeting the EPA standard by itself.

The second alternative establishes major subelements of the repository system, called barriers, and assigns minimum performance objectives to each while maintaining the EPA standard as the measure of overall system performance. This alternative has two advantages over the systems approach. First, if the barriers are chosen judiciously, multiple barriers can be prescribed which act independently and thereby enhance confidence that the wastes will be isolated. Second, by judicious choice again, the uncertainty in the evaluation of repository performance can be reduced by requiring the barriers to perform in ways which reduce their relative contribution to the uncertainty.

The third alternative, use of numerical criteria for certain engineering attributes of the system (a peak canister wall temperature, for example) has two major advantages. It provides clear guidance to designers as to exactly what is required for licensing. Secondly, the criteria can be selected to compensate directly for uncertainty by introducing conservation into the acceptable standards for each significant attribute of the system.

The approach also has several disadvantages. Of the three alternatives, it is most restrictive of design flexibility. In fact, it begins to force the regulator into a designer role. Further, to be effective, the criteria must be set on the basis of existing knowledge. Therefore, the approach cannot fully accommodate the benefits of future research and development work.

After lengthy consideration of the three alternatives discussed above, the NRC staff has selected the multiple barrier approach, and has set minimum performance standards for three separate system elements.

SELECTION OF AND PERFORMANCE OBJECTIVES FOR THE MAJOR BARRIERS

For a given initial inventory, the overall performance of a geologic repository with respect to releases to the biosphere can be described by three characteristics: (1) the length of time after closure during which radionuclides are contained, (2) the rate at which radionuclides are released to the geologic setting after containment fails, and (3) the travel time through the geologic setting for radionuclides to reach the biosphere[5]. The NRC has chosen to use these characteristics for identifying and quantifying performance standards for the system elements.

The uncertainties in evaluating the performance of the system in the near field caused by emplacement of the waste are to a large degree time dependent. Many of the perturbations that are expected to occur are the result of the increased temperature in the host rock due to radioactive decay heat. Temperatures peak and begin to fall within the first few hundred years after the waste has been emplaced. During the same period, total radioactivity of the waste decays by several orders of magnitude. As the temperature decreases many of the uncertainties in assessing near-field behavior decrease as well. The decrease in total radioactivity also represents a decrease in the source term available to be released.

Our approach for this initial period of high temperatures and radionuclide inventory is to contain the wastes within a stable package that confines the radionuclides within a physical boundary. Such "waste packages" can be designed to provide assurance of their ability to perform to specifications under anticipated near-field conditions. Thus, this alternative provides a reasonably verifiable barrier to compensate for geologic uncertainty during the period when the specific activity of the waste is high and the perturbations of the natural systems are large.

Engineered barriers can also be designed to limit the rate at which radioactive materials are released from the engineered system after the containment period and thereby supplement the geologic system in limiting the rate of release to the environment.

The rate at which radionuclides are released to the site can be limited by using waste forms and overpacks that limit the release from the package to some maximum rate, by emplacing materials (e.g., tailored backfill) around the waste that have chemical properties that retard or inhibit radionuclide transport, or by some combination of the above. Either way, in principle, the source to the geologic system can be maintained at a low level and can be tested to verify release rates under anticipated conditions.

Engineered barriers designed to minimum performance standards can provide reasonable assurance that the overall performance objective of the high-level waste disposal system will be met for an initial period. After containment failure, engineered barriers can be designed to limit the rate of release of radioactive materials from the repository. However, once materials are released from the engineered system, the site must provide whatever additional isolation is needed in order to meet environmental standards. Reliance on the geology to provide one of the major barriers to releases also introduces diversity into the waste disposal system that can compensate, in part, for unanticipated failures of the engineered system. The geologic setting is characterized by a variety of parameters that could themselves be considered individual barriers. Some examples of such parameters are permeability, interstitial groundwater velocity, and equilibrium sorption coefficients. However, all geologic parameters combine to determine two characteristics of the geologic setting: (1) the transport time of groundwater from the underground facility to the accessible environment and (2), assuming radionuclides have escaped the engineered system, the transport time of individual radionuclides from the underground facility to the accessible environment. The second characteristic differs from the first in that it takes into account the geochemical characteristics of the medium and includes retardation of the radionuclides.

After careful consideration of the uncertainties in geochemistry and its ability to predict radionuclide retardation, the NRC staff decided to place the requirement on the transport time for groundwater.

WASTE PACKAGE PERFORMANCE

In light of repository thermal conditions and waste characteristics as a function of time, the staff examined a range of containment times as performance objectives for the design of the waste package.

We examined the following alternatives for the waste package containment time:

- (i) 300 years,
- (ii) 1000 years, and
- (iii) 10,000 years.

(i) Containment of the wastes for 300 years, as suggested by DOE[6] in its comments on the NRC Advance Notice of Proposed Rulemaking[7], would prevent releases from occuring until decay causes the bulk of the fission products to disappear and the heat generation rates to decrease by about 2 orders of magnitude for waste from all fuel cycles.

A minimum containment time of 300 years has the disadvantage, however, that packages fail and release begins to occur when temperatures in the repository are near their peak and when the thermal gradients that provide the driving force for convective transport are still relatively high. Under these conditions of high temperature and high thermal gradients, hydrothermal reactions of the waste form and mineral phase changes of the backfill materials and nearfield host rock will be most severe, and the leaching and transport of radionuclides through the underground facility will be most difficult to evaluate. A containment time of 300 years permits considerable uncertainty in the prediction of the releases from the underground facility due to the effects of temperature on leach rate, hydrologic flowpaths, viscosity, rock permeability and geochemistry.

(ii) Containment for 1,000 years would prevent releases from occurring until most of the fission products have disappeared and decay heat generation rates have decreased by three orders of magnitude. More important, containment for 1,000 years has the effect of delaying releases until temperatures in the underground facility are past their peak and are decreasing and until thermal gradients in the underground facility and surrounding rock have decreased substantially from the first few hundred years. Lower temperatures and temperature gradients allow release rates and radionuclide migration rates to be predicted with greater confidence.

(iii) Containment for 10,000 years would prevent releases from occuring until the fission products have essentially disappeared and some intermediate-lived transuranics (e.g., Am-241, half life 450 yr) would have decayed to negligible levels. Heat generation rates would have decreased by over four orders of magnitude and temperatures and thermal gradients in the repository and host rock would have nearly returned to pre-waste emplacement conditions. Under these conditions, we consider that many of the transport processes can be modeled with greater confidence, and that analogies between the transport of actinides and their daughters and migration from natural ore bodies are more reasonable. However, design of a package to contain wastes for 10,000 years requires a considerable extrapolation beyond those concepts DOE has considered in the past and for which any test information exists. Costs for such a package are uncertain and may not be justified by the reduction in uncertainty that would be achieved.

The staff considers that a containment requirement for the waste package of 300 years is insufficient to increase confidence in long-term performance predictions. If packages fail and migration begins after 300 years, in order to evaluate overall repository performance it will be necessary to consider transport from the waste packages through the disturbed zone under highly uncertain environmental conditions. This situation will result in substantial uncertainties in calculation of the source term for the transport through the geologic setting. On the other hand, containment for 10,000 years would delay the onset of radionuclide migration until temperatures and temperature gradients in the disturbed zone had returned to near pre-emplacement conditions, and the source term for migration could be predicted with much less uncertainty. The staff considers that if containment for 10,000 years could be achieved, it would reduce uncertainty in prediction of long-term performance by reducing the source term available for migration, by providing better understanding of the chemical behavior of the waste when migration begins, and by delaying the start of migration until the perturbations in the geologic environment due to temperature have substantially decreased. At present the amount of the reduction in uncertainty cannot be quantified and estimates of costs to achieve containment for 10,000 years are very tenous. However, the staff considers that DOE should be encouraged to investigate the practicality of a package with a 10,000 year life. Therefore, we have framed our performance objective for the waste package such that DOE is required to design the package to provide reasonable assurance of containment for at least 1,000 years. We consider that containment for 1,000 years will substantially reduce the hazard associated with a release from the package and will increase our confidence in our ability to evaluate the effectiveness of the disposal system to maintain releases to the environment to within the EPA standard. We further consider that such a requirement is achievable at reasonable cost by a reasonably straight-forward extrapolation of current DOE programs. However, we consider containment for periods as long as 10,000 years to be a desirable goal and consider that DOE should continue to develop information on the performance and costs of packages for long-term containment and to include them in repository system if found to be reasonable achievable.

LONG-TERM PERFORMANCE OBJECTIVES FOR THE ENGINEERED SYSTEM

The NRC staff has calculated the effect of the annual release rate on the fraction of long-lived nuclides released from a repository system [8]. Limiting the release rate from the engineered system compensates for uncertainty in the prediction of long-term performance by reducing the source term that is available for transport through the hydrologic system. The calculations show that fractional release rates in the range of 10^{-5} to 10^{-7} per year result in a significant reduction in the fraction of several environmentally significant long-lived isotopes that could potentially be released from the repository, which could result in corresponding reductions in population doses.

Based on the above considerations, the NRC staff considered the following alternatives for the fraction of the waste inventory released per year from the engineered system after the containment period:

(i) a range of 10^{-3} to 10^{-4}/yr, which is typical of leach rates of many borosilicate glasses at low temperature,
(ii) a release rate of 10^{-5}/yr, and
(iii) a release rate of 10^{-7}/yr.

(i) An annual release rate of 10^{-3} to 10^{-4} of the waste inventory is insufficient to achieve much reduction in the quantities of long-lived material that would be released and would result in almost total reliance on the geology and the far field geochemistry to provide isolation for the long-lived radionuclides in the waste.

(ii) We consider that, based on technology currently being developed by DOE, annual release rates of 10^{-5} of the waste inventory are achievable at reasonable cost using combinations of waste forms and engineered barriers. In addition, a release rate after containment failure of 10^{-5} of the waste inventory per year, while not adequate to isolate waste on its own merit, is long enough that significant decay of long-lived species takes place before release. This limit will contribute to reducing doses to both populations and the maximum individual, and will substantially reduce our reliance on less certain geochemical retardation to limit releases to the accessible environment.

(iii) An annual release rate of 10^{-7} of the waste inventory after containment failure will reduce dose to individuals and releases to very low levels with little or no reliance on geochemical retardation. An engineered system that could meet this criterion would best satisfy our objective of reducing reliance on characterization and modelling of the behavior of the far-field geochemical system and placing reliance on known materials whose properties can be controlled and tested.

The staff considers that an annual release rate after failure in the range 10^{-3} to 10^{-4} of the package inventory is insufficient to achieve our objectives, since little reduction would be achieved in the quantity of long lived radioactive material released, and the repository system would rely almost entirely on the site to provide long term isolation. The staff considers that if an annual release rate from the engineered system as low as 10^{-7} of the package inventory at 1000 years could be achieved, it would compensate for uncertainty in the calculation of the transport of radionuclides through the groundwater pathway by limiting the source term to a relatively low value. Maintaining the release rate at a value this low would result in decay of most radionuclides within the engineered system. At present the amount of the reduction in uncertainty cannot be quantified, and the costs to achieve a release rate this low are very uncertain. However, the staff considers that DOE should be encouraged to investigate the practicality of maintaining release rates at very low levels. Therefore, the staff developed a minimum performance objective of an annual release rate of individual nuclides of at most 10^{-5} of the package inventory. This requirement is placed only on nuclides that contribute more than 0.1% to the release rate. We consider that a release rate if 10^{-5} per year is low enough that appreciable benefit will be gained by radioactive decay before release, and is achievable at reasonable cost by methods currently being developed by DOE. However, we consider a release rate of as low as 10^{-7} per year to be a desirable goal and consider that DOE should continue to develop information on materials and costs to achieve such low release rates and should include them in the repository system if found to be reasonably achievable.

MINIMUM PERFORMANCE OBJECTIVES FOR THE GEOLOGIC SETTING

After deciding that the appropriate performance objective for the geologic setting is a minimum groundwater travel time between the disturbed zone and the accessible environment, the staff next considered what that minimum travel time should be. Travel times of one hundred years or less would require considerable reliance on the geochemical system to ensure that the overall performance objective for the repository is met. While geochemical retardation is expected to be a strong factor in providing waste isolation, there will be considerable uncertainty in the magnitude of its contribution. This uncertainty results from the fact that it is very difficult to know how much geochemical retardation will occur. There is currently no concensus within the scientific community on how such an evaluation can be made. This situation would likely cause geochemistry to be a major source of contention in a licensing proceeding. A travel time of only one hundred years does not provide margin to compensate for uncertainties. Further, from groundwater dating studies, travel times well in excess of 100 years are known to be achievable in a variety of hydrogeologic environmentals; we would not consider a travel time for an unperturbed site as low as 100 years to be suitable for a repository. We therefore considered longer times, viz 1000 and 10,000 years.

A travel time for groundwater from the repository to the accessible environment of 10,000 years would be sufficient for many shorter-lived nuclides to meet the system's overall performance objectives with no reliance on site geochemistry. For several long-lived nuclides, e.g., Pu-239, Tc-99, some reliance on geochemical retardation would be required; but generally a considerable margin would exist between Kd's measured in the laboratory and those required to meet the release limits of the EPA standard[9]. We are uncertain, however, to what extent such a groundwater travel time is achievable. We do not want to rule out otherwise good repository sites by unnecessarily restrictive requirements. However, such a travel time could be used as a goal.

In sum we have framed our site performance objective so that the travel time from the repository to the accessible environment shall be at least 1000 years; and we intend that DOE consider during site screening that sites with longer water travel times are preferred.

RETRIEVABILITY

The repository will be developed in several stages. Design and operating decisions for each step will build on the information learned from each previous step. The decision to permanently seal the repository will be based on information on repository performance obtained during the operating period. The NRC staff considers that the option to retrieve the wastes must be preserved long enough to complete a program of monitoring and verification of repository performance prior to permanent closure. The design must also ensure that the option is preserved long enough to permit a decision to decommission the repository or take corrective actions based on the evaluation of the results of the verification program, including the time required to retrieve all or part of the wastes, if shown to be necessary by the results of the program. Since some of the assumptions and issues that will need to be verified and resolved by the verification program may not be identified until the underground facility is excavated, it is not possible to specify prior to construction the content of the verification program, or how long it will take. We expect the verification program to evolve throughout the operating lifetime of the repository. On the other hand, important design decisions will need to be made prior to submitting an application. Some of these design decisions will affect the length of time available to take corrective action or conduct retrieval if found to be necessary. For example, the thermal loading of the waste in the emplacement areas will affect the temperature of the host rock and the stability of the underground structure. These phenomena will have a large effect on retrievability since the structure could become too unstable or the rocks too hot to safely recover the wastes. Therefore, we concluded that a retrievability period must be chosen early in the design process to permit the design to go forward.

A monitoring period of only 10 to 15 years after emplacement, as suggested by some, may not be sufficient to provide the information needed to make a decision to decommission. The design must also allow for the time required to thoroughly investigate problems that may be identified during the verification program, to evaluate the results of the program, and to take corrective actions, including retrieval of part or all of the waste, if found necessary. The design of the facility must provide access for the time necessary to carry out these operations, or the ability to conduct them will be precluded.

Therefore, we have required that the repository be designed so that the waste could be retrieved on a reasonable schedule starting at any time up to 50 years after waste emplacement is complete. We can foresee no situation where protection of the public health and safety would require the waste to be removed very rapidly. Removal operations could be performed over a period of years or decades without an imminent health and safety hazard. We therefore consider that a reasonable schedule is one where the waste could be retrieved in about the same overall time that the repository was constructed and wastes were emplaced. We do not intend to preclude a decision to decommission the repository before 50 years has elapsed, if sufficient data are available to support an earlier decision, and if the people charged with the decision to seal the repository are satisfied. However, we do not want the underground facility design to be such that retrieval would be so expensive or difficult or entail such high occupational exposures that the option is foreclosed and needed corrective actions could not be taken.

Maintaining the option to retrieve the wastes does not entail keeping the mined areas open, although DOE may choose to do so in some geologic media. A design in which the emplacement rooms are backfilled and sealed but corridors and shafts were kept open and surface handling facilities are maintained could be acceptable, provided that the rooms could be remined and the wastes removed, if necessary. Remining of the backfill should not be precluded because of high temperatures or because backfill is needed for structural stability.

SUMMARY

The NRC is responsible for determining that any proposed deep geologic repository for high level waste complies with appropriate EPA standards. To reduce the uncertainties in making such a determination, the NRC has adopted the multibarrier approach, and has defined performance objectives for three barriers. These are the waste package, which shall contain the waste for 1,000 years, the engineered system, which shall limit annual releases of waste after 1,000 years to 10^{-5} of the inventory of waste in the repository, and the geologic setting, which shall provide a groundwater travel time of at least 1,000 years. As a further step to protect public health and safety, the NRC considers that the repository should be designed to permit retrieval of waste for a period of up to 50 years after completion of emplacement operations.

Subsequent to the oral presentation of this paper on March 10, 1981, the NRC formalized these objectives in the technical portion of 10CFR60 by publishing it as a proposed rule in the Federal Register [10] on July 8, 1981 for a public comment period which ended on November 5, 1981.

REFERENCES

1) Bredehoeft, J.D., England, A.W., Stewart, D.B., Trask, N.J., and Winograd, I.J., Geologic Disposal of High-Level Radioactive Wastes - Earth Science Perspectives, Geological Survey Circular 779, (1978), U.S. Geological Survey, Arlington, VA.

2) Draft Environmental Impact Statement on the Management of Commercially Generated Radioactive Waste, DOE/EIS-0046-D, (1979) U.S. Department of Energy, Washington, D.C..

3) Implementation of Long Term Environmental Radiation Standards: The Issue of Verification, (1979), National Academy of Sciences, Washington, D.C.

4) Report to the President by the Interagency Review Group on Nuclear Waste Management, (1979), National Technical Information Service, TID-29442, Springfield, VA.

5) Burkholder, H.C., Management Perspectives for Nuclear Fuel Cycle Wastes, Nuclear Waste Management and Transportation Quarterly Report, January through March, 1976, (1976), Battelle Northwest Laboratory for USNRC.

6) Comments on NRC Advanced Notice of Proposed Rulemaking, 10CFR60, Technical Portion, Department of Energy, July 15, 1980, Letter to Secretary, Nuclear Regulatory Commission.

7) Advanced Notice of Proposed Rulemaking, 10CFR60, Technical Portion, Federal Register , May 13, 1980 (45 FR 31393)

8) White, L.A., Bell, M.J., and Rohrer, D.M., "Regulation of Geologic Repositories for the Disposal of High-Level Radioactive Wastes", in Scientific Basis for Nuclear Waste Management, V.2, 1979, C.J. Northrup, Jr., Ed., Plenum Press, New York.

9) Isherwood, D., Geoscience Data Base Handbook for Modelling a Nuclear Waste Repository, NUREG/CR-0912, (1981), Lawrence Livermore National Laboratory for USNRC.

10) Proposed Rule, 10CFR60, Technical Portion, Federal Register, July 8, 1981 (46 FR 35280).

WRITING STANDARDS FOR GEOLOGIC DISPOSAL OF HIGH-LEVEL WASTE

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Disposal of high-level radioactive waste has been and continues to be a matter of high national interest. The Congress, Federal agencies, the States, and public interest groups have all actively reflected the public's continuing concern over whether these wastes can be safely disposed. Because of the long period of time over which the wastes remain potentially hazardous, their disposal presents formidable technological and institutional problems -- many persons would argue that the latter will be more difficult to overcome than the former.

The purpose of this conference is not to continue the debate, but rather to get on with the problem-solving -- and from a perspective that may be quite different from your customary view of this problem. We ask that during the next few days you view the problem of safe disposal of high-level radioactive waste from the Nuclear Regulatory Commission's perspective.

What do I mean by "NRC perspective?" When Congress passed the Energy Reorganization Act of 1974, establishing the Nuclear Regulatory Commission (NRC) and the Energy Research & Development Administration (ERDA) - the ancestor of the Department of Energy (DOE) - it gave ERDA/DOE the job of solving the commercial high-level waste (HLW) disposal problem technologically and NRC the job - in effect - of overseeing that problem-solving through the mechanism of licensing the disposal of HLW. The practical effect is that whatever solutions to the problem are decided on must not only be technologically acceptable, but also must be capable of effective regulatory oversight. The oversight is part and parcel of any technologically <u>AND</u> institutionally acceptable solution. That is to say that, in order to succeed, the approach chosen must be sound technologically and be demonstrable in a way that is credible to the public that the public health and safety and the environment will be protected. Therefore, both the DOE and NRC have roles that require the exercise of leadership if this venture is to suceed; DOE must define the solution technically and NRC must establish the basis for assuring that the public health and safety will be protected.

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The problem of finding such an acceptable solution is confounded by the fact that we are dealing with a newly developing activity for which there is virtually no previous experience. We recognize that some people would take issue with that statement and argue that relevant experience has been collected, in some instances over hundreds of years. They cite for example mining research, archeological entombment, relics of Celtic age salt mines, Project Salt Vault, Canadian experiments at Chalk River, Swedish tests at STRIPA and the Oklo uranium site in the Republic of Gabon. However, remember that we are viewing the problem from NRC's perspective and there is no denying the fact that the permanent disposal of HLW has never been licensed. So how shall we go about it? How shall we establish a regulatory framework within which licensing and regulation can be conducted and how are we going to establish criteria which will guide licensing decisions and regulatory actions? Again, how are we going to do it in a way credible to the public? The point is, given the high national interest and concern over safe disposal of HLW and lack of regulatory experience with this enterprise, how are we going to establish the framework and criteria in a manner that permits/allows/encourages/demands rigorous scrutiny of all that we do and active participation by all interested parties - the Congress, State, local and tribal governments, industry and public interest groups and individual citizens - the stakeholders in this venture?

It is quite clear that standards are needed; both for credibility and to give DOE's development program a target. When it comes to developing these standards we are clearly in a bootstrap mode - bootstrap because we usually accumulate our knowledge about criteria and procedure from licensing experience and codify it in the form of regulations, regulatory guides and other standards. These standards then provide the framework and criteria for licensing decisions and by which regulation can be accomplished.

However, the usual practice of accumulation followed by codification will not work in the case of HLW disposal. First of all, there will not be many disposal facilities, so there will be little opportunity to accumulate licensing case experience. Secondly, because the "operation" of a HLW repository begins after we walk away and extends over long periods of time, we will not be accumulating much "operational" experience either. Third - and from the institutional aspect, perhaps most important of all - the stakeholders will not permit it: the stakeholders want the regulator out there in front in this venture providing the framework and the criteria within and by which DOE can operate. A reactive NRC role is just not acceptable. Not only would the credibility of the disposal process be jeopardized, but chances of success for DOE's costly development program would be diminished as well.

Therefore, we are faced with the problem of developing the regulatory practice through standards-setting in the absence of experience by creating framework and criteria which in their application develop and establish credible regulatory practice -- "the bootstrap." Further these standards, in particular the regulations portion of the framework and criteria, must allow processes which expose the innards of the particular licensing decisions to the stakeholders and invite their active participation. (We are reminded of the "strange loops" which form the theses of the best-seller, "Godel, Escher, and Bach.") Stated differently, NRC must establish processes by which the issues which surround technologically and institutionally acceptable methods of safe geologic disposal can be identified, debated and understood; and that understanding provide the mechanism by which the uncertainties which inhere in these issues and solutions are identified and explored. This in turn is the process by which licensing decisions are made - to the satisfaction of the stakeholder.

I think the NRC has made significant progress in this area. We are attacking the problem in two basic pieces: the licensing procedures and the technical criteria. We have further partitioned each piece so that we could deal with the issues and uncertainties. We have established the framework for licensing in the form of a regulation which sets out a multi-stage licensing process which actively involves the stakeholders at each step and which keys the level of assurance embodied in each decision to the level of knowledge available at the time the decision is made.

Four stages in the "preoperational" life cycle of a geologic repository have been identified as warranting Commission review; site characterization, construction authorization, emplacement of wastes, and permanent closure. Although essentially the same features would be addressed at each stage, there would be a progressive increase in knowledge regarding these features and a corresponding increase in confidence in a decision whether HLW can be disposed of safely at a repository at the site. The process is structured to key important decisions to the availability of information; accordingly, as information develops over time, decisions will reflect increasing levels of assurance of protection of the public health and safety and the environment. But also the procedures require certain actions upon the part of DOE to assure that the information needed to make each decision will be available when the decision is to be made -- again, the bootstrap. Thus, the licensing process reflects an underlying recognition of inherent uncertainties. Further, the procedures provide for active participation by the stakeholders. We would like to add parenthetically that the public and the final rule itself (February 1981), is a strange loop in itself. Statement of Policy on Licensing Procedures for Geologic Disposal of HLW (November 1978), a proposed rule on the Licensing Procedures (December 1979) and the final rule itself (February 1981)

You received for the symposium a copy of the NRC final regulation on HLW licensing procedures which was published in the Federal Register on February 25, 1981. We trust that you all will read those procedures; however, let us provide you here with a brief description of the procedures. In the first stage when the DOE has formulated plans for a prospective repository to the extent that it wishes to begin site characterization, it will be required to submit a site characterization report which contains, among other things, the program plan by which the DOE will investigate and characterize sites. The report will address the process by which the media and site(s) were chosen for characterization and the DOE's program for further development of alternatives. The report also will contain a description of the media and site(s) to be characterized and the site characterization program. The report will be reviewed by the NRC staff with opportunity for public comment on analysis of the report.

The NRC will notice the governors of interested States and local and tribal authorities of the availability of the report, will analyze the report, publish the results of that analysis, and obtain public comment, both written and via local public forums in the vicinity of the site. In keeping with the preliminary nature of information available, there will be no formal licensing approval, but NRC's Director of Nuclear Material Safety and Safeguards will issue an opinion either of "no objection" to the program of characterization planned or offer comment on weaknesses in terms of developing the information needed to license.

The second stage begins with the submission by the DOE of a license application for a particular site selected from among those characterized. Subsequent to staff review and preparation of an Environmental Impact Statement, it is anticipated that a licensing board will be appointed and the license application will undergo the first formal review, including public hearings. If the Commission finds after considering reasonable alternatives that the benefits of the proposal exceed the cost under NEPA and that there is reasonable assurance that the types and amounts of wastes described in the application can be received, possessed, and disposed of in a repository of the design proposed at the site without unreasonable risk to the health and safety of the public or being inimical to the common defense and security, construction of the repository will be authorized. It is expected that sites selected will be from a slate of sites among the best that can reasonably be found and that there will be no obviously superior site to the one preferred. Many questions concerning the ability of the site to host a repository will have been answered and so the decision at this stage will reflect a greater degree of certainty. The decision will be based upon the record established in a mandatory hearing.

Stage three is a further review at the application prior to receipt of wastes at the repository. When construction of the repository is substantially complete, the waste emplacement decision can be made. The Commission will issue a license to the DOE if it finds, among other things, that the issuance of the license will not constitute an unreasonable risk to the health and safety of the public. The findings would be based upon a review of an update of the application submitted for construction authorization and an updated environmental report if needed. Among items to be considered in the review will be additional data acquired during construction, conformance of construction with design, and resolution of questions not answered during the construction authorization review. It is expected that adjudicatory hearings would be held to consider appropriate issues. Again, the waste emplacement decision would be based upon the greater information with respect to both scope and quality and the attendant reduction in uncertainties, that will have come from construction activities, completion of R&D activities, and in situ verification and validation (confirmatory assessment) programs. It will reflect that confidence that indeed the step of waste emplacement can be undertaken.

At some point the DOE may submit an application to close the repository permanently, and the final review of repository activities will begin. Additional geologic and hydrologic data acquired during the emplacement period as well as the results of test experiments on backfilling and a shaft sealing, along with the DOE's planned permanent closure program, will be considered by the Commission in determining whether the planned method for permanently closing is adequate.

In order to allow a decision to close the repository permanently to not inexorably follow a decision to emplace the waste, the procedures reflect a policy which allows for the temporal separation of these two decision points. As it will be discussed in a few moments, that policy is realized in the concept of retrievability - an important cornerstone of the technical criteria. This concept is predicated on the need to conduct performance verification and validation during the period of waste emplacement to reduce further and to understand better the uncertainties.

Lastly, for completeness, the procedures include a license termination stage. It is expected that much thought will be given to this question in the years ahead.

Included also in the procedures are provisions whereby states and local governments and Indian Tribal entities may participate actively in the licensing process starting with the submission of the Site Characterization Report. They may submit proposals to the Director to do so.

Let us turn now to the second piece - the technical criteria. Parallel with the development of the licensing procedures has been the development of the technical criteria against which individual licensing decisions will be made. Many of you have been involved in that activity. In late 1979, an early draft of the technical criteria was made publicly available and a series of peer reviews and meetings with experts was held to critique and refine the technical criteria. For example, meetings were held with the Keystone Radioactive Waste Management Review Group, a group hosted by the University of Arizona, the U.S.G.S., DOE, and EPA. In the spring of 1980, an Advance Notice of Rulemaking was published setting forth the approach the staff was taking in developing the criteria. The Advance Notice also included a draft of the criteria that reflected the thinking available at that point in their development. Many public comments were received on both the approach and criteria and these have been considered in the further development of the technical criteria which the staff will be recommending that the Commission approve for publishing in the Federal Register as a proposed rule. You have received a copy of the most recent draft of these criteria.

Just as with the procedures where we tried to lay out a process which allowed for approval of decisions at appropriate times according to the level of information available, to avoid premature decisions, to not foreclose options, to expose the decision processes of the NRC to scrutiny - because this is the essence of licensing in an open society - so we have tried in the technical criteria to look at the problem of waste disposal in a geologic repository from the perspective of not only what is necessary for such disposal in a repository, but what is necessary for such disposal in a licensed repository. In licensing is embodied the concept of an independent entity's (the NRC) making the decision to permit the action (disposal) based on review of the relevant information, and of independent scrutiny by others, including the Congress and the courts, of that decision, the relevance of the inforamtion and the appropriateness of the conclusions drawn therefrom.

The approach taken considered first what we were trying to do - have confidence that the waste would be disposed of safely - and record how to do that - expose all the uncertainties upfront, see what they mean, and find a way around the lack of confidence spawned by these uncertainties. Obviously easier said than done. But we think we have succeeded by redefining geologic disposal of HLW into containment for a time and isolation thereafter. Thereby, during the period when the heat and radiation from the wastes are highest, disposal means containment within the waste package; and confidence that the wastes remain within the waste packages. Following the containment period, disposal means a controlled release of radionuclides to the environment in quantities and concentrations which meet applicable standards; and confidence means that the engineering is controlling the rate of release to the extent that, despite the uncertainties in the transport of nuclides through the geology, realistic calculations made with conservative assumptions will yield results within applicable standards. Further, the engineering continues to control releases sufficiently long until the radioactivity within the repository is so diminished that the uncertainties in transport through the geology no longer need to be compensated by a controlled release calculated within prescribed bounds.

We are supplementing the development of these two rulemakings with an active program of regulatory guide development and the development of other standards, some with national standards setting groups. For example, the procedural rule will be supplemented by major regulatory guides which will provide the NRC staff position on the scope and content of the site characterization report and the environmental and safety analysis reports to be submitted by DOE in support of its application for a license. These will be issued for public review and comment with the first, the site characterization guide, scheduled for publication shortly.

In summary, DOE and NRC have separate but interdependent objectives in the disposal of HLW. Both must succeed if the overall problem is to be solved. NRC is charged with providing regulatory oversight; that is assuring that the public health and safety will be protected. Assurance must be determined before a geologic repository is allowed to begin construction and reconfirmed at key states in its development and operation. As long as the waste can be retrieved, the decisions are not irreversible. Decisions which reach beyond the period of retrievability would be based on confirmation that the protective systems will perform as required. Information needed to make the confirmation would be collected during the lifetime of the repository. Standards are required to give the endeavor direction and credibility. We have organized this symposium to utilize the wisdom and expertise you bring to this subject. We ask that you share freely of your views in regard to these questions:

- Have we identified the right aspects of the problems from the perspective of regulation?
- Have we identified the sources of uncertainty?
- How might the tools of modeling assist us in exploring, reducing, hence in overcoming uncertainties in reach licensing decisions?

EPA'S ENVIRONMENTAL STANDARDS FOR MANAGEMENT AND DISPOSAL OF HIGH-LEVEL RADIOACTIVE WASTE

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(Paper Not Submitted)

HOW MUCH INSURANCE?

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ABSTRACT

As the technological assessment of alternative waste disposal options reaches the point where criteria are developed for site selection and repository design, it becomes important to consider the technical, social and economic framework in which decisions are to be made. This paper reviews the ultimate objectives of waste disposal, the steps that must be taken in evaluating expected repository performance and the areas where input data are either of low quality or have to be chosen somewhat arbitrarily. Present tendency in developing performance criteria is to develop an ALARA approach, with multiple barriers, both engineered and environmental, providing a high degree of retention. This paper emphasizes the need to use an iterative approach including cost-effectiveness considerations to guard against excessive overdesign and unnecessarily costly design features.

When a cost-effectiveness approach is coupled to sensitivity analysis, it is seen that some parameters need to be known only within an order of magnitude, whereas at some steps uncertainty is introduced more by the choice of scenario than by the particular data chosen.

INTRODUCTION

The safe disposal of radioactive wastes has been made a matter of major public concern; it has been the subject of plebiscites, referendums and extensive and costly litigation in Sweden, Germany, Great Britain and the United States and is seen widely as the touchstone by which ultimately public acceptance of nuclear power may be judged. However, it is not entirely clear at this time by what criteria the public, that rather ill-defined entity, will judge that safe disposal has been demonstrated or assured. Past experiences with votes and referendums have demonstrated that this judgment invariably is interlinked in a complex fashion with questions of political leadership, economic pressures, alternative solutions, environmental questions and subjective distrust of government and big business.

The subject of this conference is the regulation of the geologic disposal of high-level radioactive wastes, and the purpose of my remarks is to strike a warning note before we freeze methodology and regulatory requirements too firmly in a near-irreversible fashion. I believe it is important at this stage to recall the objective of this whole operation and to place it into a somewhat broader perspective.

The purposes of development of a rather elaborate technology, assessment methodology and regulatory framework then are manifold:

1. To ensure that waste materials are emplaced in a location and in a form that will give reasonable assurance of containment for the indefinite future;

2. To ensure adherence to any proposed procedures and methods adopted for this purpose and to quality assurance at all stages;

3. To convince the public and the political decision makers that the repository methodology is safe and acceptable;

4. To provide disposal of wastes at a cost that is commensurate with the value of such operations at the end of the fuel cycle; and

5. To meet the legal requirements laid down by the National Environmental Policy Act and the various water quality laws enacted by Congress and the states.

While some may argue that the necessary technology exists to a large extent and that only political problems impede immediate action on this matter, others feel equally strongly that past dipsosal procedures have proved unsatisfactory and that much more detailed studies are needed before specific choices are embodied in a national waste repository policy. These problems are further compounded by suggestions that, for largely political reasons, defense wastes in some fashion differ from commercial wastes in their environmental impact or in requirements for cost-effectiveness of disposal.

The issue may be resolved into two questions that are somewhat interdependent:

1. How much and what kind of assurance does the public require to accept a given disposal method as safe?

2. How much insurance do we need to satisfy ourselves that risks have been reduced to a negligible level?

It is this latter question which I have chosen as the topic of this talk. I feel the situation is analogous to the decision most people make in buying life insurance or accident insurance. In each case a person is confronted with certain actuarial statistics. The decison on how much insurance to buy depends on his own perception of the risks and the consequences. Some risks are fairly evident; collision damage, theft, third-party injuries, etc., and most people are willing to pay for reasonable coverage. Others are less clear-cut; lightning, floods and other acts of God, and most people will purchase a comprehensive coverage for such risks if the premium seems low in relation to the perceived probability of such events.

RISK ASSESSMENT

In waste disposal we are confronted with a similar situation: After taking care of obvious problems that pose a relatively high probability of occurrence, much of the present preoccupation with often rather improbable scenarios tends to incur relatively high costs in relation to their probability of occurrence and the magnitude of their consequences. The difficulty that must be faced is that it is improbable that the country can afford to continue to budget hundreds of millions of dollars to "solve" problems of negligible impact, even though the magnitude of the projects and the funds committed generate their own momentum.

Nevertheless, it is important to restate what are the ultimate objectives of the waste management program. I shall focus here on three aspects:

1. What is the nature of the risk that is to be guarded against?

2. What is the magnitude of this risk, and at what level may it be considered negligible or acceptable?

3. What is the cost-effectiveness relationship in waste disposal options, and what degree of sophistication would be justified in meeting a given risk criterion?

Unless these questions are fairly answered, I believe a great many resources may be squandered and any regulatory framework may be out of focus. This fact is clearly underlined by the reevaluation of reactor accident impacts that is taking place at present in the wake of Three-Mile Island [1]. One of the consequences emerging is that the adoption of a relatively improbable design-basis accident may impose engineering design solutions that are costly, do not address the real issues, and also may lead to a public preoccupation with accident consequences that are unrealistic and merely cause fear in the community.

The present approach to the minimization of the environmental impact of prospective waste treatment methods and repository site selection consists of four steps:

a. Solidification of the waste and conversion to a matrix which is considered chemically inert and whose leaching characteristics are judged to result, under most pessimistic assumptions, in releases that are considered as low as reasonably achievable (ALARA);

b. Design of an emplacement configuration, consisting of canister, overpack, backfill, and geologic repository characteristics that would minimize water incursion and keep waste ion migration to ALARA levels;

c. Choice of a location in undisturbed mineral settings and distant from active aquifers from which migration would be negligible under normal conditions and controlled by rock-ion interaction effects to negligible ranges; and

d. A location sufficiently distant from human habitation to make uptake of any escaped radionuclide improbable and/or of negligible consequence.

Attempts are being made to make each of these procedures as effective as technically feasible regardless of cost; and in fact, the draft 10CFR60 regulations seem to demand this [2,3]. Impressive progress has been made in evaluating the performance of various repository media, waste forms and waste packages and to develop models for waste migration and dose commitment [4]. Current requirements demand absolute integrity of the waste for 500-1000 years after closure and an ALARA dose for 10,000 years. Making "conservative" assumptions regarding leach rates, migration rates and human uptake, which in practice include some rather improbable sequences, individual dose rates of well below 1 mrem/yr are estimated both for existing defense wastes [5] and anticipated commercial wastes [3,6] and dose commitments that are a small fraction of the natural radiation background. It is the subject of this conference to discuss the reliability and methodology of some of these estimates; however, it is unlikely that they would seriously underestimate the dose commitment by more than an order of magnitude, if that; an appreciable overestimate seems more likely.

The magnitude of the dose commitment obviously depends on the nature of the source material and whether or not retrievable storage is mandated. It is probably fair to state that retrievability is largely a political decision of limited technical value and would merely add appreciably to the cost of operation. Similarly, disposal of spent fuel as such would constitute a largely political approach, since reprocessing is an inherent step in any defense nuclear operation and in breeder reactor technology and, therefore, may well cover a significant fraction of waste management operations anyhow. Some economical arguments can be advanced that an open-ended fuel cycle without reprocessing may be superior to a closed one, but from the point of safety of operation and handling, reprocessed concentrated waste material seems clearly superior. In addition, burial of substantial amounts of uranium-238, instead of its consumption in fast breeders, introduces radium-226 as the long-term nuclide of critical importance for waste repository assessment. Cohen [7] has demonstrated that any one of the four steps outlined above, if conducted in accordance with 10CFR60 and ALARA design considerations, would be sufficient to meet reasonable safety criteria by itself, so that present approaches may represent gross overdesign.

This question brings me to my main topic: How do we arrive at a reasonable compromise between desire for a "perfect" solution, supposedly in accord with popular demand for absolute safety, and the need for a cost-effective solution that allows for an economically viable approach that still meets reasonable safety objectives. Such a compromise, which calls for some common sense [8], is of course the normal way in which engineering decisions are made. In the case of waste management, however, because of the extensive documentation required for every decision, a more formal methodology must be supplied.

This methodology must follow conventional risk-benefit analysis. Any such analysis consists of five steps:

1. Identification of source terms and determination of the probability of a chosen initiating event;

2. Consequence analysis, in this case in terms of leach rates, migration rates and uptake rates;

3. Risk determination; dose rates, dose commitments and population doses;

4. Sensitivity analysis; perturbation test for sensitivity to uncertainties in input parameters and identification of the more significant parameters; and

5. Cost-benefit analysis; evaluation of alternative options among the more significant operational parameters in order to identify the most economic solution that still meets specified risk criteria.

Such a process necessarily is an iterative one. It starts with a conceptual design for the waste form, package and repository site 9 . It then postulates what may be credible incursion scenarios, a subject that is highly contentious at this time [e.g., 10-12]. Migration through an aquifer may be separated for sorbable ions, nonsorbable compounds and particulate carriers [3]. Some distinction may be made whether transport would be essentially horizontal along existing aquifers or vertical along fractures or fissures produced by geological or geochemical processes. Finally, the uptake scenario, mainly via drinking water or the food chain, must be specified with a credible path to man. Official approaches have tended to emphasize perfect containment [14], with an assumption that whatever escapes will appear in the food chain. Some of the more detailed calculations by the INFCE Committee [3,14] shown in Tables 1 and 2 result in individual maximum doses of the order of 10^{-2} mrem/yr for the assumed hard rock repository and about 0.4 mrem/yr for the breached salt dome repository for spent-fuel disposal. Much lower doses have been estimated for separated fission products stored after reprocessing. Cohen [10], with slightly different assumptions, has presented data that imply 0.0074 cancer deaths/GWe-yr for fission and activation products or 0.0174 CD/GWe-yr for spent fuel, if radium-226 effects are included.

Since these levels are well below doses from natural radiation background, it becomes necessary to reach a reconciliation between the ALARA criterion and cost-effectiveness requirements. Present anticipated costs for a national waste repository are in the price range of a billion dollars, clearly out of range for the old NRC cost criterion of $1000 per man-rem reduction.

At this time we lack a clear-cut criterion for an acceptable radiological impact both for the near term and over the postulated 10,000-year period of responsibility. It is evident that the risk projected should be low compared with other risks, such as natural radiation background, both for "normal" undisturbed conditions or any natural disruptions that can be predicted with any reasonable probability. The question before this conference then is threefold:

1. Can we predict such effects with sufficient assurance within at least an order of magnitude?

2. How great a margin of error can be tolerated if the projected effects are indeed insignificant?

3. Assuming that present predictions can be made accurate to within an order of magnitude, which portion of the protective technology can be scaled back most profitably to ensure low enough impact at the lowest overall cost?

These questions are a little different from those answered by most of the papers presented at this conference. Regarding the first question, the greatest sources of uncertainty seem to be in the assumptions for the water incursion into the repository that determine the source term for the migration step in any model [15] and the actual retardation factors to be assumed (Table 3).

The second question is also difficult to answer in the absence of a rigorous sensitivity analysis and a general unwillingness to discard negligible pathways. I have proposed before [16] that it is sensible and necessary to discard terms that differ from zero by a statistically insignificant amount to avoid excessive calculational baggage and an inflation of effectively zero doses into apparent significance. Black and Niehaus [17] have argued that there is an irreducible minimum level of risk, in that additional sophistication of equipment and facilities to yet further reduce risks, will itself introduce new risks. This has been well illustrated in reactor technology where installation of risk-reducing measures, e.g., for seismic precautions, has at times resulted in a greater immediate risk by increasing occupational exposures during maintenance. Similar effects may attend insistence on retrievability of waste materials, especially the disposal of spent fuel in unreprocessed form.

CONCLUSION

In summary, it is proposed that the time is ripe to look not only at the dose reduction features of emplacement technology but also at their cost-effectiveness. This requires a close look at the degree to which relevant parameters are known to a sufficient accuracy to permit detailed predictions or a decision that more accurate determinations do not, in fact, add anything to the accuracy with which the predicted dose commitment must be known.

This is not to say that at this time all the relevant parameters are well known; rather it means that most of them are probably sufficiently well known to proceed with an engineered repository design and that it is becoming important to determine priorities in establishing performance data for those components and pathways that are most cost-determining or for which large uncertainties lead to most sensitive changes in estimated impact.

It is important to the nuclear industry and to the country at large to answer the question, whether high-level waste can be disposed of safely, as surely and expeditiously as possible. To do this, it behooves all concerned to concentrate on the main issues: to select a technology that provides adequate, but not lavish, safeguards and to assure disposal by methods that provide insurance against future risks with sufficient peace of mind at affordable cost.

REFERENCES

1. M. Levenson and F. Rahn. Realistic Estimates of the Consequences of Nuclear Accidents. Electric Power Research Institute, Palo Alto, (1980)

2. U. S. Nuclear Regulatory Commission. Disposal of High-Level Radioactive Waste in Geologic Repositories. Draft version, 10CFR Part 60, 1980

3. U. S. Dept. of Energy. In the Matter of Proposed Rulemaking on the Storage and Disposal of Nuclear Waste (Waste Confidence Rulemaking). Statement of Position of the U. S. Department of Energy. Rept. DOE/NE-0007 (1980)

4. C. Klingsberg and J. Duguid. Status of Technology for Isolating High-Level Radioactive Wastes in Geologic Repositories. Rept. DOE/TIC-11207 (Draft), U. S. Dept. of Energy, (1980)

5. Draft Environmental Statement, Waste Management Operations, Savannah River Plant, U. S. ERDA, Washington (1976)

6. Draft Environmental Impact Statement, Management of Commercially Generated Radioactive Waste. Rept. DOE/EIS-0046D, U. S. Dept. of Energy (1979)

7. B. L. Cohen. High-Level Radioactive Waste from Light-Water Reactors. Rev. Mod. Phys. 49, 1-20 (1977)

8. Radioactive Waste Management: Perspectives Reexamined. Atom No. 290, 306-311, December 1980

9. A. D. Little, Inc., Technical Support of Standards for High-Level Radioactive Waste Management. Rept. EPA 520/4-79-007A-C, 1977

10. B. L. Cohen, Analysis, Critique and Evaluation of High-Level Waste Repository Water Intrusion Scenario Studies. Nuclear Technology 48, 63-69 (1980)

11. M. D'Allesandro, C. N. Murray, G. Bertozzi and F. Girardi. Probability Analysis of Geological Processes: A Useful Tool for the Safety Assessment of Radioactive Waste Disposal. Radioactive Waste Management 1, 25-42 (1980)

12. H. C. Claiborne, L. D. Rickertsen and R. F. Graham. Expected Environments in High-Level Nuclear Waste and Spent Fuel Repositories in Salt. Rept. ORNL/TM-7201, Oak Ridge National Lab., 1980

13. G. G. Eichholz and T. F. Craft. Subsurface Migration of Radioactive Waste Materials by Particulate Transport. Draft Report to Battelle Memorial Institute, 1980

14. Report to the President by the Interagency Review Group on Nuclear Waste Management, Washington, 1979

15. A Pritzger and J. Gassmann. Application of Simplified Reliability Methods for Risk Assessment of Nuclear Waste Repositories. Nuclear Technology 48, 289-297 (1980)

16. G. G. Eichholz. Statistical Cut-off Criterion. Health Phys. 38, 434 (1980).

17. S. C. Black and F. Niehaus. How Safe is "Too" Safe? IAEA Bull. 22 (10, 40-50 (1980)

Table 1. Maximum Discharges, Times of Maximum Discharge, and Maximum Dose Rates From a Hypothetical Spent-Fuel Repository-Breached Salt Dome (Ref. 3)

RADIO-NUCLIDE	MAXIMUM NUCLIDE CONCENTRATION (micro Ci/ml)	TIME THAT MAX. NUCLIDE CONCEN. OCCURS (years)	MAX. INDIVIDUAL WHOLE-BODY DOSE RATE (mrem/yr)[(a,b)]
C-14	3.4×10^{-8}	45,000	5.0×10^{-4}
Tc-99	7.7×10^{-4}	58,600	1.1×10^{-0}
I-129	2.0×10^{-7}	50,500	7.7×10^{-4}
Cs-135	2.7×10^{-6}	45,000	4.8×10^{-2}
U-236	9.6×10^{-7}	440,000	5.4×10^{-2}
Th-232	1.2×10^{-13}	2,000,000	1.1×10^{-8}
Np-237	7.4×10^{-6}	44,000	1.9×10^{-1}
U-233	1.9×10^{-7}	1,690,000	1.2×10^{-2}
Th-229	6.8×10^{-9}	1,500,000	1.4×10^{-3}
U-238	8.2×10^{-7}	438,000	4.1×10^{-2}
U-234	2.0×10^{-6}	437,000	1.2×10^{-1}
Th-230	2.1×10^{-8}	520,000	6.3×10^{-4}
Ra-226	6.3×10^{-8}	490,000	$1.5 \times 10^{+1}$
Am-243	1.7×10^{-6}	45,000	4.5×10^{-2}
Pu-239	8.5×10^{-10}	80,000	6.8×10^{-6}
U-235	1.9×10^{-8}	470,000	1.0×10^{-3}
Pa-231	3.0×10^{-10}	460,000	1.3×10^{-1}

[a] The maximum individual dose rate generally occurs in the child age group.

[b] For some isotopes, the dose rate to individual organs may be higher.

SOURCES:

1. Reference 700: M. A. Harwell et al., Reference Site Initial Assessment for a Salt Dome Repository, PNL-2955, (Working Document), Appendixes G and H, Pacific Northwest Laboratory, Richland, WA, December 1979
2. Reference 700: Ibid., pp. 11-47 to 11-49
3. Reference 699: W. C. Arcieri, J. G. Feinstein, and J. D. Freeman, Methods Used to Compute the Dose to Individuals and the General Population Due to Geotransport of Radionuclides from a Spent-Fuel Repository, NUS-TM-327 (Draft Report), NUS Corporation, Bethesda, MD, February 1980

Table 2. Maximum Discharge Rates, Times of Maximum Discharge, and Dose Rates Received by People from a Hypothetical Spent-Fuel Repository--Hard Rock (Ref. 3)

RADIO-NUCLIDE	TIME OF MAX. DIS-CHARGE (years)	MAX. DIS-CHARGE RATE (Ci/yr)	MAX. DOSE RATE FROM SPENT FUEL (rem/yr)	MAX. DOSE RATE FROM ASSOCIATED WASTE (rem/yr)
I-129 (a)	1.1×10^{4}	5.6×10^{-4}	-	-
I-129 (b)	1.1×10^{4}	1.3×10^{-2}	5.1×10^{-6}	-
Cs-135	7.1×10^{8}	6.3×10^{-11}	1.5×10^{-14}	-
Ra-226 (c)	4.1×10^{8}	9.4×10^{-5}	1.0×10^{-7}	1.6×10^{-6}
Th-230	4.1×10^{8}	1.0×10^{-4}	2.8×10^{-8}	4.5×10^{-7}
Pa-231	4.1×10^{8}	2.2×10^{-5}	5.7×10^{-5}	5.5×10^{-5}
Th-232	8.1×10^{8}	3.5×10^{-8}	4.7×10^{-10}	-
U-234	4.1×10^{8}	2.0×10^{-4}	1.4×10^{-8}	7.0×10^{-8}
U-235	4.1×10^{8}	1.0×10^{-5}	3.0×10^{-9}	3.0×10^{-9}
Th-236	4.1×10^{8}	2.0×10^{-9}	2.7×10^{-13}	-
U-238	4.1×10^{8}	2.0×10^{-4}	2.4×10^{-8}	1.2×10^{-7}
Th-230/Ra-226 (d)	4.1×10^{8}	-	2.2×10^{-6}	1.1×10^{-5}
U-234/Ra-226 (e)	4.1×10^{8}	-	5.5×10^{-8}	2.7×10^{-7}
Max. annual total dose	4.1×10^{8}		5.9×10^{-5}	6.9×10^{-5}

a. Bound in fuel matrix.
b. In gap between fuel cladding and matrix.
c. Ra-226 reaching the biosphere directly from ground water.
d. Ra-226 produced by the decay of Th-230 in the biosphere.
e. Ra-226 produced by the decay of U-234 (via Th-230) in the biosphere.

SOURCES: Reference 683: Working Group 7, International Nuclear Fuel Cycle Evaluation Committee, Release Consequence Analysis for a Hypothetical Geologic Radioactive Waste Repository in Hard Rock, INFE/DEP/WG7/21, Table XIX, December 1979

Reference 684: Working Group 7, International Nuclear Fuel cycle Evaluation Committee, Release Consequence Analysis for a Hypothetical Geologic Radioactive Waste Repository in Hard Rock, INFE/DEP/WG7/21, Table XXV, December 1979

Table 3. Summary of Water Intrusion Scenario Studies (Ref. 14)

REFERENCE	SCENARIO	TIME OF INITIAL RELEASE AFTER CLOSURE (yr)	LEACH TIME (yr)	GROUND WATER TRAVEL TIME TO BIOSPHERE (yr)	RETARDATION FACTOR (v_{water}/v_{nuc})	REPOSITORY NUCLIDE (Ci)	CRITICAL NUCLIDE	50-YEAR ACCUMULATED DOSE TO INDIVIDUAL (m rem)	CRITICAL ORGAN	TIME AFTER CLOSURE THAT DOSE IS RECEIVED (yr)
Burkholder et al.	High-Level; Leach Incident; Discharge to river	100	30,000	150	1 10 100 500	2.86×10^6 1.03×10^6 8.07×10^6 0.347	Tc-99 C-14 Np-237 Ra-226	200 3,000 120 2,400	 Bone Bone Bone	250 1,700 15,000 2,100,000
Burkholder et al.	Spent Fuel; Leach Incident; Discharge to river	Same as Above	3,000	Same as Above	Same as Above	Same as Above	Same as Above	2,600 30,000 1,200 240,000	Same as Above	Same as Above
Swedish Safety Study	High-Level; Leach Incident; Discharge to well	1000	30,000	400	1 700 43 260	140×10^3 20 750 6400	Tc-99 Ra-226 U-233 Np-237	150 130 130 650	 Bone Bone Bone	13,000 80,000 80,000 130,000
Girardi et al.	High Level; Leach Incident; Discharge to river	1000	17,000	-0-	-0-	5.3×10^6 5.2×10^5	Am-241 Am-243	100 250	Bone Lung	1,000
Hill and Grimwood	High-Level; Leach Incident; Discharge to river	1000	3,500	100	1 100 500	1.6×10^5 1.2×10^4 ?	Tc-99 Np-237 Ra-226	5% MPAI 3% MPAI 2% MPAI	 Bone Bone	100 ? ?

Invited Session:

PUBLIC AND PRIVATE INTEREST GROUP PERSPECTIVES

Chairman

David C. Kocher

Oak Ridge National Laboratory

THE GEOLOGIST AND RADIOACTIVE WASTE DISPOSAL, OR, CAN GEOLOGISTS AGREE?

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ABSTRACT

Science advisors to federal programs in radioactive waste disposal have historically played the role of intervenor by reviewing agency work and research. These advisors tended to lend credibility and support to federal research. Today, intervenors are more commonly viewed as casting doubt on federal programs by emphasizing the uncertainties in technical data and underscoring the lack of consensus in the technical community. The Waste Isolation Pilot Plant (WIPP) serves as an example of a radioactive waste repository which has been both supported and opposed by earth scientists on the basis of site characteristics. The Environmental Evaluation Group (EEG), funded by the Department of Energy (DOE), illustrates how the state of New Mexico used scientists to review the relevant information on the WIPP site. The EEG failed to affect DOE decisions at WIPP because responding to EEG's requests for more studies and information would have forced DOE to alter its program deadlines and goals. The Nuclear Regulatory Commission (NRC) can be expected to respond more willingly to intervenor questions since its goal is to protect the public safety. The NRC will need to distinguish between issues for which more information will decrease uncertainty and those for which uncertainty will always exist. In making this distinction and related value judgements, the NRC will benefit from responding to intervenor concerns.

INTRODUCTION

When David Kocher invited me to speak here today, he asked me to discuss the role of the intervenor in radioactive waste disposal. Gagging somewhat on the word "intervenor," he apologized for using the term. Indeed, "intervenor" seems to be something of a dirty word among federal agencies. The dictionary defines an intervenor as "one who comes between two things by way of hindrance or modification." Illustrating this definition was this sentence: "We were enjoying our picnic when a thunderstorm intervened." Not surprisingly, regulatory agencies view intervenors with some trepidation. It sometimes appears that we ruin their picnic. My goal is to demonstrate that the real menace at the picnic is not the intervenors, but the difficulty of answering the questions raised by intervenors.

Historically, federal agencies have sought the advice of scientific experts to review agency programs and proposals. This form of intervention is viewed as a way of guiding new programs as well as increasing credibility and support. The National Academy of Sciences (NAS) is probably the most well-known and best example of scientific experts who often serve as reviewers. In the mid-1950s, the Atomic Energy Commission requested the formation of a Committee on Waste Disposal to study the possibilities of disposing of radioactive waste on land [1]. This committee of earth scientists recommended salt as a repository host medium, noting the ability of salt to conduct heat, to "self-heal" in the event of fracture, and its demonstrated dryness [1]. This recommendation guided the Department of Energy (DOE) and its predecessors for many years. Coming from a respected group of scientists, this recommendation provided excellent support for agency work meticulously reviewed by Congressional budget committees and the public. But, more recently, research on salt as a host medium for radioactive waste has led some to question the wisdom of using salt. In 1978, a group of NAS members addressed a letter to their colleagues urging support for recommendations to study other geologic media and to assess realistically "the uncertainties about a salt site"[2]. By publicly raising technical issues which remain unsolved, these respected scientists cast doubts on existing programs which featured salt as a prime candidate for radioactive waste disposal. Responding to these intervenors is a difficult problem which federal agencies must learn to overcome.

A CASE STUDY

The question of where to dispose of radioactive wastes has been considered by many geologists [3,4]. Incorporating the views of earth scientists who cannot agree is a formidable task, yet it appears that any site will invite such controversy. The Waste Isolation Pilot Plant (WIPP) is a proposed geologic repository which has been both supported and opposed by leading earth scientists on the basis of site characteristics. The debate over the adequacy of this site in New Mexico to contain radioactive wastes has been raging almost since the site was first proposed. The Department of Energy has encouraged this debate by funding scientists to study the WIPP site and by providing detailed scientific data in support of the environmental impact statement prepared for WIPP [5]. Yet agreement has never been reached over controversial issues such as site hydrology, mineral resource protection, and future human intrusion of the repository. Below, we examine the questions asked by scientists, the responses that were given, and the reasons for the continued dispute over the WIPP site.

WIPP is a proposed defense radioactive waste repository for transuranic contaminated waste currently being stored at the Idaho Nuclear Engineering Laboratory. The site preferred by DOE for WIPP is deep in the Salado Formation of the Delaware Basin in southeastern New Mexico. This formation is 85-90% pure halite with minor interbeds and constitutes a 2000' thick sequence whose base is 2850' below the surface. The geologic controversy at the site can be examined by studying the formation and role of the Environmental Evaluation Group (EEG) in reviewing WIPP for the state of New Mexico.

The EEG is completely funded by the Department of Energy, and has offices in Santa Fe, New Mexico. It represents a well-organized effort to include scientists in the public review process of the Department of Energy's work in radioactive waste disposal. Most importantly, the EEG represents an effort to institutionalize intervention so that views outside of the Department can be heard and incorporated early in the decision-making process. The EEG staff includes geologists, mathematicians, health physicists, radiation specialists, and environmental engineers. It is part of the New Mexico Health and Environment Department, and is charged with conducting independent technical evaluations of the potential radiation exposure to people from the proposed Waste Isolation Pilot Plant [6].

In carrying out its charge, the EEG has been involved in three major reviews over an 18-month period. They conducted extensive reviews of the draft and final environmental impact statements for the WIPP [7,8], they organized a two-day meeting of earth scientists concerned about the WIPP project to discuss uncertainties regarding the site geology [9], and they organized a follow-up field trip for geologists to look at the field evidence supporting major opposing views of hydrologic problems at the WIPP site [10]. In each of these efforts the EEG tackled site-specific problems which could cause unacceptable radiological hazards to the local population. The work of the EEG provides an excellent model for how a group of scientists representing a wide range of disciplines can review the available data on a specific site. The EEG produced provocative and useful reports which broadened and emphasized the scientific questions related to safe operation of the WIPP. However, based on recent decisions at the WIPP site, it appears that the EEG was unable to influence the Department of Energy.*

The EEG failed to be effective because they sought redefinition of DOE's program and underscored the lack of consensus on technical issues at the site. For example, in their review of the Final environmental impact statement, the EEG raised four major issues. First, the EEG felt the Department of Energy did not investigate thoroughly the anomolous seismic reflection data at the site. Second, the EEG was not satisfied that the DOE had adequate acceptance criteria for the high-level waste to be used in experiments conducted at WIPP. Third, the EEG wanted DOE to provide more details on future control of mineral and hydrologic resources at WIPP. And finally, the EEG identified potential scenarios for release of radioactivity which the Department of Energy had not investigated in its impact statement [8].

In raising each of these issues, the EEG appeared to be concerned with aspects of the WIPP which it felt were not sufficiently analyzed to ensure safety. For example, the EEG commented on the inadequate discussion of the Zone of Anomolous Seismic Reflection in the Final environmental impact statement:

> "[The discussion] should have more clearly reflected the uncertainty, controversy, and concern regarding the potential implications of this zone to the future integrity of the repository. ... The FEIS has not adequately addressed [our earlier comments on this problem] ... The EEG believes that a more definitive explanation is necessary before the site is judged acceptable for the repository." [8]

In sum, the EEG identified specific areas which it felt needed further study before the site could be judged acceptable. In addition, the EEG expressed disappointment that previous attempts to raise this issue had not been addressed to its satisfaction. Thus the EEG questioned DOE's attention to a specific technical matter and, more significantly, the EEG questioned DOE's ability to make a decision based on existing data.

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*The Department of Energy and the Department of the Interior signed a cooperative agreement on April 3, 1981 which would allow DOE to begin drilling an exploratory shaft at the proposed site in June 1981. This agreement and DOE's program are now the subjects of two lawsuits filed by the State of New Mexico and the Citizens for Alternatives to Radioactive Dumping, respectively.

Although the EEG was successful at raising relevant technical issues through its series of meetings and reviews of DOE documents, it was significantly less successful at resolving those issues for which a technical consensus did not exist. Illustrative of this problem was the field trip to the proposed WIPP site which the EEG organized. The trip spanned three days, included 23 participants, and covered 1500 square miles, including the WIPP site and the surrounding area [10]. Before the trip there was little consensus on the question of whether dissolution of salt occurs largely by surface-induced water circulation in the top of the evaporite sequence or whether water from deep aquifers flowing into the lower parts of the sequence can remove significant volumes of salt at depth. On the field trip, the major proponent of each point of view led the participants to field evidence supporting his theory. The field trip itself was highly praised by all participants as well-organized, very instructive, and important for airing differing views. But, when it was all over, the trip did not produce a consensus of scientific opinion.

To characterize the differing viewpoints, we can examine the letters received by the EEG in response to the field trip. Wendell Weart, the leader of the Sandia Laboratory's site characterization work for the Department of Energy, wrote to the EEG commenting on the unclear connection between the geologic features visited at the site and the integrity of the site as a radioactive waste repository. Weart's letter stressed that without clarification there would be a tendency to assume that an adverse connection existed when, in fact, site features may have little or no effect on site safety. Another participant said:

> "the concepts of salt dissolution and breccia pipes were well expressed. ... I see plausibility in both arguments, and as with most geologic processes there is a strong likelihood that both are correct!" [10]

And yet another participant said:

> "Does deep dissolution represent a serious threat to the long-term integrity of the WIPP repository? My conclusion is that we lack sufficient information at the present time to answer this question with a high level of confidence." [10)

Though most participants sought greater resolution, the Department of Energy response to the field trip acknowledged differences in interpretation but suggested that "in matters of geologic phenomena and processes, questions will always remain incompletely answered." The Department also stated "that a point must be reached where we say that the next step should be taken with an acceptable level of risk. That step is the [Site Preliminary Design Validation*], which involves no risk to the public." [10] The Department's response indicates a desire to move on with its original plans. It does not mention a willingness to work out the remaining uncertainties, but tries to decrease the importance of uncertainty by asserting that geology always contains unanswered questions. In order for the Department to respond otherwise would have required a reconsideration of its program and its goal to dispose of waste by 1987. The lack of consensus which emerged from the field trip made it impossible for DOE merely to absorb a new idea or recalculate its existing data. Instead, a new set of questions and suggestions for more research were posed, leaving DOE in the difficult position of proceeding without consensus or once again redefining its program.

THE VALUE OF INTERVENTION

The work of the EEG and similar mechanisms to encourage greater public involvement will often force the Department of Energy to proceed more slowly and more cautiously in its efforts towards providing radioactive waste repositories. However, questions and suggestions made by the public during the process of site selection and characterization can improve investigations by helping to avoid costly mistakes. The ill-fated Lyons, Kansas project illustrates the value of participation by local interests in the decision-making process. The Atomic Energy Commission's plan to dispose of the nation's radioactive waste in the Carey salt mine in Lyons met with substantial opposition from the Kansas State Geologist. His requests for more information and better tests were largely ignored until an adjacent mine lost 175,000 gallons of fresh water which the mine operators expected to retrieve as brine as part of a solution mining process. This mishap emphasized the importance of the State Geologists' earlier requests to study the hydrology of the local area in greater detail. If the studies had been done, the project might have been terminated for technical reasons instead of in a cloud of political embarrassment.

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*The Site Preliminary Design Validation is the Department's underground exploration program which it would like to begin in June 1981.

Radioactive waste disposal is a problem which contains a high degree of uncertainty. As illustrated above, public intervention often emphasizes that uncertainty, casting doubt on federal plans and decisions.

A lack of technical consensus, like a lack of public consensus, is very troubling for decision makers. As summarized recently by a Federal judge familiar with health-related lawsuits:

> "Uncertainty detracts from simplicity of presentation, ease of understanding, and uniformity of application. To focus on uncertainties is to invite paralysis; to disclose them is to risk public misunderstanding, loss of confidence, and opposition. Even though some uncertainty is inevitable, pointing it out will always create pressures for 'just one more study.'" [11]

Yet radioactive waste disposal is predicated on predictions which are inherently uncertain. Public confidence could suffer much deeper blows from ignoring uncertainty than from facing it. The Environmental Evaluation Group work described above indicates that as long as intervenors raise uncertainties which challenge agency programs and force changes in deadlines, there is little likelihood that they can or will be listened to. Unfortunately, though intervenors with technical expertise may improve the outcome of federal projects, there does not seem to be time to respond to their concerns. The goal to choose a waste disposal site in the next few years blinds many decision makers to the important site characterization work which must be done first.

THE REGULATORY PROCESS

The discussion above focuses on the process of providing waste disposal facilities and the inherent difficulties of responding to uncertainty while trying to meet specific goals for choosing sites. However, the goals and activities of the Nuclear Regulatory Commission (NRC) are quite different. The Commission is neither in the business of providing waste disposal nor is it dependent on the availability of waste disposal facilities. The goals of the NRC, different from the goals of DOE, are not questioned by uncertainty. The Commission's goal is to regulate radioactive waste disposal so that the public health and safety are protected.

The Commission appears already to recognize that intervention in the form of opportunities for public comment is an important part of its regulatory process. Among the Commission's procedural requirements for licensing geologic repositories is one for a Site Characterization Report. The Report is to be completed early in the consideration of a specific site and constitutes the jumping-off point for formal technical review of the geology at the site. This Report, to be submitted to the NRC by the Department, covers a range of important issues relating to the site integrity as well as a description of the research and development being conducted by DOE on waste form and packaging. The Department has said that it will allow public review of the Site Characterization Report and the Commission will provide at least a 90-day comment period on its official analysis of the Report. Although these provisions provide formal mechanisms for intervention, the real question is how will the Commission respond to the comments it receives? How will the Commission view those portions of the site characterization which make predictions that are inherently uncertain? What levels of uncertainty can it tolerate in predictions regarding geologic processes over a thousand years?

In analyzing the Department's site characterization work, the NRC has a responsibility to develop a process which can not only identify areas of technical uncertainty, but which also distinguishes between uncertainties for which more information will eliminate or significantly decrease uncertainty, and those technical problems for which uncertainty will always exist. The process devised by the NRC to make this distinction will benefit from open exchange with technical professionals outside of agency contractors and a willingness on the part of the NRC to research and consider the questions raised by intervenors. The process depends on a large technical community with broad-based support and respect to participate in the discussions which will ultimately help the Commission to face uncertainties. The NRC, like DOE, should fund independent reviews so that states which contain potential waste disposal sites can evaluate the licensing process.

Ultimately, the Commission will be faced with geologists who cannot agree. As in the EEG example, technical consensus may be impossible to attain and, as a result, the uncertainty caused by the complexities of radioactive waste disposal will be compounded by the uncertainty of proceeding without consensus. For those problems which will always contain uncertainty, the NRC must be called upon to make a value judgement regarding how much uncertainty is tolerable. This decision must be made by informed decision makers who have availed themselves of the opportunity to be influenced by a range of opinions including those of intervenors who may raise questions which force changes in agency decisions. This is the critical role fulfilled by intervenors, in harmony with the dictionary definition of "one who comes between ... ."

Author's Note: The views expressed in this paper are solely those of the author and do not necessarily reflect the opinions of the Natural Resources Defense Council.

REFERENCES

1. National Academy of Sciences, The Disposal of Radioactive Waste on Land, Report of the Committee on Waste Disposal, Publication 519 (April, 1957).

2. J.W. Berg, Jr., Memorandum to Members of the Geology and Geophysics Sections, National Academy of Sciences (May 25, 1978).

3. J.D. Bredehoeft, A.W. England, D.B. Stewart, N.J. Trask, and I.J. Winograd, Geologic Disposal of High-Level Radioactive Wastes - Earth-Science Perspectives, U.S. Geological Survey Circular 779 (1978).

4. U.S. Environmental Protection Agency, State of Geological Knowledge Regarding Potential Transport of High-Level Radioactive Waste From Deep Continental Repositories - Report of an Ad Hoc Panel of Earth Scientists, EPA/520/4-78-004 (1978).

5. U.S. Department of Energy, Final Environmental Impact Statement, Waste Isolation Pilot Plant, DOE/EIS-0026, 2 volumes (October, 1980).

6. D. Goad, A Compilation of Site Selection Criteria, Considerations and Concerns Appearing in the Literature on the Deep Disposal of Radioactive Waste, EEG-1 (June, 1979).

7. R.H. Neill, J.K. Channell, C. Wofsy, M.A. Greenfield (eds.), Radiological Health Review of the Draft Environmental Impact Statement (DOE/EIS-0026-D) Waste Isolation Pilot Plant, U.S. Department of Energy, EEG-3 (August, 1979).

8. Environmental Evaluation Group, Health and Environment Department, State of New Mexico, Radiological Health Review of the Final Environmental Impact Statement - Waste Isolation Pilot Plant, Volumes 1 and 2, DOE/EIS-0026, EEG-10 (January, 1981)

9. Environmental Evaluation Group, Health and Environment Department, State of New Mexico, Geotechnical Considerations for Radiological Hazard Assessment of WIPP, EEG-6 (April, 1980).

10. L. Chaturvedi, WIPP Site and Vicinity Geological Field Trip, EEG-7 (October, 1980).

11. D.L. Bazelon, "The Judiciary: What Role in Health Improvement?" Science 211 792-793 (1981).

"NOT IN MY BACKYARD": THE SOCIAL, POLITICAL AND INSTITUTIONAL UNCERTAINTIES ASSOCIATED WITH WASTE DISPOSAL

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ABSTRACT

The management and disposal of radioactive waste is not simply a technical problem. The success of DOE's waste disposal program will depend as much upon its social, political and institutional features as on its technical designs and engineering. For the initial phase of the program - the siting and operation of the first waste disposal facility - the major social and political obstacles are public opposition and lack of trust, unanswered questions of equity, conflicts in regulatory policy, and managerial and regulatory uncertainties. For the second phase of the program - expansion from one facility to a system capable of accommodating wastes from an expanding nuclear industry - the major problems will be the difficulty of maintaining near perfect human performance and the impact of the expanded system on the social structure of the communities affected. For the third, or long term management phase, the major social uncertainty will be whether the institutional arrangements made will function over long periods of time.

DOE HAS NOT ADDRESSED THE SOCIAL, POLITICAL AND INSTITUTIONAL ISSUES INVOLVED IN IMPLEMENTATION OF A PUBLICLY ACCEPTABLE WASTE DISPOSAL PROGRAM

The management and disposal of radioactive waste is not simply a technical problem. Technologies are not self-implementing. The success of any waste management program depends as much upon its social, organizational, and institutional features as on its designs and engineering. A task force of the Nuclear Regulatory Commission (NRC) concluded in 1978 that the "past failures of proposed radioactive waste management systems have stemmed in large part from neglect of nontechnological necessities in [the] implementation...of systems [1]. In 1979, the Interagency Review Group on Nuclear Waste Management reported to the President that "the resolution of institutional issues...is equally as important as the resolution of outstanding technical issues and problems" and that such resolution "may well be more difficult than finding solutions to remaining technical problems" [2].

Despite these warnings DOE has not come to grips with the significance of these issues. It has only just recognized that these issues exist, as is reflected in its request to the National Research Council to "attempt to identify social and economic issues to be considered in selection of repository sites" in order to "recommend ways in which to take various social and economic impacts into account in site selection..."[3].

As a result, DOE has not confronted and resolved the social, political, and institutional problems inherent in implementation of a waste disposal program. The Institute of Governmental Studies of the University of California at Berkeley cautions that failure to consider social and political issues as an integral part of the planning process for waste disposal is to "run the risk of serious political opposition," which may doom an otherwise acceptable program [4].

To be successful, the DOE program must meet dual objectives. It must be a program that the public sees as legitimate and in which it has confidence, as well as one that provides reliable and safe waste disposal operations. If the first objective cannot be met, the nation may be unwilling to commit the necessary political, technical, and economic resources to carry out the chosen method, and thus the method will fail.

Achievement of the first objective requires the identification and assessment of the relevant social and institutional obstacles to implementation of the major phases of the waste disposal program: the initial phase of siting, construction and licensing of the first waste repository; the second phase of program expansion to cope with the increased volume of wastes produced by the current and near future generation of light water reactors; and the third, long-term management phase in which the technological and institutional arrangements previously created will be tested over long periods of time. In each of these phases, new issues will present themselves for resolution, and social, political, and organizational arrangements appropriate to an earlier phase may require modification.

A. Phase One--Start Up

The phase of greatest concern at present is the initial start-up of the waste disposal program. The DOE is compelled by the decision in Minnesota v. Nuclear Regulatory Commission, 602 F.2d 412 (D.C. Cir. 1972), to develop a waste disposal program that can be implemented successfully before the expiration of the licenses of currently operating nuclear plants. Failure to do so threatens the continued viability of the domestic nuclear program, the substantial investment made by utilities and the industry, and, to a significant extent, public confidence in the ability of government to act decisively on a major social issue. The historical development of nuclear power in the United States has linked inextricably the federal government to the nuclear industry. Thus, the implementation of a waste disposal system is seen both by the public and the industry as a governmental responsibility.

The initial phase presents the greatest number of social and political uncertainties. Many of these have been identified and discussed in the 1977 Report of the Task Force for Review of Nuclear Waste Management (referred to as the Deutch Report) [5] and in the work of G.I. Rochlin and R. Kasperson, among others [6]. Key social and political obstacles are discussed below.

1. Public Opposition and Lack of Trust.--Foremost among the obstacles to implementation of the DOE program is the serious level of public opposition to nuclear power in general and waste disposal locations in particular [7]. This opposition is coupled with an increasing lack of trust in the ability of institutions and persons charged with protecting the public from the hazards of radiation to carry out that responsibility.

The unwillingness of the public to accept a waste management program manifests itself in the efforts of towns, counties, and states to restrict federal authority to transport and store wastes within their political boundaries. By October 1980, seven states had enacted laws banning nuclear waste importation for terminal disposal and twenty-five others had passed laws restricting nuclear waste disposal. Thirty-one states have limited or banned the transport of nuclear wastes within their boundaries. Indeed, by August 1980, only eight states had no laws relating to the control of radioactive wastes [8].

Former President Carter, in his statement of February 12, 1980, outlined a "consultation and concurrence" process as a means of resolving differences between the states and the federal government over the siting of waste disposal facilities [9]. Implicit in this policy is the idea that the sharing of information will lead to agreement on siting questions. However, there is doubt that simple information sharing will eliminate or even reduce the increasing reluctance of the states to be chosen as waste dumping grounds. The states are not willing allies of DOE or other federal waste management agencies. They are unlikely to side voluntarily with the federal government on waste disposal issues.

The consultation and concurrence concept was included in the nuclear waste legislation which passed the Senate on July 30, 1980 [10]. The bill provided for federal consultation with state governments concerning decisions to site waste repositories and spent fuel storage facilities. States were also given an opportunity to oppose a DOE decision to site a facility within their boundaries. The bill, however, set up three different procedures for federal override of state objections. If a state rejected the siting of a spent fuel storage facility, the bill provided for an override by a presidential directive that the facility was in the national interest. If a state objected to the location of a waste repository, the project would proceed unless the state was able to convince one house of Congress that its objections were justified. Finally, state objections to the disposal of military waste could be overridden by a declaration from the President that disposal was necessary for national security.

The states are likely to regard such consultation as inadequate participation in waste disposal decisions. Their continued opposition to siting threatens to frustrate the federal ability to implement a program, regardless of which disposal method is chosen; yet this matter has not been addressed by DOE.

2. Questions of Equity.--Closely tied to the problems of public acceptance are questions of equity. Because the benefits and risks of nuclear power are not shared equally around the nation, some members of the public will be asked to bear the risk of waste disposal for others. The degree of opposition at the local level indicates how the public feels about this burden.

The success of the waste disposal program will depend upon the development of siting principles that reflect both a systematic analysis of various social, political, and economic environments*, and a determination of fairness and justice in the allocation of the risk. No such systematic

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*/ See Rochlin, Demchak, Hershberger, Hoberg, Jr., La Porte and Windham, Social and Institutional Aspects of Radioactive Waste Management: Some Preliminary Findings, S. 3 (195/RW001, Oct., 1979). The editors include in their study a table which lists the kinds of information that should be collected about the social, economic, and political characteristics of representative or potential repository sites. Some examples include:

- -sociological data:
  - -urban/rural mix;
  - -professional/non-professional mix;
  - -racial and ethnographic data;
  - -age, sex and family data.

(continued on next page)

analysis has been conducted by DOE. Considerations of fairness and justice must be applied both spacially and temporally. The latter relates primarily to the intergenerational transfer of the risks associated with waste disposal, the former to the "not in my backyard" syndrome. A comprehensive approach to considerations of justice must also address the issue of compensation of persons who live near a waste repository.

DOE has failed to consider any of these issues in a direct or comprehensive way. Its views must be inferred from its adoption, as one of its program's objectives, of President Carter's requirement that "[t]he responsibility for resolving military and civilian waste management problems shall not be deferred to future generations", [11] and its meager discussion of "Social Concerns" in the Statement of Position submitted in the NRC's Rulemaking Proceedings on the Storage of Nuclear Waste (known as the "Waste Confidence" Proceedings.) This discussion alleges that "there is growing public recognition that nuclear waste management is a national problem and that solutions to the problem should not be postponed for future generations" [12].

3. Conflicts in Regulatory Policy.--The history of waste disposal program in the United States is a story of fits and starts and major changes of direction and focus, from geologic disposal to retrievable surface disposal and back again. DOE and its predecessor agencies have seized upon a single waste disposal solution, only to be forced to begin almost anew when the solution proved not to be feasible. It is likely that developments nationally, particularly in Congress, will result in further redirections of the program.

Although DOE has determined that geologic waste disposal is the method of choice, Congress has not made a similar commitment to this option. The Senate bill which passed on July 30, 1980, [13] provided for long-term away from reactor

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*/ (footnote continued from previous page)

- political profile:
  - attitude towards nuclear power generally;
  - sensitivity to local, extended and global environmental issues;
  - attitudes towards remote, centralized authority (state and/or federal).
- social profile:
  - activities;
  - mobility;
  - degree of social stratification;
  - lifestyle preferences;
  - median education level;
  - typical wages/salaries;
  - seasonal and migratory labor patterns, if any.

(AFR) storage and retrievable surface storage of high-level wastes. The bill also provided for the rapid development of unlicensed "demonstration" waste repositories on federally owned sites. All of these provisions would divert resources and efforts away from the development and implementation of a safe geologic disposal system.

Because the House was unable to pass its waste disposal legislation [14] before adjourning, differences between the House and Senate bills were never resolved. The House bill did not provide for retrievable surface storage of wastes or for an AFR program. It did require waste repositories to be licensed and subject to full review in accordance with the National Environmental Policy Act.

The Science and Technology Committee of the House of Representatives reported versions of Title I and Title VII of the DOE Authorization Act, which seriously undermine the DOE geologic disposal program [15]. The Committee's amendments provided for the reprocessing of commercial spent fuel and storage of reprocessed wastes. Accordingly, the Committee eliminated critical funding for geologic disposal activities and instead provided funding for development, virtually irrespective of geologic conditions. Moreover, the Committee's approach to geologic storage called for four demonstration repositories, the first to be in operation by 1986. Contrary to the recommendations of the Interagency Review Group, these repositories would not be licensed by the NRC and the opportunities for state and local participation in siting decisions would be limited.

The development and implementation of a safe geologic waste disposal program requires a commitment from Congress as well as the executive branch. Both must share a view of what is required to solve the waste disposal problem, and Congress must provide adequate funds to complete the task. At present, it appears that DOE and Congress are at cross purposes.

4. Managerial and Regulatory Uncertainties.--In Roger Kasperson's view, "management and regulatory issues constitute perhaps the most formidable obstacles to a timely resolution of the radioactive waste problem" [16]. Of particular concern is the absence of a mechanism for the coordination of all the departments within the federal government that have responsibility for nuclear waste.*/

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*/ The DOE Statement of Position stated that arrangements are being made for interagency cooperation among a few of the organizations concerned with waste management. §III.D.2., at III-42. These are far from complete, however. The necessary memoranda of understanding have not been prepared, nor have the substantive procedures required for collaboration and implementation of the program been (continued on next page)

Ten different institutions share responsibility for radioactive waste matters,**/ three of which were created in 1980.***/ Each of these organizations has its own mandate and agenda and its own views on the appropriate shape and course of the waste disposal program. There is no consensus that the program will produce a safe method of disposing of wastes within a reasonable time period. The U.S. Geological Survey, for example, has expressed doubts about the adequacy of the technical information supporting the program and the validity of the geological assumptions used [17]. The Office of Science and Technology Policy has stated its opinion that "the knowledge and technology base available today is not sufficient to permit complete confidence in the safety of any particular repository design or the suitability of any particular site." [18]

A significant reason for the lack of confidence in DOE by other federal agencies is that DOE still has not determined what must be done to design and implement a waste disposal program. DOE is presently trying "to define the technical efforts required for successful mined geologic waste disposal....[These include] site identification and characterization, rock mechanics, repository sealing, waste/media interactions and repository performance assessment", matters which should have been the subject of research efforts at the beginning of the waste disposal program. It is astonishing to find DOE attempting to "define the technical efforts required" for achievement of a goal that is more than twenty years old.

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*/ (Footnote continued from previous page) developed. *Id*. III.D.2.1.1., at III-42. Furthermore, the existence of cooperative arrangements does not supplant the need for a means of over-all coordination of the waste management effort. As "lead agency" for the development of a waste disposal method, DOE should function in this capacity.

**/ These institutions are the Nuclear Regulatory Commission, the Department of Energy, the Environmental Protection Agency, the Department of Transportation, the U.S. Geological Survey, the Council on Environmental Quality, the Office of Science and Technology Policy, the Federal Radiation Policy Council, the Nuclear Safety Oversight Committee, and the State Planning Council.

***/ The Federal Radiation Policy Council is responsible for the development of federal radiation protection policy. It will review actions of the NRC which affect public and occupational health. The Nuclear Safety Oversight Committee is charged with overseeing industry and government programs for improving reactor safety. The State Planning Council is responsible for coordination of waste policy between the federal and state and local governments.

A potentially more serious problem for the achievement of DOE's geologic waste disposal program is the lack of consistency in the programs and schedules of various agencies. For example, DOE has not designed a program that will meet the licensing requirements of the Nuclear Regulatory Commission. Unless Congress enacts legislation exempting waste disposal facilities from the provisions of the Atomic Energy Act, the NRC will be responsible for licensing waste repositories.*/ Licensing requires a determination by the NRC that the site chosen for the facility is suitable and that the facility can and will be constructed and operated at the site without endangering the health and safety of the public [19]. Such a determination must reflect findings by the NRC that the DOE waste disposal plan, including the site chosen for the location of the repository, meets NRC criteria and standards of performance. As presently constituted, the DOE plan does not fulfill even NRC's draft siting criteria and performance requirements. DOE's approach to siting and performance [20] is different from that of the NRC and may result in a conflict which will bring the waste disposal program to a standstill.

The NRC's draft siting criteria prohibit the location of repositories in areas with geologic features that could threaten their safe operation. These features include active faults, geothermal anomalies, acquifers of potable water that could be disrupted or contacted by the repository, known or potential mineral resources attractive to humans, and fractures that provide pathways for fluid movement. As stated in the NRC draft technical criteria:

> Unfavorable site characteristics are identified to eliminate from consideration sites which would not be acceptable under any circumstances for a HLW geologic repository or which would present insuperable difficulties in terms of understanding the geology and hydrology of the site or would introduce or compound uncertainties which would affect negatively confidence in any licensing decision [21].

DOE has not incorporated an identification of unfavorable geologic characteristics into its site selection process. Rather than specify features that would make a site unacceptable, the DOE calls for an assessment of the risk created by the existence of these features at the site. No site will be rejected unless the risk to the repository is judged to be "unacceptable." How an "unacceptable risk" will be defined and what degree of engineering and expense will be

*/ Recently promulgated final regulations make clear that waste repository licensing will continued to be the responsibility of the NRC. See 46 Fed. Reg. 13971 (1981).

tolerated in reducing the risk inherent in a site have not been determined by the DOE.

The NRC has promulgated provisional, technical performance requirements for waste repositories which are not satisfied by the DOE program plan. DOE's program is geared instead to a set of vague and flexible "objectives." For example, the NRC's draft performance standards require the waste package to provide containment of all radionuclides for the first one thousand years after decommissioning of the geologic repository [22]. The DOE objectives, on the other hand, call only for containment to be "virtually complete during the period when radiation and thermal output are dominated by fission product decay," and further state that this containment will be carried out only "to the extent reasonably achievable." [23] DOE also suggests that exposures of ten or more millirems per year would be permissible: "Radiological consequences should be maintained within the level of variations in natural background radiation associated with geographic location and domestic activites." [24] Finally, DOE imposes an economic standard to govern the operation of a repository: "[T]he environmental impacts associated with waste disposal systems should be mitigated to the extent reasonably achievable. 'To the extent reasonably achievable' means that which is shown to be reasonable considering the costs and benefits associated with potential mitigative measures...." [25]

With respect to the problem of human intrusion into a repository, the NRC draft technical criteria require the establishment of siting principles that will minimize the potential for such an occurrence. Since the most likely activities resulting in repository intrusion will concern exploration for natural resources and investigations of geophysical anomalies, the NRC criteria prohibit the location of a repository in an area with attractive natural features [26].

As noted, earlier, DOE has ignored the NRC's recommendations for siting criteria that would avoid the use of sites with valuable natural resources. The DOE continues to consider salt to be an acceptable repository host, [27] despite the fact that bedded salt and salt domes are far more attractive resources than granite, shale, or basalt. A special ad hoc panel of earth scientists commented on this consideration in its report to the Environmental Protection Agency, calling the resource value of salt "an important negative socio-economic factor" in the use of certain potential repository sites [28]. The report stated that

> [t]he most likely targets for near-term exploitation...are salt domes because of the potential productivity of petroleum, halite, and sulfur; and bedded salt deposits because of their potash, halite, and gypsum. The United States has only 4% of the world's total proven

> potash reserves, and most of these are concentrated in the New Mexico area now being evaluated as an HLW repository. Future conflicts between the demand for HLW repositories in bedded salt and the needs of agriculture for potash seem inevitable, and may even now constitute an important negative socio-economic factor in the development of some repositories [29].

The Waste Isolation Pilot Plant (WIPP) site in New Mexico is another example of DOE's disregard for formulating siting principles that would reduce the repository hazards for future generations. The site includes known accumulations of potash, natural gas, and oil, all of which are valuable now and are likely to become increasingly important in the future. NRC criteria would prohibit the location of a repository at a site with such valuable resources.

DOE acknowledges that the NRC's procedural requirements for licensing a waste repository could have a "major impact on costs and schedule." [30] In fact, these requirements could mean the failure of the DOE program. If the two federal agencies with the greatest responsibility for waste disposal do not share a view of how to achieve a safe, reliable waste disposal program, the confidence of states and the public in the federal government's ability to provide a timely solution will be further diminished.

B. The Second Phase--Scaling Up

The entire focus of the present DOE program is on the location, construction, and operation of one repository, designed to accommodate the nuclear waste DOE anticipates will be produced by the year 2000 [31]. DOE has yet to address the technical and organizational problems of "scaling-up" from one facility to a disposal system capable of accommodating the wastes from an expanding nuclear industry. The need to solve these problems is far from theoretical. The roughly 200 GWe (gigawatts electric) of nuclear power already on the books--that is, in plants in operation, under construction, ordered, or publicly announced--will produce enough high-level radioactive waste to fill two repositories, if the DOE capacity figure of 100,000 tons of waste per a 2,000 acre repository is used, [32] or six repositories, if the California Energy Commission figure of 35,000 tons per repository [33] is relied upon. If a nuclear commitment of 300 GWe by the year 2000 is assumed,* these numbers increase to three and

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*/ The Electric Power Research Institute has argued that about 400 GWe by the year 2000 reflects a minimum growth figure for the nuclear industry to survive. See EPRI, "Nuclear Waste Management Status and Recent Accomplishments", Final Report (NP-D87, May, 1979).

nine repositories respectively.

One fundamental problem with expansion of the waste disposal system is that it must be essentially error-free from the outset. "[T]he incremental approach to perfect performance...is explicitly not an option for the waste management program." [34] In other words, the public will not tolerate a "learning curve" for waste disposal operations.

Second, the organization required to support an expanded network of disposal sites will have different and more serious problems than those confronting the location and operation of a single repository. The organizational complexity of an expanded waste disposal program is not linear with its size. As more waste repositories are needed, the problems associated with site selection, facility design, security, and transportation are multiplied, wholly apart from the purely technical problems involved. Furthermore, as the waste disposal system expands, public confidence in its ability to perform without malfunction is likely to decrease. In part, this is due to the application of that "bit of organizational folklore, Murphy's Law":

> The larger the volume of waste materials and the more varied its composition, the larger and more complicated the total system is likely to be; and the more complicated the system, the more we are prone to imagine that, if anything can go wrong, invariably it will at some time or another [35].

As the accident at Three Mile Island demonstrated, the least reliable factor in an elaborate scheme to control nuclear dangers is the human factor. This factor will become increasingly crucial as the program expands. According to one commentator,

> [a]s the volume of wastes increases, the most crucial scarce resource may well become the people who are highly skilled and who can be motivated sufficiently to perform continuously at extraordinarily high levels of reliability, even though it is likely that the jobs will generally be routine and boring on a day-to-day basis [36].

Increased dependence on human reliability requires that the organization be equipped with an "error detection mechanism" that will "reward detection and correction of error rather than its denial or cover-up." [37] Nothing in the DOE program is responsive to this problem.

Third, DOE has failed to analyze the impact of an expanded waste disposal system on the social structure of the communities directly affected by transportation and repository siting. DOE has assumed that "social concerns" about the safety of nuclear waste disposal will be resolved because of the "growing public recognition that nuclear waste managment is a national problem." [38] This attitude ignores a critical set of issues that could lead to rejection of a waste management program. For example, DOE has not determined whether it will locate a series of waste repositories at one site or region, or spread them out in various locations across the nation. The social, economic, and political implications of these two strategies differ, yet DOE has not assessed them.

Finally, DOE has not prepared a detailed cost estimate of a comprehensive waste management program. The need for organizational refinement and superior personnel necessarily will lead to a high cost program--a cost which may be disproportionate to the "benefits" of nuclear power production. Moreover, the cost to civil liberties resulting from an authoritarian waste disposal bureaucracy which decides which communities become perpetual hazardous waste dumping grounds may be too great for society to bear.

C. The Long-Term Management Phase

The final phase of the waste disposal program, which must be assessed in terms of the social, economic, and political obstacles to its implementation, is the long-term management phase. In this phase, the disposal technology and institutional arrangements will be tested over long periods of time.

It is impossible to make any predictions about the stability of the social fabric or social and political institutions for the length of time during which the nuclear wastes generated today will remain hazardous. As a consequence, it may not be possible to design any system other than an engineered one for the protection of future generations. This does not, however, excuse consideration of the fundamental question of whether society has a right to subject future generations who may share none of the benefits of nuclear energy to the risks inherent in its waste. DOE's continuing failure to address seriously this issue is a clear indication of its lack of understanding of the social and political obstacles to the implementation of its program.

CONCLUSION

The history of the federal government's efforts to find a solution to the problem of nuclear waste disposal provides no basis for confidence on the part of the American public that nuclear wastes will be managed safely in the future. It is a history of "unbroken failure to produce an acceptable method of waste disposal,"[39] a history of fits and starts and major changes in direction and focus. Along the way the federal government has adopted and then been forced to abandon disposal sites, media, and technologies, in large part because of social and political obstacles to its program.

The history of the government waste disposal effort shows that little has been learned in the past twenty years. DOE has yet to identify or address the social, political, and economic issues involved in the implementations of its waste disposal program. It has not developed a plan that will meet even the NRC's draft performance criteria for geologic repositories. In numerous instances, DOE's program objectives are in conflict with the NRC's criteria. Even when its objectives are not in conflict, there is no evidence that the NRC criteria will be met by the DOE program.

DOE's emphasis continues to be on the technical features of the waste disposal system, although the resolution of social and institutional issues is of equal importance. Indeed, failure to properly resolve these obstacles to implementation may doom an otherwise acceptable program.

For these reasons, many segments of the American public have substantial doubts about DOE's continued promises of a prompt solution to this fundamental and long-standing problem with the application of nuclear technology. They have doubts about the ability of DOE to provide a safe and reliable waste disposal system. DOE must confront the fact that a waste disposal facility is as welcome as a skunk at a lawn party. Until the "not in _my_ backyard" syndrome is understood and countered, the gravest uncertainties exist as to the success of the geologic waste disposal effort.

REFERENCES

1. W. Bishop, N. Hilberry, I. Hoos, D. Metlay, and R. Watson (eds.), Nuclear Regulatory Commission, Essays on Issues Relevant to the Regulation of Radioactive Waste Management 57 NUREG-0412 (1978).

2. Interagency Review Group on Nuclear Waste Management, Department of Energy, Report to the President (NTIS Report, TID-29442, Mar., 1979), at 87.

3. National Academy of Sciences, 30 News Report.

4. Rochlin, Demchak, Hershberger, Hoberg, Jr., LaPorte & Windham, Social and Institutional Aspects of Radioactive Waste Management: Some Preliminary Findings, S.3 (195/RW001, Oct., 1979).

5. Office of Energy Research, (DOE/ER-004/D, UC-70, Feb., 1978).

6. Testimony of Roger Kasperson, "Institutional and Social Uncertainties in Radioactive Waste Management", presented to the Ohio Power Siting Commission (June 27, 1978).

7. Comptroller General of the United States, Report of the General Accounting Office, Nuclear Energy's Dilemma: Disposing of Hazardous Radioactive Waste Safely (Sept. 1977).

8. Nuclear Regulatory Commission, Office of State Programs, Information Report on State Legislation (1978-79); Brookhaven National Laboratory Division of Regional Studies, Summary of Nuclear Waste Legislation (Unpublished draft document) (Nov. 1980).

9. President's Message to Congress, "Comprehensive Radioactive Waste Management Program", reprinted in [1980] U.S. Code Cong. & Ad. News 541, 541-45.

10. S.2189, 96th Cong., 2d Ses. (1980).

11. Department of Energy, Statement of Position, In Re Proposed Rulemaking on the Storage and Disposal of Nuclear Waste, NE-0007 (Apr. 15, 1980) § II.A.1.3, at II-18.

12. Id. § III.F.2.2.2., at III-87.

13. S.2189, 96th Cong., 2d Sess. (1980).

14. H.R. 8378, 96th Cong., 2d Sess. (1980).

15. H.R. 6627, 96th Cong., 2d Sess. (1980).

16. Testimony of Roger Kasperson, *supra* ref. 6, at 20.

17. J. Carter, "Nuclear Wastes: The Science of Geologic Disposal Seen as Weak", 200 *Science* 1135 (1978).

18. T. Bedehoeft, A. England, D. Stewart, N. Trask, & I. Winograd, "Geological Disposal of High-Level Radioactive Wastes--Earth Science Perspectives", *U.S. Geological Survey Circular* 779 (1978).

19. 42 U.S.C. §§ 2011, 2133(d) (1954); 10 C.F.R. §§ 50.57(a)(6) (1975), 50.91 (1974).

20. Nuclear Regulatory Commission, "Draft Technical Criteria for Regulating Geologic Disposal of High-Level Radioactive Waste", 45 *Fed*. *Reg*. 31, 393 (1980).

21. *Id*.

22. *Id*.

23. Department of Energy *Statement of Position*, *supra* ref. 11, at II-6.

24. *Id*.

25. *Id*. at II-16.

26. *Draft Technical Criteria*, *supra* ref. 20, at 5.

27. Department of Energy, *Management of Commercially Generated Radioactive Waste* (Final EIS, DOE/EIS-0046F, Oct., 1980).

28. T. English, "An Analysis of the Back End of the Nuclear Fuel Cycle with Emphasis on High-Level Management", *Jet Propulsion Laboratory Pub. No. 77-59*, viii, Aug. 12, 1977; J. Bredehoeft, A. England, D. Stewart, N. Trask, & I. Winograd, "Geologic Disposal of High-Level Radioactive Waste--Earth Science Perspectives", 3 (*Geological Survey Circular* 779, 1978); Ad Hoc Panel of Earth Scientists, "The State of Geological Knowledge Regarding Potential Transport of High-Level Radioactive Waste from Deep Continental Repositories", 32 *Environmental Protection Agency*, June, 1978).

29. *Id*.

30. DOE Statement of Position, supra ref. 11, at III-73.

31. Id. at III-22, 57-59, 77.

32. Opinion of Dr. Thomas Cochran, technical expert at the National Resources Defense Council, Washington, D.C.

33. Id.

34. Social and Institutional Aspects, supra ref. 4, at 3.47.

35. T. LaPorte, "Nuclear Waste: Increasing Scale and Socio-Political Impacts", 201 Science 26 (1978).

36. Id. at 23.

37. Social and Institutional Aspects, supra ref. 4 at 3.47.

38. DOE Statement of Position, supra ref. 11 at III-87.

THE PERSPECTIVE OF STATE GOVERNMENT CONCERNING THE SOCIAL AND INSTITUTIONAL UNCERTAINTIES ASSOCIATED WITH THE REGULATION OF THE GEOLOGIC DISPOSAL OF HIGH-LEVEL RADIOACTIVE WASTE

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ABSTRACT

States should have a meaningful role in the siting and licensing of a geologic high-level radioactive waste disposal facility. This inclusion of states into the siting and licensing processes will add social and institutional uncertainties which must be resolved if the processes are to be successful. These uncertainties arise from: 1) the failure of Congress to clearly define the states' role in these processes, 2) existing state laws which apply to the siting and licensing of a nuclear waste disposal facility, and 3) the variations among states as to their willingness to "host" a disposal facility and their capabilities to be directly involved in reviewing and licensing such a unique and complex facility. The Nuclear Regulatory Commission is urged to encourage and facilitate state participation in the siting and licensing processes and to identify and resolve conflicts as early as possible.

INTRODUCTION

There is a great deal of discussion going on today concerning the regulation of high-level radioactive waste disposal which deals with the question of whether states ought to have an important and visible role in the process that is designed to lead to the siting and licensing of a geologic disposal facility. Much of this discussion can be found in the record of the generic proceeding now pending before the U. S. Nuclear Regulatory Commission (NRC) known as the "Waste Confidence Proceeding" (Docket No. PR-50, 51) in which the primary issue is: "Can the NRC reasonably conclude that it is reasonably confident that radioactive wastes produced by nuclear facilities will be disposed of safely?"

The record in the Waste Confidence Proceeding indicates that, at a minimum, the following parties believe that states should have a meaningful role in the siting and licensing of a geologic high-level radioactive waste disposal facility: former President Carter, who established a State Planning Council to deal with disposal issues; the U. S. Department of Energy (DOE), which states at page II-11 of its Cross Statement (September 5, 1980) that it is fully committed to giving state and local governments an "important role" in the process; the states, which, if selected as repository hosts, have a direct stake in the outcome of the process; and environmental groups, who want the process carefully monitored by local authorities.

The necessary inclusion of a host state into the siting and licensing processes will add social and institutional obstacles and uncertainties which must be resolved if the processes are to be successful. The discussion which follows will attempt to identify these uncertainties and will attempt to identify, where possible, the point at which these uncertainties can be addressed by the NRC. The social and institutional uncertainties identified arise from three sources: 1) what states are able to do, 2) what states have done in the past, and 3) what states are likely to do in the future.

WHAT CAN STATES DO?

The largest uncertainty which faces both states and the federal government at this time stems from the fact that everyone knows that Congress must address the states' role in site selection as soon as possible. The last session of Congress tried and failed to address this issue, and Senate File 2189 which was eventually enacted by Congress and signed by President Carter is silent on this issue. Everyone seems to recognize this necessity, and yet it is an issue which has so many different possible solutions that the members of Congress have not been able to agree upon one.

In light of Congressional inaction, federal agencies have held back from taking an aggressive leadership role defining how much ultimate authority states should have with respect to repository site selection. DOE officials in their Cross Statement in the Waste Confidence Proceeding claim to have an aggressive "consultation and concurrence" program to involve state officials in their exploration activities, and they predict active efforts to involve states in the siting and licensing processes. However, these actions and predictions are now nothing more than good intentions and certainly do not settle the question of what states can do with respect to siting and licensing decisions.

The NRC has an opportunity to take an aggressive leadership position at least with respect to state involvement in the licensing stage of repository approval. The NRC should take advantage of this opportunity and should not only provide opportunities for state involvement in licensing but should also encourage and facilitate state participation in the process. NRC efforts should go farther than contacting the highest state and local officials and should include an active attempt to identify those state agencies and groups with an historical or present interest in nuclear issues. Such efforts will be advantageous to the NRC in the long run.

WHAT HAVE STATES DONE?

Successful siting of a geologic high-level radioactive waste facility in many states is uncertain at this time due to the fact that many states have existing laws that either prohibit siting of a nuclear waste disposal facility in that state or require the approval of the state legislature or other state body.

The State of Minnesota is an example of a state that requires legislative approval prior to construction of a nuclear waste repository. Minn. Stat. §116C.72 (1980) provides that "no person shall construct or operate a radioactive waste management facility within Minnesota unless expressly authorized by the Minnesota legislature." At least twenty-three other states have laws directly applicable to high-level radioactive waste disposal. These states are as follows:

1. Alabama (Ala. Code §22-14-16 (Supp. 1980)).

2. Colorado (Colo. Rev. Stat. §§25-11-103, 25-11-201, 25-11-202, 25-11-203).

3. Connecticut (Conn. Gen. Stat. §22a-137(a) - (f)).

4. Indiana (Ind. Code §4-21-7-2.1(6)).

5. Iowa (Iowa Code §455B.88 (applies to private persons only)).

6. Kentucky (Ky. Rev. Stat. §211.852).

7. Louisiana (La. Rev. Stat. Ann. §30:1115).

8. Maine (Me. Rev. Stat. tit. 1, §15-A; Me. Rev. Stat. tit. 38, §361-D).

9. Maryland (Md. Ann. Code art. 43 §689B).

10. Michigan (Mich. Comp. Laws Ann. §325.491).

11. Mississippi (Miss. Code Ann. §17-17-49).

12. Montana (Mont. Rev. Code §75-3-302).

13. New Hampshire (N. H. Rev. Stat. Ann. ch. 125 §77).

14. New Mexico (N. M. Stat. Ann. §§74-4A-5, 74-4A-6, 74-4A-7).

15. New York (1980 N. Y. Laws ch. 260).

16. North Dakota (N. D. Cent. Code §23-20.2-09).

17. Oregon (Or. Rev. Stat. §469.525).

18. South Dakota (S. D. Codified Laws Ann. §34-21-1.1).

19. Texas (Tex. Rev. Civ. Stat. art. 4590-f).

20. Utah (S.B. No. 18, approved March, 1981).

21. Vermont (10 Vt. Stat. Ann. §6501).

22. Washington (Initiative 383, 1981 Wash. Laws ch. 1, to be codified within Wash. Rev. Code tit. 70).

23. West Virginia (W. Va. Code ch. 16, art. 27 §2).

In addition, the North Carolina legislature has passed a nonbinding resolution opposing the storage or disposal of out-of-state radioactive wastes unless approved by the legislature.

All of these state laws, of course, might be subject to challenge on the grounds of federal preemption. However, their very existence brings uncertainty into the siting process. The federal government may not want, for political or other reasons, to take a state to court to challenge the state law. Even if a court action is begun, it takes time and resources before a final decision can be obtained. Thus the NRC cannot view these state restrictions lightly or expect them to go away. These restrictions represent a serious uncertainty in the siting and licensing processes which cannot really be resolved until a specific state is chosen for siting of a repository.

Even if a state does not have a repository siting law it may have other regulatory schemes in effect which it may wish to impose on siting activities. In Minnesota, for example, there is a statute requiring issuance by the Minnesota Energy Agency (MEA) of a certificate of need prior to the siting or construction of a nuclear waste disposal facility. Minn. Stat. §116H.13 (1980) and Minn. Stat. §116H.02, subd. 5(k) (1980). While this statute may be subject to the same attacks on preemption grounds as the siting restrictions discussed above, its existence remains a potential obstacle in the siting and licensing process.

Once a potential site is identified and proposed for construction, there are a number of state approvals, at least in Minnesota, that would be required, assuming the absence of a federal statute explicitly preventing states from requiring such approvals. Minnesota's existing law requires the following:

1. Preparation of a state environmental impact statement (EIS) (in addition to the federal EIS, although the federal EIS may be incorporated into the state EIS and thus shorten the process). Minn. Stat. §116D.04, subd. 2a (1980).

2. Submission of plans and design to and obtaining construction approval from the Commissioner of Health. Minn. Rule MHD 185 (7 MCAR §1.185).

3. Obtaining a National Pollutant Discharge Elimination System Permit and State Disposal Permit from the Minnesota Pollution Control Agency (MPCA) for any discharge of pollutants to the waters of the state during construction and/or operation of the facility. 33 U.S.C. §§1311(a), 1342 (Supp. IV, 1974); Minn. Stat. §115.07 (1980) and Minn. Rule WPC 36 (6 MCAR §4.8036).

4. Obtaining air emission facility installation and operating permits from the MPCA for emission of air pollutants into the outdoor atmosphere. Minn. Stat. §§116.04, subd. 4a and 116.081 (1980).

5. Obtaining a permit for the appropriation of water from the Department of Natural Resources, if water appropriation is necessary. Minn. Stat. §105.41 (1980).

6. Obtaining access permits from the Department of Transportation, if necessary. Minn. Stat. §160.18, subd. 3 (1980) and 4 MCAR §1.5036.D.

7. Obtaining miscellaneous minor permits, such as permits for burning construction wastes, as needed. Minn. Stat. §§116.07, subd. 4a and 116.081 (1980).

Again, federal preemption under the Atomic Energy Act comes into question if Minnesota applies its laws to a federal waste repository.

Another legal dispute that will inevitably arise if a state such as Minnesota insists upon federal compliance with all its existing laws is the constitutional question known as the "federal enclave exclusion," which arises from Article I, section 8, clause 17 of the U. S. Constitution. Under current plans the nuclear waste repository is to be constructed by DOE on federal lands as required by 10 C.F.R. Part 50, Appendix F (3). The federal enclave doctrine prevents states from regulating federal installations except when Congress has clearly and unambiguously authorized such regulation. Under current federal law there are at least two types of state permits previously listed that Minnesota can insist upon requiring: the air emission and water discharge permits.

The source of the state's authority to require air emission permits for federal facilities is the act of Congress amending the Clean Air Act. The Clean Air Act was amended in 1977 so that section 118 (42 U.S.C. §7418(a)) now provides in relevant part:

> Each department, agency and instrumentality of the executive, legislative, and judicial branches of the Federal Government... engaged in any activity resulting, or which may result, in the discharge of air pollutants...shall be subject to, and comply with, all Federal, State, interstate, and local requirements,

> administrative authority, and process and sanctions respecting the control and abatement of air pollution in the same manner, and to the same extent as any nongovernmental entity. The preceding sentence shall apply (A) to any requirement whether substantive or procedural (including any recordkeeping or reporting requirement, any requirement respecting permits and any other requirement whatsoever).

The legislative history clearly indicates that this section was designed to overturn a 1976 U. S. Supreme Court decision (Hancock v. Train, 426 U.S. 167 (1976)) holding federal facilities exempt from obtaining state permits concerning air pollution control. 1977 U. S. Code Cong. and Ad. News at 1276-1280.

The air pollutants of concern in a mined geologic repository are the radioactive air emissions and also the fugitive particulate emissions which will result from the construction and/or operation of the facility. Particulate emissions have traditionally been regulated by states. Radioactive air emissions are now subject to state regulation under the 1977 amendments to the Clean Air Act. P.L. 95-95 §§302(g), 116, 42 U.S.C. §§7602(g) and 7416. See also P.L. 95-95 §122(a), 110 and 112(d)(1), 42 U.S.C. §§7422(a), 7410 and 7412(d)(1). See also H.R. Rep. No. 95-564, 95th Cong., 1st Sess. 141, reprinted in [Sept. 1977] U.S. Code Cong. and Ad. News at 2207, 2653-2654. Therefore existing federal law allows states to require air pollution permits from federal facilities.

The Clean Water Act, P.L. 95-217, 33 U.S.C. §466 et seq. was also amended in 1977 to add new language identical to that of the Clean Air Act with respect to the requirement that federal agencies obtain state permits respecting control and abatement of water pollution. P.L. 95-217 §313(a), 33 U.S.C. §1323(a). States are already exercising permitting authority with respect to nonradiological water pollutants from nuclear reactors, and it would not be stretching the states' capabilities to review the nonradiological water impacts of a proposed federal facility.

In light of the existing federal statutes concerning air and water pollution, the NRC cannot discount state permitting procedures as an institutional uncertainty in the siting and licensing processes.

The NRC will have an opportunity to limit the uncertainty as to what states will do under their existing law as soon as a state is targeted for siting of a repository. The NRC should, as early as possible, sit down with state officials and begin to define the state's position with respect to the dual regulation. It may well be that some states will be willing to exercise their regulatory powers in a manner that will not conflict with federal rules and policies. If conflicts are identified, conflict resolution should be started as early as possible. In this manner uncertainties arising from what states have done in the past can become manageable.

WHAT WILL STATES DO?

There is uncertainty as to what any given state will actually do when faced with a concrete possibility that a high-level radioactive waste disposal facility will be sited within its borders. What the states will do will depend not only upon the existing state laws that could be applied to the facility but also upon several other factors.

One factor that will influence state actions is the amount of incentives offered by the federal government to the host state, such as additional revenues or aid to the affected community for the social displacements that will result from the construction and operation of the facility.

A second and probably the most important factor is the political climate in the state. This factor will be influenced by public opinions and fears about the project, which is in turn a function of the perceived risks to the people compared to the perceived benefits to the state. Public opinion will also be influenced by the nature of the land chosen by the federal government to be a waste disposal site. For example, siting a facility in the Boundary Waters Canoe Area in Minnesota is likely to meet with opposition from the people who value it as a recreational resource.

A third factor that will determine the state's handling of the matter involves the capabilities of the state to handle review of such a unique and complex facility. For example, some states have staff people who already deal daily with problems related to nuclear facilities; others may not have any expertise on staff in this area. The same discrepancies in state expertise could exist in other subject areas related to the construction and operation of the facility, such as transportation of nuclear waste to the site.

The uncertainty of what any given state will do once chosen as a host state cannot be fully resolved until the state is chosen. That does not mean that the NRC should stand idle until a site is chosen. It should start creating mechanisms immediately to address these factors in advance to the greatest possible degree. For example, it is possible at this time to identify incentives that have been offered to states in the past for large federal facilities that are associated with a degree of risk to the local population. The NRC can begin to sort out which of these incentives are suitable for the siting and licensing processes. The NRC should actively work with states as early in the process as possible and use its experience in working with Agreement States as a basis for future programs.

CONCLUSION

Both federal and state governments are currently floating on a sea of uncertainty as they look into the future toward the siting and licensing of a high-level radioactive waste disposal facility. Much of the uncertainty stems from unresolved questions about what federal and state governments are capable of doing and what they will actually do in the future. In fact, as the states participating in the Waste Confidence Proceeding have pointed out, inability to resolve social and institutional difficulties may ultimately be the most difficult hurdle to be faced in the entire process.

The NRC at this point in time has an opportunity to take an aggressive leadership role in defining the states' role in the siting and licensing processes. NRC, as the regulatory agency, also has an opportunity to review DOE's plans for involving the states and can determine to some degree the rules of the two federal agencies as they relate to the states. The states would urge that this leadership role be taken and that states be given a maximum opportunity to be involved very early. Taking the states very seriously will have benefits for all concerned. It will reduce uncertainties and distrust and will optimize the chances for developing mutual respect among the players. Early resolution of conflicts, even if that resolution is a result of litigation, is desirable because it allows time for necessary adjustments to be made by both sides.

Finally, in adopting regulations relating to geologic disposal of radioactive waste, the NRC should remember that there is a great deal of information in the record of the Waste Confidence Proceeding which is relevant to the question of the social and institutional uncertainties which face the NRC. The NRC should consider this record in obtaining a more complete review of these uncertainties.

INDUSTRY VIEW ON MANAGEMENT OF HIGH-LEVEL WASTE

Joseph A. Lieberman
Nuclear Safety Associates

(Paper Not Submitted)

INTERGOVERNMENTAL ISSUES IN RADIOACTIVE WASTE MANAGEMENT

John Stucker
Executive Director,
State Planning Council on Radioactive Waste Management

(Paper Not Submitted)

Session I:

THE SANDIA RISK ASSESSMENT METHODOLOGY

Chairman

Paul S. Rohwer

Oak Ridge National Laboratory

THE USE OF PROBABILISTIC RISK ASSESSMENTS IN WASTE MANAGEMENT

Michael C. Cullingford
U. S. Nuclear Regulatory Commission

(Paper Not Submitted)

STRUCTURE AND OVERVIEW OF THE SANDIA WASTE ISOLATION RISK ASSESSMENT METHODOLOGY

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ABSTRACT

The Fuel Cycle Risk Analysis Division of Sandia National Laboratories is funded by the Nuclear Regulatory Commission (NRC) to develop a methodology for assessment of the long-term risks from radioactive waste disposal in deep geologic media. Analytical models have been developed to represent the processes by which radioactive waste might leave the waste repository, enter the surface environment and enventually reach humans. A hypothetical reference system has been developed to provide a realistic setting for exercise of the models in the risk assessment. An overview of this methodology is presented here.

The Fuel Cycle Risk Analysis Division of Sandia National Laboratories is funded by NRC to develop a risk analysis methodology for nuclear waste. The objective of the program is to develop a methodology to examine the long-term risks from radioactive waste disposed in deep, geologic formations and to demonstrate this methodology by application to a hypothetical reference site.

The structure of this methodology is shown in Figure 1 [1]. The "Site Description" and "Radioactive Waste Description" blocks in the diagram represent information and numerical data describing the waste form and the repository setting that must be supplied to the computer models, which are shown as the four, central blocks of the diagram. They include geological, geochemical, hydrological, environmental as well as waste-related information. The broken squares represent output from the preceding model and input to the next model.

The first model block is the "Potential Waste Release Mechanism Models." These models attempt to provide insight into the evolution of the repository site and a set of potential disruptive scenarios that might lead to significant radiologic consequence are postulated [2]. These disruptive scenarios include natural processes, for example faulting and earthquakes, waste-induced processes, for example thermally induced fractures in geomedia, as well as human activities, for example borehole drilling. With these, the release time and time-dependent probability of local release of waste from the immediate surroundings of the repository are estimated. These results, which are scenario dependent, are then fed into a "Groundwater Transport Model."

The "Groundwater Transport Model" calculates the movement of radionuclides in groundwater from the time of their first contact with groundwater to the time of their appearance in the environment. This model accounts for radioactive decay and production of daughter radionuclides for a chain of up to ten members. It also accounts for sorption of the radionuclides in the geologic medium. The Network Flow and Transport (NWFT) model was developed to provide solutions to the flow equations for a simple flow network and from these a flow path is determined [3]. Then a Distributed Velocity Method (DVM) is used to simulate the migration of radionuclides along this flow path [4]. With this method, rather than tracking individual particles, ensembles of particles are considered. The output from this model is the discharge rate of each radionuclide at pre-selected points in biosphere.

The Pathway to Man model [5] is the bridge between the Groundwater Transport model and the Dosimetry model. The purpose of the model is to represent the physical and biological processes that result in the transport of radionuclides through the earth's surface environment and in man's eventual exposure to these radionuclides. This model takes the discharge rates from the Groundwater Transport Model as input and calculates the distribution and accumulation of radionuclides in the environmental compartments, e.g., soil, river, etc. From these, it then calculates the rates of radionuclide ingestion and inhalation by humans.

Finally, the probability of latent somatic effects on humans (on an individual basis) are estimated by a "Dosimetry and Health Effect" model.

In order to exercise this methodology, we have defined a hypothetical reference site of bedded salt. This reference site is entirely hypothetical, yet its physiographic setting and its geologic and hydrologic properties are analogous to several regions in the U.S. The site is located in a symmetrical upland valley, half of which is shown schematically in Figure 2. The crest of the ridge surrounding the valley is at an elevation of 6000 ft. The crest is a surface and groundwater divide so that the only water moving in the valley falls in the valley itself. The valley is drained by a major river, River L, which is at elevation 2500 feet opposite the surface structures of the repository. The valley receives a mean annual rainfall of 40 inches of which 16 inches are lost by evapotranspiration and the remaining 24 inches recharge the groundwater system. The geology of the area near the site is shown in cross section in Figure 3. The depository is located in the middle of a salt layer which is surrounded by shale. Above and below the shale layers are aquifers. The groundwater at this reference site is recharged updip and flows through the two layers of aquifer and finally discharges into River L.

In exercising this methodology, a set of variables are input into the analytical models, e.g., distribution coefficients and solubility limits of radionuclides, hydraulic conductivities of sandstones, leach time, release time, etc. There are, however, large uncertainties associated with most of these input variables. Therefore, ranges rather than point values are specified in our methodology. Figure 4 shows the structure of this technique. Here, each of the input variables has a range of values and a distribution. With a sampling technique called Latin-Hypercube Sampling [6], sets of input values can be generated by sampling the input variables from their ranges and distributions. Consequently, sets of results from our models are obtained by using the sets of input variables generated. With this scheme, the uncertainties in input variables are associated with a range of consequences in our methodology and the result from a set of input values represents the risk associated with a set of possible environmental conditions.

Finally, an important part of our risk assessment methodology is the study of the sensitivity of the risk to the input variables. Sensitivity analysis [7] is therefore always performed to identify the important contributors to risk. In summary, the products and applications of this methodology are the following:

-- Models to evaluate prospective disposal sites

-- Statistical methods for treating uncertainties and identifying major contributors to risk

-- Probabilistic methods for risk calculations

-- Provide technical basis for licensing criteria and facilitate review of licensing applications

-- Identify research and data needs.

REFERENCES

1. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P. E. McGrath, F. J. Pearson, H. R. Shaw, J. C. Helton, and F. A. Donath, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, Sandia Laboratories, SAND78-0029, October 1978.

2. R. M. Cranwell, et al., Risk Methodology for Geologic Disposal of Radioactive Waste: Scenario Formulation, Development, and Screening, SAND 80-1429, NUREG/CR-1667, to be published.

3. J. E. Campbell, P. C. Kaestner, B. S. Langkopf, and R. B. Lantz, Risk Methodology for Geologic Disposal of Radioactive Waste: The Network Flow and Transport (NWFT) Model, Sandia Laboratories, SAND79-1920, NUREG/CR-1190, February 1980.

4. J. E. Campbell, D. E. Longsine, M. Reeves, Risk Methodology for Geologic Disposal of Radioactive Waste: The Distributed Velocity Method of Solving the Convective-Dispersion Equation, Sandia National Laboratories, SAND 80-0717, NUREG/CR-1376, July 1980.

5. J. C. Helton and P. C. Kaestner, Risk Methodology for Geologic Disposal of Radioactive Waste: Model Description and User Manual for Pathways Model, Sandia National Laboratories, SAND78-1711, NUREG/CR-1636, Vol. I, March 1981.

6. R. L. Iman, J. M. Davenport and D. K. Ziegler, Latin Hypercube Sampling (Program User's Guide), Sandia Laboratories, SAND79-1473, January 1980.

7. R. L. Iman, J. C. Helton and J. E. Campbell, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis Techniques, Sandia Laboratories, SAND78-0912, NUREG/CR-0390, October 1978.

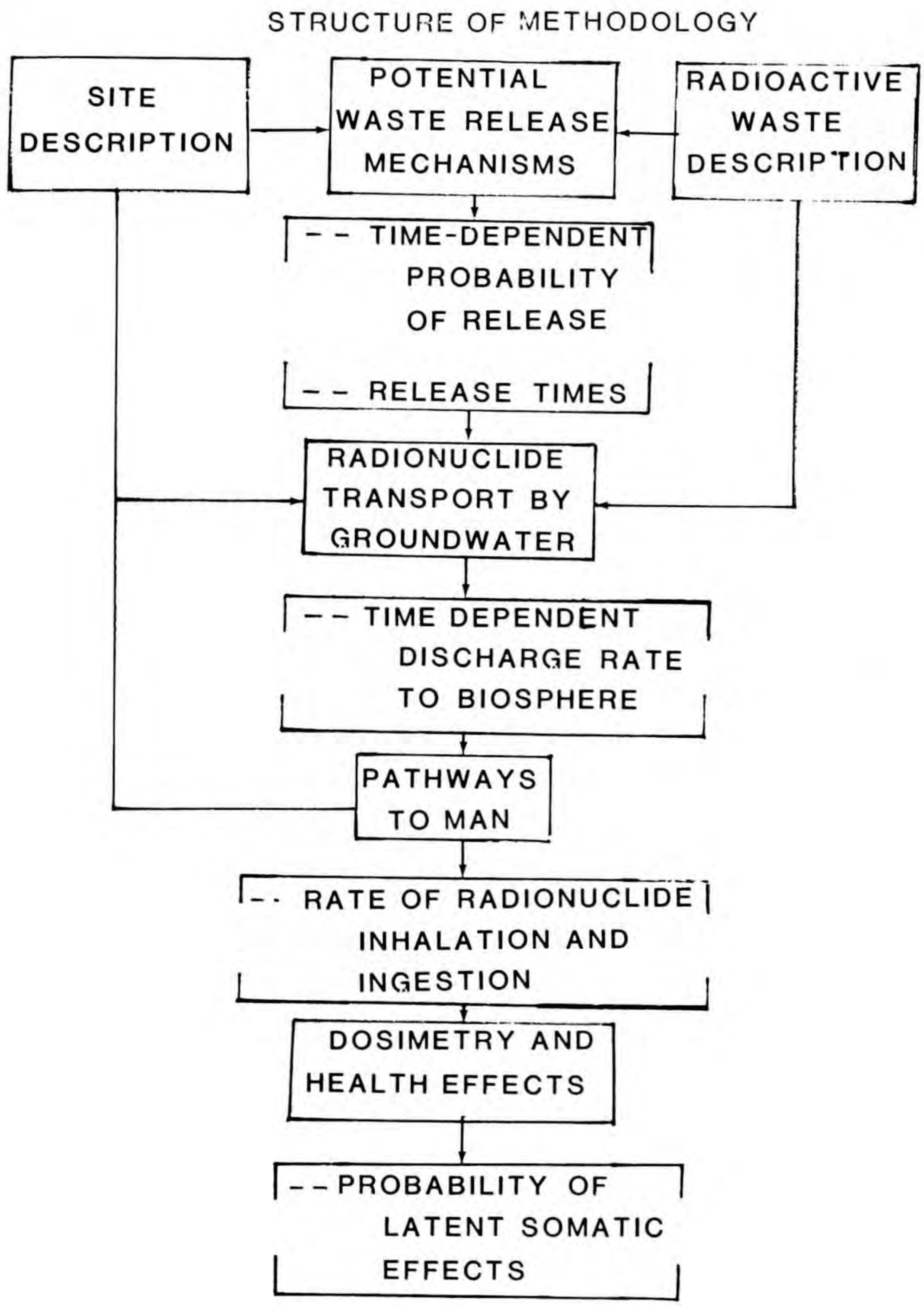
STRUCTURE OF METHODOLOGY
SITE DESCRIPTION
POTENTIAL WASTE RELEASE MECHANISMS
RADIOACTIVE WASTE DESCRIPTION
-- TIME-DEPENDENT PROBABILITY OF RELEASE
-- RELEASE TIMES
RADIONUCLIDE TRANSPORT BY GROUNDWATER
-- TIME DEPENDENT DISCHARGE RATE TO BIOSPHERE
PATHWAYS TO MAN
-- RATE OF RADIONUCLIDE INHALATION AND INGESTION
DOSIMETRY AND HEALTH EFFECTS
-- PROBABILITY OF LATENT SOMATIC EFFECTS

FIGURE 1

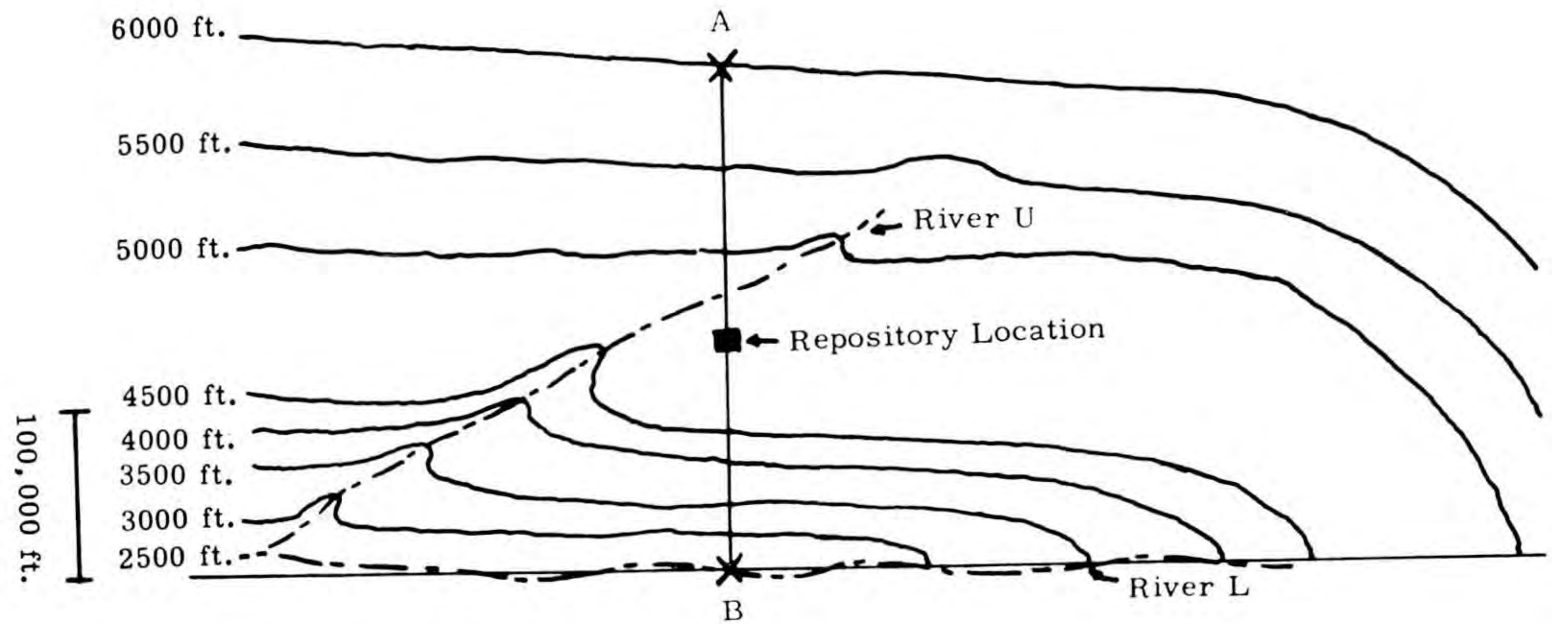
A
6000 ft.
5500 ft.
River U
5000 ft.
Repository Location
4500 ft.
4000 ft.
3500 ft.
3000 ft.
2500 ft.
100,000 ft.
River L
B

FIGURE 2

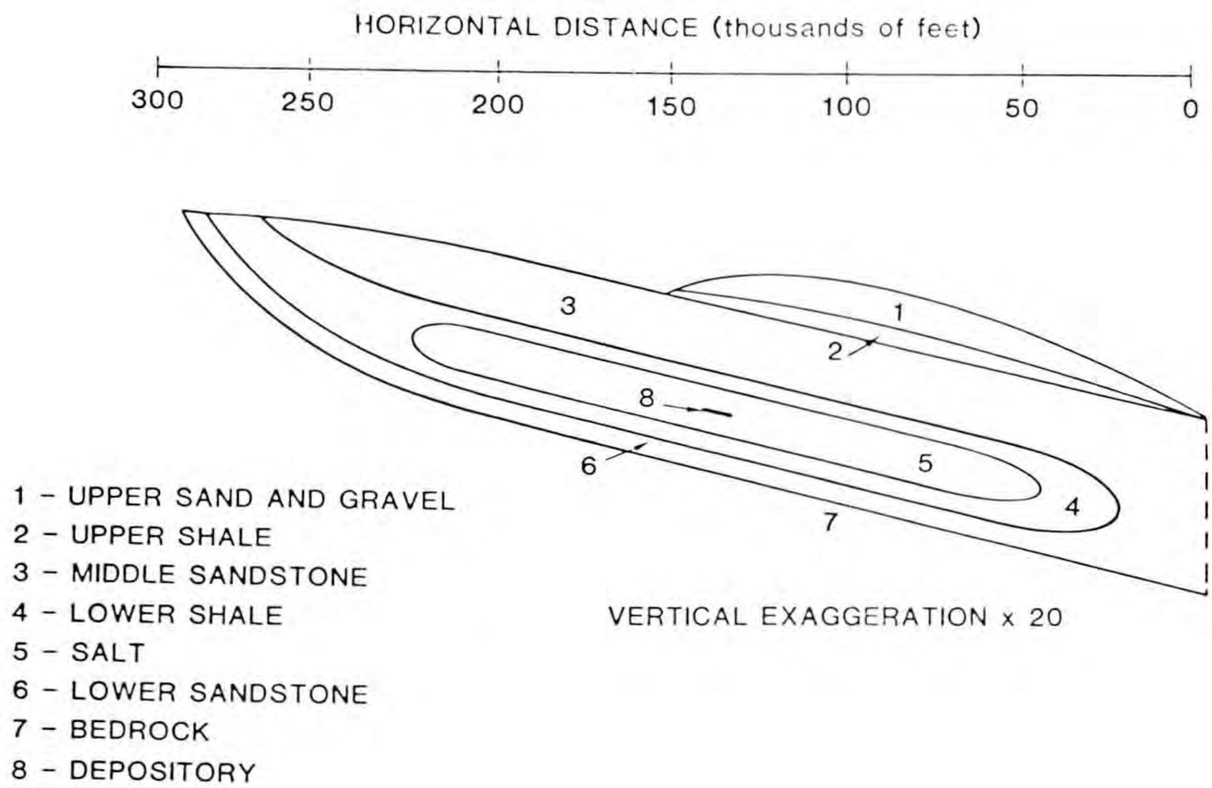
HORIZONTAL DISTANCE (thousands of feet)
300
250
200
150
100
50
0
1
2
3
4
5
6
7
8
1 - UPPER SAND AND GRAVEL
2 - UPPER SHALE
3 - MIDDLE SANDSTONE
4 - LOWER SHALE
5 - SALT
6 - LOWER SANDSTONE
7 - BEDROCK
8 - DEPOSITORY
VERTICAL EXAGGERATION x 20

FIGURE 3

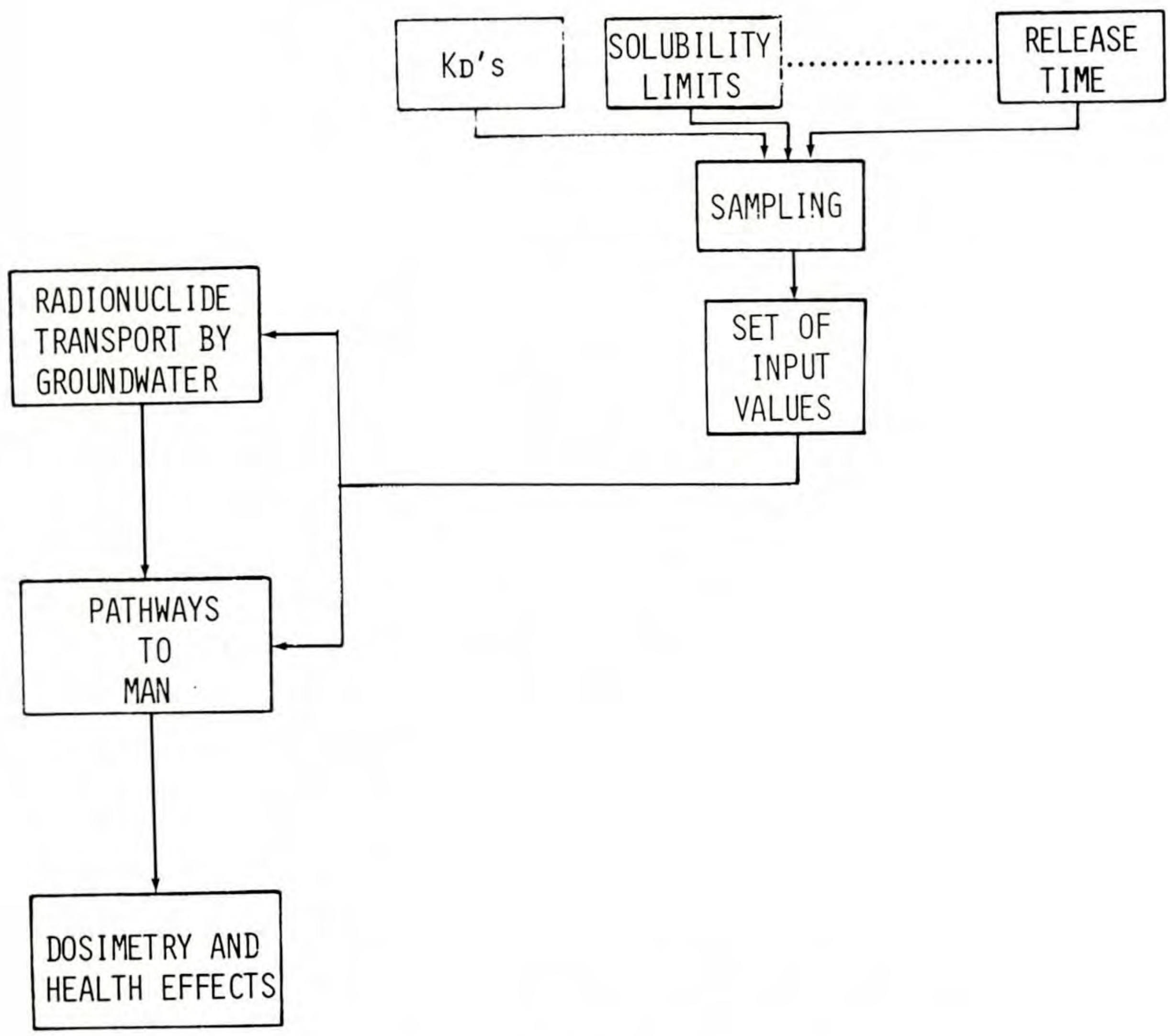
KD's
SOLUBILITY LIMITS
RELEASE TIME
SAMPLING
SET OF INPUT VALUES
RADIONUCLIDE TRANSPORT BY GROUNDWATER
PATHWAYS TO MAN
DOSIMETRY AND HEALTH EFFECTS

FIGURE 4

DNET: A MODEL FOR INCORPORATING FEEDBACK EFFECTS IN SALT DISSOLUTION PROCESSES

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ABSTRACT

For nuclear waste isolation in deep, geologic formations, transport in groundwater appears to be one of the more likely means for radioactive waste to migrate from the depository to the biosphere. With respect to a depository in bedded salt, transport in groundwater would, for most breachment scenarios, have to be preceded by dissolution of all or portions of the salt layers surrounding the depository. The Dynamic Network (DNET) model provides a capability for investigating the rate of salt dissolution associated with a variety of disruptive events and processes, and also provides a capability for investigating the effects of feedback mechanisms such as thermal expansion, subsidence, fracture formation and salt creep.

INTRODUCTION

A methodology for assessing the risk from geologic disposal of radioactive waste is being developed by the Fuel Cycle Risk Analysis Division at Sandia National Laboratories [1]. This methodology is to be demonstrated by application to a reference site -- a hypothetical waste repository in bedded salt. As part of this methodology, the Dynamic Network (DNET) model was developed to investigate processes near the depository such as salt dissolution and salt creep that could affect the release of radioactive waste to circulating groundwater. The DNET model also provides a systematic means for investigating the effects of feedback mechanisms such as thermal expansion, subsidence, fracture formation and fracture closure. These mechanisms can act to accelerate or decelerate the salt dissolution process and thus increase or decrease the potential for release of radioactive waste.

STRUCTURE OF DNET

DNET uses a network flow model similar to that used in the Network Flow and Transport (NWFT) model [2]. NWFT was developed to simulate far-field transport of contaminants dissolved in groundwater. Therefore, the flow system hydraulic properties in NWFT are assumed static. DNET, on the other hand, was developed for investigation of feedback mechanisms in the near vicinity of the depository. Thus in DNET, the hydraulic properties of the system are allowed to vary with time. DNET simulates several physical processes icluding the following: (1) fluid flow, (2) salt dissolution, (3) thermal expansion, (4) fracture formation and closure, and (5) salt creep. Because of the complexity involved in treating the several processes in DNET, the governing equations cannot be solved in an implicitly coupled fashion (i.e., simultaneously). Thus, the submodels which represent the various processes treated in DNET are applied sequentially.

Computational Sequence

The computational sequence in DNET is indicated in Figure 1. Computation begins at time T_0 after depository closure. The initial conditions input in DNET are assumed to be the conditions of the system at time T_0. However, for purposes of the thermal calculations, the radioactive waste heat source is decayed from depository closure at time $T = 0$ to time T_0 at which the analysis of DNET is begun. Fluid properties are functions of temperature and brine concentration. For the first time step, the brine concentration is input as an initial condition. In subsequent time steps, the brine concentration is calculated. Once fluid density and viscosity are determined (all other hydraulic properties are initialized in the input), fluid flow can be calculated. The salt solution model calculates salt removal by dissolution from appropriate portions of the flow system and determines brine concentrations throughout the system. Brine concentrations will be used in the following time step to determine fluid density and viscosity. System hydraulic properties are altered based on salt removal as well as several other processes as indicated in Figure 1. Once output information is printed, the time is incremented by Δt and DNET loops back as indicated for the next time step. The sequential application of the various submodels in DNET implies the assumption that the system is static over the time interval Δt.

The Network Flow Model

The construction of the flow network used in DNET is loosely based on a hypothetical flow system which serves as a reference site for the risk methodology program. The reference site is discussed in detail in Campbell, et al. [1]. Groundwater flow calculations have been performed for the reference site using the Sandia Waste Isolation Flow and Transport (SWIFT) model [3]. These calculations have shown that flow is essentially one-dimensional in the middle and lower sandstone aquifer. The SWIFT simulation of the reference site is shown in Figure 2.

The recommended geometry for the DNET network is represented by the darker lines in Figure 2. Figure 3 shows the leg and junction numbering system used for this network. The junction numbers are circled and the arrows represent the direction of positive flow. As DNET was developed to simulate salt dissolution and feedback effects near the depository, a smaller network representation could have been developed. However, DNET requires constant pressure boundary conditions at the aquifer inlets (Junctions 1 and 2) and at the discharge point to River L (Junction 3). These boundary conditions are valid if the aquifer inlet and discharge points are sufficiently far removed from the simulated disruption near the depository. The flow network in Figure 3, by representing the full (or nearly so) reference site flow system, assures that the disruptions near the depository have small effect on the boundary pressures.

Legs 1, 2 and 3 of the network are placed at the middle shale/sandstone interface and are used to represent the middle sandstone aquifer. Similarly, Legs 4, 5 and 6 are placed at the lower shale/sandstone interface and are used to represent the lower sandstone aquifer. Legs 15 and 17 are shown at the salt/middle shale and salt/lower shale interfaces, respectively, in Figure 2. However, these legs have the flexibility of being placed at any desired location between Legs 2 and 5. Similarly, Leg 16, shown at the depository level in Figure 2, can be placed at any location between Legs 15 and 17. Legs 7, 9, 11 and 13, as well as Legs 8, 10, 12 and 14, represent vertical legs through the salt and shale and must maintain a total length of 1100 ft. These legs are used to represent various disruptive features which affect the salt and shale layers near the depository. Leg 18 represents discharge from the lower sandstone aquifer to River L.

Flow Calculations

The following properties are assumed known for the flow calculations:

1. P_1, P_2, P_3	Pressure boundary conditions for aquifer inlet and discharge points (Junction 1, 2, and 3)
2. k_1, k_2, ..., k_{18}	Permeability for Legs 1 to 18
3. A_1, A_2, ..., A_{18}	Cross-sectional area for Legs 1 to 18
4. L_1, L_2, ..., L_{18}	Lengths of Legs 1 to 18
5. D_1, D_2, ..., D_{14}	Elevations above datum for Junctions 1 to 14
6. ρ_1, ρ_2, ..., ρ_{18}	Average fluid density for Legs 1 to 18
7. ϕ_1, ϕ_2, ..., ϕ_{18}	Porosity for Legs 1 to 18

Once input has been read, initial temperatures, fluid viscosities and fluid densities are calculated for each leg of the network. With these quantities determined, fluid discharge and interstitial velocities are calculated for each leg. Fluid discharge in Legs 1 through 18 is given by the following equations:

$$q_1 = \theta_1 [P_1 - P_4 + \rho_1 (D_1 - D_4)] \tag{1}$$

$$q_2 = \theta_2 [P_4 - P_5 + \rho_2 (D_4 - D_5)] \tag{2}$$

$$q_3 = \theta_3 [P_5 - P_3 + \rho_3 (D_5 - D_3] \tag{3}$$

$$q_4 = \theta_4 [P_2 - P_{12} + \rho_4 (D_2 - D_{12})] \tag{4}$$

$$q_5 = \theta_5 [P_{12} - P_{13} + \rho_5 (D_{12} - D_{13})] \tag{5}$$

$$q_6 = \theta_6 [P_{13} - P_{14} + \rho_6 (D_{13} - D_{14})] \tag{6}$$

$$q_7 = \theta_7 [P_6 - P_4 + \rho_7 (D_6 - D_4)] \tag{7}$$

$$q_8 = \theta_8 [P_7 - P_5 + \rho_8 (D_7 - D_5)] \tag{8}$$

$$q_9 = \theta_9 [P_8 - P_6 + \rho_9 (D_8 - D_6)] \tag{9}$$

$$q_{10} = \theta_{10} [P_9 - P_7 + \rho_{10} (D_9 - D_7)] \tag{10}$$

$$q_{11} = \theta_{11} [P_{10} - P_8 + \rho_{11} (D_{10} - D_8)] \tag{11}$$

$$q_{12} = \theta_{12} [P_{11} - P_9 + \rho_{12} (D_{11} - D_9)] \tag{12}$$

$$q_{13} = \theta_{13} [P_{12} - P_{10} + \rho_{13} (D_{12} - D_{10})] \tag{13}$$

$$q_{14} = \theta_{14} [P_{13} - P_{11} + \rho_{14} (D_{13} - D_{11})] \tag{14}$$

$$q_{15} = \theta_{15} [P_6 - P_7 + \rho_{15} (D_6 - D_7)] \tag{15}$$

$$q_{16} = \theta_{16} [P_8 - P_9 + \rho_{16} (D_8 - D_9)] \tag{16}$$

$$q_{17} = \theta_{17} [P_{10} - P_{11} + \rho_{17} (D_{10} - D_{11})] \tag{17}$$

$$q_{18} = \theta_{18} [P_{14} - P_3 + \rho_{18} (D_{14} - D_3)] \tag{18}$$

where

$$\theta_i = \frac{k_i A_i}{\mu_i L_i}, \qquad \mu_i = \text{fluid viscosity of Leg i.}$$

The following conservation equations are applied at the leg junctions:

$q_1 + q_7 = q_2$	Junction 4	(19)
$q_2 + q_8 = q_3$	Junction 5	(20)
$q_9 = q_7 + q_{15}$	Junction 6	(21)
$q_{10} + q_{15} = q_8$	Junction 7	(22)
$q_{11} = q_9 + q_{16}$	Junction 8	(23)
$q_{16} + q_{12} = q_{10}$	Junction 9	(24)
$q_{13} = q_{11} + q_{17}$	Junction 10	(25)
$q_{17} + q_{14} = q_{12}$	Junction 11	(26)
$q_4 = q_{13} + q_5$	Junction 12	(27)
$q_5 = q_{14} + q_6$	Junction 13	(28)
$q_6 = q_{18}$	Junction 14	(29)

Equations 19 through 29 are solved simultaneously to determine the unknown pressures P_4 through P_{14}. Fluid discharge by leg is calculated using Equations 1 through 18. Interstitial velocities, v_i, are calculated using the equation

$$v_i = \frac{q_i}{A_i \phi_i} , \quad i = 1, 18 \tag{30}$$

Pore volume changes and brine concentrations due to salt dissolution are then determined for appropriate portions of the flow system. Hydraulic properties are then altered based on salt removal as well as other processes such as thermal expansion, subsidence and salt creep.

Additional information on the DNET model can be found in either [4] or [5].

REFERENCES

1. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P. E. McGrath, F. J. Pearson, H. R. Shaw, J. C. Helton and F. A. Donath, 1978, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, Report No. SAND78-0029, Sandia Laboratories, Albuquerque, NM.

2. J. E. Campbell, P. C. Kaestner, B. S. Langkopf and R. B. Lantz, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: The Network Flow and Transport (NWFT) Model, Report No. SAND79-1920, Sandia National Laboratories, Albuquerque, NM.

3. R. T. Dillon, R. B. Lantz and S. B. Pahwa, 1978, Risk Methodology for Geologic Disposal of Radioactive Waste: The Sandia Waste Isolation Flow and Transport (SWIFT) Model, Report No. SAND78-1267, Sandia Laboratories, Albuquerque, NM.

4. J. E. Campbell and R. M. Cranwell, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: DNET: A Model for Incorporating Feedback Effects in Salt Dissolution Processes, Report No. SAND80-0067, Sandia National Laboratories, Albuquerque, NM.

5. R. M. Cranwell, J. E. Campbell and S. E. Stuckwisch, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: The DNET Computer Code User's Manual, Report No. SAND81-1663, Sandia National Laboratories, Albuquerque, NM.

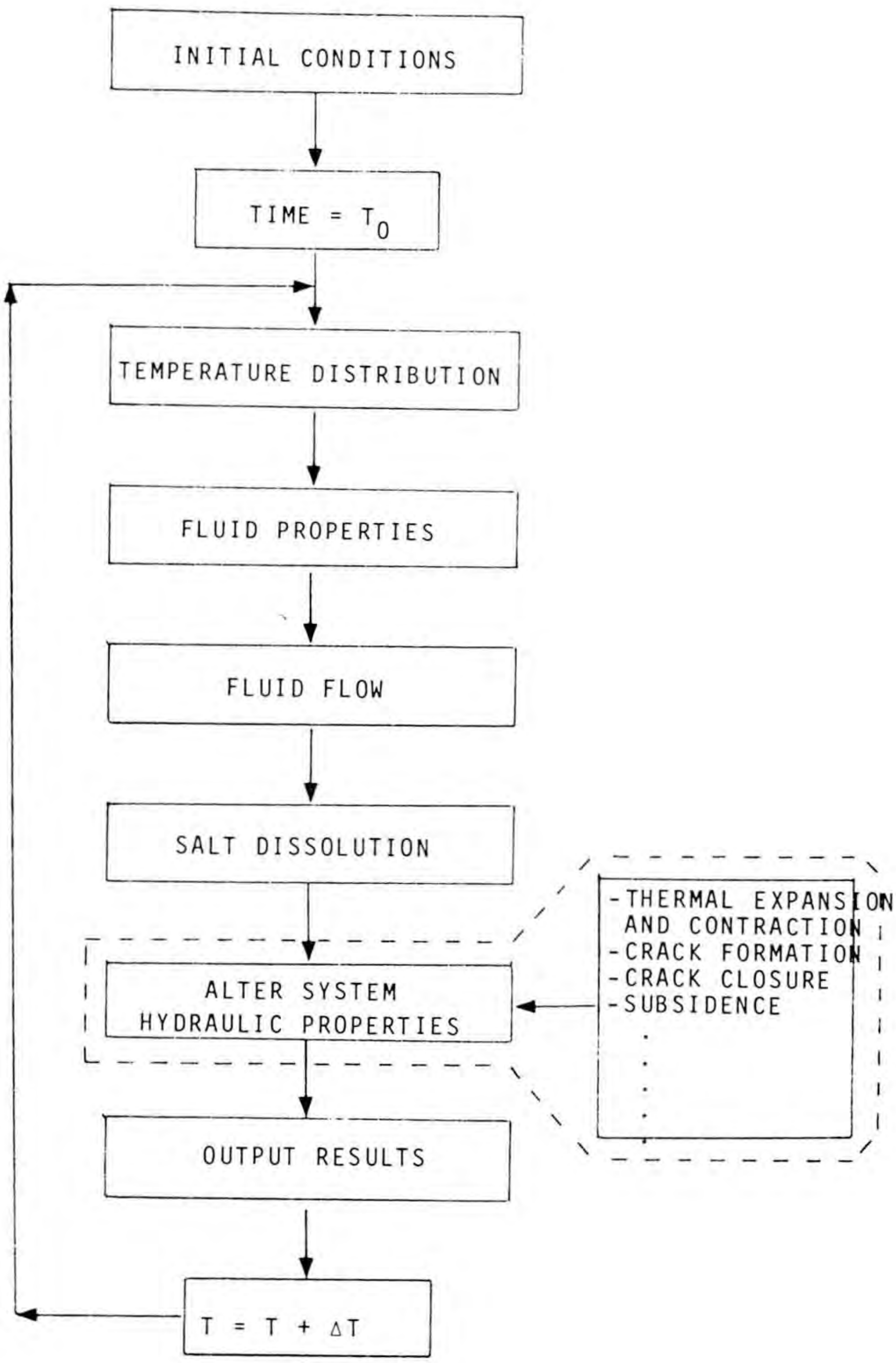

FIGURE 1. COMPUTATIONAL SEQUENCE IN DNET

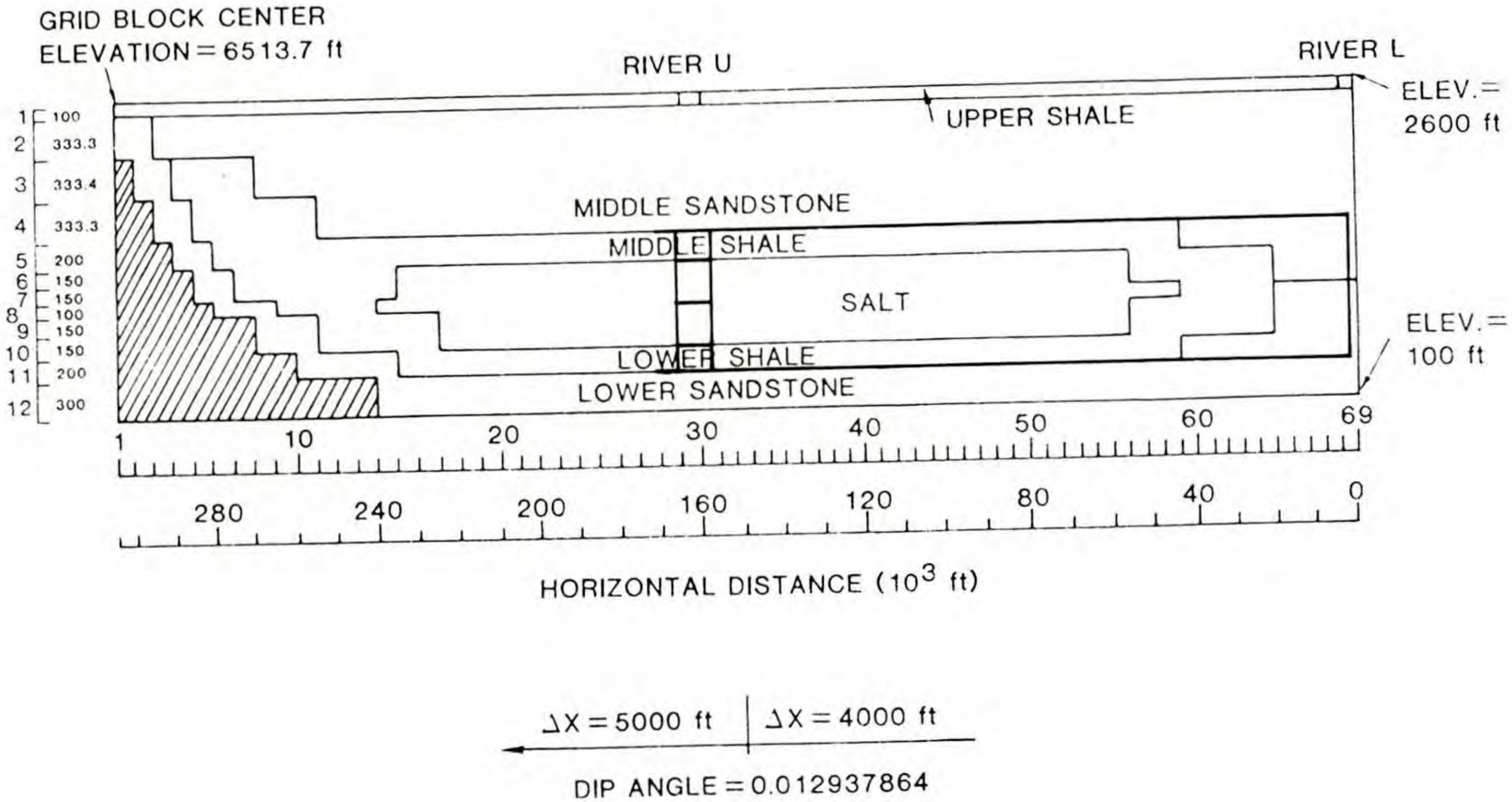

FIGURE 2. SWIFT SIMULATION OF THE REFERENCE SITE

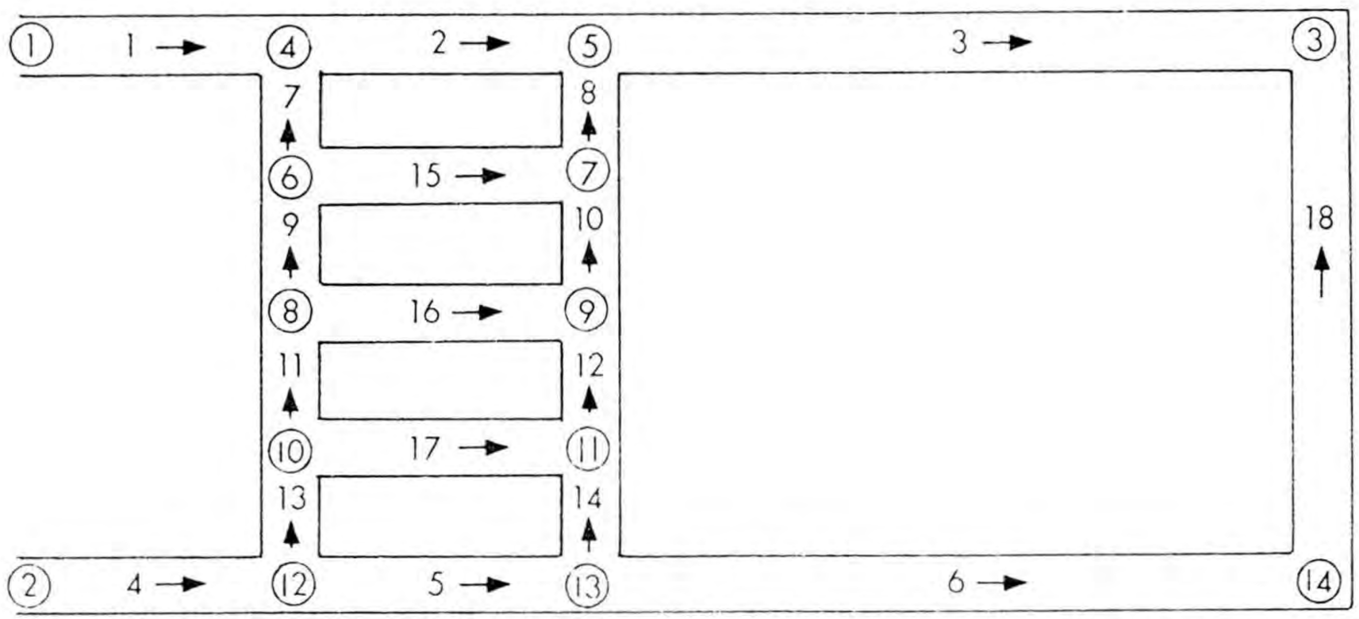

FIGURE 3. NETWORK FLOW REPRESENTATION USED IN DNET.
POSITIVE FLOW DIRECTIONS ARE INDICATED BY ARROWS.
JUNCTION NUMBERS ARE CIRCLED.

THE DISTRIBUTED VELOCITY METHOD OF SOLVING THE CONVECTIVE-DISPERSION EQUATION

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ABSTRACT

The motivation for developing the Distributed Velocity Method arises from the demands of performing a risk assessment for a nuclear waste repository. These demands include computational efficiency over a relatively large range of Peclet numbers, the ability to handle chains of decaying radionuclides with rather extreme contrasts in both solution velocities and half lives, and the ability to treat leach- or solubility-limited sources. To the extent it has been tested to date, the Distributed Velocity Method (DVM) appears to satisfy these demands.

INTRODUCTION

The motivation for this work arises in the context of a risk analysis methodology for nuclear waste repositories [1]. In such analysis, results are calculated using mathematical models which describe a number of processes. One of these processes is radionuclide migration in groundwater from the depository to a discharge point to the surface environment. Risk analysis necessarily involves large numbers of calculations. Furthermore, radionuclide migration times from the depository to the surface environment are typically long so that radionuclides in the actinide chains are likely to be significant contributors to risk. Thus a radionuclide transport model for use in risk assessment must be computationally efficient and must provide the ability to transport chains of radionuclides. To the extent it has been tested to date, the Distributed Velocity Method (DVM) appears to satisfy the demands of a risk assessment methodology.

DVM directly simulates the migration of representative particles of the trace constituent. However, tracking of individual particles is avoided by treating ensembles of particles. In the numerical implementation of DVM, the spatial extent of an ensemble of particles is taken to

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be one grid block. This grid block averaging introduces some numerical dispersion which is discussed later in this paper.

In this initial investigation of DVM we use a one-dimensional, constant velocity system which is the usual procedure for establishing error criteria. Furthermore, the resulting model is directly applicable in risk assessment for nuclear waste repositories [2]. To qualify for more general site analysis, however, the method must be extended to variable-velocity and multi-dimentional systems. Such extensions will be examined in future work.

MATHEMATICAL THEORY

To illustrate basic concepts, a single species is considered to be transported in one dimension via the mechanisms of convection and dispersion. Radioactive chains and sorption are not difficult to treat, but their inclusion tends to obscure the simplicity of the Distributed Velocity Method (DVM). Although decay chains and sorption are not considered in this section, they are included in the numerical implementation of DVM.

Direct Simulation with DVM

The thinking underlying DVM is as follows: Consider a receiver point located at x and donor points located at some typical coordinate x'. Taking the density of an ensemble of particles at time t' to be $\rho(x',t')$, the density $\rho(x,t)$ at x for t > t' may be determined by introducing a velocity distribution.

The concept here is that, due to heterogeneity of the flow field, [3] a number of alternate paths exist for migration of particles from x' to x. Such paths may be characterized by a continuum of migration times and average velocity components v in the direction of flow. The distribution of such velocities is P(v). Thus for the donor point x', only those particles with average velocity $v = (x-x')/(t-t')$ arrive at point x at time t. The density of particles at point x may therefore be obtained by summing over all possible donor points in the following manner:

$$\rho(x,t) = \int_{-\infty}^{\infty} dv\ P(v)\ \rho(x-v\Delta t\ ,\ t-\Delta t\ ) + \int_{t'}^{t} d\tau \int_{v_1}^{v_0} dv\ P(v)S(x',\tau) \qquad (1)$$

where $\Delta t = t - t'$

For convenience, P(v) is represented as a function of velocity only. However, it is certainly possible for P to be a function of other variables such as position x or time increment Δt. Otherwise, the functional form of the distribution is completely general at this point. In the next section, P(v) is specialized to a Gaussian form which is appropriate for a conventional treatment of dispersion.

The first term in Eq. (1) gives the propagation of initial conditions, at time t', to time t. The second term represents an integration over source or "injection" time where $x' = x - v(t-\tau)$. If the source function $S(x',\tau)$ is nonzero only for $x_0 \leq x' \leq x_1$

then the velocity limits for the second term of Eq. (1) are

$$V_0 = (X-X_0)/(t-\tau) \quad \text{and} \quad V_1 = (X-X_1)/(t-\tau).$$

Connection with Green's Function

The inhomogeneous convective dispersion equation in one dimension can be written as

$$\frac{\partial \rho}{\partial t} = D\frac{\partial^2 \rho}{\partial X^2} - \bar{V}\frac{\partial \rho}{\partial X} + S$$
$$D = \alpha \bar{V} \tag{2}$$

where D is the dispersion coefficient and α is the dispersivity. For null conditions on ρ at the infinite boundaries, the complete solution to Eq. (2) contains two terms

$$\rho(x,t) = \int_{-\infty}^{\infty} dx' \, \rho(x',t')G(x-x', t-t') + \int_{t'}^{t} d\tau \int_{X_0}^{X_1} dv \; S(x',\tau)G(x-x', t-\tau) \tag{3}$$

The Green's function of Eq. (3) can be written as

$$G(x-x', t-t') = \frac{1}{\sqrt{2\pi}\,\sigma_X} \exp\left\{-\frac{[(x-x') - v(t-t')]^2}{2\sigma_X^2}\right\} \tag{4}$$

where the variance is

$$\sigma_X^2 = 2D(t-t') \tag{5}$$

Comparing Eq. (1) with Eqs. (3) and (4), it is straightforward to identify the velocity distribution P(v) as

$$P(v) = \frac{1}{\sqrt{2\pi}\,\sigma_V} \exp\left\{-\frac{(v-\bar{v})^2}{2\sigma_V^2}\right\} \tag{6}$$

where

$$\sigma_V = \sigma_X/(t-t')$$

In making this identification, we have specialized DVM to the conventional Fickian treatment of dispersion.

NUMERICAL IMPLEMENTATION

This section briefly describes the numerical implementation of DVM. First consider the propagation of the density function $\rho(x',t')$ for a single radioactive species from time t' to time t. Initially, the space-velocity domain is gridded as in Figure 1. There are N_X equal space increments Δx and the time increment Δt is taken to be a constant. The velocity dimension is divided into N_V increments based on equal probability. The implementation of DVM can be generalized to variable spatial and time increments but such generalization is not considered in this report.

Propagation of densities over time-step Δt for velocity subgroup j may be written

$$\Delta\rho(i,j,t) = \mathcal{DW}(j)\left\{ M(j)\,\rho(t-k_j,t') + [1-M(j)]\,\rho(i-k_j-1,t')\right\} \tag{7}$$

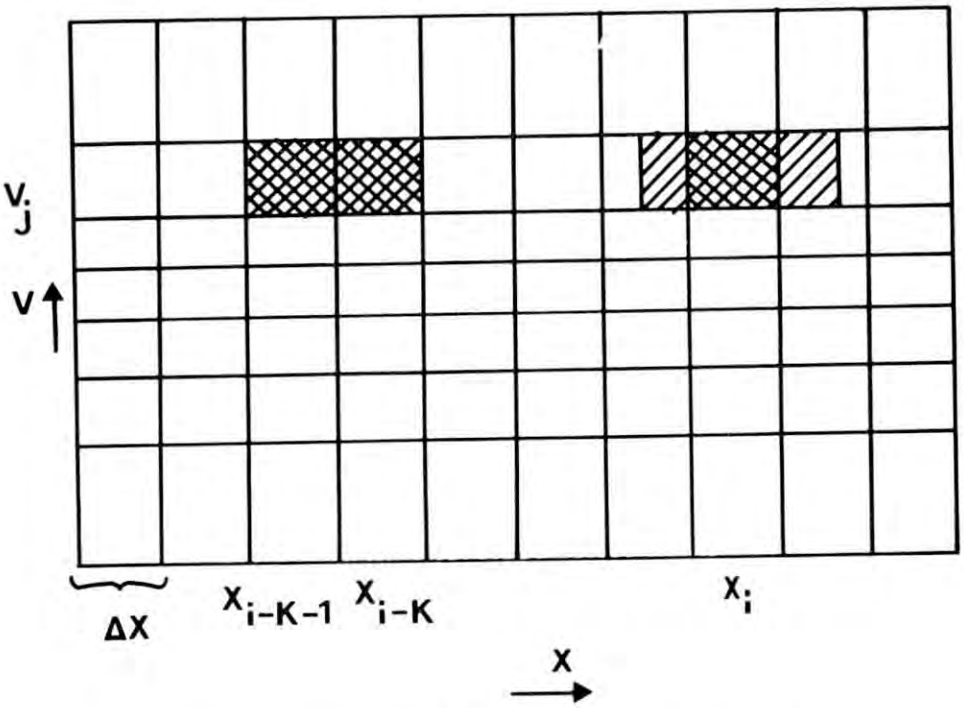

Fig. 1. Gridded Velocity-Space Domain.

Although the argument list in Eq. (7) appears formidable, it can be readily understood. As indicated there, the contribution to receiver block i for velocity interval j is determined by three fractions: a mixing fraction M, a velocity-interval fraction W, and a decay fraction D.

The mixing fraction may be understood by reference to Figure 1. Looking at velocity interval j, we see that there are two contributions to receiver block i. One is a packet of particles coming from donor block i-k and the other is a packet of particles from donor block i-k-1. As is also indicated in Figure 1, there is generally only partial overlap of the propagated block contents with receiver block i. The donor block index k_j is

$$k_j = [\![v_j \Delta t/\Delta x]\!]$$

where

$$[\![z]\!] = \text{greatest integer} \leq z$$

and v_j is the velocity assigned to the j^{th} velocity interval. The corresponding mixing fraction is

$$M(j) = 1 - (v_j \;\; t/\Delta x - k_j)$$

As the velocity dimension is divided into intervals based on equal probability, the weight $W(j)$ assigned the j^{th} velocity interval is just

$$W(j) = 1/N_v \tag{8}$$

The decay fraction is taken to be

$$D = e^{-\lambda \Delta t} \tag{9}$$

With these three fractions defined, Eq. (7) is summed over all velocity intervals to obtain the total particle density in grid block i:

$$\rho(i,t) = \frac{\mathcal{D}}{N_v} \sum_{j=1}^{N_v} \left\{ M(j)\,\rho(i-k_j,t') + [1 - M(j)]\,\rho(i-k_j-1,t') \right\} \tag{10}$$

As a last step in this analysis, we express Eq. (10) in a more computationally efficient manner. To do this, we note the possibility of degeneracy with respect to the index k_j. This could mean either

$$k_{j+1} = k_j \quad \text{or} \quad k_{j+1} = k_j+1 \tag{11}$$

Taking such degeneracies into account yields the expression

$$\rho(i,t) = \sum_{j=1}^{N_B(i)} B(j)\,\rho(i-k_j,t) \tag{12}$$

Quantities $N_B(i)$ and $B(j)$ are most easily obtained by a computational procedure which makes the tests of Eq. (11) and accumulates the coefficients to form the B matrix.

For a chain of radionuclides, analysis similar to the above discussion for a single radionuclide leads to the following expression

$$\rho(i,r,t) = \sum_{p=0}^{N_p(r)} \sum_{j=1}^{N_B(i,r,r-p)} B(j,r,r-p)\,\rho(i-k_j,r-p,t) \tag{13}$$

In Eq. (13), $\rho(i-k_j, r-p, t')$ represents the density of species r-p in grid block $i-k_j$ at time t'. The quantity B(j,r,r-p) is the fraction of species r-p which decays to species r and ends up in grid block i at time $t = t' + \Delta t$. The product of these two quantities is then summed over all contributing grid blocks and all parent species.

In addition to the transport model, represented by Eq. (13), source and discharge models have also been developed. The source model is capable of treating both leach- and solubility-limited sources. For a single time step, the computational sequence involves injection of source material into designated source grid blocks, transport of dissolved material through the system and calculation of the quantity of dissolved material discharged during the time step.

ERROR ANALYSIS

The usual method for making error analyses is to select a simple one-dimensional model problem for analysis [4]. The expectation is that the numerical errors present in more complex implementations will manifest themselves in a quantitatively similar manner to the simplified problem. We also used this strategy. However, rather than a theoretical error analysis, ours was a numerical analysis. Thus we had to choose numerical values for our physical problem. We chose a one-dimensional flow system with the length and velocity parameters

$$L = 100{,}000 \text{ ft} \quad \text{and} \quad v = 1 \text{ ft/yr}$$

Within this flow system, we considered transport of a stable, non-retarded contaminant subject to the boundary conditions

$$\rho(x=0,t) = 1 \quad \text{and} \quad \rho(x \geq 0, t=0) = 0$$

To examine the numerical dispersion introduced by DVM, we allowed the space step, time step, number of velocity intervals and dispersivity to vary randomly over the following ranges: (1) $\Delta x(100,4000)$ ft; (2) $\Delta t(100,4000)$ yr; and (3) $\alpha(1,500)$ ft. A series of calculations was performed and the dependence of numerical dispersion on Peclet number ($P = \Delta x/\alpha$) and Courant number ($C = v\,\Delta t/\Delta x$) was examined. Results of the analysis indicated that numerical dispersion introduced by DVM increases with increasing Peclet number and decreases with increasing Courant number. For comparison, consider that a standard finite-difference scheme, which is centered in space and time, requires that

$$P \leq 2 \quad \text{and} \quad C \cdot \left(1 + \frac{2}{P}\right) \leq 2 \tag{14}$$

to prevent overshoot [5]. Similar criteria hold for centered-in-time finite element implementations [6]. With DVM, the direct dependence of numerical dispersion on Peclet number and thus on grid-block size is a rather conventional relation, compared to other methods. However, the inverse dependence of numerical dispersion on Courant number and thus on time increment is quite irregular. The implication is that, consistent with the desired resolution of the breakthrough curve and the shortest radionuclide half-life, the time increment should be made as large as possible. This has the dual benefit of decreasing numerical dispersion and decreasing computer running time.

The results in Figure 2 were selected from the series of calculations

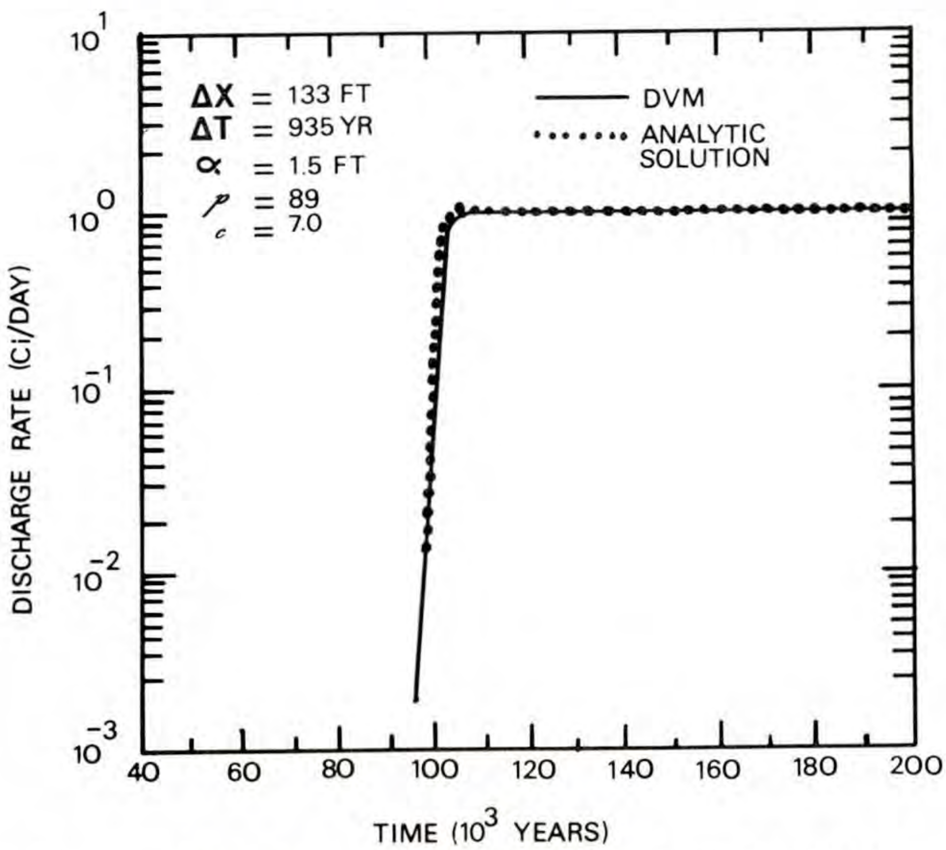

Fig. 2. Comparison Between DVM and an Analytic Solution.

performed for the error analysis. In Figure 2, despite the small dispersivity (α = 1.5 ft.) and the large Peclet number (P = 89), the comparison between DVM and the analytic solution is excellent. The reason for this good comparison is the relatively large Courant number (C = 7).

EXAMPLE RESULTS

The present implementation of DVM is able to treat radioactive decay chains of any length and leach- or solubility-limited sources. However,

Table 1. Problem Parameters for Comparison of DVM and Analytic Solution

ISOTOPE	HALF LIFE	RETARDATION FACTOR
A	10^6	1000
B	10^3	1
C	10^7	10

Path Length = 1000 ft.
Dispersivity = 100 ft.
Fluid Velocity = 10 ft./yr.

Leach Time = 10^6 yr.
Δx = 200 ft.
Δt = 4000 yr.

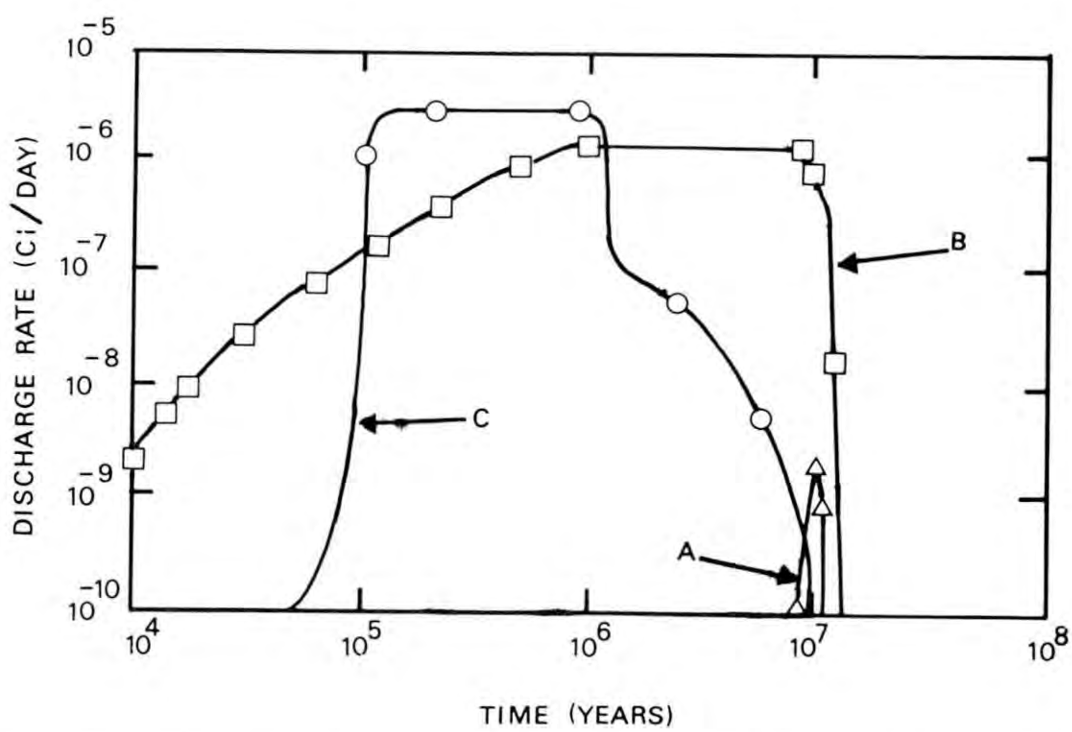

Fig. 3. Comparison Between DVM and an Analytic Solution for a Three-Member Chain.

presently available analytical solutions can only treat leach-limited sources and chains of three nuclides or less. Therefore, a comparison between DVM and an analytic solution is presented for a hypothetical, three member chain (A→B→C). Problem parameters for the comparison are shown in Table 1. Results of the comparison are shown in Figure 3. The comparison between DVM and the analytic solution is excellent.

SUMMARY

A new method has been developed to solve the convective-dispersion equation. The Distributed Velocity Method (DVM) has been implemented in one dimension to provide a basis for error analysis. Results of the error analysis indicate that, with DVM as with other numerical techniques, numerical dispersion increases with increasing Peclet number. However, the inverse dependence of numerical dispersion on Courant number is quite irregular. The implication is that, consistant with other time step requirements, the time step should be made as large as possible. This has the dual benefit of decreasing numerical dispersion and decreasing computer running time. Furthermore, because of its ability to treat decay chains of any length, leach- or solubility-limited sources, and large Peclet numbers, the one dimensional implementation of DVM is directly applicable in risk analysis of radioactive waste repositories.

REFERENCES

1. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P.E. McGrath, F. J. Pearson, H. R. Shaw, J. C. Helton, and F. A. Donath, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, NUREG/CR-0458 (SAND78-0029) (October, 1978).

2. J. E. Campbell, D. E. Longsine, and M. Reeves, Risk Methodology for Geological Disposal of Radioactive Waste: The Distributed Velocity Method of Solving the Convective-Dispersion Equation, NUREG/CR-1376 (SAND80-0717) (July, 1980).

3. F. W. Schwartz, "Macroscopic Dispersion in Porous Media: The Controlling Factors", Water Resources Research, Vol. 13, No. 4, pp. 743-752 (1977).

4. R. B. Lantz, "Quantitative Evaluation of Numerical Diffusion (Truncation Error)," Soc. Pet. Eng. Journal, (1971).

5. H. S. Price, R. S. Varga, and J. E. Warren, "Application of Oscillation Matricies to Diffusion Convection Equations," J. of Math. and Physics, Vol. 45, pp. 301-311, (1966).

6. O. K. Jensen and B. A. Finlayson, "Solution of the Convection-Diffusion Equation Using a Moving Coordinate System," Finite Elements and Water Resources II, C. A. Brabbia (Editor), pp. 4.21-4.32, (1978).

ASYMPTOTIC BEHAVIOR OF A MODEL FOR THE ENVIRONMENTAL MOVEMENT OF RADIONUCLIDES

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ABSTRACT

The Environmental Transport Model is used to represent the surface movement of radionuclides. This presentation provides an indication of the results obtained in a study of the asymptotic behavior of this model. The nature of the Environmental Transport Model is indicated and its asymptotic behavior is discussed. Then, an approach to sensitivity analysis of this behavior is outlined and the partial results of such an analysis are presented.

ENVIRONMENTAL TRANSPORT MODEL

In the Environmental Transport Model, radionuclide movement is represented with a compartment or mixed-cell model. With this approach, radionuclides in an area under consideration are divided into a number of "compartments" and then radionuclide movement between these compartments is represented by a system of linear differential equations. The basic idea is to place radionuclides which are in physical regions with different characteristics in different compartments and then to determine the distribution which results from radionuclide decay and from radionuclide movement between regions.

The Environmental Transport Model uses an algorithm to construct the radionuclide transport equations indicated in the preceding paragraph. The basis of this algorithm is a building block called a zone. A zone corresponds to a physical region in an area to be modeled. Further, each zone has four subzones (groundwater, soil, surface water and sediment) and various movements of water and solid material associated with these subzones are possible. It is the movements of water and solid material which are assumed to be responsible for radionuclide transport. In such movements, it is assumed that radionuclides are partitioned between the liquid and solid phases on the basis of distribution coefficients. A system of transport equations incorporating decay chain characteristics and regional diversity is constructed by linking a suitable number of these zones together. In this construction, each subzone has one uniformly mixed compartment associated with it for each radionuclide in the decay chain under consideration. A single zone is represented in Figure 1 and the linkage of a sequence of zones to represent the movement of a decay chain is represented in Figure 2.

The Environmental Transport Model was constructed as part of a project at Sandia National Laboratories to develop a methodology to assess the risk associated with the geologic disposal of radioactive waste. A more detailed description of the model is provided in Chapter 4 of the project's interim report [1]. Further, a user's manual which describes the model and the computer program which implements it is also available [2].

ASYMPTOTIC BEHAVIOR

The asymptotic behavior of the Environmental Transport Model is now considered. Mathematically, this model is a system of differential equations of the form

$$dq_i/dt = h_i + \sum_{\substack{j=1 \\ j\neq i}}^{n} k_{ij}\, q_j - \Big(k_{0i} + \sum_{\substack{j=1 \\ j\neq i}}^{n} k_{ji}\Big)\, q_i\ , \qquad (1)$$

$i = 1, \ldots, n$. When M zones and N radionuclides are considered, $n = 4\ MN$. In the preceding equations, h_i corresponds to a rate of radionuclide input from outside the system being modeled, k_{0i} corresponds to a radionuclide movement out of the system being modeled and k_{ij} corresponds to a radionuclide movement between two compartments within the system. The radionuclide movements referred to in the previous sentence could result from a change of location due to physical movement or a change of state due to radioactive decay. The system of equations in (1) can be expressed in matrix notation as

$$dq/dt = h + Kq, \qquad (2)$$

where q and h are column vectors of the q_i and the h_i, respectively, and where K is the matrix whose off-diagonal elements are the k_{ij} and whose diagonal elements are given by

$$k_{ii} = -\Big(k_{0i} + \sum_{\substack{j=1 \\ j\neq i}}^{n} k_{ji}\Big). \qquad (3)$$

In the following, the elements of h and K are always assumed to be constants.

For the system in (1) and the equivalent matrix formulation in (2), the expression "asymptotic behavior" is used in reference to the performance of $q(t)$ as $t \rightarrow \infty$. For such systems, it is possible to obtain various characterizations of asymptotic behavior. The paper by Thron [3] provides a good discussion of such behavior. The result which is most useful in characterizing asymptotic behavior for the present study will be stated. However, several definitions are needed first. A compartment system is said to be open if material can move out of the system. Conversely, a system is said to be closed if it is not open; that is, a system is closed if material cannot move out of it. Finally, a system is said to be completely open if it is open and contains no closed subsystem. For example, if all the arrows in Figure 1 represent nonzero flows, then the corresponding system is completely open. The desired result is now stated; a proof can be obtained in Thron [3].

Theorem. For any completely open compartment system satisfying (2), K^{-1} exists and a unique constant-valued solution is given by $q = -K^{-1}h$. Further, (i) if q is any solution to (2), then $\lim_{t \rightarrow \infty} q(t) = -K^{-1}h$ and (ii) if $h_i \geq 0$ for $i = 1, \ldots, n$ and $q(0) = 0$, then each component of $q(t)$ increases monotonically to the corresponding component of $-K^{-1}h$.

APPROACH TO SENSITIVITY ANALYSIS

For this analysis, we were interested in the following questions: How does variation in a site's properties affect variation in the asymptotic behavior of the site? Is it possible to determine the relative importance of individual variables in affecting the asymptotic behavior of the site? The general approach taken was to define a hypothetical site and to postulate a set of conditions at that site. Some of the conditions were ill-defined in the sense that they were not given fixed values. Instead, they were assumed to be described by variables with specified ranges and distributions.

After the site was suitably described, it was desired to determine the amount of variation in its asymptotic behavior and to determine the variables which were most important in influencing this behavior. An approach to sensitivity analysis based on regression analysis was found to provide a means of investigating the preceding questions. The overall approach is described in Iman et al. [4]. Basically, the idea is (a) to start with a set of input variables with selected ranges and distributions, (b) to select model inputs from these variables according to their ranges and distributions, (c) to generate model output with the selected inputs, and (d) to assess the relationship between model input and output by stepwise regression. Special techniques found to be useful include (a) Latin hypercube sampling to select values of input variables [5,6], (b) the rank transform to reduce the effects of nonlinearity in the relationships between model input and output [7,8], and

(c) the PRESS (predicted error sum of squares) criterion to indicate overfit during regression analysis [9].

The site for the sensitivity analysis is described in the next section.

SITE FOR ANALYSIS

This paper is derived from a more extensive study [10] in which four variations of a hypothetical river receiving a radionuclide discharge are considered. In this presentation, some of the analysis results are given for one of these four variations. This particular variation corresponds to a lake and a region around the lake which is being irrigated with water from the lake. Specifically, the surface-water subzone consists of a lake and the suspended sediments within the lake, the sediment subzone consists of a layer of stationary sediment beneath the lake, the soil subzone consists of a region about the lake which is being irrigated with water withdrawn from the lake, and the groundwater subzone consists of a shallow aquifer beneath the soil subzone which discharges into the lake. The flows between the subzones are similar to those shown in Figure 1. The limited space available here does not permit extensive description of this site; however, such information can be obtained from Helton et al. [10]. There, the site now under consideration is designated as the Lake Zone in Analysis A.

For this site, some of the data required for the Environmental Transport Model is fixed. However, some of the input data depends on one or more variables which describe variation in the site's description. Specifically, the following variables are considered: X1 - regional erosion rate (units: cm/ 1000 yr; range: 3, 15; sampling dist: uniform), X2 - fraction of suspended sediments entering the lake each year that are trapped and remain in the lake permanently (units: unitless, when sediment entering the lake is expressed in kg/yr; range: .2, .9; sampling dist.: uniform), X3 - scale factor such that the product of X3 and the mass of solids contained in the sediment subzone is equal to the rate of sediment exchange between the sediment subzone and the surface-water subzone (units: yr^{-1}, when sediment mass is expressed in kilograms; range: 10^{-2}, 10^{0}; sampling dist.: log uniform), X4, X5, X6, X7 - distribution coefficient for Cm-245 in groundwater, soil, surface-water and sediment subzone, respectively (units: L/kg; range: 10^{1}, 10^{5}; sampling dist.: log uniform). Some of the other sites considered in Helton et al. [10] have a greater number of variables associated with them.

This site is assumed to receive a release of Cm-245. The effects of such a release into each of the four subzones are investigated. In each case, the release rate is assumed to be 1 unit/yr; however, as the solution of the equation in (2) is linear with respect to h, the convergence to steady state is unaffected and the asymptotic concentrations can be scaled to obtain the results of other release rates. For each subzone, the following two dependent variables are considered: time to reach 90% of steady state concentration (units: yr) and amount of radionuclide present in subzone at steady state (units: units). Specifically, Y1, Y3, Y5 and Y7 correspond to time to reach 90% of steady state concentration for the groundwater, soil, surface-water and sediment subzone, respectively, and Y2, Y4, Y6 and Y8 correspond to amount of radionuclide present at steady state for the groundwater, soil, surface water and sediment subzone, respectively.

For the analysis, 1000 samples were generated from X1, ..., X7 by Latin hypercube sampling with respect to the indicated ranges and distributions. Then, each of these samples was converted into a set of input data for the Environmental Transport Model. The model was run with each such set of data four times, once for radionculide input to each subzone, and the resultant values for Y1, ..., Y8 were recorded.

VARIATION IN MODEL PREDICTIONS

Table 1 provides a synopsis of the calculated results indicated at the end of the preceding section. There is great variation in time to 90% steady state and concentration at steady state for the various possibilities considered. This variation depends on both the subzone for which these quantities are calculated and the subzone receiving the radionuclide input. For example, when radionuclide input is to the soil subzone, the time to 90% of steady state for the surface water subzone varies between 100 years and 24,000 years. However, when radionuclide input is to the surface water subzone, the same time varies between 1 year and 50 years.

SENSITIVITY ANALYSIS OF MODEL PREDICTIONS

As indicated earlier, sensitivity analysis techniques based on stepwise regression were used. Initially, regression on raw data was tried. However, this was not very successful; the regression models generated were limited in their capability to reproduce the predictions made by the Environmental Transport Model. This is not surprising as the solution to the system of differential equations associated with this model involves a matrix exponential and some of the variables under consideration were given large ranges.

It was then decided to try rank regression [7,8]. The idea here is simple: the independent and dependent variables to be used in a regression are replaced by their ranks and then the regression is performed on this new data set. With such an approach, the resulting regression equations predict ranks rather than actual values for the dependent variables. To convert from a predicted rank to a predicted raw value, linear interpolation on the original set of dependent variables is used. The results of such an analysis for radionuclide input to the surface-water subzone is given in Table 2. For each dependent variable included in this table, the independent variables selected in the stepwise regression are listed in the order in which they entered the regression. Further, also listed are the Rank R^2 and Raw R^2 values obtained with each such entry and the standardized regression coefficients for the final regression model. The designation Rank R^2 refers to the R^2 value calculated for the actual regression on ranks; the designation Raw R^2 refers to a "normalized" R^2 value obtained by first converting the predicted ranks to predicted raw values by interpolation and then calculating R^2 with these predicted raw values. The Rank R^2 provides an indication of how successful the rank regression is in predicting the rank of the dependent variable; the Raw R^2 provides an indication of how successful the rank regression is in predicting the dependent variable itself.

For the situation considered, rank regression was successful in picking out the dominant independent variables with respect to time to 90% steady state and amount at steady state. The Rank R^2 values are close to or above 0.900. Indications of the importance of individual variables are provided by the R^2 values, the absolute values of the standardized regression coefficients and the order in which the variables entered the regression. For example, on time to reach 90% of steady state in the groundwater subzone (Y1), all three of the preceding considerations indicate that the two most important variables are the distribution coefficients for the soil and groundwater subzones (X4 and X5).

SUMMARY

It is felt that the techniques indicated in this paper and presented in greater detail in Helton et al. [10,11] can be used to investigate the asymptotic behavior of radionuclide releases to the surface environment. However, such investigations are conditional. They are based on a situation with some set of processes and some set of variables which influence these processes. As the processes and variables are changed, the behavior of the model, and hence the importance of individual variables which influence this behavior, can change.

REFERENCES

1. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P. E. McGrath, F. J. Pearson, H. R. Shaw, J. C. Helton, and F. A. Donath, 1978, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, Report No. SAND78-0029, Sandia Laboratories, Albuquerque, NM.

2. J. C. Helton and P. C. Kaestner, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: Model Description and User Manual for Pathways Model, Report No. SAND78-1711, Sandia National Laboratories, Albuquerque, NM.

3. C. D. Thron, 1972, "Structure and Kinetic Behavior of Linear Multicompartment Systems," Bull. Math. Biophys. 34, pp. 277-291.

4. R. L. Iman, J. C. Helton, and J. E. Campbell, 1978, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis Techniques, Report No. SAND78-0912, Sandia Laboratories, Albuquerque, NM.

5. M. D. Mckay, W. J. Conover, and R. J. Beckman, 1979, "A Comparison of Three Methods for Selecting Values of Input Variables in the Analysis of Output from a Computer Code," Technometrics 21, pp. 239-245.

6. R. L. Iman, J. M. Davenport, and D. K. Ziegler, 1980, Latin Hypercube Sampling (Program User's Guide), Report No. SAND79-1473, Sandia National Laboratories, Albuquerque, NM.

7. R. L. Iman and W. J. Conover, 1979, "The Use of the Rank Transform in Regression," Technometrics 21, pp. 499-509.

8. R. L. Iman, J. M. Davenport, E. L. Frost, and M. Shortencarrier, 1980, Stepwise Regression with PRESS and Rank Regression (Program User's Guide), Report No. SAND79-1472, Sandia National Laboratories, Albuquerque, NM.

9. D. M. Allen, 1971, The Prediction Sum of Squares as a Criterion for Selecting Predictor Variables, Report No. 23, Department of Statistics, University of Kentucky.

10. J. C. Helton, J. B. Brown, and R. L. Iman, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: Asymptotic Properties of the Environmental Transport Model, Report No. SAND79-1908, Sandia National Laboratories, Albuquerque, NM.

11. J. C. Helton and R. L. Iman, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis of the Environmental Transport Model, Report No. SAND79-1393, Sandia National Laboratories, Albuquerque, NM.

Table 1

Minimum and Maximum Values for Time to Reach 90% Steady State and Amount of Cm-245 Present at Steady State[a]

		Y1	Y2	Y3	Y4	Y5	Y6	Y7	Y8
Ground-water	Min. Value	.20E+03	.43E+02	.20E+03	.33E-03	.20E+03	.54E-02	.20E+03	.92E-03
	Max. Value	.27E+05	.12E+05	.38E+05	.19E+02	.27E+05	.82E+00	.27E+05	.20E+02
Soil	Min. Value	.20E+03	.43E+01	.10E+03	.77E+01	.10E+03	.34E-01	.10E+03	.84E-02
	Max. Value	.38E+05	.77E+04	.21E+05	.92E+04	.24E+05	.82E+00	.24E+05	.20E+02
Surface Water	Min. Value	.20E+03	.45E-02	.10E+03	.39E-02	.10E+01	.11E+00	.40E+01	.24E-01
	Max. Value	.38E+05	.16E+02	.21E+05	.19E+02	.50E+02	.83E+00	.94E+02	.20E+02
Sediment	Min. Value	.20E+03	.12E-02	.10E+03	.42E-03	.40E+01	.16E-01	.30E+01	.96E+00
	Max. Value	.38E+05	.14E+02	.21E+05	.16E+02	.94E+02	.81E+00	.93E+02	.40E+02

[a]The leftmost column indicates the subzone receiving radionuclide input at the rate of 1 unit/yr.

Table 2
Sensitivity Analysis Results for Radionuclide Input to the Surface-Water Subzone

	Time to Reach 90% of Steady State				Amount of Cm-245 at Steady State			
Subzone	Variable	Rank R^2	Raw R^2	St. Reg. C.	Variable	Rank R^2	Raw R^2	St. Reg. C.
Ground-water	4*4[a]	0.767	0.792	0.558	4	0.803	0.643	1.340
	5*5	0.940	0.930	0.609	5*5	0.910	0.815	-0.622
	4	0.947	0.936	0.338	6*6	0.937	0.879	-0.067
	1*5	0.950	0.942	-0.061	4*4	0.952	0.902	-0.471
	2*5	0.953	0.944	0.080	5	0.958	0.909	.0291
	5	0.955	0.949	-0.212	2*6	0.962	0.920	-0.152
					1*5	0.965	0.925	-0.066
					2*5	0.967	0.931	0.083
Soil	5	0.981	0.937	0.989	5	0.953	0.811	1.148
	2*5	0.985	0.963	0.076	6*6	0.978	0.890	-0.212
	1*5	0.988	0.979	-0.072	1*5	0.982	0.928	-0.084
					2*6	0.985	0.926	-0.170
					5*5	0.986	0.934	-0.118
					6	0.987	0.934	0.168
					2	0.988	0.947	0.070
Surface Water	3*6	0.698	0.609	0.362	6	0.895	0.760	-1.006
	6	0.765	0.633	0.975	2*6	0.962	0.936	-0.313
	1*6	0.833	0.740	-0.497	1	0.979	0.952	-0.214
	1*3	0.853	0.755	0.178	6*6	0.984	0.970	0.279
	2*6	0.863	0.753	-0.217	1*1	0.985	0.972	0.130
	2*3	0.867	0.757	0.100	1*2	0.986	0.972	-0.062
	3*3	0.873	0.775	-0.579				
	3	0.888	0.808	0.597				
Sediment	3	0.482	0.268	-1.521	6	0.949	0.724	1.442
	3*6	0.677	0.367	0.684	1*6	0.980	0.883	-0.333
	1*2	0.869	0.717	-0.315	6*6	0.985	0.918	-0.260
	1*6	0.894	0.813	-0.365	1	0.986	0.914	0.088
	6*6	0.904	0.817	0.454				
	1*3	0.911	0.820	0.141				
	2*3	0.919	0.828	0.423				
	2	0.937	0.845	-0.320				
	6	0.941	0.850	-0.276				

[a] I ~ Rank XI, I*J ~ (Rank XI)*(Rank XJ)

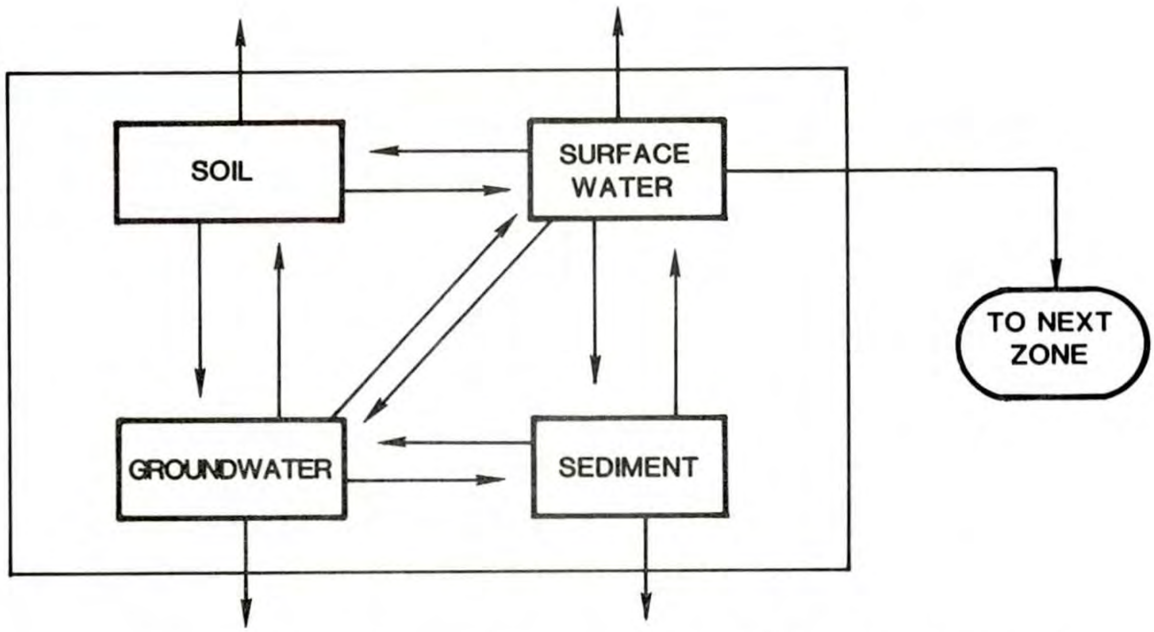

Fig. 1 Division of a zone into subzones. Arrows represent possible movements of water and solid material.

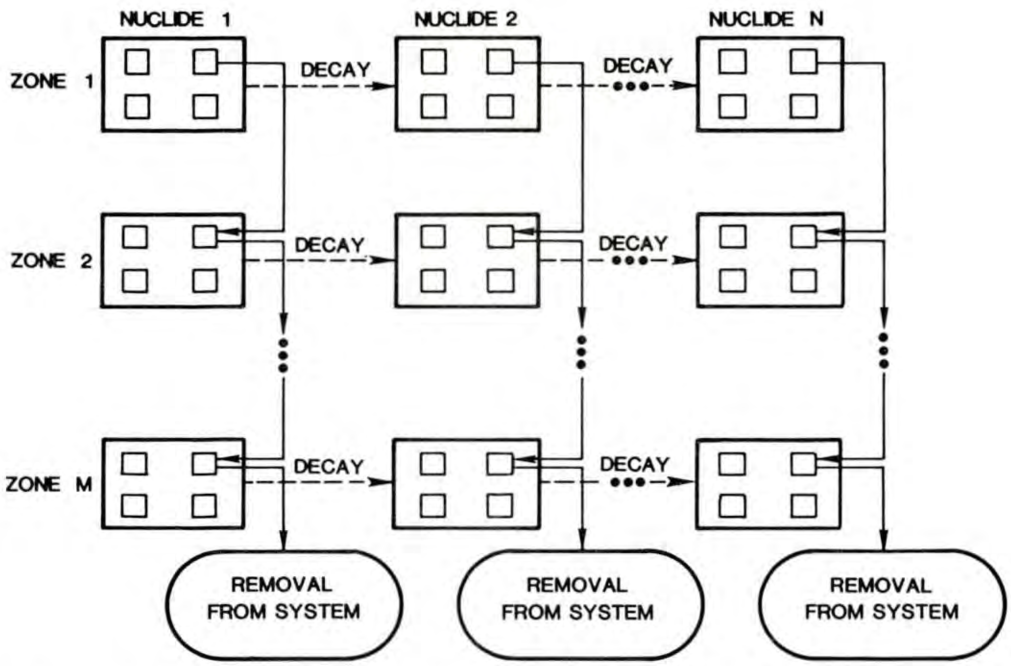

Fig. 2 Flows associated with the movement of a chain of N radionuclides through a system of M zones. Figure 1 represents the flows associated with a single zone in greater detail.

A METHOD FOR ESTIMATING THE DOSE AND HEALTH EFFECTS FROM GEOLOGIC WASTE DISPOSAL

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ABSTRACT

The development of a methodology for assessing the risk from geologic disposal of radioactive wastes includes a model for estimating the individual dose and the resulting health effects from potential radioactive releases from a depository. Possible pathways to the human include ingestion of contaminated food and drinking water, inhalation of suspended radionuclides and external exposure rates in two zones. The exposure rates for the two zones are subsequently converted to dose and estimates of adverse health effects are made by the Dosimetry and Health Effects Model. Since the releases represent chronic lifetime exposures, 70 year intake/70 year dose commitment factors were developed from the work of J. K. Soldat and are used to convert the becquerel values to a dose commitment for the various body organs. Individual risk estimators based on BEIR II and the NRC recommendations from BEIR III are used to estimate the health effects from these dose levels. The ingestion pathway dominates the risk of fatal cancers in the scenarios analyzed to date. The highest risk estimates in Zone 1 result from the scenarios with well discharge directly from the groundwater aquifer to the biosphere.

INTRODUCTION

Included in the methodology developed under the Waste Management Program at Sandia National Laboratories is a series of models to simulate the movement of radionuclides from a geologic depository through geologic media to the biosphere and ultimately to the human. Geologic transport is handled by models such as SWIFT (Sandia Waste Isolation Flow and Transport) and NWFT (Network Flow and Transport) to trace radionuclide movement from the depository until it reaches the biosphere [1].

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In our reference case the biosphere begins when the groundwater discharges into the river or in a shortened pathway situation where there is well withdrawal. At this point the Environmental Transport Model handles the movement of the radionuclides [2]. It takes into account the movement between surface water, sediment, soil and feedback to shallow groundwater.

Using these radionuclide concentrations, the human exposure is calculated from nine pathways. These becquerel (2.7×10^{-11} curie) values are converted to dose expressed in sieverts (100 rem) and risk estimates of adverse health effects from these dose levels are made. The concepts of the models handling human uptake, dose estimates and potential health effects will be detailed in this paper.

METHODOLOGY

The Environmental Transport model traces the movement of the radionuclides through the environmental media. The four compartments of the model, shallow groundwater, soil, surface water and sediments, are handled as a mixing cell with interchange between the compartments.

There are two zones, each with four compartments, in our reference case. Zone 1 describes the area of the river at the discharge point and is designed to display the effects of well discharge. Zone 2 is downstream from Zone 1 and takes into account environmental dilution and retardation, resuspension from soil and irrigation with river water. The only connection between these zones is the surface water compartment.

The radionuclide concentrations in soil, sediment and water of each zone are the final output of this segment of the Pathways Model. These radionuclide concentrations are then multiplied by concentration ratios to determine the radionuclide level in various food sources.

The ingestion pathway considers milk, beef, plants and drinking water. These terrestrial sources may be calculated with or without irrigation. If a scenario analysis includes a well, it is assumed that drinking water is obtained from the well discharge and that the surrounding land is irrigated with well water. Otherwise drinking and irrigation water is obtained from the river. These various food concentrations are combined with uptake rates to calculate consumption from the ingestion pathway. The soil concentration of each radionuclide is combined with a resuspension factor to determine the air concentration. The standard breathing rate of $8000\,m^3/yr$ is multiplied by the concentration to determine the annual inhalation exposure level. The external exposure is calculated by combining the environmental media concentrations and the average exposure times to soil, sediment, air and water. These exposure modes do not account for changes in the eating and living habits of future generations.

In order to present risk in terms of adverse health effects the becquerel per year values for the inhalation and ingestion pathways and the becquerel per

m^2 or m^3 for the external exposure need to be converted to dose expressed as sieverts. The exposure to potential releases from a geologic depository is expected to be low level spanning the lifetime of an individual.

We have developed 70 year intake/70 year dose commitment factors in units of Sv/Bq. These dose factors were derived from the ICRP-II model and the basic INREM model. The input parameters such as absorbed fractions and average energies were taken from Hoenes and Soldat [3]. The basic equations were extended to calculate the 70 year intake/70 year dose commitment factors. A data base of dose factors for 170 radionuclides is available. For our calculations we assume that the concentration level calculated by the Environmental Transport and the Human Exposure Model at a given time step is present for a 70 year lifetime.

These dose commitments expressed in sieverts (100 rem) must be combined with risk factors to estimate the adverse health effects from these exposure levels.

The BEIR [4] committee considers the most important health effects from low ionizing radiation to be the latent somatic effects expressed as latent cancers. We have used the risk factors expressed as deaths per 10^6 rem year from BEIR [5]. Since population statistics are everchanging and our predictions are extending out to 10^4 to 10^5 years after closure of the depository, we have two alternatives (1) to assume population fractions based on populations today or (2) to calculate a risk to an individual with the knowledge that the BEIR risk factors have been derived from population statistics.

We have replaced the population fractions used in calculating risk factors based on person-rem with an age fraction. The age fraction is defined as the fraction of a 70 year lifetime within each age group. For example consider the age segment between 20-30, the age fraction is 0.14 versus a population based fraction of 0.16 [6]. These age fractions are similar to the population fractions and the risk estimators calculated by our method do not vary significantly from the population based estimators. Our final risk estimators express the probability of an individual dying of latent cancer. These estimators are in agreement with the preliminary estimators calculated by NRC for use in their programs. These NRC estimators are based on the ranges presented in BEIR [4].

In addition to the estimators for latent somatic effects, we compare the average annual dose received from the potential releases from the depository and a background level of 0.001 sievert (0.1 rem) per year for estimating genetic and embryonic effects.

METHODS FOR PRESENTING RESULTS

Several methods have been developed to illustrate the results from our analysis. In Figure 1 the individual cancer risk curve is given for a borehole scenario for the reference high level waste depository [7]. The curves are the mean value for thirty-five separate runs of the computer code. Each of these thirty-five runs had a set of input parameters that were sampled from the range of each input variable. The ingestion pathway dominates the cancer risk with the external exposure falling approximately two orders lower. The inhalation path is two orders lower than the external pathway and only the peak appears on this graph.

Another method for illustrating some of the uncertainty of our analysis is given in Figure 2. There are thirty-five individual points at any given timestep. Each point represents the risk to an individual from the set of input parameters that were sampled for that computer run. In this graph only $\sim$ 50% of the thirty-five values are shown with the remainder falling below 10^{-8}. The mean values are represented by the solid line. The mean falls within the upper range of the risk values. The median value is below the 10^{-8} cut off.

The effect on the risk estimates of varying the range of several of the input parameters to the geologic transport model is illustrated in Figure 3. The retardation factor (Kd) ranges of our reference case is given in curve #1. This is the same mean curve previously shown in Figures 1 and 2. Curve #2 illustrates the effect that increasing the upper range of the Kd value by 1000 for all the radionuclides considered in our analysis. The mean risk decreased when the thirty-five computer runs were analyzed. In case #3 the upper range of the Kd values was increased by 1000 and the lower range of the hydraulic conductivity was decreased by 100. The results of this analysis are below the 10^{-8} cut off and for our purpose considered negligible.

From this analysis we can see the sensitivity of the risk results to the uncertainty of the input values. If the regulator is aware of these sensitivities, further analysis and experimental measurements of the appropriate input parameters may be required.

Another type of output from our methodology is in the form of a statistical analysis to present the radionuclides that contribute most to the risk at each time step. The results of a partial rank correlation of the individual radionuclide discharges from the Geologic Transport Model and the total cancer risk is given in Table 1. This is a relative ranking of the contributors to risk with time. There are many radionuclides contributing at early times. Radium-226 is the major contributor to risk after 25,000 years post closure.

APPLICATIONS AND CAPABILITIES

The Pathways and Dosimetry and Health Effects Models discussed in this paper may be used to identify the radionuclides that are the major contributors to risk. Our risk estimates are made on an individual basis with no dependence on population statistics. However, if a population density is known or can be estimated, these individual estimates are easily converted to the population based estimates.

In addition the Pathways and Dosimetry and Health Effects Models are independent of the Geologic Transport Model. It is possible for the user to define any type of source and execute these models independently. The pathways and human exposure parameters, dose factors and risk estimators are easily substituted. Therefore the user is not bound to those that we have used in our analysis.

REFERENCES

1. J. E. Campbell, P. C. Kaestner, B. S. Langkopf and R. B. Lantz, Risk Methodology for Geologic Disposal of Radioactive Waste: The Network Flow and Transport (NWFT) Model, NUREG/CR-1190, SAND79-1920, Sandia Laboratories, Albuquerque, NM (1980).

2. J. C. Helton and P. C. Kaestner, Risk Methodology for Geologic Disposal of Radioactive Waste: User Manual Pathways Model, NUREG/CR-1636, SAND78-1711, Sandia National Laboratories, Albuquerque, NM, (1981).

3. G. R. Hoenes and J. K. Soldat, Age Specific Radiation Dose Commitment Factors for a One-Year Chronic Intake, NUREG-0172, Battelle Pacific Northwest Laboratories, Richland, WA (1977).

4. BEIR, (Biological Effects of Ionizing Radiation -- Advisory Committee), The Effects on Populations of Exposure to Low Levels of Ionizing Radiation, National Academy of Sciences, Washington, DC (1980).

5. BEIR, The Effects on Populations of Exposure to Low Levels of Ionizing Radiation, National Academy of Sciences, Washington, DC (1972).

6. USNRC (U.S. Nuclear Regulatory Commission), Reactor Safety Study, Appendix VI, WASH-1400, NUREG-75/014, U.S. Nuclear Regulatory Commission, Washington, DC (1975).

7. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P. E. McGrath, F. J. Pearson, Jr., H. R. Shaw, J. C. Helton and F. A. Donath, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, NUREG/CR-0458, SAND78-0029, Sandia Laboratories, Albuquerque, NM (1978).

Table 1

Partial Rank Correlation of the Transport Discharge Values and the Individual Cancer Risks

Isotope Time (yrs)	Pu 240	U 236	Th 232	Ra 228	Cm 245	Pu 241	Am 241	Np 237	U 233	Th 229	Cm 246	Pu 242	U 238	Pu 238	U 234	Th 230	Ra 226	Ph 210	Am 243	Pu 239	U 235	Pa 231	Ac 227	Sr 90	Cs 137	Tc 99
4000.0	60						60				60	45		45					60	45		60				
7000.0	45						45	45	45	45	45	45					45	45	45	60	60	60	45			
10000.0	60					60						60	45		45	45	60	60		60	75	60	60			
13000.0																										
16000.0	45																			60	75	60	60			
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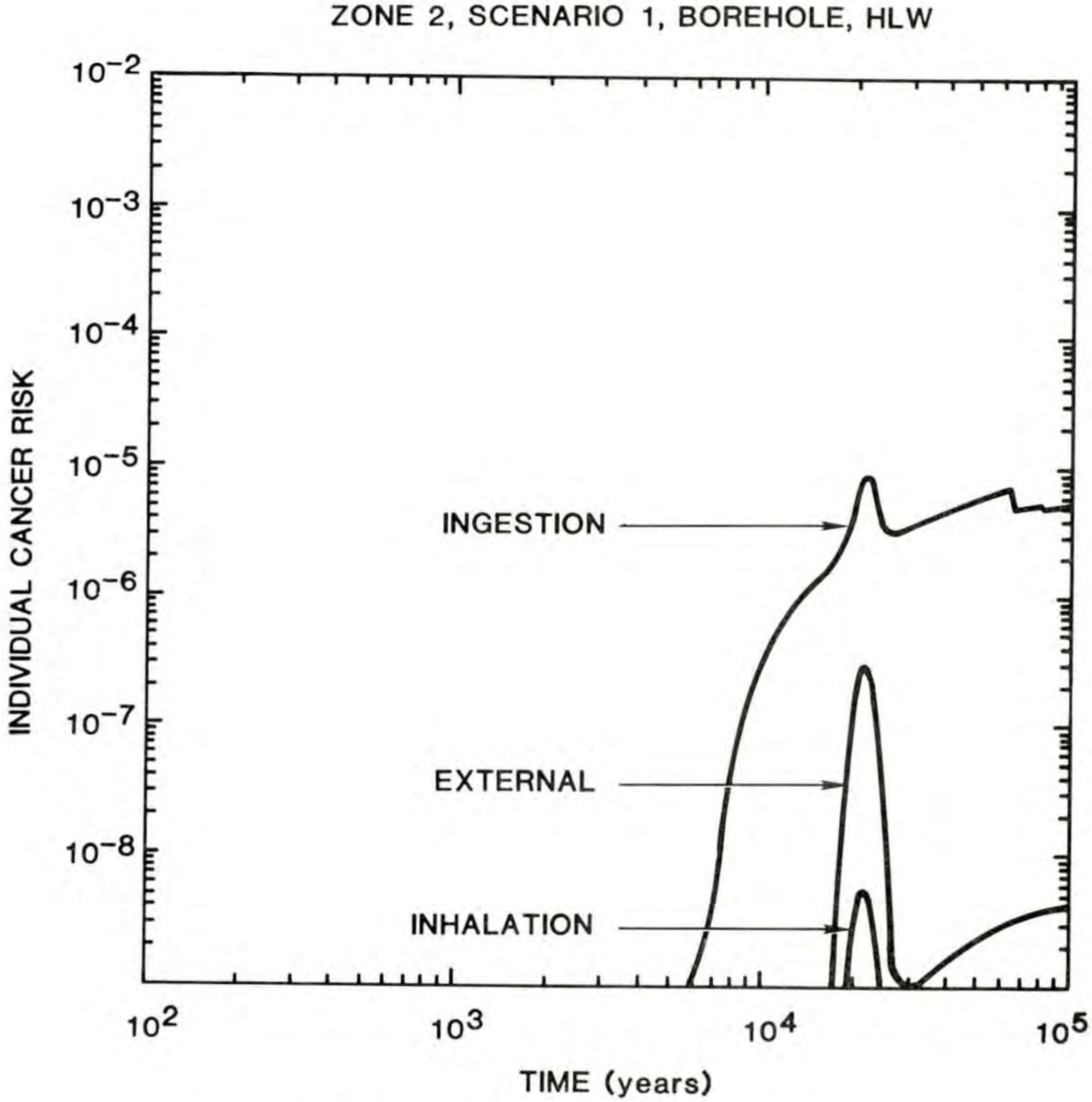

Fig. 1. Conditional Probability of an Individual Dying of Latent Somatic Cancer in Zone 2.

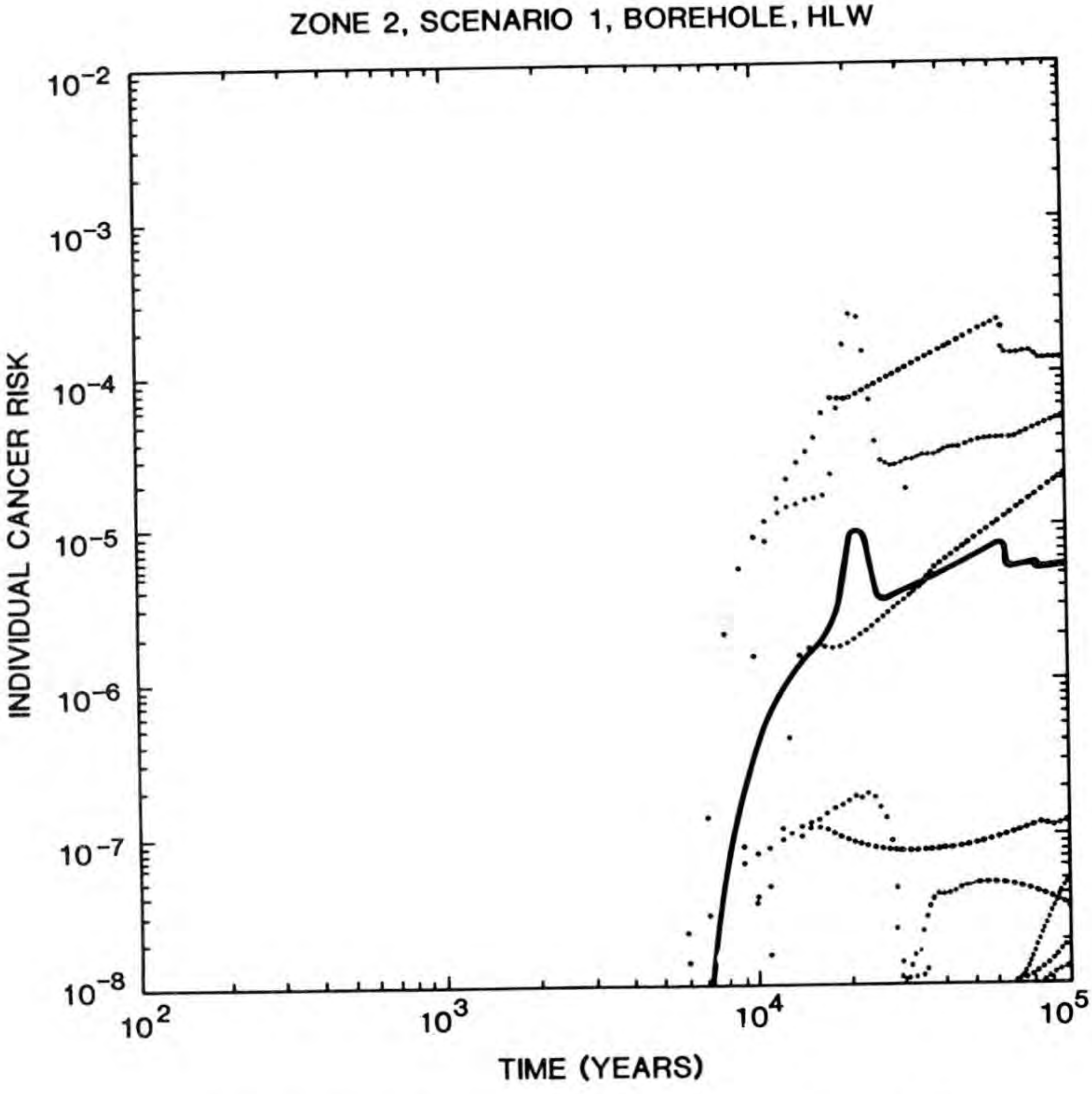

Fig. 2. Individual risk from 35 separate runs of the model at each time step. The dots represent the summation of the risk from the ingestion, inhalation and external paths.

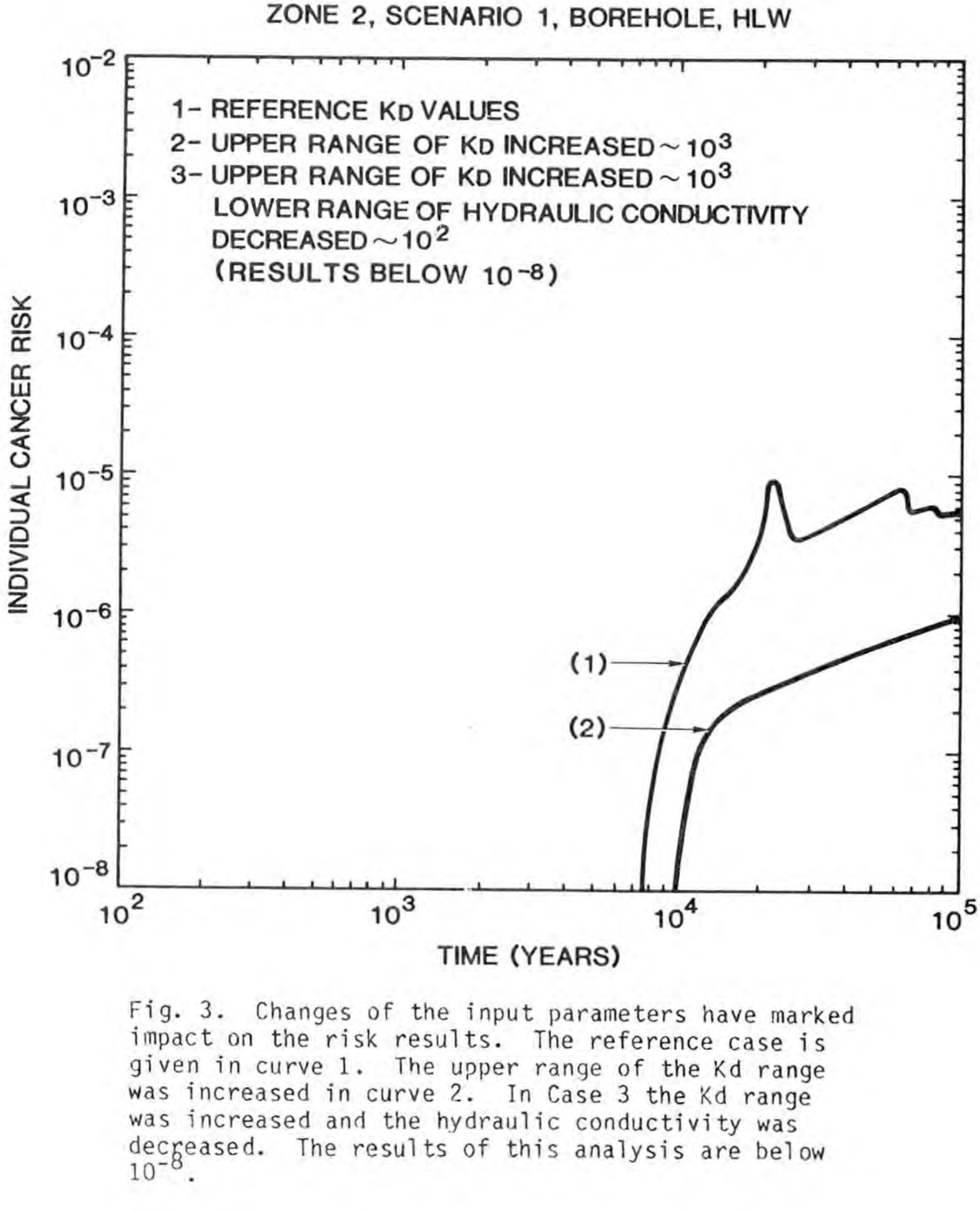

Fig. 3. Changes of the input parameters have marked impact on the risk results. The reference case is given in curve 1. The upper range of the Kd range was increased in curve 2. In Case 3 the Kd range was increased and the hydraulic conductivity was decreased. The results of this analysis are below 10^{-8}.

UNCERTAINTY ANALYSIS FOR GEOLOGIC DISPOSAL OF RADIOACTIVE WASTE

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ABSTRACT

The incorporation and representation of uncertainty in the analysis of the consequences and risks associated with the geologic disposal of high-level radioactive waste are discussed. Such uncertainty has three primary components: process modeling uncertainty, model input data uncertainty, and scenario uncertainty. The following topics are considered in connection with the preceding components: propagation of uncertainty in the modeling of a disposal site, sampling of input data for models, and uncertainty associated with model output.

INTRODUCTION

This paper is derived from a project at Sandia National Laboratories to develop a methodology to assess the risk associated with the geologic disposal of high-level radioactive waste. The project consists of three major parts: the development of models to represent physical processes associated with the disposal of radioactive waste in geologic formations, the development of techniques for the assessment and use of these models, and the application of these models and techniques to a hypothetical waste repository. The development of uncertainty analysis techniques belongs in the second category. Here, the designation "uncertainty analysis techniques" is used to mean methods by which the inexactness of our knowledge with respect to the occurrence of events and processes at a disposal site and the inexactness of our capability to describe such events and processes can be translated into probabilistic statements (e.g., expected values, variances, distributions, uncertainty bounds) about their consequences.

As indicated in the abstract, uncertainty in the analysis of geologic waste disposal has three primary components. Process modeling uncertainty arises from problems associated with selection of processes to be modeled, determination of appropriate parameters for use in model construction, mathematical formulation of models, and numerical techniques used in conjunction with the mathematical formulation of models. Model input data uncertainty arises from problems associated with selection of appropriate values for model input, data interpretation and possible misuse of data, and variation of data. Scenario uncertainty arises from problems associated with the "completeness" of scenarios, the definition of parameters which describe scenarios, and the rate or probability of scenario occurrence. The preceding sources of uncertainty are discussed in the following three sections.

In general, the obtainable information for a potential waste disposal site does not provide immediate insight into the consequences and risks associated with a depository at that site. Models must be used to process the obtainable information into forms which do provide insight with respect to the repository. This leads to the following questions: How should uncertainty be propagated through the various models required to represent a disposal site? How should input data for models be selected as part of an uncertainty analysis? How should the combined uncertainty from models, data and scenarios be represented and analyzed? The preceding questions are discussed in the last three sections of the paper.

UNCERTAINTY ASSOCIATED WITH PROCESS MODELING

The time periods over which the performance of waste repositories in geologic media must be assessed are long - from at least a few thousand years to perhaps a few hundred thousand years. Obviously, experiments and monitoring to gain information on system behavior cannot be carried out over such time periods. Predictive modeling provides the only available way to evaluate candidate sites, waste forms and repository designs and to assess the safety of repositories. Process models are required in the following general areas: waste/rock interaction and feedback effects, groundwater flow and contaminant transport, surface transport and human uptake, and dosimetry and health effects. Discussions of modeling efforts in these areas in the Sandia waste isolation project can be found in the following reports: Dillon, et al. [1], Cranwell and Campbell [2], Campbell, et al. [3,4,5], Helton and Kaestner [6], and Runkle, et al. [7].

A radioactive waste depository will experience a continuous evolution of state due to forces which are independent of the presence of the depository (i.e., externally-induced forces), forces which are caused by the presence of the depository (i.e., self-induced forces) and, perhaps, human intrusion. The modeling of such processes is never exact. Uncertainties arise from a lack of understanding of the processes, a limited capability to mathematically represent the processes, and an insufficient data base with which to describe a system or the processes acting on it. Each of these contributes to uncertainty in the results generated by a process model.

In general, a modeling effort can be thought of as a five step process: recognition of features and processes which must be modeled, recognition of suitable properties and parameters which can be used in the construction of models, development of appropriate mathematical models, development of appropriate numerical algorithms and computer programs to implement the models, and validation of models and their implementation. Uncertainty in the calculated properties of a disposal site can be introduced at each step of the preceding process. Unfortunately, it is also difficult to make general statements about the quantification of such uncertainties.

UNCERTAINTY ASSOCIATED WITH MODEL INPUT DATA

Once appropriate models have been selected to represent processes at a disposal site, one is confronted with the problem of obtaining suitable input data for these models. There is uncertainty associated with such data for several reasons: measurement error, spatial variation, misinterpretation of data. Further, there is the problem of quantifying these uncertainties for use in later uncertainty analysis. Several methods of representing this uncertainty are possible: point estimates, interval estimates, probability distribution functions.

There are several possible sources of measurement error. First, there is the possibility that the measuring technique is either incorrect or misapplied. For example, laboratory tests to determine distribution coefficients might be conceptually incorrect or conceptually correct but misapplied. Also, measurement error could have a physical source due to the treatment of the material to be studied. For example, a specimen is sampled from depth, removed to a laboratory and then tested. In the course of this, the ambient stresses on the specimen are released and the specimen may be damaged. A new stress and thermal state are then applied and measurements are taken. The result is that the measured properties often differ from those in existence in the field. Finally, measurement error could have a statistical source. For example, commonly used estimators for the autocovariance of spatial variability may be statistically biased.

Data measurements often display significant scatter across a site due to spatial variation of rock properties. These properties would vary in space even if it were possible to measure them without error. Uncertainty is introduced by replacing such spatial variability by lumped-parameters (i.e., averages) or by distributed but deterministic parameters (e.g., trend surfaces). Spatial variation is a serious problem. Better characterization of a site by more sampling (i.e., more drill holes) may lead to compromising the integrity of the site. There are various techniques available for spatial interpretation and extrapolation of data. The most prominent technique of this type is Kriging.

Uncertainty can also arise due to misinterpretation of data. For example, even for similar rock and groundwater conditions, measured data for distribution coefficients may vary over several orders of magnitude.

A possible explanation for this variation is an overly simplistic interpretation of distribution coefficients and a resultant misinterpretation of field data. For this example, more detailed models for the causes of radionuclide partitioning may be required for more meaningful interpretation and use of field data.

The representation of uncertainty in model input data is now considered. One possibility is to use point estimates. Here, a single value is selected which reflects the uncertainty in a given model input. Such an estimate might be an average value, a "best" estimate or a conservative estimate. Such quantities usually come from some combination of laboratory measurements, field measurements, theoretical considerations and expert opinion. Another possibility is to use interval estimates. Here, a range of values is assigned to a parameter. This range might represent measurement error, uncertainty in exact value or spatial variation. As before, such a range might be based on laboratory measurements, field measurements, theoretical considerations, or expert opinion. Finally, uncertainty with respect to a variable might be described with use of probability distribution functions. However, sufficient information for suitable definition of such functions may not be available.

UNCERTAINTY ASSOCIATED WITH SCENARIOS

To perform an analysis of a disposal site, it is necessary to determine the various scenarios which could affect the performance of the site. A scenario is defined to be a collection of related events, features and processes potentially affecting radionuclide movement away from a depository and eventual human exposure to these radionuclides. There are several types of uncertainty which arise in the consideration of scenarios: uncertainty associated with "completeness" of scenarios, uncertainty associated with screening of scenarios, and uncertainty associated with analysis of scenarios.

First, there is the question of "completeness": Are all possible scenarios being considered? To provide some confidence as to completeness, a systematic method of compiling scenarios is needed. Further, as it is usually not possible to immediately ascertain the consequences associated with individual scenarios, it is necessary to describe the scenarios in suitable detail and then to use models to predict their consequences. The organizational method must work to group events, features and processes into scenarios in a manner that facilitates the use of available models to predict consequences. Unfortunately, it is not possible to prove "completeness" in the sense of unequivocally establishing that all possible scenarios have been compiled. Through care in scenario development and appropriate independent review, assurance can be sought that a collection of scenarios is acceptably complete. However, this cannot be proved.

The organizational technique that is being used in the Sandia project is now briefly indicated. The events, features and processes that are important with respect to the behavior of a repository are organized into three categories: (1) those which could influence release of radionuclides from the depository to a nearby aquifer system, (2) those which could influence radionuclide movement in groundwater to some surface discharge location, and (3) those which could influence radionuclide movement in the surface environment and resultant human exposure. Each of the preceding categories has a number of sets of conditions associated with it. Appropriate unions of these sets are referred to as scenarios and are the basic organizational units for the analysis of a disposal site. This technique is intended to operate in conjunction with the physical models indicated earlier. Additional discussion of scenario development, definition and application is provided in Cranwell, et al. [8].

Next, there is uncertainty associated with the screening of scenarios. The scenario generation technique will probably generate more scenarios than can be incorporated into the final analysis of a site. Indeed, the first effort at scenario development will probably be to generate as comprehensive a collection of scenarios as possible. Then, a suitable subcollection of these scenarios must be selected for use in a comprehensive site analysis. With the assumption that the scenario development process disallows physically unreasonable scenarios, there are two criteria left which can be used to screen scenarios for inclusion in the final site analysis: consequence and probability. Scenarios with very low consequences can be omitted because of their small potential to affect risk and to cause uncertainty in the analysis of risk. Similarly, scenarios with very low probabilities can also be omitted. It is also possible that scenarios with "intermediate" consequences and probabilities may be screened on the basis of risk. Due to the large computational effort required to perform a site analysis, it is important to reduce the number of scenarios as much as possible. An additional technique that may be useful is to seek out scenarios which are "similar" and to find ways to pool such collections into single scenarios.

Finally, there is the uncertainty associated with the analysis of individual scenarios. This uncertainty has two components: the probability that the scenario will occur and the state of the repository system after the occurrence of the scenario. The preceding are needed to screen scenarios on probability and consequence and to perform risk calculations. Determination of the variables needed to describe the scenarios associated with a disposal site is dependent on both the individual scenarios and the particular site. It is difficult to give specific techniques for their determination in a general paper such as this; indeed, the thrust of this paper is, given that these variables can be determined, how can the uncertainty which they impose on assessments of a site be studied? However, the following six general approaches might be used: application of known physical relationships, laboratory measurements of properties and processes, field measurements of geologic conditions and processes, investigation and interpretation of past historic and geologic records, synthesis of expert opinion, and deliberate conservatism. All of these techniques are most useful when their application is as site-specific as possible. Various of these techniques have been applied in the Sandia waste isolation project. Such applications can be found in the following papers and reports: Beckman and Johnson [9], Cranwell [10], Cranwell and Donath [11,12,13], Donath, Schwartz and Cranwell [14], and Helton and Iman [15].

This completes the first part of the paper. In the preceding sections, three general causes of uncertainty in the analysis of waste disposal sites are discussed. Now, the analysis of such uncertainty is considered. The following discussion is divided into three sections. First, the propagation of uncertainty in the analysis of a disposal site is considered. Next, the selection of the actual variable values which are used in an uncertainty analysis is considered. Finally, the representation and analysis of uncertainty in model output is considered. For the following discussion, it is assumed that the models and scenarios required for a site have been selected and that there is no computational uncertainty associated with the models (i.e., the models are working properly). Further, it is assumed that all uncertainty associated with the behavior of a site can be incorporated into probabilistic statements about either the occurrence of scenarios or the values taken by input data for models used to represent scenarios.

PROPAGATION OF UNCERTAINTY

In geologic waste disposal and many other problems, one generally has a network of interconnected models (e.g., groundwater transport models, biosphere transport models, health effects models, etc.). The process of coupling these models introduces the question of how does one incorporate the propagtion of uncertainties from one model into the calculations of another? A possible approach to this problem is a direct calculation of the distribution function of the output variable. Specifically, given that the distribution functions of the input variables to the models are known, what is desired is knowledge of the distribution function of the output variable. In principle, it is possible to perform this calculation. However, in practice, such an effort would be formidable. Generally, the best that can be obtained is an approximation of the distribution function of the models' output variable. Then, this approximation can be analyzed to gain information with respect to the uncertainty associated with the predictions made by the models. This approximation could be determined through use of a procedure to sample values of the input variables in such a way as to enable the corresponding output values to produce desirable estimates of the mean, variance and distribution function of the models' output.

Thus, one initially has a model D which is a function of the variables $v_1, \ldots, v_n$. Generally, D would consist of submodels $D_1, \ldots, D_k$ which operate in sequence in the sense that D_i generates input for D_{i+1}. An approximation to the distribution function for D might be constructed in the following manner. First, a sequence of samples from the v_i would be generated by some appropriate procedure. Then, D_1 would be managed by a program P_1 which read the samples from the v_i, converted them into input for D_1, supplied this input to D_1, and recorded the results generated by D_1. Similarly, D_2 would be managed by a program P_2 which read the samples from the v_i and the results generated by D_1, converted this information into input for D_2, supplied this input to D_2, and recorded the results generated by D_2. This process would continue until a file containing the actual information of interest (i.e., an approximation of the distribution function of the coupled models) was generated. Thus, the approximate distribution function would be determined through operation of the sampling procedure, the data handling programs $P_1, \ldots, P_k$, and the original models $D_1, \ldots, D_k$.

GENERATION OF INPUT DATA FOR MODELS

Given that the models to be used have been selected and the variables which describe input data for these models have been defined, it is necessary to generate input data from the given variables and their associated ranges for actual use as input to the models. Then, the output generated must be analyzed to determine the uncertainty in predictions about consequence and risk associated with the site. Before a possible sampling technique is considered, several desirable properties for such a method are listed: provide for estimates of mean, variance and distribution of model output, provide for estimates that have small mean square error, provide for estimates of confidence intervals, permit investigation of different distribution assumptions, provide for assessment of relative importance of each input variable, be numerically efficient with respect to the amount of calculation required, permit correlation among input variables.

There exist a number of sampling techniques which might be used to generate input data for models: random sampling, factorial stratified sampling, Latin hypercube sampling, quadrature-based sampling. Several of these methods are compared in McKay, Conover and Beckman [16]. Of the possible sampling methods, the one used in the Sandia project is Latin hypercube sampling. This technique to select n different values from each of k variables $X_1, \ldots, X_k$ operates in the following manner. The range of each variable is divided into n nonoverlapping intervals on the basis of equal width or equal probability. One value from each interval is selected at random (for equal probability, random sampling means with respect to the probability density in the interval). The n values thus obtained for X_1 are paired in a random manner (equally likely combinations) with the n values of X_2. These n pairs are combined in a random manner with the n values of X_3 to form n triplets, and so on, until n k-tuples are formed. This is the Latin hypercube sample. A computer program for generating Latin hypercube samples has been developed and documented by Iman, Davenport and Zeigler [17].

For input data generated by Latin hypercube sampling, the following statements can be made about the analysis of the associated model output: Unbiased estimates of the mean, other moments and the distribution function are possible [18, Theorem 1, p. 11]. Variances of the preceding estimators may be smaller than variances of estimators arising from other sampling techniques; however, this result is related to monotonicity properties of the model. The sample variance provides a biased estimate of the population variance; however, the bias is often small [18, Sections 2.5 and 2.6]. Sensitivity analysis techniques based on partial correlation and stepwise regression can be used to determine the dominant independent variables with respect to the dependent variable [19]. The effects that different distribution assumptions for the independent variables have on the dependent variable can be investigated without rerunning the model [18, Chapter 3]. Also, correlations between variables can be considered [20]. A variation of Latin hypercube sampling known as replicated Latin hypercube sampling can be used to obtain confidence intervals for estimators with respect to the dependent variable (personal communication from R. L. Iman with respect to ongoing work). Determination of the preceding information is efficient in that it can be accomplished with less calculation (i.e., the generation of fewer sample values for the dependent variable) than with other sampling techniques which have been considered [16; 18, Chapter 4). It is felt that all of the preceding considerations fall under the general heading of uncertainty analysis.

UNCERTAINTY ASSOCIATED WITH MODEL OUTPUT

As discussed in previous sections, there are three sources of uncertainty in consequences or risks predicted for a possible disposal site through predictive modeling: uncertainty associated with process modeling, uncertainty associated with model input data, and uncertainty associated with scenarios. In practice, the three categories of uncertainty may blend together and be difficult to separate. For example, uncertainty with respect to the appropriate value of a variable such as hydraulic-conductivity could be interpreted as being due in part to each of the three sources of uncertainty. In the two preceding sections, the propagation of uncertainty in the modeling of a disposal site is discussed. For these sections, and probably in the analysis of any disposal site, it is assumed that the uncertainties in the analysis can be incorporated into ranges and distributions for variables which describe data required in modeling the site. Such ranges and distributions might reflect uncertainty with respect to the value of a physical constant; however, they might also reflect uncertainty with respect to the suitability of a model or the realism of the description of a particular scenario.

By suitable description of input data and orchestration of model operation, it is possible to generate a collection of model predictions with respect to the performance of a site. If the values used in the generation of these predictions have been selected in a suitable manner (i.e., the entire range of each variable has been sampled and it is possible to associate a probability with each of the samples), then it is possible to generate a distribution function which represents the uncertainty in the prediction of interest. Further, one can calculate confidence intervals for this distribution function and its associated mean and variance. However, these confidence intervals are with respect to the uncertainty in the estimates generated by the sampling technique.

The reduction of uncertainty is now considered. There are several approaches which may contribute to this: sensitivity analysis, additional data analysis, additional scenario analysis, model validation, peer review. By suitable sensitivity analysis, the dominant variables in influencing consequences or risk at a site may be determined. Then, additional effort can be directed at understanding the behavior and influence of these variables. In turn, this refined knowledge could result in reduced importance for these variables due to more realistic ranges or modeling or perhaps alterations in depository design or location. Such refinements could come from additional data collection and analysis and also from additional scenario analysis. There have been several applications of sensitivity analysis in the Sandia project: Iman, et al. [19], Campbell, et al. [21], Helton, et al. [22], and Helton and Iman [15]. Additional reductions in uncertainty could be achieved by model validation and refinement. Assurance can be sought that models are both conceptually and computationally correct. In some instances, use of more appropriate or more sophisticated models may reduce uncertainty with respect to the suitability of model input data and the meaningfulness of model predictions. Finally, there is peer review. Unfortunately, many of the areas of modeling and data needs for the analysis of geologic disposal of radioactive waste are not well-defined. Standard techniques and data do not exist. Thus, peer review is an important process in the development of a consensus with respect to the appropriateness of models and data used and the appropriateness of uncertainty measures used in conjunction with such models and data.

We conclude by noting that uncertainty analysis for a waste disposal site will probably be an iterative process. It is unlikely that an overall analysis for a site will be planned and then performed once, thereby giving the information needed to conclude whether or not the site is suitable. What is more likely is that there will be a sequence of analyses, with each analysis providing more information on the behavior of the site but also indicating areas in which understanding is insufficient. Each such area will require additional analysis to increase the understanding of it and thereby reduce the uncertainty in the overall site analysis. It is anticipated that a sequence of analyses would continue until some form of consensus was reached as to the acceptability of the site.

Additional discussion of some of the ideas contained in this presentation can be found in another paper by the authors on uncertainty analysis [23].

REFERENCES

1. R. T. Dillon, R. B. Lantz and S. B. Pahwa, 1978, Risk Methodology for Geologic Disposal of Radioactive Waste: The Sandia Waste Isolation Flow and Transport (SWIFT) Model, Report No. SAND78-1267, Sandia Laboratories, Albuquerque, NM.

2. R. M. Cranwell and J. E. Campbell, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: A Model for Incorporating Feedback Effects in Salt Dissolution Processes, Report No. SAND80-0067, Sandia National Laboratories, Albuquerque, NM.

3. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P. E. McGrath, F. J. Pearson, H. R. Shaw, J. C. Helton and F. A. Donath, 1978, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, Report No. SAND78-0029, Sandia Laboratories, Albuquerque, NM.

4. J. E. Campbell, P. C. Kaestner, B. S. Langkopf and R. B. Lantz, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: The Network Flow and Transport (NWFT) Model, Report No. SAND79-1920, Sandia National Laboratories, Albuquerque, NM.

5. J. E. Campbell, D. E. Longsine and M. Reeves, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: The Distributed Velocity Method of Solving the Convective-Dispersion Equation, Report No. SAND 80-0717, Sandia National Laboratories, Albuquerque, NM.

6. J. C. Helton and P. C. Kaestner, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: Model Description and User Manual for Pathways Model, Report No. SAND78-1711, Sandia National Laboratories, Albuquerque, NM.

7. G. E. Runkle, R. M. Cranwell and J. D. Johnson, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: Dosimetry and Health Effects, Report No. SAND80-1372, Sandia National Laboratories, Albuquerque, NM (to appear).

8. R. M. Cranwell, J. E. Campbell, F. A. Donath, J. C. Helton and N. C. Finley, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: Scenario Development, Report No. SAND80-1429, Sandia National Laboratories, Albuquerque, NM (to appear).

9. R. J. Beckman and M. E. Johnson, 1980, A Probabilistic Model for Estimating the Effects of Volcanism on a Nuclear Waste Repository, Report No. LA-UR80-2166, Los Alamos Scientific Laboratory, Los Alamos, NM.

10. R. M. Cranwell, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: Probabilistic Study of a Waste Repository Being Penetrated by Drilling Activity, to appear as a Sandia National Laboratory technical report.

11. R. M. Cranwell and F. A. Donath, 1980, Application of Geometric Probability to the Existence of Faults in Anisotropic Media, to appear in Proceedings of the International Symposium on the Scientific Basis for Nuclear Waste Management.

12. R. M. Cranwell and F. A. Donath, 1980, Probabilistic Treatment of Faulting in Geologic Media, J. Amer. Geophys. Un., Special Handin Volume.

13. R. M. Cranwell and F. A. Donath, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: Probability of Faulting in a Repository System, Report No. SAND80-0164, Sandia National Laboratories, Albuquerque, NM.

14. F. A. Donath, F. W. Schwartz and R. M. Cranwell, 1980, Probabilistic Aspects of Faulting and Its Possible Influence on the Isolation of Nuclear Waste, paper presented at 26th International Geologic Congress, Paris, France.

15. J. C. Helton and R. L. Iman, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis of the Environmental Transport Model, Report No. SAND79-1393. Sandia National Laboratories, Albuquerque, NM.

16. M. D. McKay, W. J. Conover and R. J. Beckman, 1979, A Comparison of Three Methods for Selecting Values of Input Variables in the Analysis of Output from a Computer Code, Technometrics 21, pp. 239-245.

17. R. L. Iman, J. M. Davenport and D. K. Zeigler, 1980, Latin Hypercube Sampling (Program User's Guide), Report No. SAND79-1473, Sandia National Laboratories, Albuquerque, NM.

18. R. L. Iman, W. J. Conover and J. E. Campbell, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: Small Sample Sensitivity Analysis Techniques for Computer Models, with an Application to Risk Assessment, Report No. SAND80-0020, Sandia National Laboratories, Albuquerque, NM.

19. R. L. Iman, J. C. Helton and J. E. Campbell, 1978, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis Techniques, Report No. SAND78-0912, Sandia Laboratories, Albuquerque, NM.

20. R. L. Iman and W. J. Conover, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: A Distribution-Free Approach to Inducing Rank Correlation Among Input Variables for Simulation Studies, Report No. SAND80-0157, Sandia National Laboratories, Albuquerque, NM.

21. J. E. Campbell, R. L. Iman and M. Reeves, 1980, Risk Methodology for Geologic Disposal of Radioactive Waste: Transport Model Sensitivity Analysis, Report No. SAND80-0644, Sandia National Laboratories, Albuquerque, NM.

22. J. C. Helton, J. B. Brown and R. L. Iman, 1981, Risk Methodology for Geologic Disposal of Radioactive Waste: Asymptotic Properties of the Environmental Transport Model, Report No. SAND79-1908, Sandia National Laboratories, Albuquerque, NM.

23. R. M. Cranwell and J. C. Helton, 1981, Uncertainty Analysis Associated with Radioactive Waste Disposal: A Discussion Paper, to appear in Proceedings of the 1980 DOE Statistical Symposium, pp. 186-196.

STATISTICAL METHODS FOR INCLUDING UNCERTAINTIES ASSOCIATED WITH THE GEOLOGIC ISOLATION OF RADIOACTIVE WASTE WHICH ALLOW FOR A COMPARISON WITH LICENSING CRITERIA

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ABSTRACT

A project funded at Sandia National Laboratories by the Nuclear Regulatory Commission has as its charter to develop a methodology for evaluating applications for nuclear waste repositories. Since the Sandia methodology has the capability of expressing the output variable (for example integrated discharge rates) as a distribution, this report illustrates how to put uncertainty bounds on the output distribution. Additionally this approach permits a comparison against licensing criteria. The licensing criteria used in this paper while hypothetical in nature did involve guidance from experts.

INTRODUCTION

Several reports [1, 2, 3, 4, 5, 6, 7, 8, 9, 10] document the methodology developed at Sandia National Laboratories for using a given set of parameters and random variables as input variables in a computer code that simulates repository behavior, in order to obtain output as a function of time. The sheer volume of output from the code prohibits any comparison with licensing standards as it stands, without some condensing. On the other hand, the complexity of the computer code output permits enormous flexibility in the types of questions that may be answered, and in the types of licensing criteria that may be used.

This paper describes a set of hypothetical licensing criteria and shows how the output data may be summarized in order to permit a comparison with those criteria. It begins on the premise that the output variable is available, having been obtained using methodology documented in the reports mentioned previously. In particular the following three assumptions are made as a basis for the methodology presented in this paper.

1. Computer models are available which model the various physical, chemical and biological phenomena which could affect the discharge of radioactive waste to the accessible environment.

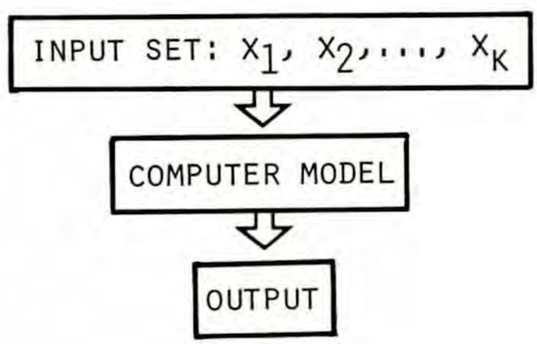
2. A representative data base is available for use with these models.

3. A sensitivity analysis has been performed on these models to identify dominant variables.

These assumptions should not be taken lightly as a great deal of time and effort is needed to satisfy them. It is expected that the computer models would be modified as in situ data is obtained and research programs provide further insights on the important physical and chemical processes. This will lead to an iterative procedure which includes model validating and refinement of the data base and risk predictions. Space limitations of this paper precludes further discussion on this point.

MODEL INPUT AND SUMMARY OF OUTPUT

The Sandia methodology uses several computer models in series to model various physical and chemical processes. The running of each of these models requires that values of the input variables be selected. Additionally, the output of each model is used as input for the next model in line. However, to simplify the process let us assume that there is only one model which requires input and this model in turn produces output we can process. The following schematic diagram illustrates such a case where k input variables have previously been identified as important. Each computer run of the model requires that specific values of each of the k input variables have been selected.



The output could have several possible forms. For example, two possibilities for the output are as follows.

1. Total integrated discharge to the accessible environment over some time period on a per isotope basis.

2. Cancer risk per person over some time period.

Certainly the output could take many other terms and will almost certainly have a time history associated with it. However, again for sake of simplicity let us assume that the output is in the form of total integrated discharge on a per isotope basis and that the integration takes place only to a specific point in time. This means that for a given set of computer runs the output can be summarized in a simple matrix where the rows represent the number of computer runs and the columns the individual isotopes. Cell (i,j) contains the total integrated discharge to a specific point in time for the ith computer run and the jth isotope for a given scenario. Such a matrix is represented in Figure 1 where the number of computer runs has been set equal to 30.

If attention is now directed at any one column in Figure 1 there are 30 estimates of the total integrated discharge for the jth isotope at some specific point in time. The reason that these 30 estimates may differ from one another is that the values of the k input variables have been changed from run to run. These 30 estimates can be exhibited graphically by first ordering the values from smallest to largest and then plotting them as a complimentary cumulative distribution function (i.e. discharge versus probability of discharge). Such a graph appears as a stairstep function in Figure 2 where the integration has taken place over 10^4 years and each step height is equal to 1/30. (Note in Figure 1 that there are actually 4 steps at the value 10 on the horizontal axis where the graph was truncated.) From Figure 2 it can be seen that about 63% of the time this isotope showed a discharge.

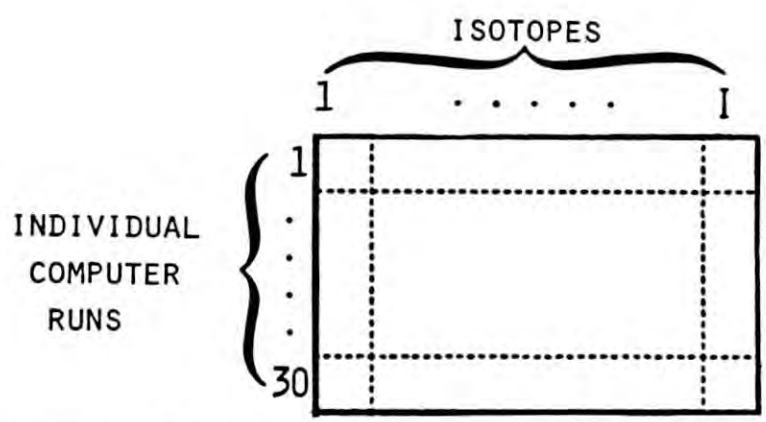


Figure 1. Cell (i,j) contains the output variable at a specific point in time on the ith run for the jth isotope for a given scenario.

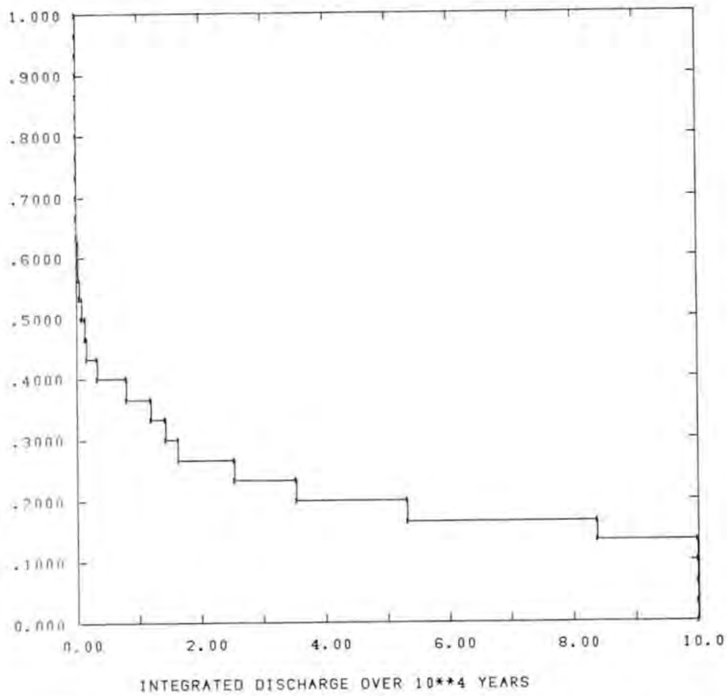


Figure 2. Example of output for 1 isotope based on 30 computer runs.

REPLICATION OF SAMPLES TO OBTAIN UNCERTAINTY BOUNDS

Although Figure 2 does show discharge versus probability of discharge, there are no uncertainty bounds that can be associated with this curve. The reason for this is that a Latin hypercube sample was used to generate the input values and techniques do not exist for placing a bound on a distribution function estimated from a single Latin hypercube sample. The nonparametric Kolmogorov bounds are appropriate for use with random samples, but the Latin hypercube sample is not random and it has never been shown what type of bounds would be provided if the Kolmogorov bounds were used with Latin hypercube samples. However, the distribution function estimates will usually have smaller variability when using Latin hypercube sampling rather than random sampling and this smaller variability translates directly into fewer computer runs. Therefore, it would be desirable to retain Latin hypercube sampling but still have uncertainty bounds. This is accomplished by repeating (replicating) the computer runs by taking new Latin hypercube samples. For example if it were initially decided that 90 computer runs could be made, then these runs should use input from 3 Latin hypercube samples each of size 30 (3 x 30 = 90). This means that the summary matrix of Figure 1 needs to be expanded to accommodate more runs. Such an expanded matrix is shown in Figure 3.

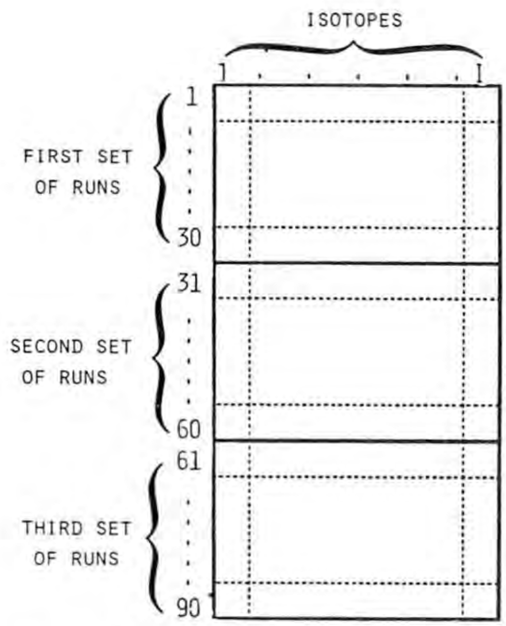


Figure 3. Cell (i,j) contains the output variable <u>at a specific point in time</u> on the <u>ith</u> run for the <u>jth</u> isotope <u>for a given scenario.</u>

Each column in Figure 3 provides 3 estimates of the complimentary cumulative distribution function such as appeared in Figure 2. That is, each set of runs depicted in Figure 3 can be used to plot a complimentary cumulative distribution function. Such plots are given for 1 isotope in Figure 4. The variability demonstrated in Figure 4 is due to the different values used for the input variables from one set of runs to the next.

The value of having the 3 curves in Figure 4 is that they enable the computation of an average curve by averaging step heights at a given value of discharge. In addition to the average, a measure of the variability is also obtained at each value of discharge. For example, consider the discharge value of 4.00 in Figure 4. A dashed vertical line has been added to the graph at this point which intersects the three curves at 4/30 = .13, 6/30 = .20, and 8/30 = .27. Each of these three values is an estimate of the probability of the integrated discharge exceeding 4.00. The average of these three values, (4/30 + 6/30 + 8/30)/3 = .20, is used to plot the average curve at the discharge value of 4.00. The average curve is completed in a like manner by averaging step heights for all other values of discharge. The average curve is the middle curve given in Figure 5 along with uncertainty bounds, which are the upper and lower curves.

The average value computed at each discharge value x is a point estimate of the probability of the discharge exceeding x. The standard error of this point estimate at the discharge value of 4.00 is obtained as

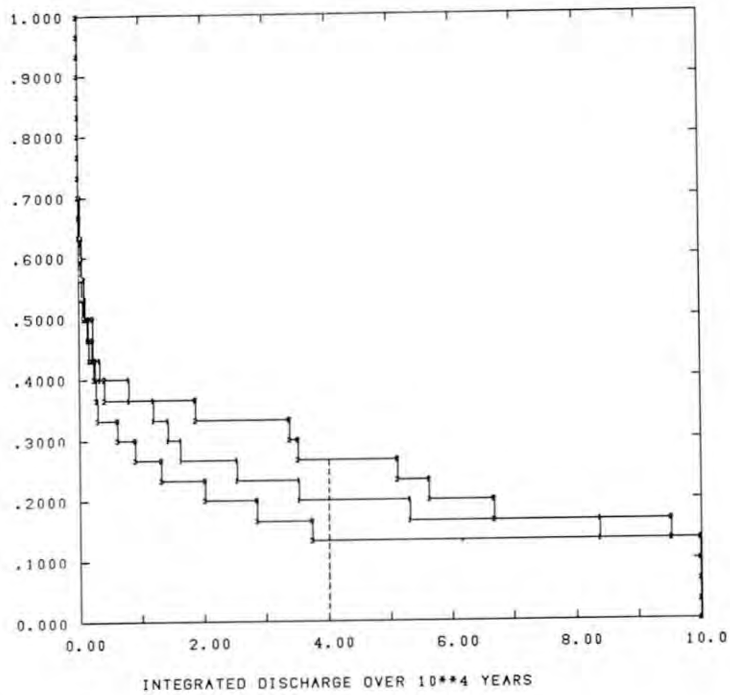


Figure 4. Example of the variability in the output for 1 isotope based on 3 runs of size 30 each.

$$\left[\frac{(4/30\ -\ .20)^2 + (6/30\ -\ .20)^2 + (8/30\ -\ .20)^2}{3\ (3\ -\ 1)}\right]^{1/2} = .04$$

The standard error is computed at other discharge values in a similar manner. This is just the usual way of computing the standard error of the estimate with independent identically distributed values such as these are.

These standard errors can be used to provide uncertainty bounds around the average curve. The reason for this is that the average value calculated at a given discharge value x is the mean of 3 independent and identically distributed observations, and each of these 3 observations is the mean of 30 zero-one random variables, (zero if the discharge value for a given curve is < x and one if the discharge value is > x). Therefore, the Central Limit Theorem may be used to justify using the normal theory limits of $\pm$ 2.92 standard errors as approximate 90% uncertainty bounds. (The value of 2.92 comes from a student's t-distribution with 2 degrees of freedom.) Of course the width of the uncertainty bounds is influenced by the number of computer runs used in each set in Figure 3 as well as the number of sets used and the desired confidence in the bounds, i.e. 90%, 95%, 99%, etc.

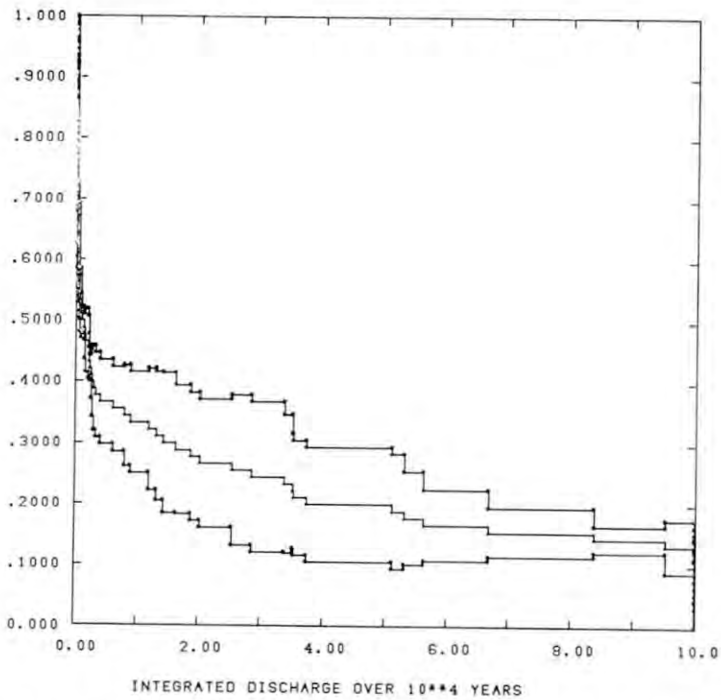


Figure 5. Average curve over 3 runs of size 30 each along with 90% uncertainty bounds.

There is one minor adjustment needed to obtain the uncertainty bounds in Figure 5. This adjustment is necessitated by the discreteness of the step function in Figure 4 and the likelihood that the graphs will coincide at some point such as those in Figure 4 do for discharge values close to 10. When the curves coincide the standard error is zero. In the meantime nearby discharge values may result in considerably different values of the standard error. Acting under the belief that neighboring values of x should have approximately equal standard errors, a moving average (over 3 adjacent observed discharge values) was used to smooth the values of standard error.

HYPOTHETICAL LICENSING CRITERIA

If a licensing criterion was to specify a release limit for each radionuclide over some time period then one may want to consider "normalized discharges." That is, if the total integrated discharge over some time period is denoted by D_j for the jth isotope and the licensing criterion specifies a release limit of L_j for the jth isotope then

$$\text{Normalized Discharge} = D_j/L_j$$

would define the normalized discharge value for the jth isotope. Furthermore, the licensing criterion may specify that a comparison be made of the summation of the normalized discharges over all k isotopes on a given computer run against some upper limit. That is, for the ith computer run, the value

$$T_i = \sum_{j=1}^{k} D_j/L_j$$

would be compared against some upper limit. However, since there are many computer runs involved it may be meaningful to construct a complimentary cumulative distribution function of normalized discharges for each set of computer runs. Figure 6 contains such curves for 3 sets of computer runs when the upper limit has arbitrarily been set equal to 1.

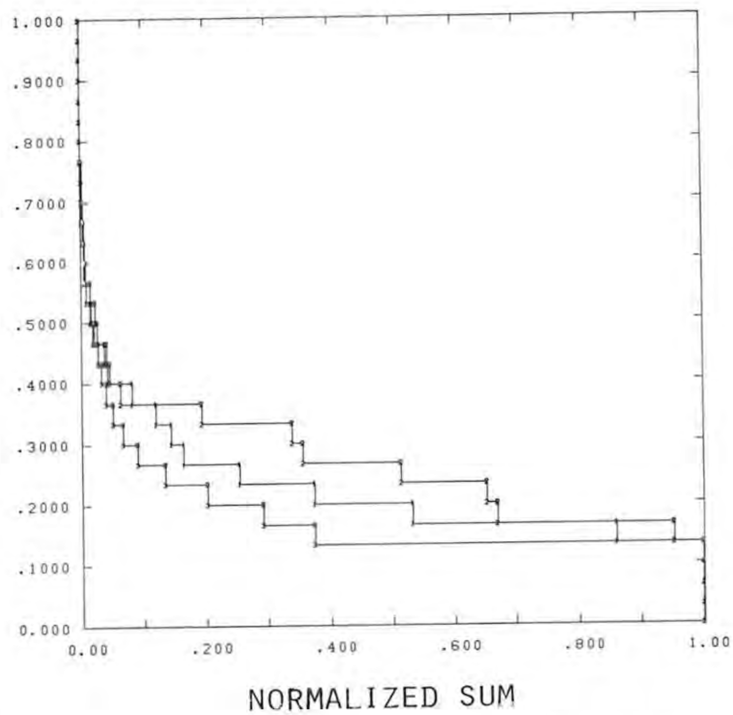


Figure 6. Normalized sum over isotopes for each of 3 runs of size 30 each when the upper limit for this sum has been arbitrarily set equal to 1.

Note that the normalizing and summing procedure effectively removes the dimension of "Isotopes" from Figure 3 and that Figure 3 collapses to a single column containing 90 values. These 90 values form 3 sets of 30 each from which the curves in Figure 6 were plotted.

The three curves in Figure 6 can be treated like those in Figure 4 to produce an average curve with uncertainty bounds as appeared in Figure 5. Such a curve for normalized discharges appears in Figure 7.

Figure 7 brings up a very important point. It seems clear that if the curves in Figure 7 had fallen short of the arbitrary upper limit of 1 then the licensing criterion would be met. However, what happens when the upper limit is exceeded? Another way of stating this is to ask about the interpretation of this upper limit. That is, given the uncertainties that are incorporated into the input and that the output is expressed as a distribution, can the upper limit be exceeded with probability .01 or .05? Or should the licensing criteria be stated to provide values for these two quantiles?

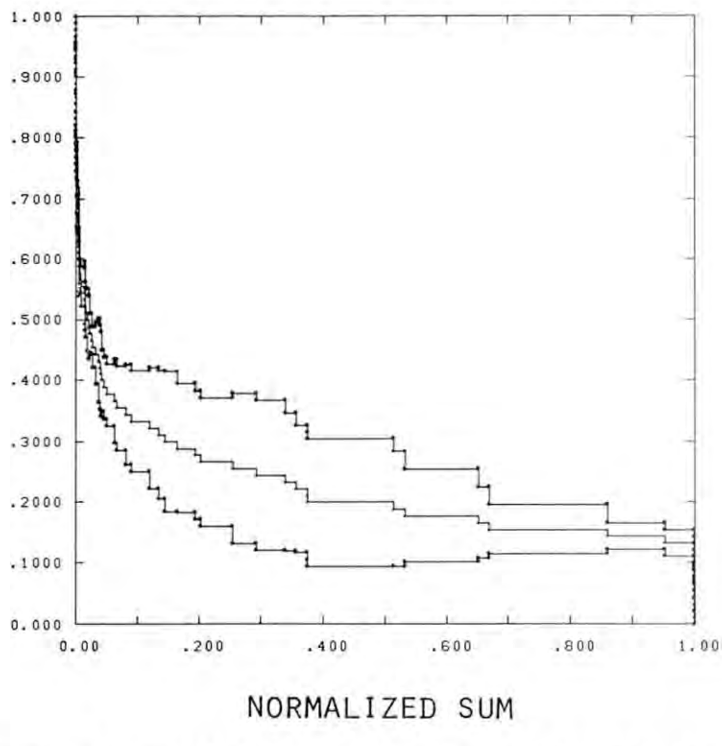


Figure 7. Average normalized sum over 3 runs of size 30 each along with 90% uncertainty bounds for a given scenario.

POST-SENSITIVITY ANALYSIS

Early in this paper an assumption was made that a sensitivity analysis had been performed on the computer models to identify the dominant variables. Equally important with this pre-sensitivity analysis is a post-sensitivity analysis. That is, whereas the pre-sensitivity analysis condenses perhaps a very large set of variables down to a smaller list which are potentially important the post-sensitivity analysis identifies the main contributors to discharge or risk. The post-sensitivity analysis pinpoints those variables which are influencing the placement of the curve in Figure 7. If the curve is showing exceedance of a carefully defined upper limit then perhaps the cause of this can be identified as a variable for which some very conservative assumptions have been made (conservative means in the direction more likely to exceed the licensing criteria). On the other hand if the upper limit is not exceeded this may be attributed to a variable for which a very liberal (non-conservative) assumption has been made.

The post-sensitivity analysis is based on calculating partial rank correlation coefficients given that Latin hypercube sampling was used to select the values of the input variables. Both of the concepts are explained in [2]. If a time history is available on the output variable then a graph of the values of the partial rank correlation coefficient can be made over time for each input variable. This allows the behavior of each input variable to be observed over time. Two such plots appear in Figure 8. The top graph in Figure 8 shows a variable which gradually becomes more important up through about 1500 to 2500 years and then its importance tapers off. The bottom graph shows a variable which continues to gain in importance throughout time.

INCORPORATION OF SCENARIOS

All of the procedures demonstrated thus far have been demonstrated for a given scenario. However, if it is desired to associate weights w_1, $w_2, \ldots, w_s$ with each of s scenarios then this is easily folded into the procedure. First with the addition of scenarios the matrix of Figure 3 takes on a 3-dimensional aspect as given in Figure 9. The procedure starts by normalizing the discharges in each cell by dividing by the appropriate release limit for the isotope in the cell. Next the normalized discharges are summed over isotopes within a given scenario. This removes the dimension of "Isotopes" from Figure 9 and leaves a two dimensional figure (runs by scenarios). The weights w_k are then multiplied times the normalized discharge for each scenario. The summation across scenarios for each computer run then reduces Figure 9 down to 90 values as before. Once this point is reached the uncertainty bounds are formed as was done previously and illustrated in Figures 4 and 5.

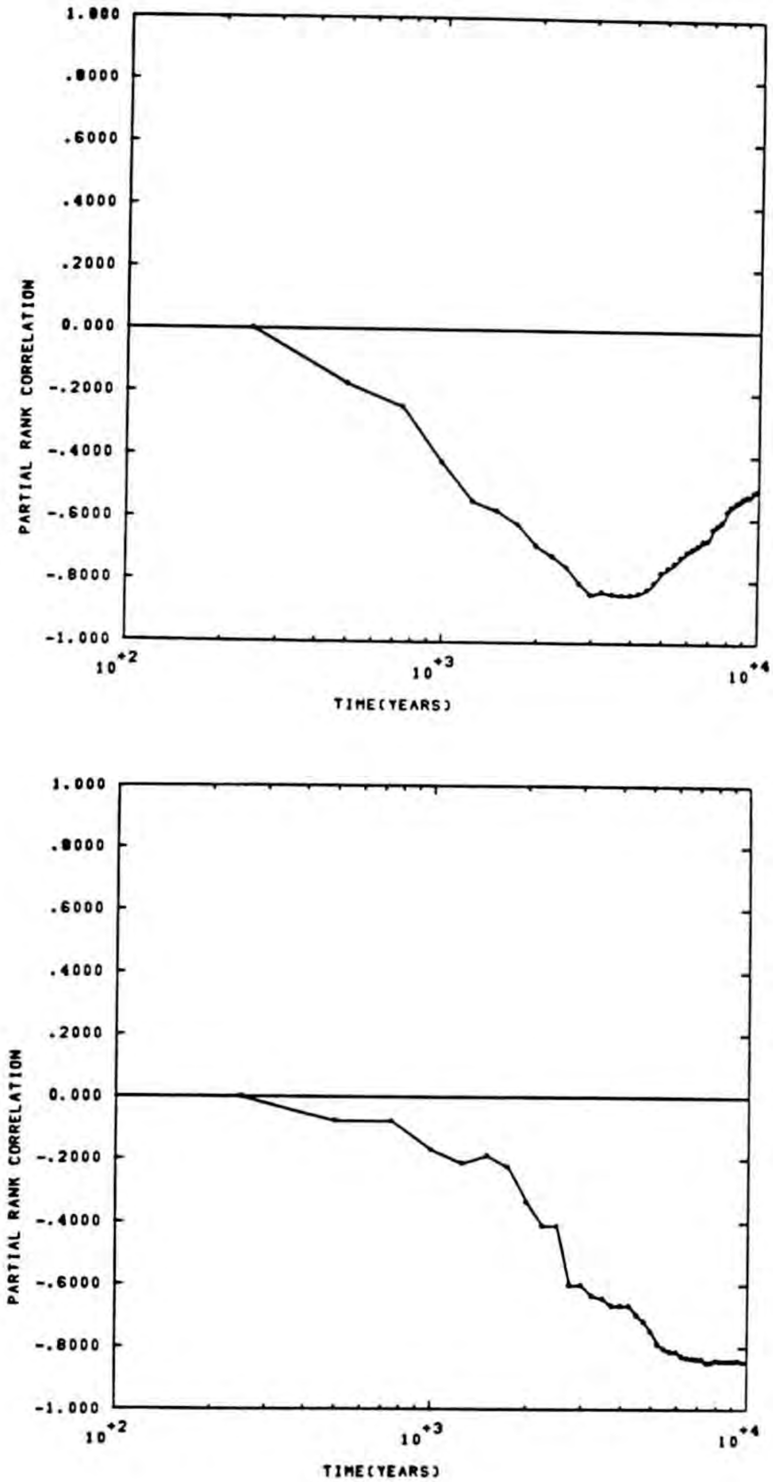


Figure 8. Partial rank correlation plotted over time for two different input variables.

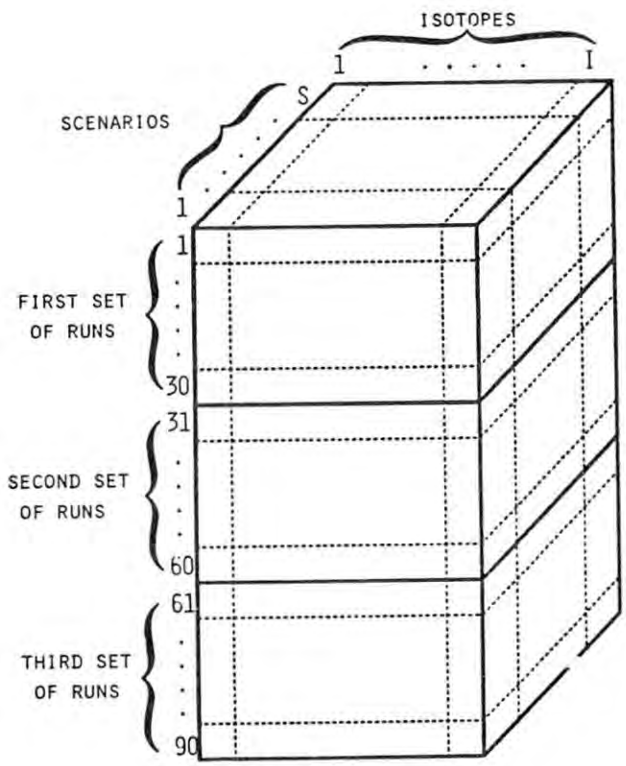


Figure 9. Cell (i,j,k) contains the output variable at a specific point in time on the ith run for the jth isotope under the kth scenario.

ACKNOWLEDGMENT

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REFERENCES

1. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P. E. McGrath, F. J. Pearson, H. R. Shaw, J. C. Helton, and F. A. Donath, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, NUREG/CR-0458 (October 1978).

2. R. L. Iman, J. C. Helton, and J. E. Campbell, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis Techniques, NUREG/CR-0394 (October 1978).

3. R. T. Dillon, R. B. Lantz, and S. B. Pahwa, Risk Methodology for Geologic Disposal of Radioactive Waste: The Sandia Waste Isolation Flow and Transport (SWIFT) Model, NUREG/CR-0424 (October 1978).

4. J. E. Campbell, P. C. Kaestner, B. S. Langkopf, and R. B. Lantz, Risk Methodology for Geologic Disposal of Radioactive Waste: The Network Flow and Transport (NWFT) Model, NUREG/CR-1190 (February 1980).

5. R. L. Iman and W. J. Conover, Risk Methodology for Geologic Disposal of Radioactive Waste: A Distribution-Free Approach to Inducing Rank Correlation Among Input Variables for Simulation Studies, NUREG/CR-1262 (March 1980).

6. R. L. Iman, W. J. Conover, and J. E. Campbell, Risk Methodology for Geologic Disposal of Radioactive Waste: Small Sample Sensitivity Analysis Techniques for Computer Models, With an Application to Risk Assessment, NUREG/CR-1397 (March 1980).

7. J. E. Campbell, R. L. Iman, and M. Reeves, Risk Methodology for Geologic Disposal of Radioactive Waste: Transport Model Sensitivity Analysis, NUREG/CR-1377 (June 1980).

8. J. E. Campbell, D. E. Longsine, and M. Reeves, Risk Methodology for Geologic Disposal of Radioactive Waste: The Distributed Velocity Method of Solving the Convective-Dispersion Equation, NUREG/CR-1376 (July 1980).

9. J. C. Helton, R. L. Iman, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis of the Pathways Model NUREG/CR-1632, Vol. 2 (December 1980).

10. J. C. Helton, J. B. Brown, and R. L. Iman, Risk Methodology for Geologic Disposal of Radioactive Waste: Asymptotic Properties of a Model for the Environmental Movement of Radionuclides, NUREG/CR-1637, Vol. 3 (February 1981).

RISK ASSESSMENT METHODOLOGY FOR HIGH LEVEL WASTE: ASSESSING COMPLIANCE WITH EPA STANDARD INCLUDING UNCERTAINTIES

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(Paper Not Submitted)

ISOTOPIC TECHNIQUES IN RADIOACTIVE WASTE DISPOSAL SITE EVALUATION: A METHOD FOR REDUCING UNCERTAINTIES I. T, T/^{3}He, ^{4}He, ^{14}C, ^{36}Cl

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ABSTRACT

This paper introduces five of the isotopic techniques which can help reduce uncertainties associated with the assessment of radioactive waste disposal sites. The basic principles and practical considerations of these best known techniques have been presented, showing how much additional site specific information can be acquired at little cost or consequence to containment efficiency. These methods, and the more experimental methods appearing in the figure but not discussed here, should be considered in any detailed site characterization, data collection and analysis.

INTRODUCTION

An important attribute of potential sites for the geologic disposal of radioactive wastes is their low hydraulic permeability. This causes difficulties in evaluating their flow systems. Flow times are too long for large artificial tracer experiments. Experiments of acceptable duration yield data on only a few meters of geomedium. Extensive drilling which may seriously hinder isolation effectiveness would be required to characterize the site in sufficient detail for detailed hydrologic modeling. This implies that a large uncertainty must be accepted in modeling flow regimes and transport times in poorly permeable geomedia.

Certain isotopic methods exist which can help reduce this uncertainty by directly determining groundwater ages. The age of ground water sampled at the level of a potential depository indicates the mean time water has taken to flow from the recharge area. The age of groundwater further down gradient along a flow tube indicates how long ago that water was recharged and the difference in ages reflects the travel time from the proposed repository. Naturally occurring radioisotopes (either dissolved in ground water, making up part of the water itself, or in the geologic material through which the ground water travels) may be used to help evaluate the relative groundwater age. This has bearing on nuclear waste disposal site evaluation since it can identify dynamic flow sytems which may rapidly discharge released waste nuclides to the environment or stagnate systems which would aid in containing wastes. Very small amounts of these radioisotopes may be detected in ground water due to their radioactivity, although direct counting for some of the isotopes is now possible by accelerator mass-spectrometry. If one accounts for all the sources and sinks of the isotope, the time since the sample had its initial activity may be determined because radioisotope concentration decreases according to the radioactive decay equation. Groundwater age may be found either by determining the amount of radioisotope which has decayed or by measuring the amount of decay product which has accumulated. Examples of both approaches are discussed here. Five isotopic methods, indicated by arrows in the accompanying figure, will be summarized.

The concept of radiometric "age" of ground water is a difficult one. "Age" implies the time elapsed between recharge and sampling and suggests that no mixing has occurred. A no-mixing system under natural conditions is highly improbable. Recharge is usually not from single events but is an integration of inputs from many events over many seasons. Thus "mean age" is a more useful concept in groundwater dating. Only tritium, since it is part of the water molecule itself, actually reflects the "age" of water. The other isotopes are influenced by different chemistries and may behave differently than the water itself. The isotopic dating of ground water should thus be considered a tool for determining the relative age of water bodies in the context of other hydrologic methods rather than a way to independently determine the number of years since the sample was recharge water.

Tritium (^3H or T)

In nature, tritium ($t_{1/2}$ = 12.46 $\pm$ 0.05 years) is produced by the reaction ^{14}N (n,T) ^{12}C in the upper atmosphere. Secondary neutrons from cosmic ray spallation reactions provide the necessary neutron flux. Tritium enters the water cycle as HT after reaction with O_2. The atmospheric testing of nuclear weapons also creates large quantities of tritium in the stratosphere. The tritium content of rainfall increased from a natural level of about 10 TR* to over 2000 TR during peak testing in 1963, and decreased rapidly to about 55 TR since the enactment of the atmospheric nuclear test ban treaty. This provides an ideal environmental tracer pulse for analyzing hydrologic systems with transit times up to about 50 years. Unfortunately, the tritium content of rainwater is not uniformly distributed in space or time. Due to variations in the shape and intensity of the terrestrial magnetosphere, higher natural tritium production occurs in the polar stratosphere. Further, the anthropogenic tritium was introduced with varying frequency and intensity into the stratosphere at mid to high northern hemisphere latitudes. The tropopause retards its transfer from the lower stratosphere, where it has a mean residence time of 1 to 10 years, to the troposphere where it remains 5 to 20 days before it rains out. The polar and equatorial tropopauses overlap in the winter, encouraging the retention of tritium in the stratosphere. In the spring, the polar tropopause moves poleward and the equitorial tropopause moves toward the equator, causing

*One tritium ratio (TR) equals one tritium atom per 10^{18} atoms of ^1H.

a discontinuity through which tritium readily passes. This results in winter tritium minima in rainfall which equal about one tenth of spring maxima. Once in the troposphere, tritiated water vapor may be removed from the atmosphere by precipitation. Continental precipitation is thus in general higher in tritium than that over the oceans. A gradual rain-out effect can be observed as precipitating air masses move inland.

Because of the complexity of the tritium input function, precise tritium groundwater dating may be done infrequently and only in very well characterized situations. Generally, tritium can be used to indicate the presence of young ground waters (mean age of less than 30 years). Waters containing over 10 TR probably contain a thermonuclear test contribution while 20 TR or more would suggest a component of water recharging since 1961. In waste disposal site evaluation, any measurable quantity of tritium in ground water would identify a dynamic flow system. In such a system, water transport occurs on a much smaller time scale than the radioactive decay of potentially discharged waste nuclides.

Water samples intended for tritium analysis are normally collected in 500 ml well-sealed gas-impermeable containers. Tritium samples are measured by either liquid scintillation or gas proportional counting. These β-counting techniques relate the activity of the sample to the amount of remaining tritium and have a detection level near 10 TR. Water samples can be enriched by hydrolysis, thermal diffusion or gas chromatography to produce samples which are measurable to about 0.1 TR with a routine precision of $\pm$ 0.2 TR.

Tritium-Helium-3 (T/^{3}He)

The tritium/helium-3 method of dating is an extension of the tritium method. Since tritium decays to stable ^{3}He, original tritium concentrations of the water can be reconstructed from the amount of ^{3}He in solution due to tritium decay. Since not all ^{3}He in solution is of this origin, the fraction of the total measured ^{3}He attributable to this source must be estimated from ^{3}He and ^{4}He mass balance. When a simple binary mixture of ^{3}He and ^{4}He are in solution, the amount of ^{3}He from T decay may be obtained from the measured ^{3}He/^{4}He ratio. The mean tritium age of a sample can then be obtained from a modified form of the radiometric dating equation.

The seasonal fluctuations in the tritium input must be assumed to be averaged by mixing processes in the subsurface if T/^{3}He ages are to be taken as representative. Further, some tritium may be produced within the aquifer when uranium or lithium are present. Fortunately this is uncommon in normal, near surface ground waters.

Water samples for ^{3}He analysis are taken in gas-tight 10 ml containers. The dissolved gases are extracted and ^{3}He and ^{4}He concentrations are measured by mass spectrometer.

To date, most $T/^{3}He$ dating of water has been used in lakes [1]. The method helps determine groundwater influences, gas exchange rates, gas renewal, turnover and vertical diffusivity in the epilimnion. Although this technique has been suggested as early as 1969, no example of using the technique to evaluate waste disposal sites has appeared in the scientific literature to date.

Helium-4 (^{4}He)

The helium-4 groundwater dating technique uses decay-product accumulation to determine how long ground water has been in contact with the surrounding geologic media. In nature, ^{4}He is produced primarily by charge neutralization of α-particles from ^{238}U, ^{232}Th and ^{235}U decay. Since the U and Th are generally bound within minerals in the geologic medium and because the mean path length of α-particles is short, most ^{4}He is formed within the solid matrix. The diffusion rate of this ^{4}He into pore water is poorly known. If a steady-state process is assumed, the ^{4}He content of a static ground water in contact with U and Th bearing minerals would increase quasi-linearly with time. The ^{4}He input function is difficult to estimate. The diffusion rate of ^{4}He in various geomedia is poorly characterized. Since in oxidizing environments U can be mobile and form fine-grained mineral coatings on fracture or grain walls, the opportunity for direct, non-migrational release of ^{4}He into solution is greatly improved. Helium-4 may also rise into a groundwater mass from an older, underlying ^{4}He-bearing groundwater body.

One to 50 liters of water are sampled for ^{4}He analysis. The dissolved gases (among them ^{4}He) are flushed from the sample and analyzed by gas chromatography/mass spectrometry. Although a newly developed technique, two studies have employed ^{4}He dating in conjunction with other geochemical and isotopic dating methods. Bath, et al. [2] obtained 17 pairs of ^{4}He and corrected ^{14}C dates for ground waters from the Bunter sandstone in England. The ^{4}He ages, which ranged from 600 to 85,600 years BP were in general greater than the corrected ^{14}C age.

In the Stripa mine granite in Sweden, Fritz et al. [3] obtained about 1.4×10^{5} and 6×10^{5} years groundwater age at the 330 m and 410 m mine levels respectively using the ^{4}He method. As in the case of Bath et al. [3] this is also significantly older than corrected radiocarbon ages. Although the method was successful in identifying very old ground waters in the fracture system, the apparent systematic error in the method observed in the two cases presented here may be due to the release of accumulated ^{4}He from the system. Such releases may be caused by the dilation of microfractures in the granite or changes in surface loading due to glacial fluctuations in the sandstone.

Radiocarbon (^{14}C)

Like tritium, ^{14}C ($t_{1/2}$ = 5730 ± 30 yrs) is produced naturally by the reaction $^{14}N(n,p)$ ^{14}C in the upper atmosphere. Free ^{14}C reacts with O_2 to form CO_2. Anthropogenic radiocarbon was introduced into the atmosphere by thermonuclear weapons testing. These tests increased northern hemisphere middle latitude atmospheric concentrations of ^{14}C from about 100 pmc* to near twice that. The tropospheric ^{14}C content peaked in 1963 and decreases quasi-exponentially thereafter. Nuclear facilities do not produce appreciable amounts of ^{14}C and thus do not influence the global ^{14}C balance. The radiocarbon content of the atmosphere prior to direct sampling may be reconstructed from the ^{14}C analysis of annual tree rings since they are made up of carbon derived from CO_2 respired during the growing seasons of the annual rings. The analysis of the ^{14}C content of dendrochronologically dated tree rings has permitted the generation of an 8000 year record of atmospheric ^{14}C variations. Fluctuations around the atmospheric mean of 100 pmc are neglected since they have far smaller influence than other uncertainties have in radiocarbon groundwater dating.

Unlike the dating of organic materials where an initial ^{14}C activity (A_o) of 100 pmc from the atmosphere can be assumed, dissolved carbonates in ground water derive carbon not only from the soil atmosphere but also from mineral carbonates. In water in contact with $CO_{2(g)}$ and a mineral carbonate (e.g. $MeCO_3$), the following equilibrium is established.

$$CO_{2(g)} \rightleftharpoons CO_{2(aq)}$$

$$CO_{2(aq)} + H_2O \rightleftharpoons H_2CO_3^o$$

$$H_2CO_3^o \rightleftharpoons HCO_3^- + H^+$$

$$HCO_3^- \rightleftharpoons CO_3^{2-} + H^+$$

$$MeCO_3 \rightleftharpoons CO_3^{2-} + Me^{2+}$$

The pH of the solution determines the equilibrium distribution as a function of the temperature-dependent chemical equilibrium constants of the reactions. At pH values below about 8.3, typical of most natural ground waters, no appreciable CO_3^{2-} exists, so that the sample taken for radiocarbon analysis is practically all from HCO_3^-. If a hydrogen ion balance occurs (i.e., if the pH remains constant, which is a reasonable assumption), the equilibrium reaction set reduces to

$$CO_{2(g)} + MeCO_3 + H_2O \rightleftharpoons 2HCO_3^- + Me^{2+}$$

*100 pmc (percent modern carbon) represents the approximate mean atmospheric ^{14}C activity before anthropogenic carbon was introduced.

in which one of the bicarbonates ion produced derives its carbon from the $CO_{2(g)}$ of the soil zone and the other obtains it from mineral carbonates in that zone. If one assumes that all the carbon in the form of H_2CO_3 and $CO_{2(aq)}$ is of gaseous origin, the fraction

$$F = \frac{mH_2CO_3^* + 1/2\ mHCO_3^-}{mH_2CO_3^* + mHCO_3^-}$$

represents the proportion of dissolved carbon which is derived from $CO_{2(g)}$. Here the symbol m represents molality and H_2CO_3* includes $H_2CO_3^0$ and $CO_{2(aq)}$. This fraction may then be used to determine A_0 by

$$A_0 = F\ A_g + (1-F)\ A_{min}$$

where A_g and A_{min} represent the ^{14}C activity of the gaseous contribution of carbon and of the carbonate mineral phase respectively. This stoichiometric model does not account for exchange processes, fractionation, precipitation, dissolution and other geochemical mechanisms. Modification of this model which do account for these processes are discussed by Fontes and Garnier [5]. The bases for these models show the complexity of predicting the initial radiocarbon concentrations in ground water and chemical changes influencing that concentration as the water travels in the subsurface. Nevertheless, the atmospheric source function for ^{14}C is well defined. Although modeling complexity makes accurate absolute dating difficult, if one admits that points along a flow path have been affected by similar physicochemical processes, then mean relative groundwater ages and flow velocities may be obtained with confidence.

Conventionally ^{14}C in ground water is sampled by removing all aqueous carbon species from a 50 liter water sample by precipitating them as $BaCO_3$ in alkaline media. The precipitate is transported to the laboratory and hydrolized to liberate CO_2. The ^{14}C activity is measured by gas proportional counting or liquid scintillation counting. Conventional high precision techniques allow the detection of ^{14}C activity to 0.2 pmc (about 9 $t_{1/2}$ or 52,000 years absolute). Thermal diffusion and laser excitation techniques can be used to concentrate ^{14}C in 5 m^3 samples of water to gain an additional 3 to 4 $t_{1/2}$ (or about 13 $t_{1/2}$ or 75,000 years absolute). Tandem accelerators have recently been used to measure ^{14}C concentrations directly by mass spectrometry. Some researchers believe that detection limits to better than 17 $t_{1/2}$ (about 100,000 years) will be achieved. Results comparable to those obtained conventional methods have already been attained at several accelerator facilities. A great advantage of accelerator techniques is the very small sample size needed: about 100 ml of water or 10 mg of carbon.

Radiocarbon analyses of waters from the nuclear waste disposal test facility at Asse mine near Brunswick, Germany, show waters containing from 54 to 84 pmc. [6] Although no bomb-test carbon can be identified, these results suggest recent waters, with a mean age no greater than 4,000 years and probably containing a much younger component. The comprehensive geochemical and isotopic investigation at the Stripa mine granite nuclear waste test facility [3] included radiocarbon measurements. Waters discharging from the fractures in the granite at the mined 330 m and 410 m level appear to have mean corrected radiocarbon ages of approximately 20,000 years. Further analyses of these preliminary results and of the hydrogeologic and geochemical systems at Stripa are necessary before further interpretations can be made.

Chlorine-36 (^{36}Cl)

Because of this long half-life ($t_{1/2}$ = 3.0×10^5 years), chlorine-36 is well suited for dating old ground waters. It is produced in the atmosphere either by thermonuclear explosions or in small quantities by isotope spallation by cosmic rays converting ^{40}Ar to ^{36}Cl plus an α'-particle. Concentrations of ^{36}Cl may be empirically obtained from site location data [7]. Concentration of ^{36}Cl are usually presented as atoms of ^{36}Cl per unit mass of water. Since total chloride concentration is relatively constant in fresh ground waters, this avoids the problem of dilution by the dissolution of ^{36}Cl-free minerals.

Once recharge waters have infiltrated, production of ^{36}Cl in the subsurface will cause ^{36}Cl ages of ground water to appear too young. Bentley [7] has calculated this contribution for various aquifer materials and has found it significant enough to require a source term correction, particularly in high uranium and thorium bearing minerals. Based on waste inventories, the release of ^{36}Cl into ground water from nuclear wastes stored in geologic formations is expected to be negligible. Calculations show that nuclear wastes such as spent fuels which have a high neutron flux will produce measurable amounts of ^{36}Cl from interaction with ^{35}Cl in the geologic environment according to the reaction ^{35}Cl (n,γ) ^{36}Cl. Because of the extreme mobility of chlorine in solution, ^{36}Cl has therefore been suggested as a means of monitoring nuclear waste repositories in salt formations. Although the aquatic chemistry of chlorine is much more simple than that of carbon, some complications arise with the technique. Some isotopic fractionation due to evapotranspiration before recharge or due to ultrafiltration effects as ground waters pass through silts or clays may change ^{36}Cl concentrations, although fractionation factors are not expected to be large at such a high mass number. Cross-formation ground water flow and mixing may also effect results and subsurface ^{36}Cl production limits the dating technique to about 10^6 years in most geologic environments.

Tandem accelerator mass spectrometry now allows the detection of one ^{36}Cl atom in 10^{16} extracted from four liters. Although interference from ^{36}S in the sample has been a problem with this analytical technique recently chemical separation techniques combined with sulphur isotope dilution has reduced the ^{36}S contamination to one ^{36}S in 10^{10} ^{36}Cl.

A joint effort between the Nuclear Structure Research Laboratory at the University of Rochester and the Department of Hydrology at the University of Arizona has developed the techniques of accelerator measurement and chemical sample processing. They have ^{36}Cl dated a number of groundwater samples from around the world. Bentley and Davis [8] have demonstrated the dilution of ^{36}Cl by old chloride of marine origin in the Carrizo sandstone aquifer in Texas. No appreciable surface production could be identified in a sample of modern salt crust from Wilcox Playa, Arizona, and no subsurface production was evident in a sample of dome salt from Clear Fork, Texas. Waters less than 20,000 years old have been found using this technique in the alluvial aquifer in the Tucson Basin, Arizona. This result is corraborated by radiocarbon results. Recent waters have also been found in the tertiary alluvium of the Madrid, Spain, basin and in the Fox Hills Sandstone, North Dakota. Waters in the 10^5 to 10^6 year range have also been identified in the Fox Hills aquifer. In applications evaluating geologic repositories for nuclear wastes, ^{36}Cl dates of about 5×10^5 years have been obtained at the Savannah River Plant, South Carolina. This agrees very well with the ^3He age. The method is being currently applied to the waters obtained from the basalts at the Hanford Reservation, Washington, and from the bedded salts near the WIPP site in New Mexico.

The Isotope Hydrology Section of the International Atomic Energy Agency in Vienna is currently preparing a publication which reviews in detail these, and other isotopic techniques used in potential nuclear waste disposal site assessment.

REFERENCES

1. Torgersen, T., W.B. Clarke and W.J. Jenkins (1979) The tritium/helium-3 method in hydrology, In: Isotope Hydrology 1978 (Proc. symposium Neuherberg, 1978) IAEA, Vienna, II:917-930.

2. Bath, A.H., W.M. Edmunds and J.N. Andrews (1979) Paleoclimatic trends deduced from the hydrochemistry of a triassic sandstone aquifer, United Kingdom, In: Isotope hydrology 1978 (Proc. symposium Neuherberg, 1978) IAEA, Vienna, II:545-568.

3. Fritz, P., J.F. Barker and J.E. Gale (1980) Geochemistry and isotope hydrology of groundwater in the Stripa granite, Lawrence Berkeley Laboratory Rep. LBL-8285.

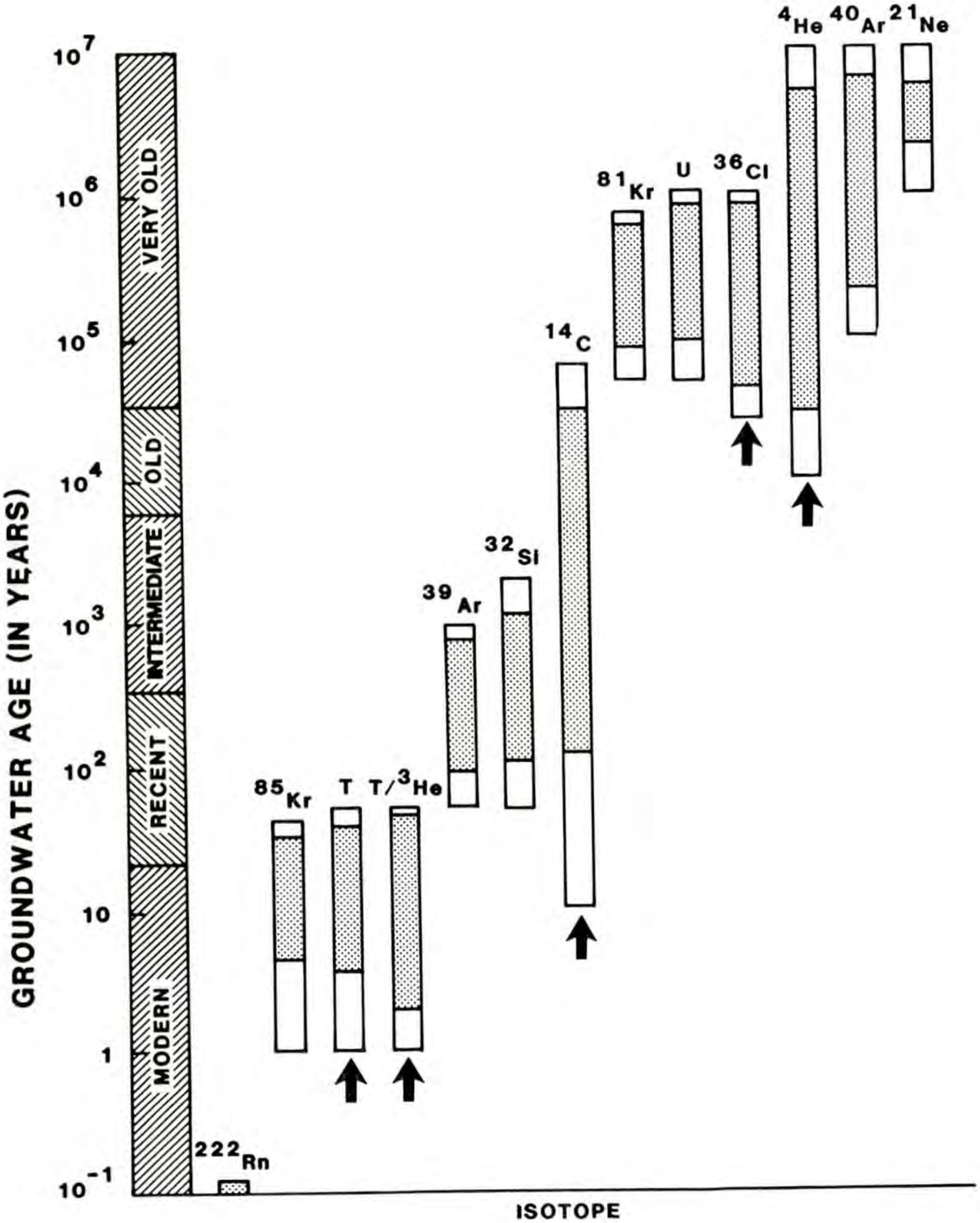
4. Marine, I.W. (1976) Geochemistry of groundwater at the Savannah River Plant, DuPont de Nemours and Co., Savannah River Laboratory Rep. DP-1356.

5. Fontes, J.-Ch. and J.-M. Garnier (1979) Determination of the initial ^{14}C activity of the total dissolved carbon: A review of the existing models and a new approach, Water Resources Research, 15,2:399-413.

6. Batsche, H., W. Rauert and K. Klarr (1980) Hydrologic research at the site of the Asse salt mine, In: Underground disposal of radioactive wastes (Proc. symposium Otaniemi, Finland, 1979) IAEA, Vienna, I:345-370.

7. Bentley, H.W. (1978) Some comments on the use of ^{36}Cl for the dating of very old groundwaters, In: Workshop on dating old groundwater (S. Davis, ed.) (Proc. symposium Tucson, AZ, 1978) Univ. Arizona and Office of Waste Isolation Rep. Y/OWI/SUB-78/55412, p102-111.

8. Bentley, H.W. and S.N. Davis (in preparation) ^{36}Cl dating of very old groundwater.



USEFUL TIME RANGES FOR VARIOUS ISOTOPIC METHODS FOR DATING GROUNDWATER

RISK ANALYSIS METHODOLOGY FOR UNREPROCESSED SPENT FUEL DISPOSAL IN BEDDED SALT

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ABSTRACT

In accordance with the decision to defer the reprocessing of commercially generated spent fuel, we are investigating the implications on risk of direct disposal of spent fuel assemblies. To the extent possible, we are using the methodology developed at Sandia for the NRC to evaluate risks from the disposal of wastes from reprocessing of spent fuel. This allows direct comparison of the risks calculated for the two waste forms. A number of differences between the two waste forms with implications on risk have been identified and investigation of their effects has begun. Among these are the presence of gases and additional plutonium and uranium isotopes, the potential for differing leach behavior, and the difference in the decay heat source which determines the overall thermomechanical response of the host media. We have analyzed a number of scenarios for a hypothetical geologic repository that have been identified as important contributors to risk from the disposal of both reprocessed and unreprocessed spent fuel. For each scenario, we employ the Groundwater Transport, Pathways to Man, and Dosimetry and Health Effects models of the High Level Waste Methodology. Risks are compared for the reprocessed and unreprocessed spent fuel wastes and the effects of uncertainty in the parameters of the various models are compared.

Several years ago, Sandia began the development of a methodology to assess risks from the disposal in bedded salt deposits of high level wastes (HLW) resulting from the reprocessing of spent nuclear fuel [1]. This project, called the HLW study, is nearing completion. Recently, spent unreprocessed fuel (SURF) has become a candidate waste form. The question then arises as to what extent the methodology resulting from the HLW study could be applied to the disposal of SURF. It is the objective of the SURF study to assess the applicability of this methodology to SURF disposal and to expand and modify the methodology, if necessary.

The use of the HLW methodology to describe SURF disposal, if successful, is desirable in several respects,

(1) It avoids duplication of the developmental effort expended in the HLW study.

(2) It allows direct comparison of results of the SURF and HLW risk calculations since both are produced using the same methodology.

(3) It exercises and extends the HLW methodology. The experience and insights thus gained will allow additional checks on the methodology.

The obvious differences between SURF and HLW that may result in differences in the risk and in the variables important to risk are such things as depository geometry and inventory differences, especially those introduced by bypassing reprocessing. In addition, we will investigate the possibility that a different scenarios dominate the risk of SURF disposal than those dominating the risk of HLW disposal. We have in mind here those scenarios which may involve the decay heat characteristics of the waste.

At the beginning of each study a design for the mined facility was assumed from the open literature. These descriptions were needed to provide room and corridor geometry, shaft locations, waste emplacement, density, etc. For the HLW study an 1100 acre design was chosen with waste canisters emplaced at a density producing 60 kW/acre from decay heat [2]. For the SURF study, a 3000 acre design was chosen with an emplacement density of 30 kW/acre [3-5].

The waste radionuclide inventories were taken from available projections of waste generation rates. For the HWL study, the Blomeke "Low growth" projections were used [6]. In the SURF study, a projection for spent fuel assembly discharge rate was used [5] along with SANDIA-ORIGEN [7] to generate the isotope-specific inventory.

With the isotope specific inventories we may begin to address one of the basic questions to be answered in the SURF study. This question addresses the similarity of the sets of scenarios chosen to analyze HLW and SURF repositories. We sort the scenarios into two general categories,

1) generic, media-dependent
2) waste or mining induced

In the first group we place any scenario that may be postulated for a repository-sized tract with the same, or similar geology, independent of the presence or absence of nuclear waste or the type of waste. Examples of scenarios in this category are exploratory boreholes and faulting in the area. In the latter category we would include such scenarios as shaft-seal failure and the thermomechanical response. This last example, the thermomechanical response, is one of potential difference between HLW and SURF which we feel should be investigated.

We have performed a limited anlaysis of the thermomechanical response of HLW and SURF repositories. This analysis was performed using the ADINAT[8] and SANCHO[9] codes available at Sandia. With these we have the capability to analyze the thermomechanical response of the reference repository and to propagate the uncertainties in the thermomechanical parameters. The creep behavior of the salt is included in this analysis. Some preliminary results are presented in Figures 1-3. For this calculation we have assumed an axisymmetric repository making the depository a thin disk of radius R_{dep}.

In Figure 1, the heat generation is plotted and has been normalized to that of the HLW repository, 60kW/acre. Initially the HLW emplacements generate heat at twice the rate of the SURF emplacements. Beyond about 90 years, the SURF emplacements generate heat at a greater rate than the HLW emplacements and continue to do so indefinitely. This results from their higher actinide content. In Figure 2 is plotted the radial dependence of the surface uplift at the time of maximum central uplift for the two repositories. Both are normalized to the depository radius. Here we see that the HLW depository has the greater uplift. In both cases, the bulk of the uplift occurs over a distance of approximately R_{dep}. In Figure 3 is shown the time dependence of the surface uplift over the depository center. Here we see that the response in the HLW depository is nearly over at the time the SURF depository is reaching the maximum uplift. The initial negative displacement results from the treatment of the backfilled drift region. This region is modeled as a volumetrically creeping zone and is intended to account for the higher pore volume in the backfilled region which induces initial subsidence as this region assumes the undisturbed state. After times long with respect to the duration of the thermal pulse, the surface assumes a displacement of about 40 cm due to compaction of the backfilled regions. We will continue to analyze the results of these and other calculations in order to better understand the implications of the stresses and strains induced by the thermal response.

In examining the results of the risk calculations, we have two basic questions to answer,

(1) What are the important variables and parameters in calculating risks from SURF disposal?

(2) If they are not those found in the HLW study, can we understand the differences?

Thus we will compute risks and perform sensitivity analyses on a number of scenarios common to both SURF and HLW disposal and will compare the results.

The first example presented postulates a U-tube hydraulic connection leading to transport of radionuclides through the upper aquifer (Figure 4). In this scenario, we also postulate a field of withdrawal wells into the upper aquifer, down-dip from the depository. The U-tube may result from some combinations of boreholes, shaft seal failures, or thermally induced fractures. In Figure 5 results of a typical risk calculation are shown. At each time a scatter of calculated consequences is shown which results from scatter in such input data as hydraulic properties and geochemical parameters. Also shown are the arithmetic mean and median consequences. The sensitivity analysis determines the variables dominating the spread in calculated consequences at a given time.

Results of this sensitivity analysis are shown in Figures 6 and 7. In Figure 6 is shown the importance to the consequence of the variable determining the duration of the release of radionuclides from the depository. The quantity plotted indicates the importance of the variable by its proximity to unit magnitude. The first observation is that the importance is time-dependent in general. Secondly, the variable is important for both HLW and SURF disposal, a frequent result for variables related to the radionuclide source term.

In Figure 7 is shown the time dependence of the importance of the sorption constant, k_d, for Am. Here we see that this variable is important for both waste repositories initially, but becomes less important with time for SURF repositories. This results from the difference in the assumed radionuclide inventories in the two studies. In the HLW the ^{245}Cm inventory is present in sufficient inventory to maintain a nearly constant ^{241}Am inventory. In the SURF inventory assumed, the initial ^{241}Am inventory is somewhat higher than in the HLW while the ^{245}Cm parent is lower. Thus, the equilibrium ^{241}Am inventory is lower in SURF and Figure 7 is showing the decay of a larger initial ^{241}Am inventory, and, along with it, the importance of its sorption constant to the risk.

In Figure 8 is shown a borehole scenario also analyzed. For this analysis we chose to look at the total integrated discharge as a measure of the risk. A similar sensitivity analysis showed distribution coefficients, leach rates, hydraulic conductivities and release initiation time to be important for both HLW and SURF. However, in the SURF analysis, solubility limits for Pu and U were also important. These last two variables were not important for either HLW or SURF in the U-tube scenario, most likely due to the higher groundwater flows in that scenario. The result may be understood by noting the higher U and Pu inventories accessed in this scenario for the SURF repository.

Preliminary conclusions from the SURF study may now be presented.

(1) The set of scenarios chosen for the analysis of SURF repositories is not expected to differ greatly from those chosen to describe HLW depositories.

(2) The variables of importance in risk analysis are not universal. They may depend on the radionuclide inventory, the scenario, the depository geometry, and time.

The study of risk from disposal of unreprocessed spent fuel in bedded salt will be completed this year.

REFERENCES

1. J. E. Campbell, R. T. Dillon, M. S. Tierney, H. T. Davis, P. E. McGrath, F. J. Pearson, H. R. Shaw, J. C. Helton and F. A. Donath, Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report, Sandia Laboratories, SAND78-0029, NUREG/CR-0458, October 1978.

2. "Waste Isolation Facility Description-Bedded Salt," prepared by Parsons, Brinckerhoff, Quade and Douglas, Inc., for the Office of Waste Isolation, Union Carbide Corp., Oak Ridge, TN, Y/OWI/SUB-76-16506, September 1976.

3. J. S. Ritchie, A National Waste Terminal Storage Repository in a Bedded Salt Formation for Spent Unreprocessed Fuel: Basis of Design Document and Conceptual Design Decisions, Report 78-59-R, Oakland, CA, Kaiser Engineers, December 1978.

4. J. S. Ritchie, A National Waste Terminal Storage Repository in a Bedded Salt Formation for Spent Unreprocessed Fuel: Conceptual Design Report, Vols. 1 and 2, Report 78-57-RE, Oakland, CA, Kaiser Engineers, December 1978.

5. J. S. Ritchie, A National Waste Terminal Storage Repository in a Bedded Salt Formation for Spent Unreprocessed Fuel: Twenty-Five Year Retrievability, Special Study, Report 78-60-RE, Oakland, CA, Kaiser Engineers, December 1978.

6. J. O. Blomeke and C. W. Kee, "Projections to be Generated," International Symposium on the Management of Wastes from the LWR Fuel Cycle, Denver, CO, July 11-16, 1976.

7. D. E. Bennett, Sandia-ORIGEN User's Manual, NUREG/CR-0987, Sandia Laboratories, Albuquerque, NM, October 1979.

8. K. J. Bathe, ADINAT: A Finite Element Program for Automatic Dynamic Incremental Non-Linear Analysis of Temperatures, Massachusetts Institute of Technology, Report 82448-5, Cambridge, MA, May 1977.

9. S. W. Key, C. M. Stone and R. D. Krieg, "A Solution Strategy for Quasi-Static Large Deformation, Inelastic Response of Axisymmetric Solids," Sandia National Laboratories, presented at the US-European Workshop on Nonlinear Finite Element ANalysis in Structural Mechanics, Ruhr-Universidata, Bochum, FGR.

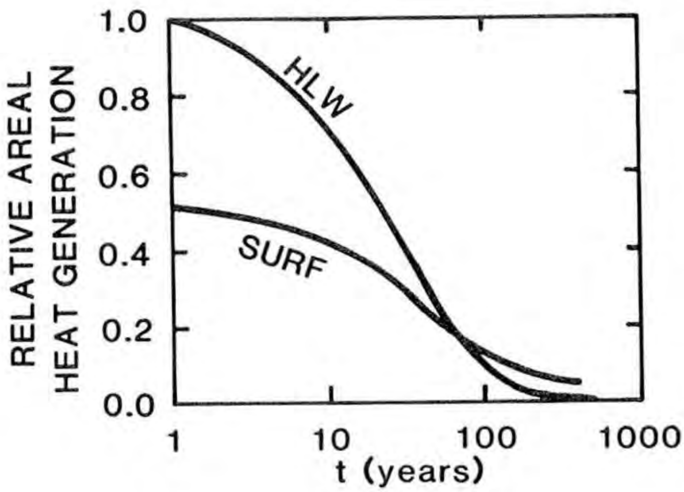


FIGURE 1

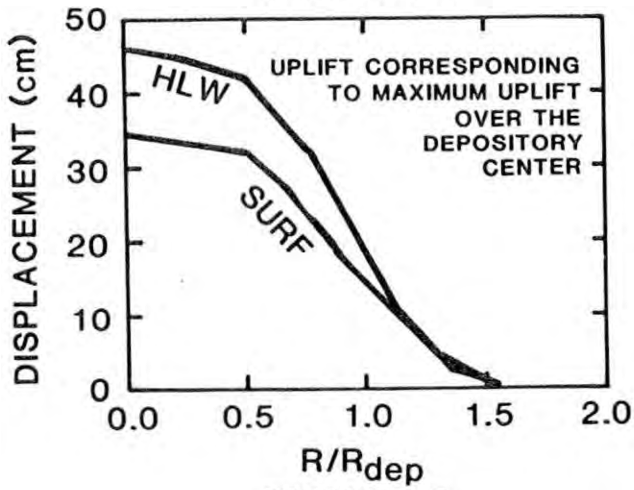
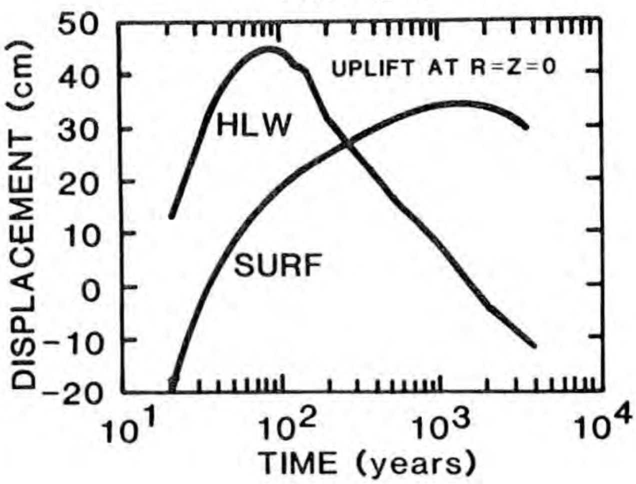


FIGURE 2



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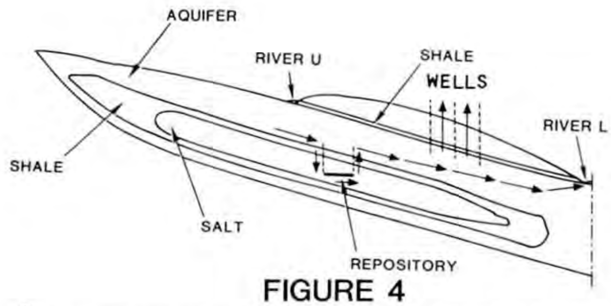

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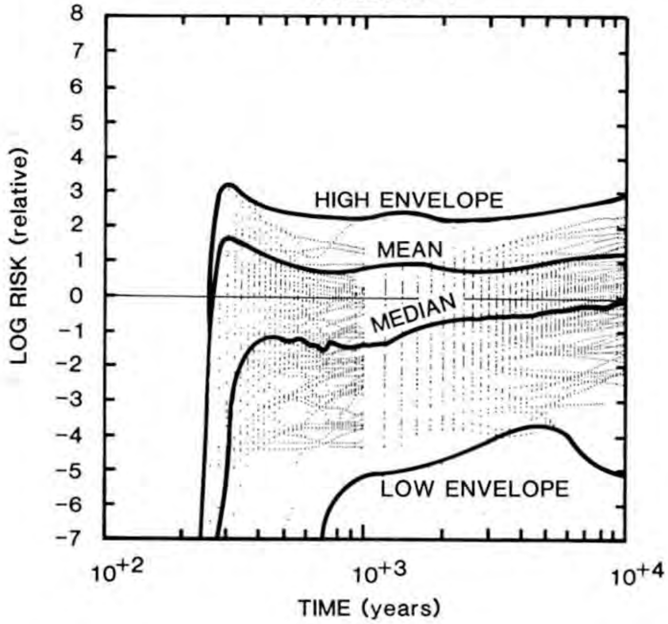

FIGURE 5

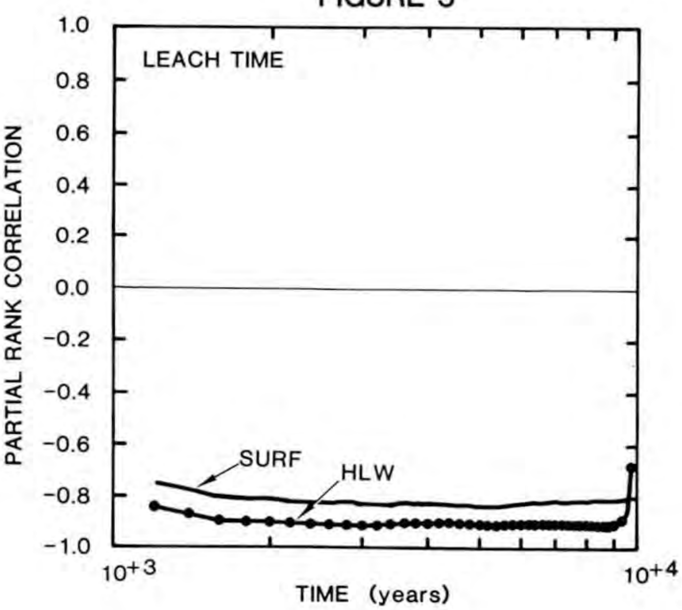

FIGURE 6

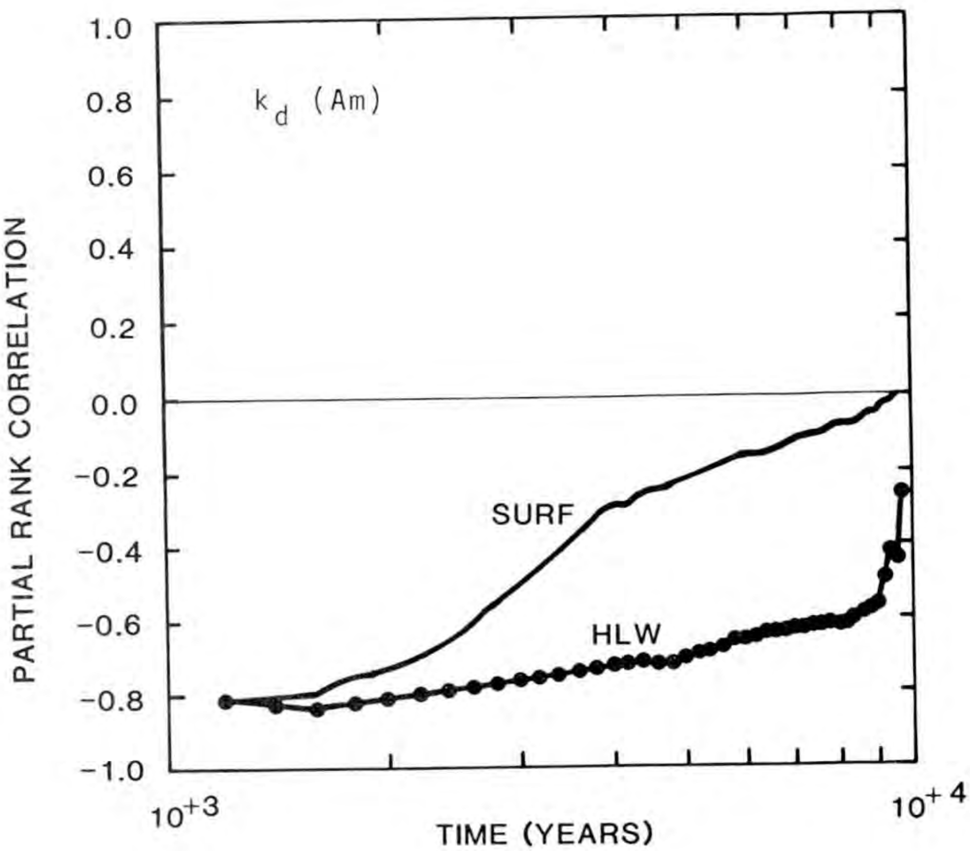

FIGURE 7

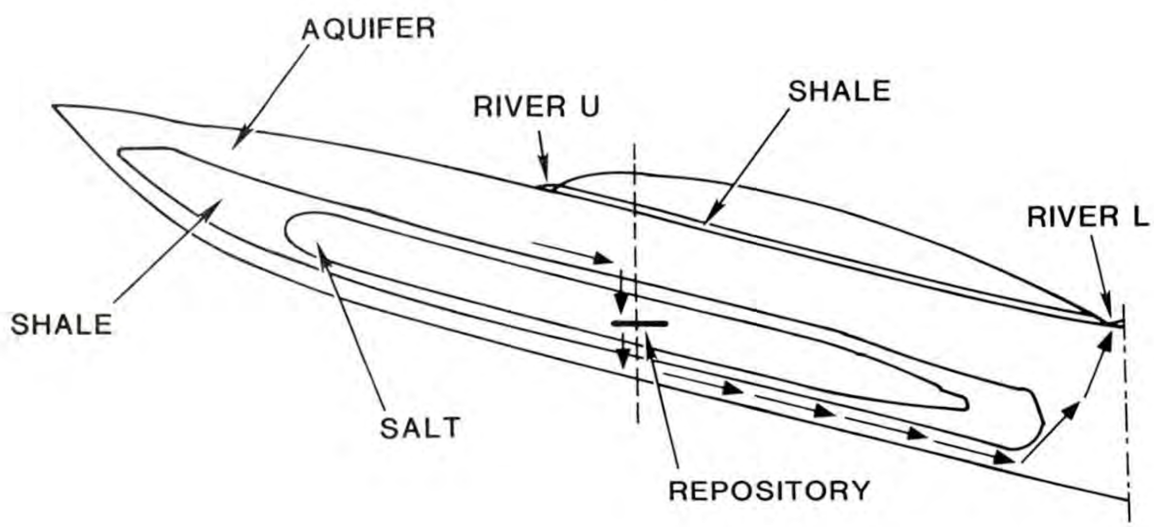

FIGURE 8

Session II:

GENERAL TOPICS

Chairman

Nestor R. Ortiz

Sandia National Laboratories

THE NUCLEAR REGULATORY COMMISSION WASTE CONFIDENCE PROCEEDING: AN OVERVIEW

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ABSTRACT

In late 1979, the Nuclear Regulatory Commission initiated what has come to be known as the "NRC Waste Confidence Rulemaking." This paper presents a general background of the proceeding, and its current status, as well as a brief discussion of positions taken by some of the participants on various significant issues.

THE NRC WASTE CONFIDENCE PROCEEDING

The U. S. Court of Appeals for the District of Columbia Circuit on May 23, 1979 rendered a decision remanding to the NRC two licensing actions involving the expansion of specific reactor spent fuel pools, for consideration of "whether there is reasonable assurance that an off-site storage solution will be reasonably available by the years 2007-2009, the expiration of the plants' operating licenses, and if not, whether there is reasonable assurance that the fuel can be stored safely at the reactor sites beyond those dates." In its decision, the court specifically rejected arguments that the NRC was required to address these issues through an adjudicatory proceeding, and, indeed held that in view of the breadth of the questions involved and the fact "that the ultimate determination can never rise beyond a prediction," the determination could be "a kind of legislative judgment for which rulemaking would suffice[1]."

In response to this decision, and also as a continuation of previous expressions of intent to periodically review the basis for its past explicit and implicit findings of confidence, the Commission published on October 25, 1979 a notice of its intent to conduct a rulemaking proceeding whose purpose is "solely to assess generically the degree of assurance now available that radioactive waste can be safely disposed of, to determine when such disposal or off-site storage will be available, and to whether radioactive wastes can be safely stored on-site past the expiration of existing facility licenses until off-site disposal or storage is available[2]."

Following this notice, the proceeding attracted more than 40 full participants, representing industry, states and cities, Federal agencies including the Department of Energy (DOE), individuals, and non-profit organizations including such entities as the American Nuclear Society and the American Institute of Chemical Engineers, and such groups as the Natural Resources Defense Council (NRDC) and the New England Coalition on Nuclear Pollution. Shortly after a late January, 1980 prehearing conference to resolve a number of procedural issues, the presiding officer issued an order which, among other things,

1. sustained DOE's position that the proceeding should deal only with spent fuel and not high-level waste from reprocessing,

2. provided clarification that the rulemaking would not address low-level wastes, mill tailings, etc., nor would it address the safety of transportation,

3. ruled that the proceeding does not represent a major Federal action having a significant impact on the environment, and thus an environmental impact statement is not required[3].

Additionally, the order provided a schedule for written submittals and responses to be filed by the participants.

On April 15, 1980, the U. S. Department of Energy filed its Statement of Position, a 740-page document which concluded:

"1. Spent nuclear fuel from licensed facilities ultimately can be disposed of safely off-site.

2. Disposal facilities will be in operation between 1997 and 2006, and the initial increment of off-site storage facilities can be in operation by 1983.

3. Spent nuclear fuel from licensed facilities can be stored safely either on-site or off-site until disposed of ultimately."

On July 7, 1980 the other participants filed their statements of positions. At the risk of oversimplification, it may be said that statements of industry groups, professional organizations and Federal agencies generally favored a finding of confidence, while, on balance, States and public interest groups questioned to varying degrees the ability of the Federal government to solve a number of alleged technical and institutional problems. The four-volume Statement of Position of the Utility Nuclear Waste Management Group, and the Edison Electric Institute, who are participating jointly, expressed support for the Department of Energy position, and provided an independent assessment of the status of storage and disposal technology, and the basis of our confidence in the long-term performance of a mined geologic disposal system.

On September 5, 1980 participants filed cross-statements on the positions taken by others, and on October 6 all participants were required to file suggestions as to the conduct of further proceedings, additional areas of inquiry or other data or studies required to reach an informed decision.

I will return to the chronology of the proceeding and its present status in a moment, but first I would like to briefly summarize for the benefit of this audience the more significant allegations made on the part of those participants who would favor a finding of no confidence in safe waste disposal, with particular emphasis on those allegations relating most directly to the subject matter of this symposium - the development of standards and criteria, and the uncertainties surrounding prediction of post-closure repository performance. In summarizing these concerns, I fear I will depart slightly from a dispassionate recital of the issues, and give you a very abbreviated UNWMG-EEI response.

Statements by some participants raise a fundamental objection in that they claim that, without NRC and EPA criteria and demonstration that these criteria will be satisfied, there can be no assurance that radioactive wastes can be safely disposed of[4,5]. Briefly stated, the UNWMG position on this issue is that in no area of human affairs does the absence of government regulations make an assessment of safety impossible, and, indeed such evaluations are routinely performed in other areas. Bearing in mind the massive amount of information in existence concerning the effects of radiation and what constitutes acceptable levels of exposure, the UNWMG-EEI submission contains many comparisons of radioactive waste hazards to other hazards which are routinely accepted, for example, variations in natural background exposure with geographic location. Further, we argue, the ongoing dialogue which DOE maintains with both the NRC and EPA gives assurance that DOE programs are structured in a sufficiently flexible manner to accommodate refinements in the developing regulatory criteria[6,7].

Related to allegations regarding the lack of regulatory standards and criteria is another argument advanced by some participants that the DOE has not demonstrated that it will be able to comply, and indeed in some cases the DOE program is currently at variance with, draft NRC technical criteria published in May, 1980[8]. Notwithstanding the fact these criteria carry the disclaimer that they do not necessarily represent even the NRC staff's tentative position on the issues, much less having been proposed on a formal basis, the UNWMG-EEI has responded to these allegations by pointing out what we believe to be misinterpretations of the draft criteria. As one example, the NRC draft criteria refer to establishing site suitability criteria which would lead to "uninteresting sites of little resource value" in order to lessen the possibility of human intrusion. At least one participant apparently interprets this as eliminating salt as an acceptable host rock[4]. A reading of the discussion accompanying the draft criteria reveals, however, that NRC is concerned with the avoidance of sites which "are sure to attract the developer or explorer." Parenthetically, the UNWMG-EEI believe the problem of human intrusion has been vastly overstated, in that failure mechanisms which conceivably could be caused by intrusion have been and are included in system safety analyses. Further, other scenarios involve small numbers of individuals and are based on unrealistically conservative assumptions regarding loss of institutional control, loss of markings and other records, and absence of radiation detection capability[7].

With respect to site selection, some participants have expressed reservations concerning the ability of earth scientists to adequately anticipate future geologic conditions. For example, one participant alleges that it is impossible "to predict long-term geologic processes," and that in view of the unknown probability of a geologic event, it is impossible to calculate "a reliable risk assessment of the impact of such an event" and that we "cannot begin to rely on engineered barriers or 'conservative assumptions' to overcome the uncertainties[9]." Closely related to these allegations concerning geologic stability are those concerning hydrology. More specifically, some participants have raised questions with respect to the ability of the state-of-the-art to adequately characterize hydrologic conditions, the extent to which groundwater can influence repository performance, and the characteristics of groundwater movement[10,11].

Without going into detail, in our view these doubts result both from a lack of perspective as to the time period over which a high degree of containment is required in relation to the geologic time scale, that is, the three centuries or so during which fission product activity is reduced by roughly three orders of magnitude, as opposed to hundreds of millenia, and misconceptions regarding the very nature of geologic processes, which by and large operate extremely slowly and at rates which are uniform over very long periods of time. In this and other areas germane to prediction of long-term repository performance, we believe that existing so-called "gaps" and "uncertainties" in our current knowledge turn out to be in our inability to forecast with precision repository performance into the distant future, not in our ability to perform those conservative or bounding calculations which can be used to demonstrate safety[7].

Some participants point to data needs concerning radionuclide transport as an obstacle to a current finding of confidence[4,9]. Indeed, in the case of measurement of rock properties, some participants have expressed the view that it is necessary to conduct the site-specific investigations, or <u>in-situ</u> testing needed to design and construct a repository in order to have confidence that radioactive materials can be disposed of safely[4,12]. Herein lies what we believe to be a fundamental misconception regarding the role of <u>in-situ</u> testing. In our view such testing is properly used chiefly to describe near-field phenomena and to determine the rock-mechanics parameters for design and construction of the mined facility itself, not to verify long-term repository performance assessments. Rather, the validity of such assessments rests on knowledge of waste form performance and geohydrology, areas well enough understood today for confidence in ultimate safety[7].

Several comments point to the absence of reliable models for the fundamental sorption process as an obstacle to confidence[9,12,13]. For example, referring to the status of modeling technology in general, one participant claims in its position statement that "(n)ecessary mathematical models are underdeveloped, undeveloped, or impossible to develop [12]." By contrast, the UNWMG-EEI take the position that available models and models undergoing refinement are in fact appropriate for safety evaluations, and are based on well-established scientific principles. They describe phenomena and processes in an appropriate level of detail, and naturally incorporate approximations for phenomena not precisely understood and, indeed, for computational ease and economy. The users of such models are well aware of the capabilities and limitations of such models, and therefore their suitable uses. In our view elegant, highly detailed, highly sophisticated models describing mechanistically every conceivable phenomenon affecting nuclide movement are simply not required for an assessment of disposal safety[6,7].

The foregoing discussion is not complete by any stretch of the imagination. I would like to stop at this point, however, to give you a brief litany of other technical and institutional issues which have been raised, but which time does not permit me to discuss at any length. Such technical issues include:

1. Assertions as to insufficient data on long-term spent fuel storage in water-filled basins, including accidents at such pools and sabotage, and allegations that higher discharge burnup fuel anticipated in future fuel cycle designs will be less suitable for long-term storage[4,5,9,11,14].

2. Allegations that current exploration technology is inadequate for identifying suitable repository sites, including assertions as to disruption of the sites' integrity during the characterization phase[9,10].

3. Claims that knowledge of host rock properties and interaction mechanisms are inadequate to assure mine stability for the emplacement and retrievability periods[9].

4. Assertions that the data base on spent fuel as a waste form is insufficient to allow adequate prediction of release rates, including claims as to the potential for recriticality[4,11,12,15].

5. Claims that the data base on corrosion of potential canister materials and sorptive capacity and thermodynamic properties of potential overpack and backfill materials is inadequate[4,9,12,13].

6. Assertions that technology for adequately sealing boreholes and shafts is not available[4,9,10].

Institutional issues include:

1. Belief that the DOE will be unable to develop and implement an effective waste management program because its internal managerial and organizational structure is inadequate[12,16].

2. Assertions that no reliance can be placed on the constancy of future Administrations and Congresses in their commitment of manpower and funds for the waste program[4,10, 12,17].

3. Doubt that the many Federal agencies with authority over various segments of the waste management program will be able to effectively coordinate their efforts and resolve their differences[12,17].

4. Allegations that the Federal government in general and DOE in particular will be unable to resolve the problem of the role of State and local governments in implementing a repository system[4,9,11,12,16, 18,19].

5. Claims that DOE has seriously underestimated the difficulties involved in achieving public acceptance of a geologic repository[11,19,20].

In the few minutes remaining I would like to inform you as to the current status of the proceeding and the implications of the ultimate Commission finding for nuclear utilities.

On December 10, 1980 the Advisory Committee on Reactor Safeguards issued a report to the NRC in which it concluded that the Commission should have a high degree of confidence that radioactive wastes can be safely disposed, and safely stored in the interim, but cited institutional issues as contributing to uncertainty as to a firm availability date for a geologic repository[21].

On January 29, 1981, pursuant to an earlier Commission order, the special NRC internal working group on the Waste Confidence proceeding filed its report on the record established to date. The purpose of this report was to summarize the record and identify key issues and controversies, and, insofar as possible, indicate how their resolution could affect the Commission's decision. Also, the report recommends areas where, in the opinion of the working group, the record should be supplemented. In this connection the working group identifies these areas in which it believes additional information may be desirable:

1. Historical and projected DOE Program expenditures and manpower commitments, and program plans which detail the relationship between those programs and specific technical problems and the timing of expected solutions,

2. Information on basalt at the Hanford site derived from recent power plant siting investigations,

3. Additional technical studies on DOE's analysis of retrievability[22].

On March 5 the UNWMG-EEI filed its comments on the working group report, pointing out instances of what we believe to be inadequate weighting of the identified major issues based on the evidence submitted by the participants, and restating our position that sufficient factual information has been supplied for the Commission to make an affirmative determination. As of this date, the Commission has not formulated its position as to the conduct of the remainder of the proceeding.

While a negative finding by the Commission in this rulemaking would not necessarily present an insurmountable obstacle to future reactor and spent fuel pool expansion licensing, depending on the reason for the negative finding, the Utility Nuclear Waste Management Group and the Edison Electric Institute believe the proceeding provides a unique forum where public concerns, as well as those of State regulatory agencies and local governments can be somewhat eased, or, alternatively, seriously aggravated. In addition, an affirmative finding on the part of the Commission could create a stronger basis of support for nuclear power on the part of the investment community and the Congress, both traditionally supportive of nuclear development but recently having come under increasing pressure due to a perceived lack of progress in resolving the high level waste disposal question. These, then, are the reasons, most particularly public acceptance, for the high degree of interest and involvement on the part of nuclear utilities in this proceeding, and this interest and involvement will continue until it is brought to a successful conclusion.

REFERENCES

1. Minnesota v. NRC, 602 F. 2d 412 (1979).

2. Storage and Disposal of Nuclear Waste, Notice of Proposed Rulemaking, 44 Fed. Reg. 61, 372 (October 25, 1979).

3. First Prehearing Conference Order (February 1, 1980).

4. Statement of Position of the Natural Resources Defense Council (July 7, 1980).

5. D. Abrahamson, Comments on Proposed Rulemaking on the Storage and Disposal of Nuclear Waste (July 2, 1980), attached to the Statement of Position of the State of Minnesota (July 3, 1980).

6. Statement of Position of the Utility Nuclear Waste Management Group and the Edison Electric Institute-The Capability for Disposing of High-Level Wastes Safely (Document 2 of 4) (July 7, 1980).

7. Cross-Statement of the Utility Nuclear Waste Management Group and the Edison Electric Institute (September 5, 1980).

8. Technical Criteria for Regulating Geologic Disposal of High-Level Radioactive Wastes, Advance Notice of Proposed Rulemaking 10CFR Part 60, 45 Fed. Reg. 31, 393 (May 13, 1980).

9. Statement of Position of Robert Abrams, Attorney General of the State of New York (July 7, 1980).

10. Statement of Position of the California Department of Conservation (July 7, 1980).

11. Statement of Position of the New England Coalition on Nuclear Pollution (July 7, 1980).

12. Statement of Position of the California Energy Commission Nuclear Fuel Committee (July 7, 1980).

13. Statement of Position of Dr. M. G. Mudrey, Wisconsin Geological and Natural History Survey (July 10, 1980).

14. Statement of Position of the State of Delaware (July 3, 1980).

15. Statement of Position of the United States Geological Survey (July 7, 1980).

16. Testimony Submitted by David A. Deese on Behalf of the State of Wisconsin in the Matter of Proposed Rulemaking on the Storage and Disposal of Nuclear Waste (July 8, 1980).

17. Statement of Position of the State of Ohio (undated).

18. Statement of Position of the State of Illinois (July 7, 1980).

19. Testimony Submitted by Dr. John E. Kelly on Behalf of the State of Wisconsin (undated).

20. Statement of Position of Safe Haven, Ltd. (July 7, 1980).

21. Letter dated December 10, 1980 from Milton S. Plesset, Chairman, Advisory Committee on Reactor Safeguards to John F. Ahearne, Chairman, U. S. Nuclear Regulatory Commission.

22. Report of the Working Group on the Proposed Rulemaking on the Storage and Disposal of Nuclear Wastes (January 29, 1981).

METHODS FOR DETERMINING SYSTEM UNCERTAINTY

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ABSTRACT

Several Monte Carlo and numerical analytic techniques have been used to estimate system uncertainty for a moderately difficult hydrology problem. Each technique was constrained to use only about 200 system evaluations because in repository analyses these evaluations are often quite costly. The results show that the Monte Carlo techniques yield reasonable estimates of system uncertainty. The numerical analytic techniques failed because of the severely "spiked" integrands that must be evaluated.

INTRODUCTION

The safe disposal of commercially generated nuclear waste requires a careful assessment of the dangers to present and future generations posed by the disposal site. A key issue in determining the potential risk from disposal is the appropriate evaluation of the effect of the uncertainty. Indeed, a large portion of this conference has been devoted to discussions of the means by which sources of uncertainty can be identified and evaluated. This paper deals with a discussion of the relative merits of methods which can be used to evaluate the uncertainty in overall system performance caused by the uncertainties in component performance.

In the past, system uncertainty studies have been based on a Monte Carlo approach. Either "random" Monte Carlo (Ref. 1) or a form of stratified sampling Monte Carlo (Ref. 2) have been employed. At the ONWI/INTERA Conference on Uncertainty held in Galveston, Texas, the panel on system uncertainty endorsed the use of Monte Carlo techniques in general and Latin Hypercube Monte Carlo in particular for system uncertainty evaluations. However, it was conjectured at that conference that traditional numerical analysis techniques might also be effective in evaluating system uncertainty.

This paper reports some work performed to determine the relative merits of several Monte Carlo and numerical analysis techniques in the evaluation of system uncertainty in a moderately complex hydrologic problem.

The principal cost incurred in evaluating system uncertainty is the expense of repetitively exercising the entire systems model. In a real case, each run of the model is likely to be quite expensive. Consequently, system uncertainty evaluation calls for a method which provides reasonable accuracy with a relatively small number of runs. In this study, even though the "systems model" is a single line of code, only uncertainty evaluation methods requiring approximately 200 system model evaluations were considered. This restriction was arbitrarily imposed in order to provide a fair comparison between the methods employed.

THE PROBLEM

The hydrologic problem for solution is depicted in Fig. 1. Groundwater flows upward from a lower aquifer to a repository through a resistance R_1. Groundwater can leave the repository moving to an upper aquifer either through the geologic formation (resistance R_2) or through a shaft with cross section W, permeability k, porosity e, and length z. The potential difference between aquifers is ΔH and there is no potential difference between the shaft and formation outlets in the upper aquifer.

All of the parameters except shaft length are assumed to be uncertain. Figure 1 contains a description of the uncertainties. Throughout this paper "logarithm" is meant to be base 10. A variable is said to be distributed log-X if the $\log_{10}$ of the variable is distributed according to the probability distribution X.

Application of Darcy's Law shows that t, the flow time in the shaft, is given by

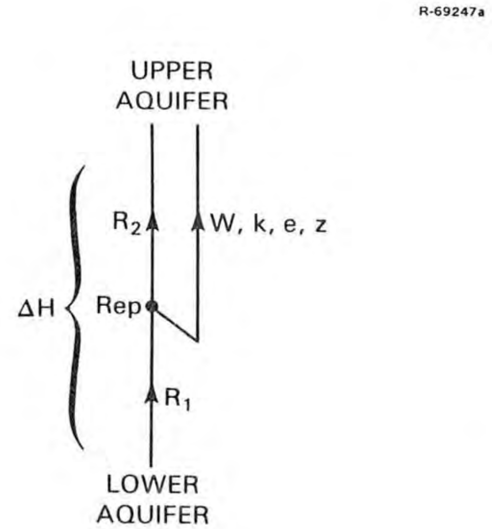
$$t = \frac{zWe}{\Delta H}\left(R_1 + \frac{z}{kW} + \frac{R_1 z}{R_2 kW}\right) \tag{1}$$

The uncertainty in the value of the flow time caused by the parameter uncertainties is to be determined.

METHODS FOR UNCERTAINTY EVALUATION

Monte Carlo Methods

Three different Monte Carlo techniques were used to evaluate system uncertainty. In the Random Monte Carlo (RMC) technique, a value for each input parameter is randomly chosen according to the probability density functions (pdf's) of the parameters. This procedure was repeated in order to form an ensemble of 200 possible problem descriptions with each individual parameter distributed according to its assumed pdf. In the Latin Hypercube (LHC) technique, each parameter's range was divided into 200 equiprobability segments according to the parameter's pdf, and a value of the parameter was randomly chosen in each segment. An ensemble of 200 problem descriptions was then formed by randomly choosing without replacement one of the set of values for each parameter for each description.



VARIABLE	RANGE	DISTRIBUTION
z (m)	400	FIXED
R_1 (yr/m^2)	(0.2, 20)	LOG-NORMAL
R_2 (yr/m^2)	(2, 200)	LOG-NORMAL
ΔH (m)	(8, 12)	UNIFORM
W (m^2)	(20, 80)	NORMAL
k (m/yr)	(3.16×10^{-3}, 31.6)	LOG-UNIFORM
e	(10^{-3}, 10^{-2})	LOG-TRIANGULAR

Fig. 1. The problem for solution.

For these techniques the uncertainty is estimated by performing statistical analysis of the flow times evaluated from the descriptions in the ensemble. Means, variances, and distribution functions can be estimated.

The final Monte Carlo technique used was the Replicated Latin Hypercube (RLHC). In this technique, ten independent twenty-description LHC's are performed. The average values of the ten means and variances for the LHC's are used as estimates of the system mean and variance.

Numerical Analytic Techniques

Several numerical analytic techniques were used to estimate system uncertainty. In each case, a numerical integration was attempted to determine the first two moments of the pdf of flow time. That is, to evaluate six-dimensional integrals of the form

$$\int t^n \, df_{R_1} \, df_{R_2} \, df_k \, df_e \, df_w \, df_{\Delta H} \tag{2}$$

where n is 1 or 2, df_X denotes the pdf of variable X, and (1) is used for t in the integrand.

Two general types of integration techniques were used to evaluate the integrals. In Gaussian Quadrature techniques (Ref. 3) the integral is approximated by a sum of the form

$$\int f(\underline{x}) \, d\underline{x} \cong \sum_{n=1}^{N} c_n \, f(\underline{x}_n) \tag{3}$$

where the constants c_n and the points $\underline{x}_n$ are chosen so that any polynomial up to a specified degree can be exactly integrated. The hope is that if the integrand looks roughly like a polynomial of appropriate degree, (3) will approximate the value of the integral. In this study methods based on polynomials up to degree 7 in six variables were attempted. Higher degree formulas exist, but the limitation to approximately 200 function evaluations ruled out their use.

The other numerical integration methods used were examples of techniques based on the theory of uniformly distributed sequences (Ref. 4). In these methods recent results in number theory are used to choose sequences of points $\underline{x}_n$ such that

$$\int f(\underline{x}) \, d_{\underline{x}} \cong \frac{1}{N} \sum_{n=1}^{N} f(\underline{x}_n) \tag{4}$$

These techniques have been used successfully in the evaluation of multi-dimensional integrals. In this study, sequences with N approximately 200 were used.

FAILURE OF THE NUMERICAL ANALYTIC TECHNIQUES

After using several integration techniques of each type, none of the approximations to (2) gave results which were physically reasonable. The difficulty is the extreme "bumpiness" of the six-dimensional integrand. Figure 2 indicates the severity of the problem. It shows a plot of the variation of the integrand of (2) with the shaft permeability, k, when the other variables are held constant, and n=1. The extreme slope of the integrand is too sharp for adequate integration in six dimensions using only approximately 200 points.

Spikes also occur in the variation of the integrand with some of the other variables. Consequently, the integration surface is far too complex for either integration method given a few hundred point limitation.

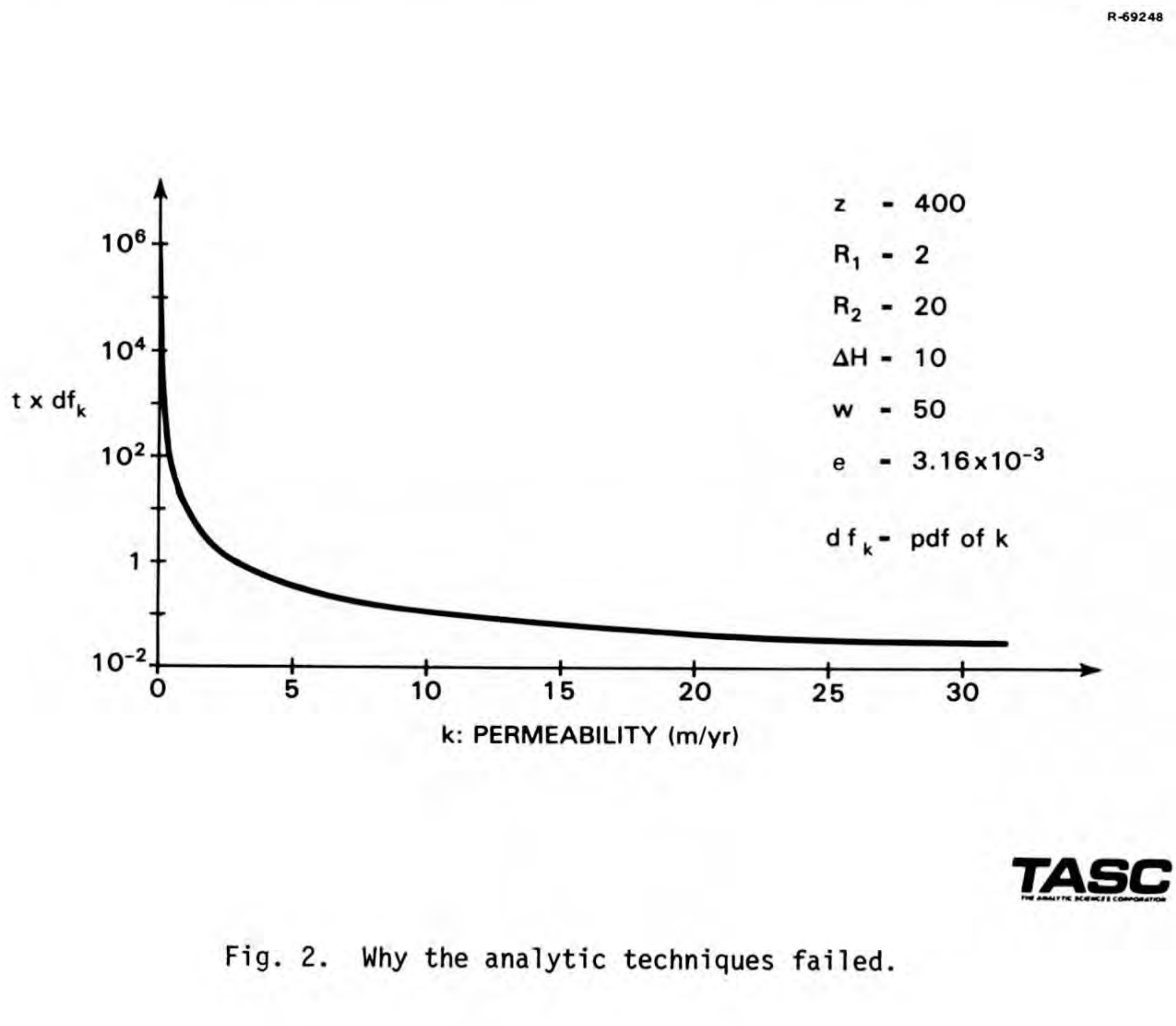


Fig. 2. Why the analytic techniques failed.

RESULTS OF THE MONTE CARLO METHODS

Means and Variances

The three Monte Carlo techniques all gave values for the mean and variance of the flow time which were in good agreement with each other and with physical intuition. These methods were further tested by repeating each fifty times in order to determine the expected scatter in results for the mean and variance of the uncertainty. Table 1 contains the mean value of the 50 means and variances calculated for log t for each technique as well as the standard error in the means and variances. These results show that with 200 samples all of the techniques provide roughly equal estimates of the system uncertainty. Moreover the results are repeatable from analysis to analysis.

Table 1. Means and Standard Errors of 50 Repetitions of Estimators of the Expectation of Flow Time, E(t), and the Variance of Flow Time, Var(t)

	Evaluation Technique		
Statistics of Estimators	RMC	LHC	RLHC
Mean of E(t)	3.426	3.438	3.439
Std. error of E(t)	0.071	0.026	0.055
Mean of Var(t)	1.112	1.125	1.127
Std. error of Var(t)	0.066	0.024	0.024

Determination of the Cumulative Distribution Function

Figure 3 shows the cumulative distribution function (cdf) for log t based on the fifty RMC analyses. The solid curve is the best estimate. The dashed curves are 95% confidence bounds on the cdf. This information allows approximate determination of the probability of exceeding any given value of log t.

An alternative approach to event probability estimation is to treat the Monte Carlos as a set of repeated experiments designed to estimate event probability. For example, suppose it is necessary to estimate the probability that shaft flow time is less than twenty years. In the 10,000 (50×200) RMC runs, 19 outcomes with $t< 20$ were observed. Using elementary statistics to estimate p, the probability of $t< 20$, we find that p is almost certainly less than 0.0034.

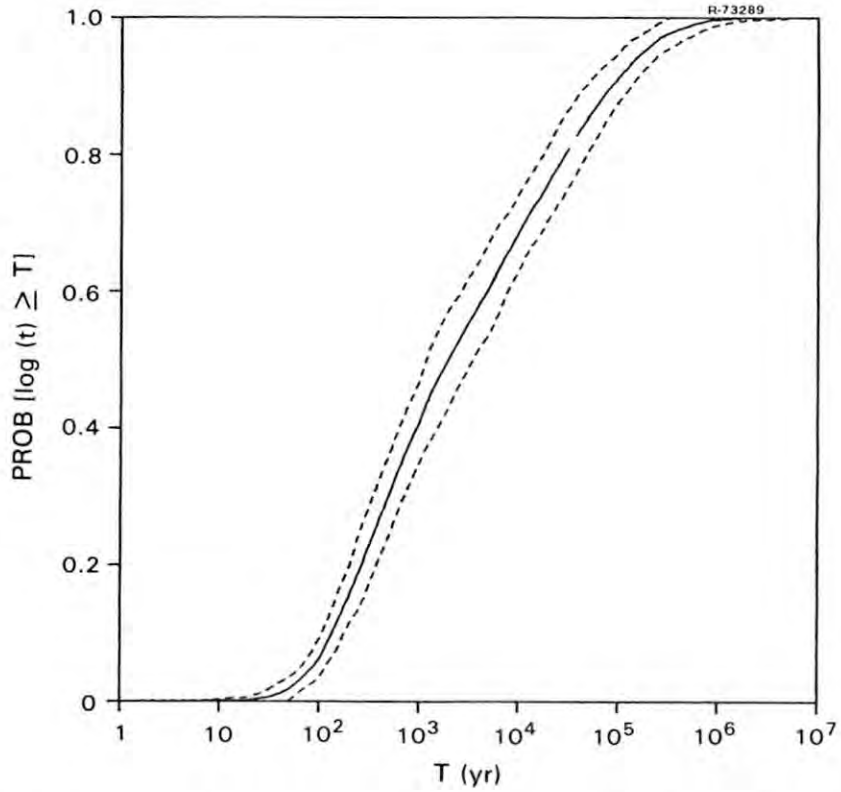


Fig. 3. The cumulative distribution function for the logarithm of flow time.

CONCLUSIONS

The results of this study lead to the following conjectures about the evaluation of system uncertainty where the number of system evaluations is limited to a few hundred.

- Monte Carlo techniques can accurately estimate system uncertainties for moderately complex problems

- Numerical integration techniques fail to accurately estimate system uncertainty.

The principal advantage of the Monte Carlo techniques is that the parameter pdf's are used only to sample the distribution. That is an easy task. However, for the numerical integration techniques one must integrate these pdf's which are often "spikes" for distributions describing geologic variables. This is significantly less easy.

REFERENCES

1. M.S. Giuffre, M.F. Kaplan, D.A. Ensminger, S.G. Oston, and J.Y. Nalbandian, Bedded-Salt Repository Analysis: Final Report, UCRL-15236 (March, 1980).

2. R.L. Iman, W.J. Conover and J.E. Campbell, Risk Methodology for Geologic Disposal of Radioactive Waste: Small Sample Sensitivity Analysis Techniques for Computer Models, With an Application to Risk Assessment, NUREG/CR-1397, (March, 1980).

3. A. Stroud, Approximate Calculation of Multiple Integrals, Prentice-Hall, 1971.

4. L. Kuipers and H. Neiderreiter, Uniform Distribution of Sequences, Wiley, New York, 1974.

ASSESSING MODEL UNCERTAINTIES

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ABSTRACT

We have examined, by Monte Carlo methods, the uncertainties of a variety of models of different mathematical forms, including an atmospheric carbon dioxide model, a marsh hydrology model, a model of plutonium movement in a forested watershed, and a model of food chain transport of iodine. When the uncertainties affecting the predictions of these models are partitioned into the sources of error we find that: (1) the relative contribution of a parameter to model uncertainty may not be reflected by sensitivity analysis; (2) the mathematical formulation of the model is critical with simpler models often having lower uncertainties; (3) deterministic solutions often give biased predictions, especially when stochastic effects are present; and (4) assumptions regarding statistical frequency distributions are often unimportant.

Results indicate that little information may be needed to reflect the error propagation properties of a model, and identify the critical portions of the model (e.g., parameters and/or mathematical structure) and make reasonable estimates of uncertainties associated with predictions.

ERROR ANALYSIS

Introduction

Error analysis is the systematic determination of uncertainties in model predictions. The determination of the causes and extent of model uncertainties encompasses aspects of model development, analysis, data collection and synthesis, and model simulation and prediction.

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Our studies on the phenomena associated with uncertainties have considered a broad range of ecological and environmental models. The results provide guidelines that permit the design of experiments and models which reduce prediction error. These results have direct application to the determination of uncertainties associated with the environmental transport, fate, and subsequent health effects of high-level radioactive wastes.

Background

Since our early explorations [1,2] and development of an unpublished document by Overton [3], error analysis in ecosystem models has become a topic of considerable interest. This interest was a natural development of work on sensitivity analysis (for example, see refs. 4-6) and stochastic modeling (see ref. 7 for a review of this field). Over a dozen studies have appeared since 1977 which are directly or indirectly related to error analysis.

Similar interests have been shown in other fields. For instance, climate models (see Frankignoul [8] for a recent paper in this field), atmospheric transport models [9-11], hydrologic models (e.g., refs 12-16), and problems of spatial variability [17,18].

One of the important results from stochastic simulation has been the application of experimental design to Monte Carlo analysis. By using statistical sampling designs (e.g., Latin Hypercube, refs. 19,20), the efficiency of Monte Carlo simulations can sometimes be increased significantly. Steinhorst [21] and Steinhorst et al. [22] have also applied experimental design to "validation" and sensitivity studies [23].

The objectives of each analysis determine the methods applied. For the majority of our studies, we have adopted a Monte Carlo methodology. For each Monte Carlo study the model parameters are described by statistical distributions (e.g., means and variances), a specific set of parameters is chosen randomly from these distributions, and a model simulation is produced. This process is repeated until the statistical properties of the model output can be determined. We have recently applied this process to radiological assessment models which were developed to provide a deterministic solution to a stochastic process [24,25]. The need to address the effect of parameter variability in radiological assessment models has been addressed by the International Commission on Radiological Protection [26], and our methods have been applied to evaluate the potential short-comings of their recommended prodedures for analyzing predictive variability or imprecision as a consequence of input variability [27].

Sources of Uncertainty

The uncertainties associated with model predictions can be summarized under three categories [28]: (1) uncertainties resulting from the structure of the model; (2) uncertainties due to natural variability of the system being modeled; and (3) uncertainties associated with the estimation of model parameters.

Effects of Model Structure

Uncertainty associated with model predictions are partly a function of the mathematical formulation chosen to represent processes in the system. Deviations of model predictions from real-world values can only be detected through experimental testing or validation of the model. However, error propagation techniques may be used to estimate the imprecision or variability associated with model predictions. In addition to predictive bias, model structure can also influence the inherent ability of the model to propagate errors due to parameter variability.

Six nonlinear models were calibrated to the same data and their prediction uncertainty compared [29]. Differences in mathematical formulations caused major differences in prediction uncertainty among the models. Parameters associated with loss terms affect primarily one state variable and the simpler the term the less the error. However, terms describing relationships between state variables can play a dominant role in the propagation of total model errors. Complex terms describing these interactions have proven to be more accurate and less uncertain than simpler terms [30].

Assumptions frequently applied in ecosystem modeling can lead to serious errors. This phenomenon is particularly evident when the modeler attempts to explain the behavior of the system from the known behavior of the system components [31]. For example, even if each individual in a population responds discontinuously to temperature, the total population may still respond in a smooth continuous curve because there is genetic variability among individuals.

In radiological assessment, consideration of individual variability will be of concern when determining compliance with dose limits for individuals, but prediction of average doses to the exposed population is more relevant for the assessment of total health effects. Therefore, when translating health effects from collective population doses, an over-estimate of predictive uncertainty may be expected if this estimate is based on uncertainty among individuals rather than errors in the population average. Less error is expected in population averages than among individuals.

Thus, uncertainties in the final predictions are often determined by the model structure and the questions to be addressed by the model. Estimation bias (deviation of predictions from some desired behavior) may be a minor component in determining the uncertainties of model predictions [29], yet most models are developed and parameterized to minimize estimation bias.

Effects of Natural Variability

Ignoring natural system variability in a deterministic model always results in prediction error. Current models used for radiological assessment are deterministic and do not explicitly account for natural variability [32]. The prediction of a deterministic model, using the mean values of each parameter, is not equal to mean behavior of the system when the parameters are allowed to vary around their means [33]. The systematic bias introduced can lead to serious error in some circumstances [34].

Natural environmental constraints place boundaries on model error. Natural processes, such as seasonal changes in light and temperature, place upper and lower boundaries on possible systems behavior. Models which consider these constraints also show bounds on their error terms. In an aquatic model [35,36], error was bounded because of seasonal temperatures well above and below the biological optima. For this reason, the variability of parameters of radiological assessment models were bounded by specification of maximum and minimum possible values [37,25].

These results have a direct effect on the design of field validation experiments. Discrimination between models will be most effective when data is collected during periods when conditions are optimal and system components are changing most rapidly [36].

Effects of Parameter Uncertainty

Parameter means and variances must be estimated with specific model objectives in mind. In our study of the Marsh model [36], variance on one parameter, W, was estimated a priori from knowledge about how this parameter varies across marshes. Although this parameter was relatively unimportant to model mechansisms, the variance on W dominated total prediction uncertainty. When its variance was reduced to the variability that would be expected within any specific marsh, its importance dropped to reasonable levels.

Another source of uncertainty is the derivation of parameter values from literature data. Data in the literature represents examples of judgment and convenience and may not be representative of a given situation for which a prediction is performed. This is especially true for parameters used for biotransport, dosimetry, and health effects in radiological assessment models. Frequently, parameter values developed from the global literature are applied in models used to predict the fate of radionuclides in specific locations [32]. One can only hope in this case that the results of error anlaysis will produce a confidence interval that includes the actual range of values representative of the conditions prevailing at the location in question.

Simultaneous measurement of parameters can be as important as an increase in accuracy of parameter measurements. Correlations obtained from simultaneous measurements can be included in the analysis with a reduction in prediction uncertainty which is often equivalent to reducing the variance on individual parameters [35,38].

Sensitivity analysis is the most common approach used to decide which parameters of a model should be measured most accurately. However, the assumptions of the sensitivity approach are seriously violated in many ecosystem studies leading to serious errors in the results. In our analysis of the marsh model [36] we were able to show that sensitivity analysis directed the researchers attention to the wrong parameters.

Statistical distribution of parameters can have an important influence on experimental design. Some parameters are insensitive to the distribution from which they are drawn, particularly if their variances are small. In this case, a triangular distribution adequately represents their contribution to prediction uncertainty. The model requires only an estimate of the mean and the upper and lower limits which can often be supplied from a small sample size. In another case, a simple radiological assessment model with known lognormally distributed parameters was analyzed by assigning a uniform distribution to each parameter. The range of these uniform distributions was determined by minimum and maximum observed values. The results surprisingly increased the predicted mean by less than a factor of two. The predicted 99th percentile increased by less than a factor of three from predictions made using the previously determined lognormal distributions (Schwarz and Hoffman, unpublished).

Discussion

Models are proving to be a valuable tool for assessing the impact of new technologies. Their ability to combine data, fact, and theory to produce a coherent synthesis provides the analyst with critical information for making decisions. Therefore, it is imperative that techniques for realistic, quantitative assessment of model predictions be available.

The application of error analysis to a variety of models has demonstrated the usefulness of this method for determining the uncertainties associated with model predictions. The identification of the relative sources of uncertainty for a variety of models has clearly shown that uncertainties related to model behavior are often unimportant compared to uncertainties generated by variability of the system being modeled. Determining the nature and extent of uncertainty from all sources is an important aspect of ensuring reliable system modeling. As error analysis techniques are applied to more models it will become possible to combine the practical with the theoretical to ensure that all sources of variability are known and uncertainties associated with model predictions minimized.

ACKNOWLEDGMENTS

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REFERENCES

1. R. V. O'Neill, "Error Analysis in Ecological Models," EDFB Memo Report 71-15. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 31 pp. (1971a).

2. R. V. O'Neill, "A Stochastic Model of Energy Flow in Predator Compartments of an Ecosystem," pp. 107-121, in G. Patil, E. C. Pielou and W. E. Waters (eds.), Statistical Ecology, Vol. 3. Penn State University Press, University Park, Pennsylvania (1971b).

3. W. S. Overton, Modelling Notebook: Sensitivity Analysis as "Propagation of Error" and Model Validation. Oregon State University. 13 pp. (1971)

4. M. Brylinsky, "Steady-State Sensitivity Analysis of Energy Flow in a Marine Ecosystem," pp. 81-101, in B.C. Patten (ed.), Systems Analysis and Simulation in Ecology. Vol. II. Academic Press, New York (1972).

5. R. B. Williams, "Steady State Equilibriums in Simple Nonlinear Food Webs," pp. 213-240, in B. C. Patten (ed.), Systems Analysis and Simulation in Ecology, II. Academic Press, New York (1972).

6. J. B. Waide, J. E. Krebs, S. P. Clarkson, and E. M. Setzler, "A Linear Systems Analysis of the Calcium Cycle in a Forested Watershed Ecosystem," pp. 261-345, in Progress in Theoretical Biology, Vol. 3. Academic Press, New York (1974).

7. R. V. O'Neill, "A Review of Stochastic Modeling in Ecology," J. Int. Soc. Ecol. Model. 1: 10-29 (1979c).

8. C. Frankignoul, "On Noise Level of Climatic Models," J. Atmos. Sci. 34: 1827-1831 (1977).

9. R. L. Bohac, W. R. Derrick and J. B. Sosebee, "Sensitivity of the Gaussian plume model," Atmos. Environ. 8: 291-283 1974.

10. Weber, A.H. Atmospheric Dispersion Parameters in Gaussian Plum Modeling, Part I, Review of Current Systems and Possible Future Developments. EPA-600/4-76-030a. U.S. Environmental Protection Agency, Washington, D.C. (1976).

11. M. G. Morgan, S. C. Morris, A. K. Meier and D. L. Shenk, "A Probabilistic Methodology for Estimating Air Pollution Health Effects from Coal Fired Power Plants," Energy Systems and Policy 2: 287-310 (1978).

12. R. P. Ibbitt, "Uncertainties in Deterministic Models," pp. 86-91, in Proc., International Symposium on Uncertainties in Hydrology and Water Resources., Vol. III (1972).

13. C. A. Cornell, "First Order Analysis of Model and Parameter Uncertainty," pp. 1245-1271, in Proc., International Symposium on Uncertainties Hydrology, Vol. III, Water Resour. Syst. (1972).

14. Y. Onishi, Mathematical Simulation of Sediment and Radionuclide Transport in the Columbia River. BNWL-228. Battelle-Northwest Laboratories, Richland, Washington. 89 pp. (1977a)

15. Y. Onishi, Finite Element Models for Sediment and Contaminant Transport in Surface Waters - Transport of Sediments and Radionuclides in the Clinch River. BNWL-227. Battelle-Northwest Laboratories, Richland, Washington. 103 pp. (1977b)

16. E. F. Wood, "An Analysis of the Effects of Parameter Uncertainty in Deterministic Hydrologic Models," Water Resour. Res. 12: 925-932 (1976).

17. H. Fluher, M. S. Aredakani, and L. H. Stolzy, "Error Propagation in Deterministic Hydraulic Conductivities from Successive Water Content and Pressure Head Profiles," Soil Sci. Soc. Am. J. 40: 830-836 (1976).

18. O. L. Smith and R. A. Freeze, "Stochastic Analysis of Steady State Groundwater Flow in a Bounded Domain: I. One-Dimensional Simulations," Water Resour. Res. 15: 521-528 (1979).

19. R. L. Inman and W. J. Conover, "Small Sample Sensitivity Analysis Techniques for Computer Models, with an Application to risk Assessment," Commun. Statist. Theor. Meth. A9(17), 1749-1842 (1980).

20. M. D. McKay, W. H. Conover and D. E. Whiteman, "Report on the Application of Statistical Techniques to the Analysis of Computer Codes," LA-6479-MS. Los Alamos Scientific Laboratory, Los Alamos, New Mexico. 15 pp. (1976)

21. R. K. Steinhorst, "Validation of Deterministic System Simulation models," ASA Proceed. STAT. Comp. 24: 283-291 (1976).

22. R. K. Steinhorst, H. W. Hunt, G. S. Innis and K. P. Haydock, "Sensitivity Analysis of the ELM Model," pp. 231-255, in G. S. Innis (ed.), Grassland Simulation Model. Springer-Verlag, New York (1978).

23. G. S. Innis, "Role of total systems models in the Grassland Biome Study," pp., 14-47, in B.C. Patten (ed.), Systems Analysis and Simulation in Ecology, Vol. III, Academic Press, New York. 601 pp. (1975).

24. R. V. O'Neill, R. H. Gardner, F. O. Hoffman and G. Schwarz, "Parameter Uncertainty and Estimated Radiological Dose to Man from Atmospheric 131-I Releases: A Monte Carlo Approach," Health Physics 40: 760-764 (1981).

25. G. Schwarz and F. O. Hoffman, "Imprecision of Dose Predictions for Radionuclides Released to the Environment: an Application of a Monte Carlo-Simulation-Technique," Environment International (1981 in press).

26. International Committee on Radiological Protection, "Radionuclide Release into the Environment: Assessment of Doses to Man," Annals of the ICRP 2(2) 75 pp. Pergamon Press, Oxford (1979).

27. F. O. Hoffman, G. Schwarz and G. G. Killough, "A Critique of the Use of ICRP-29's 'Robust Index' in Evaluating Uncertainties Associated with Radiological Assessment Models," Health Physics 39:367-369 (1980).

28. R. H. Gardner and R. V. O'Neill, "Parameter Uncertainty and Model Predictions: A Review of Monte Carlo Results," in B. Beck (ed.), Uncertainty Analysis of Aquatic Ecosystem Models. International Institute of Applied Systems Analysis, Vienna (in press).

29. R. H. Gardner, R. V. O'Neill, J. B. Mankin and D. Kumar, "Comparative error analysis of six predator-prey models," Ecology 61: 323-332 (1980).

30. R. H. Gardner, J. B. Mankin and W. R. Emanuel, "A Comparison of Three Carbon Models," Ecol. Model. 8: 313-332 (1980).

31. R. V. O'Neill, "Transmutations Across Hierarchical Levels," pp. 59-78, in G. G. Innis and R. V. O'Neill (eds.), Systems Analysis of Ecosystems. International Cooperative Publishing House, Fairland, Maryland (1979d).

32. F. O. Hoffman, D. L. Shaeffer, C. W. Miller and C. T. Garten, Jr. (eds.), Proceedings of a Workshop on the Evaluation of Models used for the Environmental Assessment of Radionuclide Releases, CONF-770901, 131 pp. Oak Ridge National Laboratory, Oak Ridge, Tennessee (1978).

33. R. V. O'Neill and R. H. Gardner, "Sources of Uncertainty in Ecological Models," pp. 447-463, in B. P. Zeigler (ed.), Simulation Models and Methodology. North Holland Press (1979).

34. R. V. O'Neill, "Natural Variability as a Source of Error in Ecological Models," pp. 23-32, in G. S. Innis and R. V. O'Neill (eds.), Systems Analysis of Ecosystems. International Cooperative Publishing House, Fairland, Maryland (1979b).

35. R. V. O'Neill, R. H. Gardner and J. B. Mankin, "Analysis of parameter error in a nonlinear model," Ecol. Model. 8: 297-311 (1980).

36. R. H. Gardner, D. D. Huff, R. V. O'Neill, J. B. Mankin, J. H. Carney and J. Jones, "Application of Error Analysis to a Marsh Hydrology Model," Water Resour. Res. 16: 659-664 (1980).

37. R. V. O'Neill, R. H. Gardner, W. W. Christensen, W. Van Winkle, J. H. Carney and J. B. Mankin, "Effects of Parameter Uncertainty in Density-Independent and Density-Dependent Leslie Models for Fish Populations," Can. J. Aquat. Sci. 38: 91-100 (1981).

38. G. Schwarz and D. E. Dunning, Jr., "Impression in Estimates of Dose from Ingested ^{137}Cs Due to Variability in Human Biological Characteristics," Health Physics (1981, in press).

MODELS AND CRITERIA FOR WASTE REPOSITORY PERFORMANCE

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ABSTRACT

A primary objective of the Waste Management Program is to assure that public health is protected. Predictive modeling, to some extent, will play a role in assuring that this objective is met. This paper considers the requirements and limitations of predictive modeling in providing useful inputs to waste management decision making. Criteria development needs and the relation between criteria and models are also discussed.

INTRODUCTION

Calculational models for predicting the consequences of waste management activities should be consistent with applicable criteria; however, definitive criteria for judging the effectiveness of waste management activities have not, as yet, been established. Despite the absence of official criteria, there has been a considerable amount of work on model development. Aside from development of definitive criteria, there are several other areas that should be defined before beginning model development. Model development requires consideration of what is to be predicted, how accurate it must be, and how the results can be validated.

Over three hundred existing models applicable to the management of radioactive waste have been identified [1]. Yet, an extensive degree of effort in model development continues. Given this situation, one might logically ask the questions: Why isn't what we have good enough?; What are the specific gaps that need to be filled?; and How can we recognize an acceptable model should one evolve? These questions should be resolved before more extensive model development takes place. Failure to do so could result in an endless quest for some unattainable state of perfection.

This paper includes a discussion of modeling approaches that are available; consideration of the limitations to modeling; and some comments on model complexity. In light of these limitations, the

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subject of uncertainty is discussed. The "worst case" approach is considered, and, finally, some concluding comments on criteria are offered.

MODELING APPROACHES

There are several major approaches to the modeling o environmental transport. These include:

- Physical (Analytical)
- Empirical (Analog)
- Statistical (Probabilistic)
- Combinations

Physical models are determined by an analytical or theoretical description of the process being modeled based on first principles. Due to the infinite complexity of real world processes, physical models can never be exact predictive tools.

Empirical models are based on experimental data. They consider the nature of conditions and events, as well as the observed consequences. These are related empirically, thereby eliminating the need for a detailed understanding of the underlying theoretical basis. They represent an acceptance of the inability to thoroughly define the process while making the best use of the data that we do have.

Statistical models use probabilistic inference to characterize the process. In some cases this is because the underlying theory is stochastic in nature, and in other cases it is a reflection of the uncertainty about input parameters or the detailed underlying physics. In most cases, models tend to be based upon a combination of these approaches.

MODEL LIMITATIONS

There are several well known limitations to theoretical modeling. Data limitations exist where the data required to support the model is unavailable. The data may be either unknown or, in some cases, unknowable. Whereas it is extrememly difficult to model the unknown, it is impossible to model the unknowable. Although models requiring unattainable data may be technically and mathematically elegant, from a practical standpoint, they are essentially useless.

Computational limitations also occur when the implementation of a conceptual model requires a greater level of computer capability than is available. Developmental limitations are a result of the fact that model development activities do not generally have unlimited resources to work with. Frequently time, personnel, economics, or other factors limit the desirability or capability for model development.

Finally, there are always differences betwen conceptual models and reality, and this is a fundamental limitation. Unfortunately, or fortunately, depending on your point of view, the world is not a uniform, infinite, homogeneous slab. The very nature of modeling assumes a predictable universe, yet the world, particularly the geologic part of it, is full of surprises. Increased model complexity attempts to account for all possibilities and to give the impression that all conceivable factors have been properly considered. Obviously, however, no degree of complexity can compensate for unanticipated events or unknown parameters. Nonetheless, model predictions can be of use in evaluating the sensitivity of outcomes to various input parameters and for providing a general perception and perspective on the nature of the phenomena.

There appears to be a tendency to equate model complexity and detail with credibility of results. In fact, models span a very wide spectrum in terms of complexity from the simplest excercise carried out in a back of the envelope calculation, to a conceptual model in which each atom of a system is described by it own series of differential equations. The fact is that, especially in light of the limitations to models, a more complex model does not necessarily yield more valid results. As Bernie Cohen [2] put it "Improvements in the handling of the complexities we do understand in no way compensates for the omission of even a single one of the many complexities we don't know how to handle."

Taking it one step further, an inappropriate model yields inappropriate results - or in the computer lingo, garbage in - garbage out (GIGO). This goes back to the difference between models and reality. An example of this is seen in the standard man calculational models used for determining MPC's. When the value for the nuclide Nd-144 is converted into mass units, the result is an MPC of 59 Kg per liter. Similar absurd results are found for other very long lived nuclides [3].

The quest for perfection in modeling may, in fact, be counterproductive, except perhaps for continued developmental funding. By putting more and more effort into uncertainty resolution, you eventually run into the law of diminishing returns. By pushing it further, you may create more uncertainty than you resolve. Since there will always be residual uncertainty, the quest for perfection will be an endless one.

UNCERTAINTY

A commonly heard goal is the resolution of remaining uncertainty. However, uncertainty exists - and it always will. The important question is "How certain do we need to be?" and not how can we resolve or eliminate uncertainty. Further, it is the magnitude of the uncertainty, rather than its ratio to the result that is important.

For example, a 50 rem dose with a factor of 10 uncertainty should be of much greater concern than a 10^{-8} rem dose with a factor of 1000 uncertainty. A recent study by SAI [4] gave results for the long term consequences of a HLW repository on the order of these latter values.

THE "WORST CASE" APPROACH

Considering the "worst case" approach, there are several major problems with this method of analysis.

First, The results are dependent largely on the imagination of the analyst. Given a worst case analysis, it is not very difficult to top it. Since probability is not considered in "worst-case" analyses, the unlikelihood of events is not a constraint in scenario development. One need only to stretch one's imagination, therefore, to determine events and consequences of greater and greater severity. Second, in the public's eye, the results of such analyses are assumed to reflect reality. An example of this is seen in Brookhaven's WASH-740 analysis which assumed a 50% release of reactor core material. This grossly conservative assumption was then subjected to exquisite detail in terms of consequence analysis. The WASH-740 results have been widely misused in arguments against nuclear power.

To counter these problems, probabilistic analysis may be used. An example of this approach is seen in the Reactor Safety Study. Such analyses clearly also have their own credibility problems. Nevertheless, for technological purposes, the probabilistic approach is preferable to the worst case approach. However, in order to be meaningful, the probabilistic approach requires probabilistic criteria against which to judge such results.

CRITERIA

In terms of criteria, it is important to determine "How good is good enough" or "How certain do we need to be?" Without this information an endless search for absolute certainty may be initiated.

Criteria should be the guiding force behind model development. Unfortunately, it seems to work the other way around.

Criteria should be logical and consistent both internally and externally. For example radiological criteria should be consistent with criteria for nonradiological activities, assuming the goal is the optimal protection of public health and safety.

Vague criteria are not useful. Statements like "As many passive barriers as feasible should be used to preclude radionuclide movement from the facility, taking into account social and technical considerations" [5] are not sufficiently defined to be capable of implementation.

Finally, the probabilistic approach to analyzing waste systems has merit, but there is a need to develop probabilistic criteria against which to judge the results.

SUMMARY AND CONCLUSIONS

Predictive modeling can be a useful tool for assessing the consequences of underground burial of radioactive waste. However, in applying this tool, certain constraints and limitations should be kept in mind. These are:

- Without definitive criteria for acceptability of radwaste management, it will be difficult if not impossible to determine the form and required outputs of the predictive model.

- Treatment of "uncertainties" should not be attempted without first defining "How certain do we need to be?"

- Acceptable levels of uncertainty should be based on a consideration of the magnitude of the difference between results and criteria or standards rather than the ratio between them.

- Basic limitations of the modeling process should be recognized and dealt with. These include:

  - Unknown or unknowable data inputs.
  - Constraints of time, money, and computer capacity.
  - Differences between the conceptual model and the reality it is intended to represent.

- Model complexity, per se, does not necessarily contribute toward validity of results.

REFERENCES

1. J. E. Mosier, et al., Low Level Waste Management: A Compilation of Models and Monitoring Techniques, ORNL/SUB-79/13617/2, April 1980.

2. B. L. Cohen, "A Super Analog Computer for Evaluating the Safety of Buried Radioactive Waste," Waste Management '80, 1980.

3. J. J. Cohen, C. F. Smith, and A. J. Toy, "The Hazard of Long-Lived Radionuclides," Health Physics Society Annual Meeting, Seattle, Washington, July 1980.

4. R. R. Fullwood, Z. T. Mendoza, and C. A. Stevens, Working Paper for a Probabilistic Radiological Safety Evaluation of a Nuclear Waste Repository, SAI/SR-220-79-PA, April 1980.

5. U.S. NRC, 10 CFR Part 61: Disposal of Low-Level Radioactive Waste and Low-Activity Bulk Solid Waste, Draft for public comment, November 1979.

VALIDATION OF PREDICTIVE MODELS FOR GEOLOGIC DISPOSAL OF RADIOACTIVE WASTE VIA NATURAL ANALOGS*

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ABSTRACT

The incorporation of toxic or hazardous material in the earth's crust is a phenomenon not unique to radioactive waste burial. Useful insights on the environmental transport and effects of underground toxic or radioactive material can be derived from comparative analysis against natural (mineral) analogs.

This paper includes a discussion of the background and rationale for the analog approach, a description of several variations of the approach, and some sample applications to illustrate the concept, focusing on Radium-226 and Iodine-129 as specific case studies.

BACKGROUND

Predictive modeling based largely upon theoretical principles has found wide application in determining potential consequences of underground burial of radioactive wastes. This approach, however, has definite limitations [1]. The analog approach can provide another assessment tool, which may be applied in parallel with theoretical modeling to gain additional insight.

The basis for the approach is field observation and data resulting from study of the movement and biological effects of underground toxic materials. Obviously, field work may be guided by theoretical considerations, and observed data may be useful in revision or verification of theory. In other words, insights useful in development and validation of theory can be drawn from natural analog investigations.

The concept of toxic or hazardous material being incorporated within the earth's crust is certainly not unique to radioactive waste

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burial. Toxic minerals and various forms of hazardous waste have resided underground for long periods of time, and resultant effects have been observed and documented [2]. The objective of the analog approach is to study and derive useful insights from these phenomena. Insights derived from such observation might also provide a more credible basis for justification of waste management programs than those derived from purely theoretical calculations.

RATIONALE

The rationale behind the natural analog approach is based on the observation that all mineral constituents of the human body had their origin in the earth. It has also been shown that bioconcentrations and geoconcentrations of mineral substances are generally related. Extreme imbalance in this relationship has frequently been manifested in disease states. For example, Itai-Itai disease is caused by excess local cadmium concentrations. Similarly, selenium poisoning has been observed in areas where excessive soil concentrations of this element occur. The opposite effect, in the form of deficiency disease, has also been noted. For example, goiter is a condition resulting from iodine deficiency in soils. Perhaps the most notable example of such phenomena was the discovery of the low incidence of tooth decay in areas of high fluoride concentration. These are among several examples indicating that a relationship exists between geologic and biologic levels for many elements.

With this in mind, the analog approach can be used to evaluate the potential effects of human activities by asking the question "how have we perturbed the natural order of things by underground burial of radioactive waste?" Although the approach cannot be used to predict maximum effects on individuals, it can be useful to assess average effects on populations.

GEOTOXICITY

One application of the analog approach is what we have termed Geotoxicity Study, which is the characterization and assessment of the harmful effects of hazardous material incorporated in the earth's crust by either man or nature.

For example, consider the overall potential toxicity of the earth's crust as a whole. Some of the major contributors to this toxicity include such elements as arsenic, barium, chromium, lead, mercury, selenium, and cadmium. Using simple toxicity indices (volume of water required to dilute the inventory of material to acceptable drinking water standards, or MPC's), if 1000 year old waste from a thousand reactors operating for 100 years (10^8 MWe-Yr) were buried underground, it can be calculated that the net toxicity of the earth's crust would increase by only one ten millionth of one percent. Although such comparison is crude, it is nevertheless an interesting example based on large scale perspective. However, a criticism of the example might be that the waste would be concentrated in a few repositories, whereas natural toxic materials are spread out homogeneously.

However, this latter observation is not entirely valid either. Nature has not distributed toxic minerals uniformly. These materials have generally been concentrated in the form of mineral deposits or ore bodies. Therefore, there may be validity in comparing waste repositories and ore deposits as analogous situations. As an example, Tonneson and Cohen [3] have shown that application of this approach to high-level radioactive waste repositories indicates that such repositories become relatively less toxic than Mercury ore deposits within a few centuries, and in about 1000 years become less toxic than typical Uranium ore deposits from which nuclear fuel is originally extracted. Comparisons based solely on toxicity or dilution volume indices are, of course, limited since they do not consider the potential for distribution of these materials in the biosphere or their assimilation by humans.

THE GEOTOXICITY HAZARD INDEX

To accomodate the other parameters known to influence hazard, the geotoxicity hazard index was developed [4]. This index includes terms for toxicity, persistence or time effects, availability or transport potential, and the buildup of toxic decay daughters. The index is defined for each component material and the total index is obtained by summation over all the toxic materials present.

In the formulation of the geotoxicity hazard index, toxicity, persistence, and buildup are factors that highlight the differences between a radionuclide and its natural analog. Availability on the other hand, is the term describing radionuclide behavior in an anlogous manner to its natural (stable element) counterpart.

As a first approximation, data have been presented by Cohen and Jow [5] which describe the relationship between human intake rates and crustal abundance for each element based on its overall gross average. This value may then be appropriately modified to reflect deviations from the gross average resulting from specific characteristics of the waste form or its geologic setting which may differ from that of the natural analog.

OTHER APPROACHES

Another analog approach is the mass extraction rate concept developed by Fleming [6]. This concept involves determination of the mass of earth from which a quantity of a radionuclide must be extracted and ingested each day to reach a level of intake defined by the maximum permissible dose. The approach has been applied to fallout from atmospheric nuclear weapons testing, and also to underground radionuclides resulting from activities in the Plowshare program.

SAMPLE APPLICATIONS

To illustrate the analog approach, let us consider some examples. First, the EPA model [7] for determination of population risks from HLW disposal predicts that release of 3 ci of Radium from a HLW repository would cause 10 health effects over a 10,000 year period. It is anticipated that this model may be utilized in specifying criteria for allowable releases of radionuclides from HLW repositories into the "accessible environment" [8]. Assuming that the top 10 meters of soil in the USA may be considered part of the "accessible environment," it can be calculated that this soil layer contains a total inventory of 2.6×10^8 ci of naturally occurring radium. Applying the EPA model, it can be calculated that natural radium from the top 10 meters of soil should, at equilibrium, cause 87,000 health effects per year in the USA. It is estimated that radium exposure accounts for approximately one percent of total natural background radiation dose in the USA [9]. Therefore based on assumptions derived from the EPA model, the total cancer rate in the USA from environmental radioactivity should be about 8.7 million cases per year. Yet the actual cancer rate from all causes is about 350,000 cases per year [10]. The total death rate from all causes, in fact, is only about 2 million per year. Something must be wrong.

As another example, consider Iodine-129. A recent ORNL report [11] indicated that the potential peak dose to the thyroid resulting from release of Iodine-129 from a HLW repository was 3.3 rem/year. Due to its long half-life and accordingly low specific activity it can be calculated, based on the standard man model, that if every atom of iodine in the human body were I-129, the resultant whole-body equivalent dose (applying ICRP-26) would be 0.6 rem/year.

Consider, for example, a High-Level Waste (HLW) repository containing the waste from 10^6 MWe-Yr of power production. Such a repository would contain a total inventory of 10^7 gm of I-129. Assume the repository is sited at a depth of 1,000 meters and within a watershed of 10^4 km^2 area. Based on average crustal abundance, the top 1,000 meter layer of the watershed would contain about 7.5×10^{13} gm of natural iodine. Further, assuming the I-129 from the repository is no more nor less available for release to water supplies than is the naturally occurring stable iodine, then the I-129/stable I ratio in water would be 1.3×10^{-7}. Anyone in equilibrium with the environment for that watershed could therefore receive an equivalent whole-body dose of only 8×10^{-8} rem/yr and a thyroid dose of 7.5×10^{-6} rem/yr. Barring the highly unlikely possibility that the body could fractionate or partition iodine radionuclides, then once again it must be concluded that something is wrong with a theoretical model predicting a thyroid dose of 3.3 rem/yr (~six orders of magnitude higher than the analog estimate).

Other recent work, for example that of Pigford [12], predicts that I-129 is the limiting nuclide for High-Level Waste (HLW) disposal, from a public hazard standpoint. Given that this is true, it might indicate something about the seriousness of the waste management problem.

The uranium ore body has been considered as a natural analog to the HLW repository. Such comparison has been suggested as a possible standard for HLW repository performance by an ANSI committee [13].

There have been many studies [3,14,15] making comparisons of the toxicity or potential hazard based on inventories of toxic ore material. In addition, there have been several recent studies that address the total hazard of ore bodies in comparison with a HLW repository. For example, a recently published EPA document [16] presents such an analysis of uranium ore body risks, which can be related to those of HLW repositories.

The Oklo phenomenon [17] is a fortunate natural analog situation which provides information of importance to determining long term transport potential of radionuclides. Unfortunately, this information has not, as yet, been widely applied to provide perspective on waste transport issues.

CONCLUSIONS

In conclusion, natural analogs can provide useful insight, not only for nuclear, but also for non-nuclear hazardous material underground disposal. The approach can also be a tool for model verification. Finally, the approach provides a focus for studying and understanding geotoxic phenomena and the role of natural minerals in human health and disease.

REFERENCES

1. C. F. Smith and J. J. Cohen, "Models and Criteria for Waste Repository Performance", Symposium on Uncertainties Associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste, Gatlinburg, Tennessee, March 1981.

2. National Academy of Sciences, Geochemistry in Health and Disease, 1973.

3. K. A. Tonnesson and J. J. Cohen, Survey of Naturally Occurring Hazardous Materials in Deep Geologic Formations: A Perspective on the Relative Hazard of Deep Burial of Nuclear Wastes, UCRL-52199, 1977.

4. C. F. Smith, J. J. Cohen, and T. E. McKone, A Hazard Index for Underground Toxic Material, UCRL-52889, June 1980.

5. B. L. Cohen and H. N. Jow, "A Generic Hazard Evaluation of Low-Level Waste Burial Grounds", Nuclear Technology, Vol. 41, December 1978.

6. E. H. Fleming, "Mass Extraction Rates of Radionuclides in Fallout Material from a 170-kt Nuclear Crater", Proceedings for the Symposium on Public Health Aspects of Peaceful Uses of Nuclear Explosives, Las Vegas, April 1969.

7. C. B. Smith, et al., Population Risks from Disposal of High-Level Radioactive Wastes in Geologic Repositories, USEPA Report 520/3-80-006, 1980.

8. USEPA, Selected Provisions of EPA's Disposal Standards for High-Level and Transuranic Radioactive Wastes, Draft for discussion, March 1981.

9. M. Eisenbud, Environmental Radioactivity, Academic Press, New York, 1973.

10. U.S. Bureau of Census, Statistical Abstract of the United States, Washington, D.C. 1975.

11. A. G. Croff, J. O. Blomeke, and B. C. Finney, Actinide Partitioning - Transmutation Program Final Report, ORNL-5566, June 1980.

12. T. H. Pigford, "Berkeley Results on the Migration of Radionuclides through Geologic Media", Seminar presented at LLNL on March 4, 1981.

13. Proposed Standard ANS-40.12, General Criteria for Radioactive Waste Disposal, Draft 5, 1980.

14. C. F. Smith and W. E. Kastenberg, "On Risk Assessment of High-Level Radioactive Waste Disposal," Nucl. Eng. and Design, 39, 293-333 (1976).

15. M. Willrich and R. K. Lester, Radioactive Waste Management and Regulation, The Free Press, New York, 1977.

16. W. A. Williams, Population Risks from Uranium Ore Bodies, EPA 520/3-80-009, October 1980.

17. D. G. Brookins, "Retention of Transuranic and Actinide Elements and Bismuth at the Oklo Natural Reactor, Gabon: Application of Eh-pH Diagrams," Chemical Geology, 23, (1978), 309-323.

RISK ASSESSMENT METHODOLOGY FOR GEOLOGIC REPOSITORY OF HIGH LEVEL NUCLEAR WASTE

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ABSTRACT

Nuclear waste presently exists and must be effectively isolated from the biosphere for the duration of its toxic hazard; additional nuclear waste may continue to be produced in the future. Deep geologic repositories apparently offer the best alternative for disposal of high level nuclear waste. Such a facility must, however, be designed and analyzed in the face of inherent uncertainty and must, at the same time, be publicly acceptable. Very conservative design techniques, in response to these requirements, typically result in unnecessarily high project costs. A risk assessment approach, however, results in cost optimization, by defining the public's utility function (or trade-off values), and, in addition, allows for public examination, by clearly exposing the judgement and decision process.

INTRODUCTION

Problem Statement

Large volumes of nuclear wastes of various types and levels of radioactivity presently exist, and increasing volumes of waste may continue to be produced in the future, especially by the nuclear power industry. This waste is toxic, with the level of toxicity varying with type of waste and generally decreasing with time; for high level nuclear wastes (HLW), such as spent fuel and waste from both reprocessing spent fuel and nuclear weapons program, this toxic hazard is high and lasts for thousands of years. The presence of this toxic nuclear waste will lead to an increased incidence in health effects among the population which comes in contact with it, either directly or indirectly through the biosphere. This increased incidence can be predicted quantitatively and will be approximately proportional to the amount of toxic nuclear waste present. Clearly, existing toxic nuclear waste must be disposed of and in a manner which reduces the predicted increased incidence in health effects as low as reasonably achievable (ALARA). For disposal of future additional toxic nuclear waste generated from continued nuclear power production, the predicted increased incidence in health affects has to be evaluated in light of the alternative of discontinuing nuclear power production; i.e., ALARA may not be sufficient. Indeed, continued nuclear power production is being threatened by public opposition pending an effective solution of the problem of disposal.

Criteria

The primary criterion, which any solution to this disposal problem must satisfy, is that the increased incidence of health effects due to nuclear waste disposal be ALARA. However, other criteria must also be considered:

- o feasibility
- o timeliness
- o cost-effectiveness
- o permanence, requiring no perpetual maintenance
- o public consensus and acceptability

If it can be clearly demonstrated to public satisfaction that the disposal solution satisfies the above criteria, i.e., a feasible, timely, and permanent scheme which has acceptable consequence (not only ALARA) and is also cost-effective, there should be no rational grounds for public opposition to continued nuclear power production on the basis of waste disposal problems.

Solutions

In order to reduce the increased incidence of health effects ALARA, the toxic nuclear waste must be effectively isolated from the biosphere for the duration of the toxic hazard. Various alternative schemes for achieving effective isolation have been previously evaluated, based on the previously stated criteria. It is generally agreed by the technical community that a deep geologic repository is feasible, timely, and permanent, and provides the best alternative, in terms of estimated costs and predicted consequences, for storage and ultimate disposal of HLW. Other solutions have been developed for disposal of other types of nuclear waste; this paper, however, will focus on HLW disposal.

It is envisioned that such a HLW disposal facility will consist primarily of surface facilities and deep underground tunnels, connected by shafts. At such a facility, manufactured HLW packages will be received at the surface facilities, temporarily stored, and possibly modified. Each package will subsequently be transported down the shaft and into the underground tunnels, which are organized into modular panels. Each package will then be emplaced in an individual receptacle drilled in the floor of the tunnel. The package receptacle will be backfilled with engineered material, and subsequently the tunnels in a full modular panel will be backfilled. Eventually, all remaining openings and shafts will be backfilled and sealed, and the facility decommissioned.

Thus, the facility, as well as the HLW package, must be designed and constructed so that HLW can be safely and efficiently emplaced, stored, and possibly retrieved for a period of time prior to decommissioning. Also, all backfills and seals must be designed and constructed to provide a barrier which, in conjunction with the manufactured HLW package itself and with the host rock, effectively isolates radionuclides from the biosphere after decommissioning.

In order to develop the public consensus necessary to implement HLW disposal in a deep geologic repository, as well as for design and licensing purposes, it will have to be shown at various licensing steps that such a HLW disposal facility will satisfy the previously stated criteria, especially that it will have acceptable consequences and also be cost-effective. For analysis, especially in regard to consequences, suitable models representing the repository and its behavior must be developed. This modeling, however, entails significant uncertainty, some of which can be reduced (as will be subsequently shown). Typically, acceptable consequences are ensured in the presence of residual uncertainty by a very conservative design approach; however, the safety margin is unknown, as the uncertainty is not quantified, and is achieved at substantial cost. A risk assessment approach, on the other hand, incorporates uncertainty and quantifies the safety margin, allowing for optimization of safety margin and costs.

MODEL

Physical Model

The repository system is relatively complex. In order to make it tractable for analysis, the system can be modelled as consisting of four basic "nested" components. These components, as shown schematically in Figure 1a, are of an increasingly larger scale, encompassing those of a smaller scale. The components do not overlap (i.e., are mutually exclusive) and equal the whole system when summed or assembled. Each component thus consists of everything within the volume defined by its inner and outer boundaries. Thus, in addition to liquids (such as groundwater) and gases, each component consists of the following

- o waste package/backfill-
  HLW waste, its manufactured canister, surrounding engineered sorptive backfill, plug, and in situ rock (possibly altered) immediately adjacent.
- o repository level -
  engineered supports, backfill and seals within underground tunnels, and in situ rock (possibly altered) immediately adjacent; excludes waste package/backfill components.
- o host rock -
  in situ rock, including backfilled/sealed shafts and boreholes contained therein, extending to the inner boundary of the biosphere; excludes the repository level component.

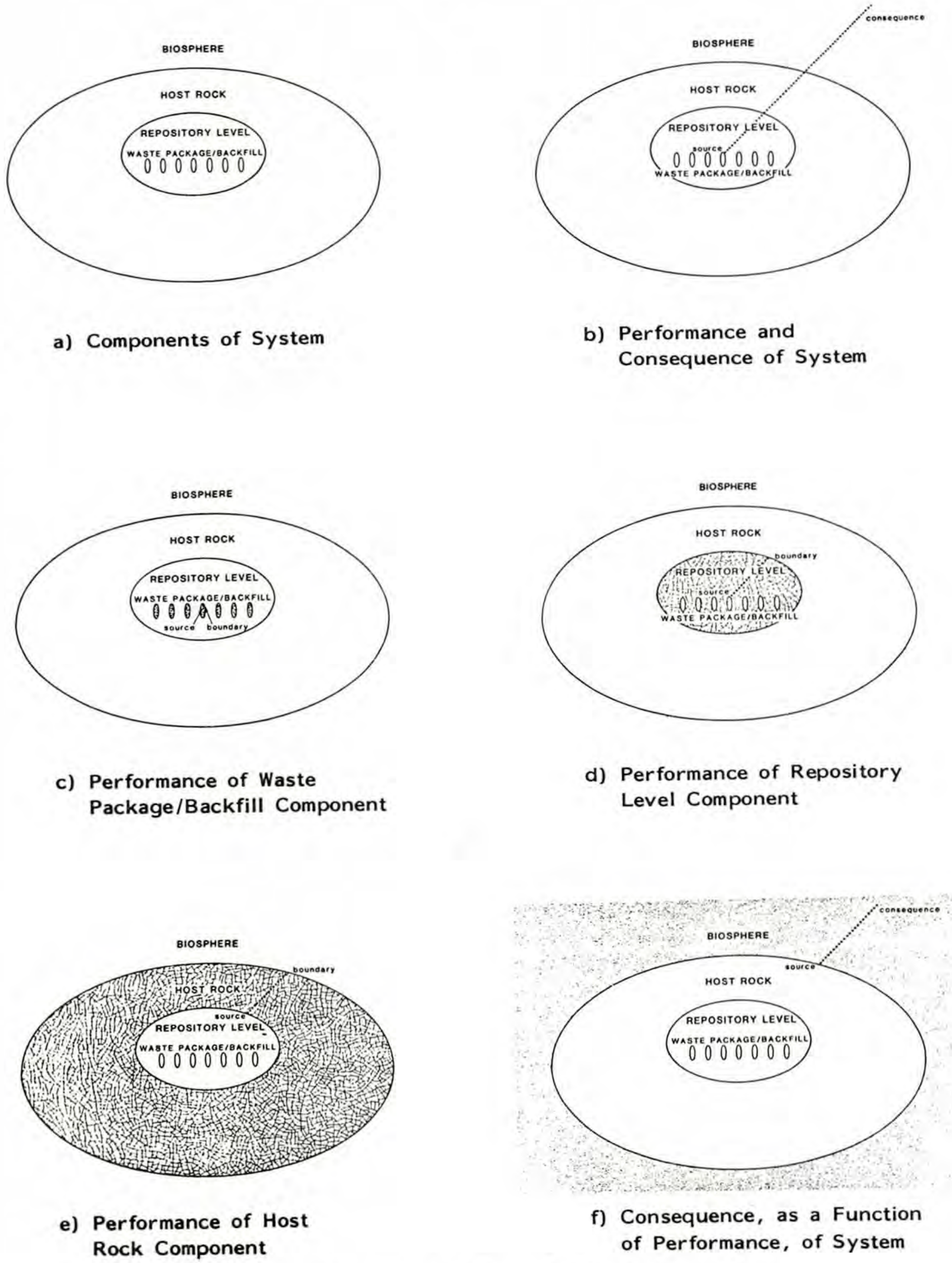


Fig. 1 Schematic of Deep Geologic Repository

o biosphere-
in-situ rock/soil, flora and fauna, humans, extending beyond the atmosphere; excludes the host rock component.

The boundaries between these various components, although somewhat arbitrary, are significant, as will be subsequently discussed.

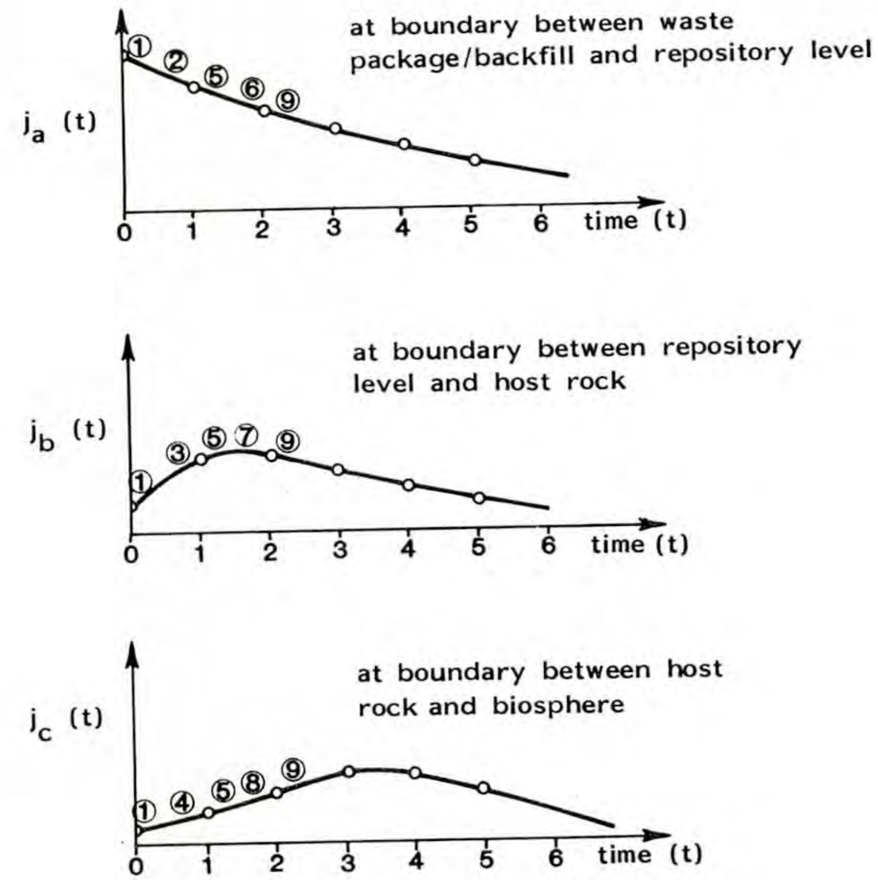
The response of the repository system, as shown schematically in Figure 1b, can be considered as the cumulative response of the various components, as shown schematically in Figure 1c-f. A function which relates the inner to the outer boundary conditions must be determined only for each component. In assembling the responses of the various components to determine the response of the system, it is necessary that at any given time the outer boundary conditions at one scale (e.g., waste package/backfill component) be identical to the inner boundary conditions at the next larger scale (e.g., repository level component), i.e., be compatible.

In addition to dividing the system into components for tractability, it is also necessary to discretize time due to the long time frame of interest. Hence, as shown conceptually in Figure 2, the conditions at each boundary can be determined at specific times. The conditions at any boundary (e.g., b) at a given time (t) are a function of the conditions at the adjacent boundaries (e.g., a and c) at that time (t), as well as the conditions at that boundary (i.e., b) and the adjacent boundaries (i.e., a and c) at the preceding time (t-1). As shown in Figure 2, this analysis is sequential (from t=o) and also often iterative during each time increment, as the conditions at adjacent boundaries at that time are generally unknown.

Numerical Model

The functions which relate the inner to the outer boundary conditions over a time increment for each component essentially comprise the numerical model. This model must sufficiently represent the physical processes which are perceived as going to occur during that time increment within that component; hence, the physical processes must be adequately conceptualized prior to defining an appropriate model. Also, properties or parameters of each component must be subsequently determined in order to quantify the model.

Generally, the thermomechanical/hydraulic response of, and radionuclide transmission through, the repository system over time is of primary interest. This is typically analyzed in a deterministic fashion by artificially coupling thermal, thermomechanical, thermo-hydraulic, and solute transport analyses for each component during the same time interval; these discrete analyses are tied together as previously shown in Figure 2.



STEP	DETERMINE
①	$j_a(0),\ j_b(0),\ j_c(0)$
②	$j_a(1) = g[j_a(0),\ j_b(0),\ j_b(1)^*]$
③	$j_b(1) = h[j_b(0),\ j_a(0),\ j_a(1),\ j_c(0),\ j_c(1)^*]$
④	$j_c(1) = k[j_c(0),\ j_b(0),\ j_b(1)]$
⑤	iterate $[j_c(1)^* = j_c(1)] \rightarrow j_b(1)$
	$[j_b(1)^* = j_b(1)] \rightarrow j_a(1)$
	$j_a(1) \rightarrow j_b(1) \rightarrow j_c(1)$
⑥	$j_a(2) = g[j_a(1),\ j_b(1),\ j_b(2)^*]$
⑦	$j_b(2) = h[j_b(1), j_a(1), j_a(2), j_c(1), j_c(2)^*]$
⑧	$j_c(2) = k[j_c(1),\ j_b(1),\ j_b(2)]$
⑨	iterate · · ·
	⋮

Fig. 2 Time Discretization Solution

Uncertainty

Due to the unusually extreme complexities and long time horizon, there are inherent uncertainites in the models, which lead to uncertainty in the prediction of response, and hence consequences, of the repository system. These uncertainties are especially due to:

- o complex interactions/perturbations or the occurrence of unexpected events/processes, especially over the long term, which have not been conceptualized or perceived as going to occur and thus not represented by the model.

- o approximation/simplification in models (e.g., artificial coupling, discretization) for tractability, and lack of complete validation/verification of each model due to time or cost limitations.

- o natural variability of properties and measurement errors/biases in the determination of properties and parameters (as illustrated in Figure 3), as well as simplification/approximation in parametric submodels.

The uncertainty in the prediction of the response of the system is a function of the uncertainty in the prediction of the response of each component. Also, this uncertainty increases with time projection; i.e., the uncertainty in the prediction of response over one time increment increases with the length of the time increment and is compounded over subsequent time increments.

Many of these uncertainties are difficult, if not impossible, to quantify objectively, and must be determined with some subjectivity by technical experts. Thus, as these subjective uncertainties are incorporated, the determination of uncertainty in the predictions of response of the repository system over time also becomes somewhat subjective. The magnitude of this composite uncertainty can, however, be significantly reduced by reducing some of the various contributory uncertainties, as follows:

- o in conceptualization and perception of physical processes by
  - demonstrations (including observation of prototypes)
  - scale models
  - judicial site selection (where processes are well understood)

- o in modeling (of conceptualized and perceived physical process) by

- validation (through comparisons provided by simplified and well defined benchmark tests)
- verification (through comparisons provided by in situ testing, prototype performance monitoring)
- judicial site selection (where processes are relatively simple)

o in determination of properties and parameters (of model) by
- utilizing a statistically valid number of sample points
- utilizing multiple types of appropriate tests with satisfactory correlations
- quality controlled, reliable tests
- judicial site selection (where properties are easily determined, reliable, and relatively uniform)

The uncertainty in the determination of properties/parameters can be reduced, as shown above, but at increasing cost. Hence, the only properties/parameters which should be extensively investigated are those whose possible range in values are neither very small nor have negligible effect on the predicted response, as shown by sensitivity analyses of each model.

The residual uncertainty inherent in the prediction of consequences can be handled in one of two ways:

o conservative design approach
o risk assessment approach

These two approaches will be discussed separately.

CONSERVATIVE DESIGN APPROACH

Conservatism

A conservative design approach has typically been taken in order to ensure acceptable consequences of a repository in the presence of uncertainty. The conservatism in this approach is due primarily to the following:

Defense in Depth - Each component of the repository is conservatively assumed to act as an independent, rather than sequential, barrier to radionuclide transmission. Conservative design limits, or performance criteria, are then specified for each component, generally in terms of response at each boundary at various times.

Parametric Design - Threshold parameter values are identified, which are bounds on adverse behavior, i.e., below this threshold value there is no evidence of adverse behavior (e.g., maximum generated temperature Tt). Conservatively, parameter values significantly below these threshold values are then utilized as bounds for design (e.g., $Td<Tt$). Where natural variability exists in properties/parameters, a bound (worst case) on determined values is utilized for design.

Using the above concepts, an upper bound on potential adverse consequences (i.e., worst case; e.g., Tu) is found deterministically with the previously described models and ensured to be acceptable (e.g., $Tu<Td<Tt$). As an upper bound, it is extremely unlikely that this value will actually be exceeded; this likelihood is not generally quantified, however. The public often perceives the upper bound as being a threshold, i.e., although the public is assured that the actual adverse consequences will not exceed the upper bound, the public perceives that these consequences might be only slightly below the upper bound. By this rationale, the upper bound, which is very unlikely to occur, must be acceptable. The actual adverse consequences will almost certainly be significantly below the acceptability limit (e.g., $Ta<<Tu<Td<Tt$). Divergence between this limit and the actual adverse consequences (e.g., Tt-Ta) is clearly a function of the conservatism, i.e., the divergence increases with increasing conservatism. The amount of expected divergence is considered an indication of the margin of safety. In the conservative approach, however, the margin of safety is not generally quantified and thus unknown.

Cost Considerations

If the cost of the repository were unrelated to the margin of safety, clearly the conservative approach which ensures acceptable consequences would be satisfactory. However, generally in order to increase the margin of safety, it is necessary to increase the cost. Thus, in order for the unlikely upper bound to be acceptable, as in the conservative approach, the margin of safety and thus the cost will be extremely high. It is very likely, however, that acceptable consequences could be achieved at significantly less cost than is possible with this conservative approach. Optimization of design safety margin and costs is suggested, which requires the following additional information not typically generated in the conservative approach:

- o quantification of safety margin
- o determination of relationship of safety margin to likelihood of occurrence of adverse consequences
- o assessment of trade-off costs of adverse consequences (i.e., defined by "utility" function).

This information is, however, generated and utilized in risk assessment.

RISK ASSESSMENT APPROACH

Incorporation of Uncertainty

The inherent uncertainties in modeling the repository system can be quantified and incorporated in analysis of any design, as follows:

- A comprehensive and mutually exclusive list of possible physical processes can be assembled, and subjective probabilities of occurrence assigned (i.e., fault tree developed).
- Models, as previously discussed, are then developed which represent those physical processes with significant probability of occurrence; an assessment of the models accuracy, or representation, can be made by "benchmarking" and a PDF for error estimated.
- Model parameters which quantify the properties of the system are sampled; sampling errors and biases are filtered and corrected, and a PDF, with correlations, for each parameter estimated.
- Discretized analyses, utilizing coupled models and model parameters, are performed; uncertainties in the input (which is the uncertain results of a previous analysis), the model accuracy, and the model parameters (all expressed as PDF's) are thus compounded to give uncertainty in the results, which is also expressed as a PDF.
- The predicted likelihood (PDF) of results, or consequences, for any design can then be evaluated with respect to various criteria, including cost considerations.

Updating and Remedial Action

The operating period of the repository, prior to decommissioning, will extend over many years and will afford an excellent opportunity to help ensure satisfactory performance. By observation and monitoring of performance, verification of processes, models and parameters can be achieved, thus significantly reducing their uncertainty. The compounded uncertainty in predicted performance is further reduced by updating, or using measured results, as input for subsequent analyses. Hence, the uncertainty in predicted performance should be significantly reduced by verification and updating afforded by performance monitoring; the licensing process specified in 10-CFR-60 implies updating and reevaluation at each of the licensing steps.

If updated predictions indicate a high likelihood of unsatisfactory performance, remedial action can be taken in order to help ensure satisfactory performance. Such remedial actions might range from removal of individual faulty waste packages to grouting to complete retrieval; the option of complete retrieval is specified in 10-CFR-60.

Incorporation of Cost Considerations

As previously discussed, the evaluation of predicted consequences should incorporate cost considerations. In order to incorporate cost considerations in the design decision process, a "utility" function must first be defined. This utility function should express the relationship, which is accepted by those people affected, between the adverse consequences and their trade-off costs, as illustrated in Figure 4a. Clearly, the definition of such a utility function is subjective and, in the case of HLW disposal where trade-off costs for cancer incidence and deaths must be defined, very emotionally charged. Although this utility function can be derived from comparable utility functions (or risk assumptions) presently accepted, perhaps implicitly, by society, in this case, this is a political decision; the responsibility for defining the utility function ultimately falls on Congress and the President (as the people's elected representatives). As illustrated in Figure 4b, a PDF of cost of adverse consequences can subsequently be derived from the PDF of adverse consequences and the utility function. An expected value of cost of adverse consequences can then be determined either directly from the PDF of trade-off costs or, without developing this PDF, by integrating over all levels of consequence the trade-off cost of each level of adverse consequence (from the utility function) times its likelihood of occurrence (from the adverse consequences).

The cost and effectiveness of various remedial actions in improving the likelihood of satisfactory performance, and thus reducng the expected cost of adverse consequences, can also be evaluated and compared to assess the cost-effectiveness of each remedial action. The optimum design would thus incorporate possible remedial actions which could reduce the expected cost of adverse consequences more than the cost of the remedial actions themselves, and give the lowest total cost.

SUMMARY

A conservative design approach ensures acceptable consequences of a deep geologic repository for HLW disposal in the presence of inherent uncertainty by designing for an upper bound, or worst case scenario, typically at unnecessarily high cost.

A risk assessment approach, which quantifies and incorporates uncertainty in the prediction of consequences, incorporates cost considerations through the definition of utility functions, and assesses the effect of various remedial actions on both consequences and costs, is feasible and can provide reasonable assurance that a deep geologic repository for the disposal of HLW will have acceptable consequences and also be cost-effective. This assurance is provided by clearly exposing, for public examination, the judgement (i.e., subjectivity) and decision process, as required by the licensing process.

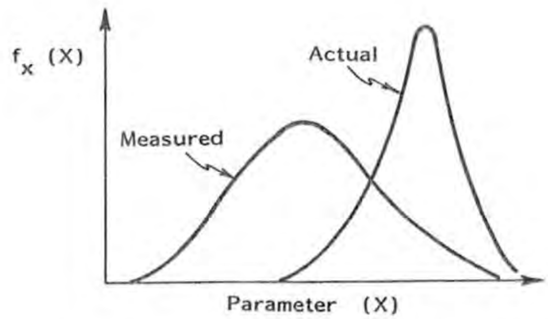
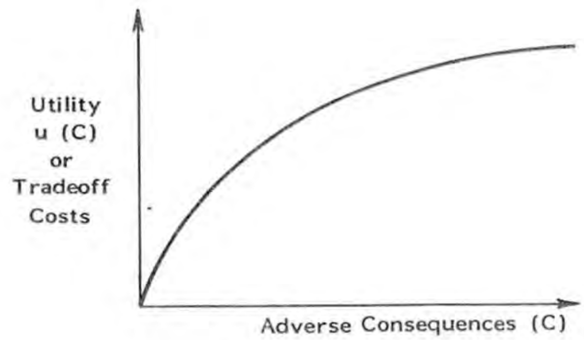
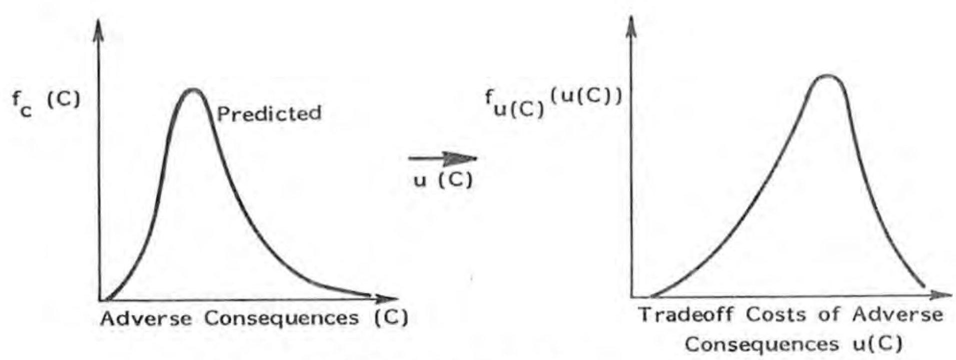


Fig. 3 Parameter Uncertainty



a) Utility Function



b) PDF of Costs of Adverse Consequences

Fig. 4 Cost Considerations

EVALUATION OF UNCERTAINTIES IN GEOLOGIC INFORMATION*

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ABSTRACT

Geologic uncertainties in earth models and maps are a fact of life for earth scientists, although the source and in fact existence of such uncertainty is often obscure to those untrained in the earth sciences. Means of systematically treating geologic uncertainty have lagged, especially in decisions that involve comparative data or locations. Our means of treating uncertainty ultimately will influence confidence and credibility in licensing a nuclear waste repository.

Most geologic information is interpretive as opposed to factual; however, in either case conclusions commonly are derived from data having unequal reliability. Decisions can take such disparity into account by using information hierarchies, by incorporating more realism into mapping, and by more rigorous quantification of models and maps. Information hierarchies can rank direct measurement (from outcrops, boreholes, etc.) as having greater confidence than indirect measurement (from geophysics, etc.). Mapping realism can be enhanced by showing individual outcrops and by more precisely defining attributes of map units.

Recognizing the evolving nature of "geologic truth" appears to undermine confidence, as does acknowledging the inherent subjectivity in our information base. On the other hand, facing and systematically treating uncertainty ultimately can lead to greater confidence in final decisions.

INTRODUCTION

Man's gathering of knowledge about the earth and its substructure has accelerated in recent years because of the increased desire to improve our well being and material comfort. This advancement has been possible because of better understanding of earth processes that affect physical safety and through the exploitation of resources. Increasing population and diminishing resources have amplified this interest and prompted more concentrated study of earth features and processes. New

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computational tools have also played a significant role, especially in data processing and earth modeling.

In spite of this revolution in geology, man's decisions about his use of the land continue to be made using information that is either incomplete, or which is not equally reliable--in short, one of uncertainty. Uncertainty affects confidence, which is significant for emplacing most civil engineered structures, including licensing a nuclear waste repository.

This paper addresses some aspects of uncertainty in geologic information that affect confidence. After the identification of those sources of uncertainty, means of treating or mitigating them are discussed with a view toward improving information credibility, and thus confidence, for decisions.

SOURCES OF UNCERTAINTY

Geology has always been a largely qualitative discipline, even though it increasingly relies on numerical data and support from other physical sciences. The nature of geology presents the opportunity for uncertainty to be manifest in several ways:

- purpose of project
- state-of-knowledge (geologic "truth")
- map characteristics
- reliability of information

The ability to accept uncertainty has been implicitly the mark of a "good geologist." One anonymous definition of good geology is the "ability to reach valid conclusions by meticulous observation of fragmentary evidence, [and] to extrapolate a reasonable projection beyond available data" [1]. This ability enabled early giants such as G. K. Gilbert, N. H. Darton, and F. L. Ransome to map large areas, in a way that is considered largely correct today [2].

Good geology could be highly inadequate when applied to the requirements for licensing a nuclear waste repository. Uncertainty is thus relative when considering information purpose. Uncertainty is also abstract semantically; there are numerous opposite terms (Table 1) to express certainty, each of which expresses a slightly different shade of meaning to each user.

Table 1. Uncertainty is Negative in Comparison With These Related Words

confidence	conviction	quality	plausibility	dependability
credibility	assurance	reliability	probability	performance
authority	certitude	accuracy	likelihood	trustworthy

Purpose of Project

Geologic maps have been constructed for different purposes over the years. Many have had rather broad objectives, e.g., filling gaps in regional knowledge or improving understanding for oil and gas potential, etc. In recent years, more and more maps have been produced for specific purposes, e.g., engineering or mining, wherein a specific course of action or decision was needed. Varnes [3] calls this purpose "decisive," in contrast with "non-decisive" purposes. These two categories are sometimes sufficiently disparate that use of a specific map for other reasons is often impossible, or at least undesirable at best (Table 2). Attempts to construct derivative maps (such as by transposing geomorphic units to "potential for landsliding" units) have frequently met with only limited success, and considerable caution must be exercised in this regard. When using archival information that was obtained for a different purpose, a conscientious effort is needed to not exceed the intent of the original product and to recognize the inherent capabilities and limitations.

Table 2. What is the Purpose? Geological Information is Obtained for Different Purposes

	NON-DECISIVE	DECISIVE
Use	• non-specific projects • advancement of knowledge <u>Examples</u>: quadrangle mapping; research projects	• specific projects • action needed pending outcome <u>Examples</u>: mineral exploration; mine planning (or repository)
Features	• format more formal (reports, etc.) • timeliness less important	• format less important and more informal • timeliness/monetary implications
Problems	• relevance often unclear • scope/style restrictive • old information sometimes obsolete	• limited application • proprietary sometimes • information lost when not published

Decisive information has value for specific purposes, and is often used to precipitate a course of action. Such action may require a threshold to be reached; the threshold could be influenced by several factors, sometimes operating together [3]:

- knowledge increase (new data)
- credibility increase (e.g., new authority enhances believability)
- recognition that information previously thought to be irrelevant is relevant
- decrease in level of effort needed by user to effect action (e.g., through increased experience)

Time can also be a precipitator. That is, an idea may not have sufficient credibility to influence a decision when first presented, but passage of time may provide the confidence needed to proceed along a given path.

Clearly perceptions are involved here; uncertainty and confidence play a significant role. These are further addressed in subsequent sections of this paper.

Geologic Truth

When in graduate school in 1959, the distinguished Dutch geophysicist, F. A. Vening-Meinesz, visited our university and guest-lectured on convection currents in the mantle and continental drifting. I shall always remember that his ideas were revolutionary at that time and how we were "de-propagandized" by several professors following his departure. Now, just 22 years later, our conceptual models of plate tectonics include petroleum and mineral emplacement schemes, and seismicity and volcanism in time and place. Regardless of how strong our beliefs are now about plate tectonics, future generations are apt to see things differently as a result of new information and new tools. Even then, truth in an absolute sense may still be a long way off.

"Truth" is not evolving, but our conceptual frameworks are, and these influence our perceptions, which in turn affect everything that we do, such as cataloging and storing information, and even locating individual drill holes. Regardless of our desire to be objective and responsible scientists, our information gathering will be biased by our conceptual frameworks. And our models are vulnerable because of the assumptions and interpretations we make on the basis of these frameworks.

Let me show how perceptions of one area in the Canadian Shield changed over 30 years. The 1928 map (Figure 1, left) relied on the conventional wisdom of that time regarding igneous rocks and it was heavily influenced by laboratory data and by the ideas of N. L. Bowen [5] and other petrologists. During the thirties and forties, geologists in many areas of the world were unable to reconcile their field observations with the early ideas derived largely from laboratory observations [6]. Subsequently, granitization (ultrametamorphism) gained more recognition by American geologists. The result of remapping the same area of Figure 1 (right) with this revised perception is a distinctly different map.

The science of geology relies on the method of multiple working hypotheses, being continually subjected to re-examination in light of new data and concepts [7,8]. This fact does not alter our view of uncertainty, but recognizes the evolutionary development of all scientific thought.

Credibility is thus a matter of perception, which is not always related to truth. Nonetheless, credibility is required for decisions; much of this is derived from how information is presented, and how reliable it is. Our goal therefore is to be accurate and to convey the why of our beliefs.

Map Characteristics

Much geologic information is ultimately stored in map format--an excellent communicative device which the human mind can readily understand. A map is essentially a two-dimensional miniaturization and generalization of the field; some maps also include a third dimension (depth) for multiple stratigraphic units [9]. Most maps are representations and interpretations of reality to some extent, and herein lies an element of value. That is, the judgement added to what is factual has worth for some specific use.

The usefulness of the map often will reflect the judgements made in developing the legend [10]. In defining attributes for map units, decisions are required that cover the entire range of concerns being discussed here. Anyone who has ever constructed a map understands the importance of the legend; those who are using the map gain confidence through recognizing the exactitude and care with which the legend was developed [1], and by using it. When honest uncertainty is revealed it should be more an aid than a detriment. Such intellectual honesty can be accomplished by showing alternative interpretations (Figure 2), or by displaying information sufficient for the user to reach his own interpretations [2].

Reliability diagrams are sometimes used by map makers to express graded confidence in information, such as on the Army Map Service topographic series. Such honesty can be expressed in map symbolism of several kinds and provide added credibility to the product.

Instituting conservatism by the inclusion of some features, e.g., faults, is sometimes essential when constructing certain maps. To omit suspected features could lead the user to the possibly unjustified belief that they do not exist [12].

The location of lines and boundaries for map units is usually judgemental, requiring that criteria or definitions be established. Errors (or degrees of unreliability) can be introduced from several sources, including interpretation and generalization (being intentionally less reliable) [11].

Reliability of Information

Earth scientists are frequently blessed--sometimes even overwhelmed--with information of all sorts. But the simple truth is that not all information has equal worth. Information reliability, or the probability of its accuracy, is highly variable. Reliability affects confidence which has been known all along, but attempts to systematically treat confidence have been few.

A hierarchy of information sources (Table 3) can be constructed and used for some types of decisions. Geologists recognize that some information is inherently more accurate and can scale estimated reliability, however qualitative, and use the relative worth to influence decisions. Caution is necessary because these estimates could be misleading when judging that a specific item of information is necessarily more likely to be accurate as a result of its source. Adding to (or detracting from) the reliability of information and its credibility are other qualifying elements such as simplicity, homogeneity, and the scientist's reputation.

Table 3. Confidence Hierarchy of Geologic Information

Type of Information		Reliability*
1. Measurements:		High
a. direct (e.g., sample or field observation)		
b. indirect (e.g., borehole geophysics or force field)		
2. Interpretations:		
a. direct (from direct measurement, e.g., empirical models)		
b. indirect (from indirect or sparse measurement base)	conceptual models	
3. Informed but poorly supported explanations (educated guess)		Low

*± Additional weighting factors:

a. agreement with intuition
b. simplicity/homogeneity (structural, lithologic)
c. amount of supporting data (see Table 4)
d. authority (scientist's reputation)
e. interpolations favored over extrapolations

Another way of evaluating the quality of information considers data interpretation, comparing data density with maturity of knowledge in a given field (Table 4). This shows that inferior data density is compensable by greater relative understanding in a given field of knowledge

Confidence Scoring

DATA DENSITY (data/sq mi)	Educated Guess	Expert Opinion in Very Poorly Understood Field	Expert Opinion in Poorly Understood Field	Expert Opinion in Well Understood Field	Expert Opinion in Very Well Understood Field
Dense (≥ 9)	55	65	75	85	95
Abundant (1-8)	45	55	65	75	85
Adequate (1/8-1)	35	45	55	65	75
Limited (1/36-1/8)	15	35	45	55	65
Isolated (< 1/36)	5	15	35	45	55

Definitions

<u>Very Well Understood</u>: Sound, stable theoretical basis, confirmed by adequate experimentation, and with few unexplained phenomena.

<u>Well Understood</u>: Theoretical basis confirmed by some experimentation and/or field examples.

<u>Poorly Understood</u>: Limited number of hypotheses and usually operable rules-of-thumb.

<u>Very Poorly Understood</u>: As many hypotheses as experts.

<u>Educated Guess</u>: Thoughtful ideas of a professional in the field <u>or</u> expert opinion based on virtually no data.

Table 4. Quality of Data Interpretation. The dashed lines suggest approximate equivalency, or tradeoffs, between the two properties.

(chart courtesy R. L. Link, Sandia National Laboratories, unpublished) and suggests that field locations can be compared with respect to types and amount of data. Similar tradeoffs involving data density can be developed for geologic simplicity, amount of rock exposure, etc. The ideas of formally assessing confidence in geologic information are being introduced into the formal area screening of the Nevada Nuclear Waste Storage Investigations, performed by Sandia National Laboratories.

In general, our earth models and information bases gain in their quality (and thus usefulness) through their predictive capacity. And this requires validation and time; the more facts that fit the old models, the greater the credibility of the models.

TREATMENT OF GEOLOGIC UNCERTAINTY

Having recognized the sources and nature of uncertainty in geological information, we are confronted with treating it in an enlightened, systematic way. With respect to the geologic disposal of high-level radioactive waste, one approach for handling uncertainty would introduce conservatism by:

- avoiding potentially adverse features where possible;
- favoring simple sites, those most stable, and those easily understood; and
- constraining design and performance of engineered components.

I have already commented on weighting information reliability. In some instances, it is also possible to evaluate the likelihood of alternative hypotheses. Such evaluations are useful in performing safety assessments which rely on probability studies. An example of interpretive differences between faulted and flexed veins is described by Brown [13]; regional statistics in that case provided the basis for determining the likelihood of the correct interpretation. Such methods have been applied extensively to mineral exploration where economic risk is involved, but are also applicable to other problems. A summary of the treatment of geologic uncertainty is shown in Table 5.

CONCLUSIONS

Uncertainties in geological information arise from several sources but can be treated. The concepts discussed here are very much interrelated, and most of the words involving certitude are overlapping in meaning. Much of this topic is subjective; much involves perceptions, and even feelings, all of which reaffirms our human nature, but does not compromise our need to be as objective as possible. Uncertainty cannot be eliminated, but it can be understood, reduced, and treated, thus lending confidence to decisions.

Table 5. Treatment of Geologic Uncertainty

Uncertainty Source	Treatment
• Purpose of project	→ Clear understanding of intent, e.g., problems and limitations
• Knowledge level	→ Evaluate consequences of alternative hypotheses
• Reliability of information	→ Weighting (scaling/grading)
• Map characteristics	
--classifying scheme	→ precision in definition
--outcrops	→ reliability designation
--interpretation	→ intellectual honesty

REFERENCES

1. Geotimes (1964), v. 8, no. 5B, p. 4.

2. Kupfer, D. H. (1966), Accuracy in Geologic Maps, Geotimes, v. 10, no. 7, p. 11-14.

3. Varnes, D. J. (1973), Some Elements of the Term "Value" as Applied to Interpretive Geology, Bull. Assoc. Eng. Geol., vol. x, no. 4, p. 289-298.

4. Harrison, J. M. (1963), Nature and Significance of Geological Maps, in Albritton, C. C. (Ed.), The Fabric of Geology, Addison-Wesley Pub. Co., p. 225-232.

5. Bowen, N. L. (1928), The Evolution of Igneous Rocks, Princeton Univ. Press, 332 p.

6. Geological Society of America Memoir (1948), Origin of Granite (James Gilluly, ed.), Geol. Soc. Am. Mem. 28, 139 pp.

7. Woodford, A. O. (1956), What is Geologic Truth?, Jour. Geol. Education, v. 4., no. 1; p. 5-8.

8. Krumbein, W. C. (1962), The Computer in Geology, Science, v. 136, no. 3522, p. 1087-1092.

9. Matula, M. (1971), Engineering Geologic Mapping and Evaluation in Urban Planning, in Nichols, D. R., and Campbell, C. C., eds., Environmental Planning and Geology, U.S. Department of Housing and Urban Development and U. S. Dept. of Interior, p. 144-153.

10. Varnes, D. J. (1974), The Logic of Geological Maps, with Reference to their Interpretation and Use for Engineering Purposes, U.S. Geol. Surv. Prof. Paper 837, 48 pp.

11. Hageman, B. P. (1968), The Reliability of Geological Maps, Internat. Jahrbuch fur Kartographic, v. 8, p. 144-154.

12. Wentworth, C. M. and J. I. Ziony, et al. (1970), Preliminary Geologic Environmental Map of the Greater Los Angeles Area, U.S. Geol. Survey TID-25363, 41 p., issued by U.S. Atomic Energy Comm., Oak Ridge, TN.

13. Brown, B. W. (1961), Stochastic Variables of Geologic Search and Decision, Geol. Soc. Amer. Bull., v. 72, no. 11, p. 1675-1686.

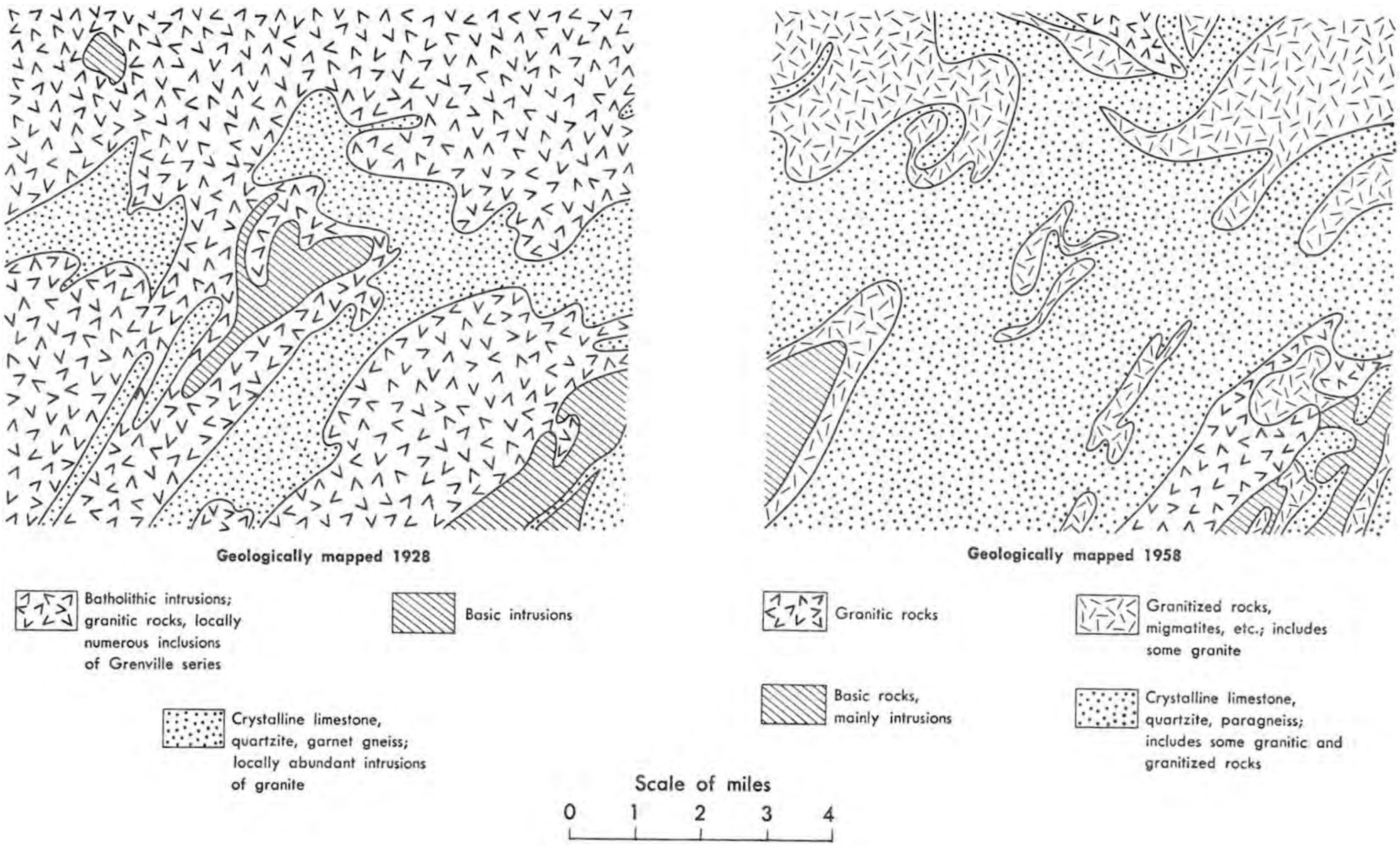

Fig. 1 Two geological maps of the same area in the Canadian Shield, mapped 30 years apart [4]. (Used by permission, Geological Society of America.)

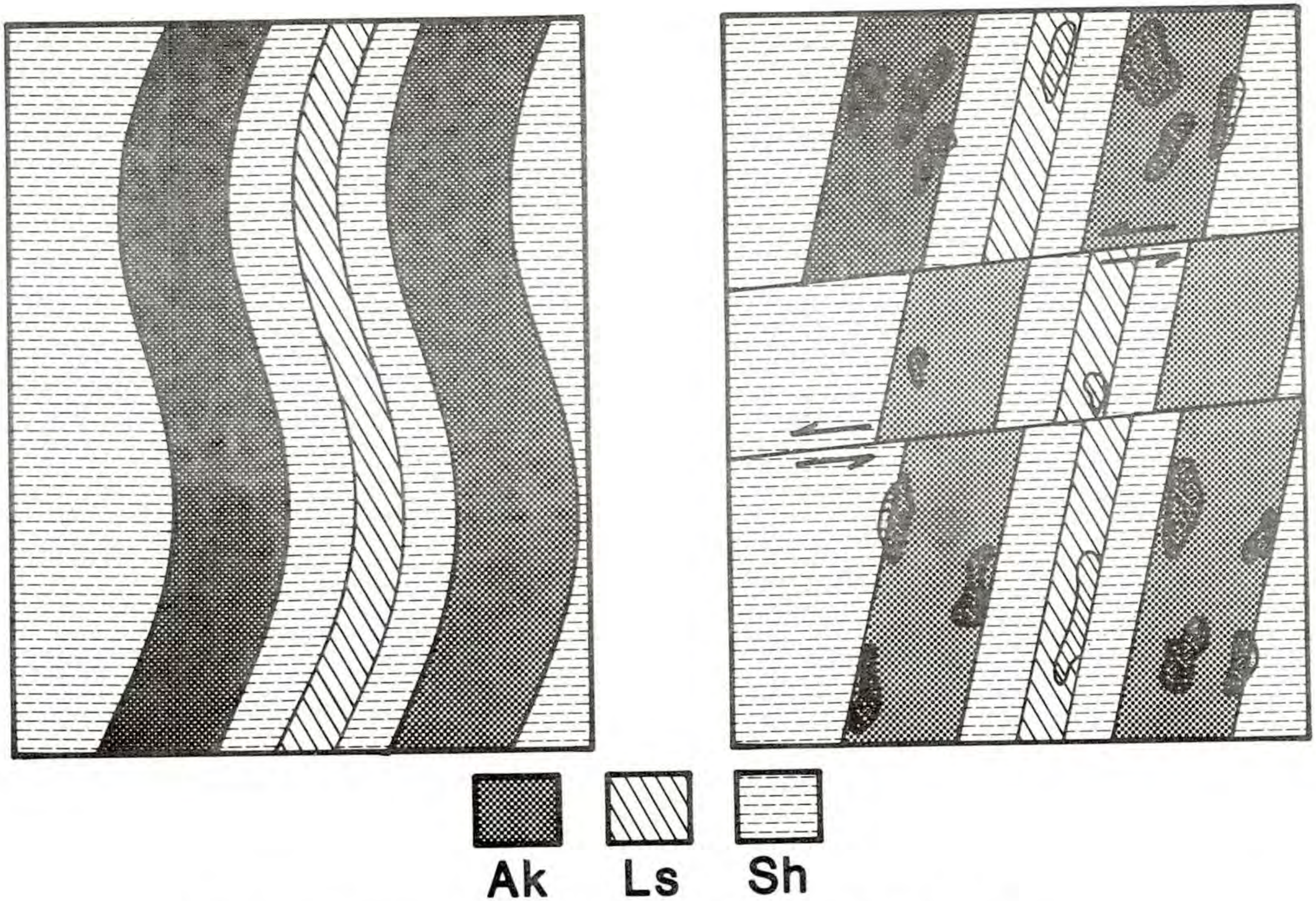

Fig. 2 Geological maps of hypothetical sedimentary sequence of arkose, limestone, and shale. Conventional geologic map is shown on left; outcrop map (right) enables alternate interpretation by user.

NATURAL GEOLOGIC PROCESSES AS SOURCES OF UNCERTAINTY IN REPOSITORY MODELING

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ABSTRACT

Repository modeling needs to consider the environmental changes that can occur from the natural geologic processes of: (1) tectonic movements, (2) igneous activity, (3) non-tectonic deformations, (4) climate changes, (5) erosion, deposition, and dissolution, and (6) groundwater movement. Rates and mechanisms of these processes are well established by the present state of knowledge in the earth sciences, indicating that they operate in an orderly manner at very low rates over thousands to tens of millions of years. Consequently, future conditions can be forecast with confidence for at least 10,000 years. Significant uncertainty does not occur until the time frame of a few hundred thousand to a million years. Selective siting can further reduce the number of these variables that must be considered in modeling and reduce the range of others. Knowledge of the geologic processes involved is important for planning and evaluating the geologic investigations at a prospective site.

INTRODUCTION

Changes caused by natural geologic processes have been considered to be potential sources of uncertainty in modeling long-term performance of a repository. In order to evaluate the significance of these processes and the dimensions of the uncertainties that would be introduced over a repository lifetime, we have conducted an extensive review of the available literature on the mechanism, rate, periodicity, and occurrence of the relevant geologic processes. Results are summarized briefly in this paper.

This study has addressed the natural geologic processes, excluding those perturbations resulting from construction and operation of the repository or other human activities. The reason for this emphasis is that the natural processes will be investigated in site characterization to evaluate the suitability of a site before considering the works of man. Additionally, the natural processes operate on a much larger scale and can influence a repository over much longer periods of time.

RATES OF GEOLOGIC PROCESSES

The types of geologic processes that can be important for post-closure performance of a repository are summarized in Table 1, along with their typical rates or recurrence times. Two main classes of these processes may be defined. Tectonic movements and climate changes may be classified as "primary" processes in that they act independently. The remainder occur as a result of or are greatly influenced by other processes and may be classified as "secondary".

Of the processes listed in Table 1, groundwater movement is particularly important because it is considered to be the principal means whereby radionuclides might be transported to the accessible environment. However, rates of groundwater movement are difficult to cite because of the number of complex factors involved. The rate at which groundwater moves through any medium is determined by the hydraulic conductivity, or permeability, of the medium and the hydraulic gradient. The critical question for a repository of how long may be required for fluids to reach the accessible environment depends additionally on the length of the prospective flow paths. Such flow paths may have complex geometry and typically would be very much longer than the depth of the repository. Permeability, hydraulic gradient, and flow-path lengths each can vary over orders of magnitude and must be determined individually for any site.

In previous generic analyses, rates of groundwater movement in the range of 3 mm per year to 60 mm per year have been assumed for rocks of low permeability [1]. However, there is substantial geologic evidence that actual rates of groundwater movement in the field may be much lower than estimated from laboratory tests. Isotope dating of groundwater at prospective repository sites has indicated that the fluids have been isolated from the larger hydrologic system for hundreds of thousands of years [2] [3]. In other environments, such as the geopressurized zones in the Gulf Coast [4], there is geologic evidence that fluid migration has been restricted for as long as tens of millions of years.

Groundwater movement is particularly influenced by the other geologic processes. For example, deformations may affect permeability by increasing fracturing in the medium or may change hydraulic gradients. Erosion/deposition or climate changes may affect hydraulic gradients by changing base levels or introducing additional sources of water. However, it is important to note that groundwater movement is a "passive" process in that the parameters determining direction and rate of flow normally do not change without outside influence. Accordingly, in areas where tectonism, volcanism, or other deformation is not occurring or expected, there is no reason to anticipate changes in hydraulic conductivity below the expected depths of erosion. Similarly, hydraulic gradients should remain within the range of climate-induced variations if there are no

Table 1. Summary of Rates for Geologic Processes

Tectonic Activity:

Plate tectonic movements (relative motion of plates occurring across plate boundaries)--1.5 to 16 cm/yr - 2.6 cm/yr average rates averaged over millions of years. [5] [6]

Rates of slip on individual strike-slip faults, averaged over thousands to millions of years - less than 0.1 cm/yr to 6.6 cm/yr. [7]

Uplift
- a) To 0.08 cm/yr over periods of 120,000 to 450,000 yr in large areas of southern California. [8] [9]
- b) 1 cm/yr over last 45,000 years in Ventura area, California. [8]
- c) 1.8 to 3.6 cm/yr in Transverse Ranges of California over periods of few years: 0.43 to 0.48 cm/yr average over last 100 years. [10]
- d) Less than 0.03 cm/yr over last 660,000 years in southeastern United States. [11]
- e) 0.00003 cm/yr differential uplift over 100,000 years in Texas Gulf Coast. [12]

Igneous Activity:

Periodicity of basalt flows in Columbia Plateau - several flows per million years up to 12 m y ago, more than a million years between flow units to 6 m y ago, and no activity since. [26]

Climate Changes:

Sea level: Fluctuates over range of 85 to 140 meters (average range estimated at 100 meters) with change occurring at rate of 0.1 to 1 cm/yr. [13] [14] [15]

Non-tectonic Deformations:

Salt Diapirism: Less than 0.003 to 0.2 cm/yr averaged over millions of years. [16] [17]

Erosion/Deposition

Erosion: Typical rates of 0.0002 to 0.08 cm/yr estimated based on historic sediment loads in various streams in the US [18]; 0.04 cm/yr (810 m in 2,100,000 years) on lower Colorado River [19]; 0.003 to 0.004 cm/yr average over 115 to 180 m y in Northern New England. [20]

Deposition: 0.01 cm/yr in Western US desert basins, averaged over 3 m y [21] to 0.2 cm/yr offshore of Gulf Coast, averaged over estimated 2.5 m y. [22]

Salt Dissolution: - 0.005 to 0.016 cm/yr of dissolution (vertical component) averaged over drainage basins in southeast New Mexico, based on present-day dissolved salts. [23]

- 0.01 cm/yr at solution-depression in southeast New Mexico, averaged over 600,000 years. [24]

Groundwater Movement:

16,000 cm/yr	(poor confining formation)
6.3 cm/yr	(good confining formation)
0.3 cm/yr	(highly confining formation)

Rates are calculated on basis of representative permeabilities and assumed hydraulic gradients [1]; actual conditions will vary.

changes from uplift, tilting, or downwarping. The conclusion that may be drawn from this relationship is that if a site can be shown to be stable with regard to overall geologic processes, there is no reason to expect changes in groundwater movement.

DIMENSIONS OF CHANGE FROM GEOLOGIC PROCESSES

A major objective in selecting and demonstrating geologic stability of a repository site is the identification of regions wherein geologic processes have operated very slowly and without significant change in rates for periods so much greater than the time of concern for the repository, that the uncertainty of their future performance is very small. In accomplishing this, some potential geologic hazards can be avoided completely; the likelihood of others can be minimized, and the impact of others can be mitigated by design or construction procedures.

Figure 1 compares the dimensions of change for some typical natural geologic processes acting over a period of 10,000 years to the typical depth of a repository and to the dimensions of a typical hydrologic regime or system. The ranges of process rates are based on the data summarized in Table 1. Rates at a well-chosen site represent the typical long-term (i.e., 10,000 years) rates at places that are considered stable with regard to the subject process. A typical range of higher, "moderate" rates also is shown for additional perspective.

For each of the processes represented on Figure 1, the dimensions of the changes that may be anticipated over a 10,000-year period are orders of magnitude smaller than the dimensions of a repository or of the surrounding hydrologic system. With careful selection, sites can be identified where the changes over such periods could be less than 0.1 percent of the repository depth, and a correspondingly smaller fraction of the extent of the hydrologic system. At such sites, there can be considerable confidence that the changes in the geologic environment from natural processes do not introduce significant uncertainty for modeling repository performance.

LENGTH OF THE GEOLOGIC RECORD

Figure 2 compares a repository lifetime of 10,000 years with the length of the geologic record in some of the geologic settings that presently are being considered for repository siting. Also included in this comparison is the range of time (35,000 to 500,000 years) over which the existing Seismic and Geologic Siting Criteria for Nuclear Power Plants [25] require investigations of the age of most recent fault movement to determine whether faults are "capable". This is shown because it is an accepted standard with which the industry has considerable experience, using existing technology to perform the required investigations successfully in many parts of the United States.

The geologic record for each of the geologic settings indicated on Figure 2 is several orders of magnitude longer than the period of concern for a repository. Although the geologic record is not complete in any one place, the ability to evaluate the history of geologic processes over such long periods of time leaves little doubt as to the nature and rate of their continued operation over the repository lifetime. Accordingly, any doubts as to the continuing operation of the natural geologic processes do not add significant uncertainty to modeling of repository performance.

CONCLUSIONS

The existing information on the mechanisms, occurrence, and rates or periodicity of the natural geologic processes can be used in selection and evaluation of prospective repository sites in order to minimize uncertainties for modeling long-term performance. The geologic investigations for siting and site characterization should be appropriate for each process acting at a prospective site. They should cover a distance from the site adequate to evaluate the relevant process and its influence; and they should address a period of geologic time adequate to confidently establish geologic history. These requirements must be determined individually for each site and for the different types of investigations performed at a single site; arbitrary standards for distance and length of geologic history are unlikely to be suitable.

In choosing a site, emphasis should be placed on identifying places where the rates of geologic processes are demonstrably low. At the same time, adverse conditions that are not suitable for modeling may be recognized and avoided in siting. In this manner, sites can be selected where the uncertainty resulting from changes in geologic processes would be insignificant for modeling repository performance over periods on the order of 10,000 years.

ACKNOWLEDGEMENTS

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REFERENCES

1. G. de Marsily, E. Ledoux, A. Barbreau, and J. Margat, "Nuclear Waste Disposal: Can the Geologist Guarantee Isolation?," Science, 197, 519-527 (1977).

2. I. W. Marine, "The Use of Stable Decay Products for Dating Groundwater," Proceedings of the National Waste Terminal Storage Program Information Meeting, ONWI-62, 17-19 (1979).

3. D. W. Powers, S. J. Lambert, S. E. Shafer, L. R. Hill, and W. D. Weart, Geological Characterization Report, Waste Isolation Pilot Plant (WIPP) Site, Southeastern New Mexico, SAND 78-1596 (1978).

4. R. H. Wallace, Jr., T. F. Kraemer, R. E. Taylor, and J. B. Wesselman, "Assessment of Geopressurized-Geothermal Resources in the Northern Gulf of Mexico Basin," U.S. Geol. Survey Circ. 790, 132-155 (1978).

5. R. Isacks, J. Oliver, and L. R. Sykes, "Seismology and the New Global Tectonics," J. Geophys. Res. 73, 5855-5899 (1968).

6. X. LePichon, "Seafloor Spreading and Continental Drift," J. Geophys. Res. 73, 3661-3697 (1968).

7. Woodward-Clyde Consultants, Report of the Evaluation of Maximum Earthquake and Site Ground Motion Parameters Associated with the Offshore Zone of Deformation, San Onofre Nuclear Generating Station, Report to the Southern California Edison Company (1979).

8. K. R. Lajoie, J. P. Kern, J. F. Wehmiller, G. L. Kennedy, S. A. Mathieson, A. M. Sarna-Wojecieki, R. F. Yerkes, and R. F. McCrary, "Quarternary Marine Shorelines and Crustal Deformation, San Diego to Santa Barbara, California," in P. L. Abbott ed. Geological Excursions in the Southern California Area, 3-16 (1979).

9. P. J. Kern, "Origin and History of Upper Pleistocene Marine Terraces, San Diego, California," Geol. Soc. America Bull. 88, 1553-1566 (1977).

10. D. A. Rodgers, Vertical Deformation, Stress Accumulation and Secondary Faulting in the Vicinity of the Transverse Ranges of Southern California, California Division of Mines and Geology, Bull. No. 203 (1979).

11. O. Wilhelm and M. Ewing, "Geology and History of the Gulf of Mexico," Geol. Soc. America Bull. 83, 575-600 (1972).

12. H. A. Bernard and R. J. LeBlanc, "Resume of the Quaternary Geology of the Northwestern Gulf of Mexico Provinces," in H. E. Wright Jr. and D. G. Frey, eds., The Quaternary of the United States, Princeton University Press, 137-186 (1965).

13. R. F. Flint, Glacial and Quarternary Geology, John Wiley & Sons, New York, New York (1971).

14. R. D. Steiner, R. S. Harrison, and R. K. Matthews, "Eustatic Low Stand of Sea Level Between 125,000 and 105,000 B.P.: Evidence from the Subsurface of Barbados, West Indies," Geol. Soc. America Bull. 83, 63-70 (1973).

15. N. J. Shackleton and N. D. Opdyke, "Oxygen Isotope and Palaeomagnetic Stratigraphy of Equatorial Pacific Core V28-238; Oxygen Isotope Temperatures and Ice Volumes on a 100,000 and 1,000,000 Year Scale," Quarternary Research 3, 39-55 (1973).

16. F. Gera, "Review of Salt Tectonics in Relation to the Disposal of Radioactive Wastes in Salt Formations," Geol. Soc. America Bull. 83, 3551-3574 (1972).

17. J. D. Martinez, "Tectonic and Hydrologic Stability of Louisiana Salt Domes," Proceedings of the National Waste Terminal Storage Program Information Meeting, ONWI-62, 80-83 (1979).

18. S. Judson and D. F. Ritter, "Rates of Regional Denudation in the United States," J. Geophys. Res. 69, 3395-3401 (1964).

19. P. E. Damon, M. Shafqllah, and R. B. Scarborough, "Revised Chronology for Critical Stages in Evolution of the Lower Colorado River," Geol. Soc. America Abstracts with Programs 10, 101-102 (1978).

20. J. T. Doherty and J. B. Lyons, "Mesozoic Erosion Rates in Northern New England," Geol. Soc. America Bull. Part I 91, 16-20 (1980).

21. A. J. Eardley, R. T. Shuey, V. Grosdetskyu, W. P. Nash, M. Dene Picard, D. C. Grey and G. J. Kukla, "Lake Cycles in the Bonnevile Basin, Utah," Geol. Soc. America Bull. 84, 211-216 (1973).

22. B. R. Sidner, S. Gantner, and W. R. Bryant, "Late Pleistocene Geologic History of Texas Outer Continental Shelf and Upper Continental Slopes," in A. H. Bouma, G. T. Moore, J. M. Coleman etds. Framework, Facies, Oil-Trapping Characteristics of the Upper Continental Margin, 243-266 (1978).

23. G. O. Bachman and R. B. Johnson, Stability of Salt in Permian Salt Basin of Kansas, Oklahoma, Texas, and New Mexico, U.S. Geological Survey Open File Report No. 4339-4 (1973).

24. G. O. Bachman, Geologic Processes and Genozoic History Related to Salt Dissolution in Southeastern New Mexico, U.S. Geological Survey Open File Report No. 74-194 (1974).

25. U.S. Nuclear Regulatory Commission, Seismic and Geologic Siting Criteria for Nuclear Power Plants, 10CFR Part 100 Appendix A, (1973).

26. D. A. Swanson, T. L. Wright, P. R. Hooper, and R. D. Bentley, Revisions in Stratigraphic Nomenclature of the Columbia River Basalt Group, U.S. Geol. Survey Bull. 1457-G (1979).

27. U.S. Department of Energy, Statement of Position: Proposed Rule-making on the Storage and Disposal of Nuclear Waste, Appendix A, DOE/NE-0007 (1980).

28. Dames and Moore, Inc., Crystalline Intrusives in the United States and Regional Geologic Characteristics Important for Storage of Radioactive Waste (Draft), ONWI-50, (1979).

FIGURE 1

DIMENSIONS OF CHANGE AT TYPICAL REPOSITORY FROM RATES OF A NATURAL GEOLOGIC PROCESSES

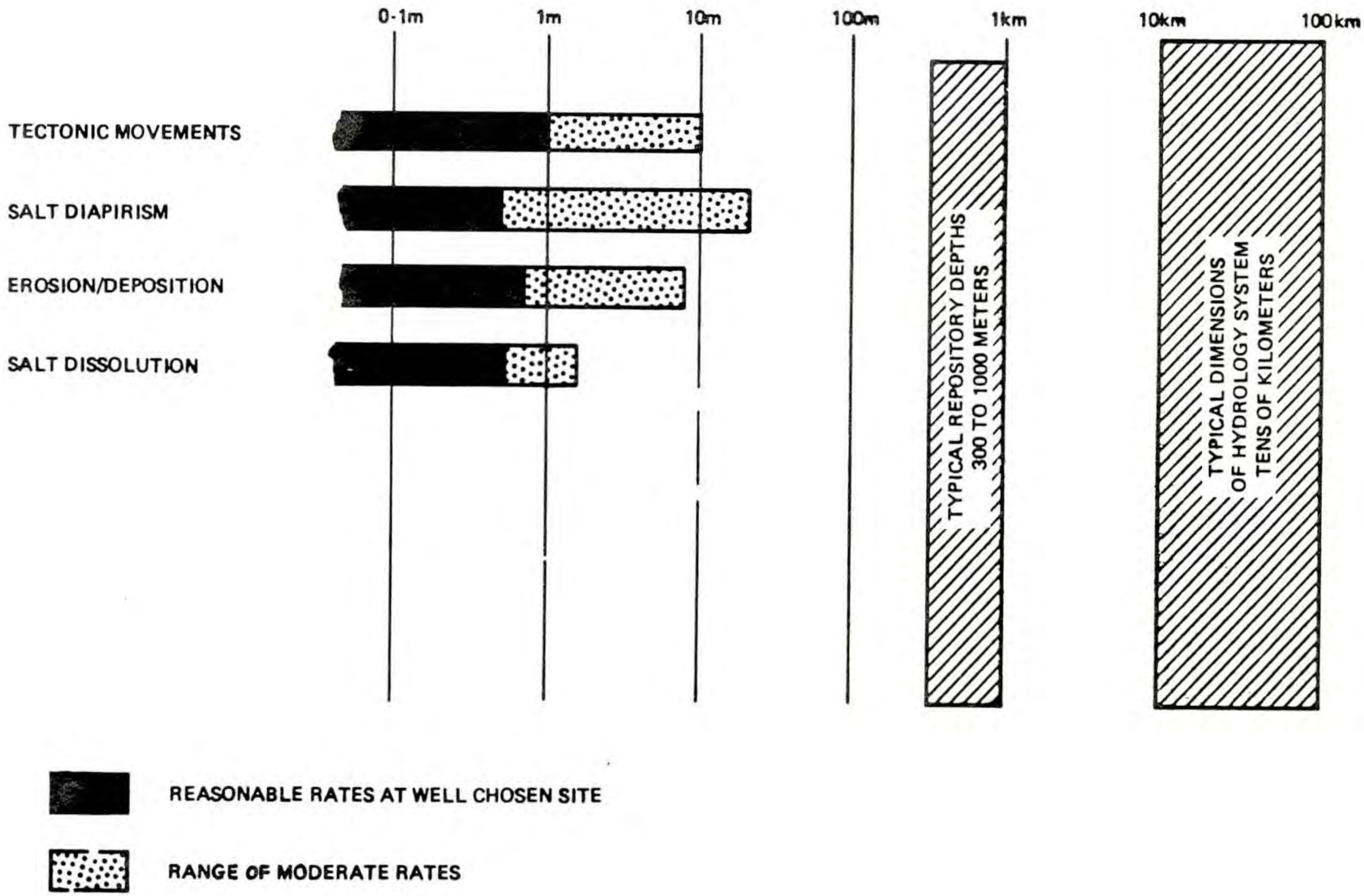
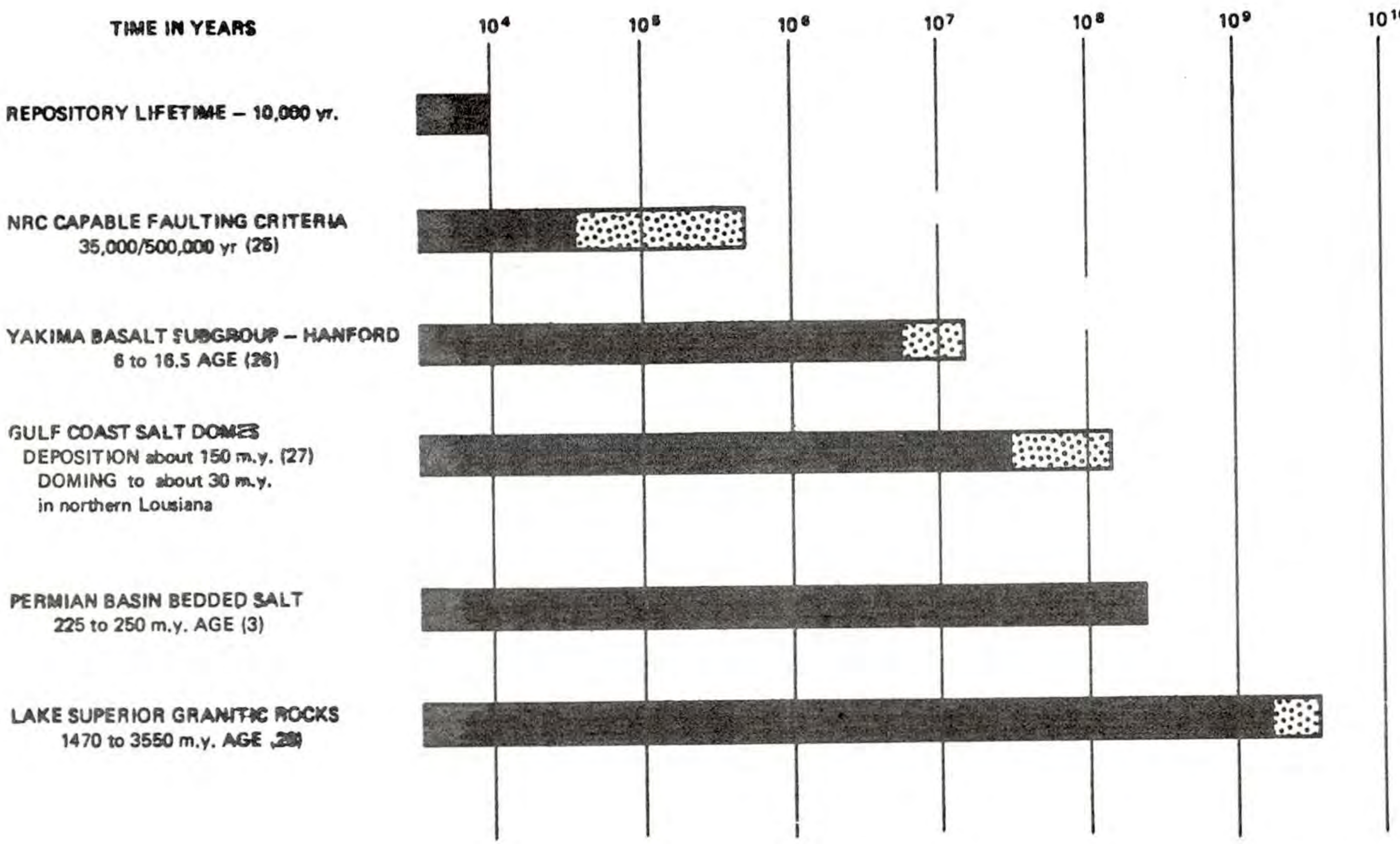


FIGURE 2
REPOSITORY LIFETIME IN RELATION TO LENGTH OF GEOLOGIC RECORD FOR SOME CANDIDATE MEDIA



THE PROBLEM OF HUMAN INTRUSION: TOWARDS A RESOLUTION OF UNCERTAINTY

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ABSTRACT

Isolation of high-level radioactive waste over long periods of time requires protection not only from natural events and processes, but also from the deliberate or inadvertent activities of future societies. This paper evaluates the likelihood of inadvertent human intrusion due to the loss of societal memory of the repository site. In addition measures to prevent inadvertent intrusion, and to guide future societies in any decision to deliberately intrude into the repository are suggested.

INTRODUCTION

In addition to the uncertainties regarding the knowledge of the physical factors relating to long-term containment of radioactive waste, there has been a concern over the uncertainty of human behavior in relation to the repository. [1] One major aspect of this concern is the possibility of human intrusion, either inadvertent or deliberate, into the buried waste at some time in the future. Inadvertent or accidental intrusion is based on the assumption that no institutional control remains at the site, that all physical markers onsite have disappeared, and that all public records and social memory of the site have been lost. Some commentators have also questioned the ability of future societies to decipher the message contained on site markers and in site records. [2] Inadvertent intrusion could then occur as a result of search for minerals, curiosity over the heat source, or as a result of the search for scientific knowledge. Deliberate or intentional intrusion is based on the assumption that future generations will make a conscious decision to breach the repository in order to recover the high-level waste itself, as a mineral associated with the site. A further assumption would be the potential attractiveness of the site as a target for terrorists and saboteurs. I would also note that the concern over human intrusion is not only limited to high-level radioactive waste, but is also of concern in the disposal of low-level radioactive waste, particularly since the shallow land burial makes the possibility of a breach easier than in the case of a geologic repository. The remainder of this paper will discuss the validity of the various assumptions on which the human intrusion scenario is premised, with the objective of reaching some conclusions on the likelihood of human intrusion occurring.

ANALYSIS

Most of our recent history is based on records that have been passed down from generation to generation, and the history of earlier periods is based on writings which have been recovered through archeological excavation. However our earliest written records are only about 5,000 years old and human development began considerably before that. In addition, some of the societies that had writing used it for only very limited purposes, or else they wrote on materials that have not survived. [3] As a result, adequate historical documentation is sometimes dependent on other sources than writing, such as oral tradition and physical artifacts. Before the invention of writing, the accumulated knowledge of a civilization was transmitted from one generation to another by word of mouth. Much of this oral record was eventually written down [4] and thus served as a mechanism for preserving knowledge of the ancient world. The ancient Greeks collected the accounts of warfare and political practices and the expedition against Troy, for example, was the basis for Homer's Illiad and Odyssey. Although these epics were eventually written down, they began as a part of the oral tradition. [5] In fact the site of the ancient City of Troy was identified by the use of Greek tradition and the details in the Homeric poems. [6] Similarly, the histories of the ancient Hebrew narrators were eventually incorporated into the Old Testament. The Celtic bards, Anglo-Saxon scops, Scandinavia scalds, German minnesingers, and the troubadours of France are all prime examples of the oral tradition. In primitive societies of more recent times, the spoken word was used to guide future generations through birth, death, marriage, hunting, and harvest. In Polynesia, verbal transmission was carefully regulated and was extremely accurate. [7] The Harepo of Tahiti, the Tuhuna of the Marquesas Islands, and the Rogo Rogo of the Gambier Islands all were trained by the priests and were required to pass examinations on the retentiveness of their memories. The Dahomeans utilized the unique recordkeeping system of appointing female officials to remember what their male counterparts did. The Shamans of the North American Indians were entrusted with the knowledge and transmission of sacred texts, as well as magic formulas for treating sickness and for successful hunting. The critical point that emerges from this discussion of the oral tradition is that even before writing, records were kept and information was transferred from generation to generation, and this information survived to later be incorporated into written records. [8]

Although the oral tradition was successful in a number of instances for passing knowledge on to future generations, the key development in terms of expanding the human ability to communicate and transmit knowledge to the future was the advent of writing and printing. The earliest known writing, cuneiform, is credited to the Sumerians, who lived in Southern Mesopotamia around 4000 B.C. The use of cuneiform was spread by the Assyrians, Babylonians, and other people of the Near East, and has brought us knowledge of a number of complex and highly developed cultures. At approximately the same time that cuneiform was being developed, the Egyptians started the writing form known as heiroglyphics. The earliest known examples date back to 3000 B.C. and have supplied us with extensive knowledge of ancient Egypt. The development of parchment suitable for writing on both sides gave rise to the copying of numerous manuscripts for the libraries of the Hellenistic World. In Europe during the Middle Ages the church became the protector and repository of knowledge. Christian doctrine was systematized and recorded at this time and many monastaries contained scriptoriums where highly skilled monks would copy and recopy sacred texts, as well as historical, literary and philosophical writings. The birth of the great universities of Europe in the 13th Century stimulated a new demand for books and the universities established their own manuscript copy centers. With the increased use of printing in the 15th Century, the widespread communication of knowledge became possible. This was the dawn of the modern era of communications. [9]

The historical record provides us with many precedents for the survival of highly complex information over long periods of time. From as early as 3000 B.C. onward abstract information in the areas of religion, science, mathematics and engineering has been transmitted to future generations. [10] We have extensive knowledge of ancient civilizations through the survival of books and written records such as:

- Akkadian - Sumerian dictionaries;
- Hammurabi's Code;
- the extensive records of Ancient Egypt;
- the Torah of the Old Testament;
- records of Minoan civilization of Crete;
- the great books of Greece and Rome: Homer, Plato, Demosthenes, Thucydides, Herodotus, Vergil, Lucretius, and many others;
- manuscripts and texts from ancient China;
- Mayan books, such as the Dresden Codex.

There are also many examples of books and records that have survived from the earliest days of Western civilization, [11] including records that have been kept continuously for centuries:

- the tax and land ownership records of England that have been kept since 1066 A.D.;
- the archives of the English courts maintained since 1156 A.D. with very few gaps;
- the specific legal rules of many European nations; [12]
- the complex body of Church doctrine;
- the record of the development of research science from the Middle Ages; [13]

All of this information from the ancient world and from modern civilization, has survived a wide range of material and manmade hazards - fire, war, weather, natural disasters, negligence and willful destruction. Even during the so-called "Dark Ages" in Europe, Christian monastaries served as centers of enlightenment and were responsible for the preservation of many varieties of information. In addition, cultures flourished in other parts of the world. [14] In other words the "Dark Ages" was a localized phenomenon.

There have been cases where information has not been passed down to us from the past. We don't have the answers to such questions as:

- who built the Megaliths?
- how were the pyramids constructed?
- what caused the disappearance of the Mayans, the Olmecs, and the Aztecs?
- who were the Tiahuanaco's, Indians, builders of fortress cities in the Andes before the time of the Incas?
- who were the Etruscans?
- what was the purpose of the huge diagrams made by the Nazca Indians of South America
- who were the "mound builders" of North America?

Although we don't as of yet know the answers to these questions, because no written records provide us with this information, we at least possess knowledge of the culture of these societies, as well as knowledge of the site itself, just as the culture of Rome and Greece, and the site of Rome and the Acropolis were not forgotten during the Dark Ages. Many of these so-called "Lost Civilizations" - the Mayans, the Khmers of Angkor Wat, the Incas of Machu Picchu - were not discovered by the outside world until centuries after they were built. However, the local natives did know about them and in fact, provided the information that eventually led to the discovery of these sites. [15] When archeologists were digging at a hill called South Glastonbury Castle in England, the possible site of King Arthur's Camelot, one old man came up and asked them anxiously if they had come to release Arthur! [16] The memory of the site had lived on in local legends for centuries. As mentioned previously, ancient legends also led to the discovery of the city of Troy. In other words, these sites were not completely forgotten and their discovery was not inadvertent and fortuitous.

McNiel, an eminent historian, compares civilizations to mountain ranges:

> "...it is only within the frame of geological paleontology and universal history that mountains and civilizations rise and pass away. On shorter time scales, they constitute enduring landmarks." [17]

He went on to show how enduring civilizations were, by comparing a culture map of Eurasia as it existed in 500 B.C. with the same map 2000 years later. The map was essentially the same, the fundamental cultural structure unchanged over 80 generations. Although the institutions within these civilizations may change or cease to exist, the specialized function of an institution, for example, information preservation and transfer, and the body of critical information to be preserved, can survive throughout the existence of the civilization. The longevity of human institutions has been a major issue in the disposal of high-level radioactive waste. However, the primary emphasis has not been on human intrusion, but rather on the extent to which human institutions should be involved in the long-term management of the site vis-a-vis natural and engineered barriers. [18] Although, one function of an institution is to transfer information, retention and transfer of information about the repository site is not dependent on the existence of the same institution over a long period of time.

Obviously, one cannot predict with certainty that the information on the repository will not be lost. However, there has never been a global blackout of knowledge and although there are examples of information that has been lost from the past, these occurred before the widespread use of writing and printing. Modern information transfer and storage systems would decrease the probability of this happening today. The preponderance of historical evidence indicates that the likelihood of information loss is small. Our efforts should be directed towards careful and deliberate marking of the site, an understandable explanation of the design and nature of the repository, as well as storage of information about the site in multiple offsite locations. This will further minimize the chance that memory of the site will be lost, and ensures that comparative texts do exist to enable future generations to decipher the site records. [19] Any event cataclysmic enough to destroy onsite markers and all offsite records would make the hazards posed by loss of the repository location small in comparison.

Onsite markings should range from very simple markers using universal symbols to warn of the danger, to complex descriptions of the mine and its contents. Both types of markers should be made of materials that are impervious to destruction by time and the elements. [20] In situ "tell tales" could also be placed in the repository media to alert would-be mineral explorers of the nature of the repository in the case that societal memory of the site was lost. [21] The specific languages to be used for marking the site should be those that have the highest potential for survival. These would be those spoken by the largest number of people--English and Chinese--and a language spoken in a place remote enough to survive a natural or man-made holocaust--for example, Spanish, which is scattered throughout the corners of South America.

It is unlikely that any future generations would utilize different languages than those language groups used at the time the repository was developed. Before writing, language change was relatively rapid. The invention of writing stabilized language to the point where radical change no longer occurred. [22] Even if a natural or manmade holocaust occurred, the survivors would not have to evolve a new system of writing, unless only young children survived, an extremely unlikely possibility. Therefore, we can assume that future generations will be able to decipher the message left at the repository.

In the event that the memory of the site is lost, there is a high probability that any future society that possesses the knowledge and capability to locate and explore for resources at 600 meters would also have the knowledge to recognize what they have found, [23] how to mitigate adverse consequences, and would establish administrative control over the area. [24]

Reliable documentation of the site location and the nature of the repository will also be key factors in minimizing the hazard that could result from deliberate intrusion into the repository. It is impossible to predict what the probability is for deliberate intrusion, but it is a possibility that cannot be ignored. Even if a repository site is an "unattractive" location in terms of existing resource potential, the high level waste itself may be an attractive resource and there may also be other site location resources and natural features that prove valuable to future generations. However, any effort to recover those resources will be based on a conscious societal decision. Our obligation to future generations is to assist them in making an informed decision on whether to breach the repository or to conduct any type of operation in its general vicinity. Documentation should present a full explanation of repository design, the nature of the waste including potential resource use, and an assessment of the potential hazards from breach of the repository to enable future generations to fully evaluate the costs and benefits of intrusion. The remaining measure to be employed to minimize the possibility of deliberate human intrusion is to evaluate the existence of site resources of current or future value in making a site suitability decision. This type of site selection criteria will also work to minimize inadvertent human intrusion. In neither case, however, will it foreclose on its own credit the possibility that human intrusion will occur.

One type of deliberate human intrusion that would not be a collective societal decision would be intrusion by terrorists or saboteurs Although, a repository might seem to be an attractive target, there is little possibility that terrorists could covertly breach a repository. Breach of the repository would require extensive use of machinery for drilling and excavating over a considerable period of time, and it is highly improbable that a terrorist group could accomplish this without being detected and stopped by government authorities.

REFERENCES

1. See DOE, FEIS Management of Commercially Generated Radioactive Wastes, V. I, p. 3.43 (1980); DOE, Statement of Position, NRC Waste Confidence Rulemaking, PR-50, 51, p. II-192 (1980). Hebert, J. A., et al., Nontechnical Issues in Waste Management: Ethical Institutional and Political Concerns, p. 43 (1978); Lipschutz, R. D., Radioactive Waste: Politics, Technology, and Risk, p. 155 (1980); Nuclear Energy Policy Study Group, Nuclear Power Issues and Choices, p. 243 (1977); Willrich, M., Radioactive Waste, p. 95 (1977); Proceedings of Conference on Public Policy Issues in Nuclear Waste Management (October 27-29, 1976); EPA Workshop on Environmental Protection Criteria for Radioactive Waste (ORP/CSD-77-1; ORP/CSD 77-2; ORP/CSD-78-2); NRC Advance Notice of Proposed Rulemaking, Technical Criteria for Regulating the Geologic Disposal of High Level Radioactive Waste (45 Fed. Reg. 31398, May 13, 1980).

2. Rochlin, Gene, "Nuclear Waste Disposal: Two Social Criteria," Science, January 7, 1977: 27.

3. Trigger, Bruce, Time and Tradition (1978).

4. McNiel, William, The Rise of the West, 58 (1963).

5. Lord, A. J., The Singer of Tales.

6. See Leaf, Walter, Troy (1912); Burr, A. R., Minoans, Philistines, and Greeks; Thorndike, Joseph, Mysteries of the Past, 66 (1977).

7. Suggs, Robert, The Island Civilization of Polynesia.

8. See, Vansina, Jan, The Oral Tradition. McCall, Dan, Africa in A Time Perspective.

9. See, Fabre, Maurice, A History of Communication From Stone Age to Space Age; Hogben, Lance, The Signs of Civilization; Sabloff, J., Ancient Civilization and trade; Kunz, Gorge, Imperishable Records of the Ancients; Posner, E., Archives in the Ancient World.

10. Supra n. 4.

11. For example, the Domesday Book commissioned by William the Conqueror in 1066; the Magna Carta; the Lindesfarne Bibles, and many others.

12. Strohl, P. "Legal, Administrative and Financial Aspects of Long Term Management of Radioactive Waste," Nuclear Energy Agency OECD, Nuclear Law Bulletin No. 21 (June 1978).

13. Hebert, et al., "Nontechnical Issues in Waste Management: Ethical, Institutional, and Political Concerns," Battelle Pacific Northwest Division, Human Affairs Research Center (May 1978).

14. Supra n. 4, see also Wells, H. G., An Outline of History.

15. Thorndike, Joseph, Discovery of Lost Worlds (1979); Thorndike, Joseph, Mysteries of the Past (1977); Bacon, Edward, Lost Civilizations (1963); Daniel, G. E., Man Discovers His Past; Daniel, G. E., The Idea of Prehistory; Daniel, G. E., The Origins and Growth of Archeology; Daniel, G. E., 150 years of Archeology; Trigger, Bruce, Beyond Prehistory; Trigger, Bruce, Time and Tradition. The destruction of the Alexandrian Library has also been cited as an example of the loss of societal memory. However, most of the volumes in the library were duplicates of originals stored in other parts of the waved, and furthermore, many of the library's volumes we're shipped out in the 4th century to Rome and Constantinople, before the burning of the Alexandrian libraries. See, Parsons, Ed., The Alexandrian Library; Dunlap, Les, Alexandria: The Capital of Memory.

16. Thorndike, Joseph, Discovery of Lost Worlds, 196, (1979).

17. Supra n. 4, at 249.

18. The EPA (see 43 Fed. Reg. 53263 11/15/78) and the NRC (see 45 Fed. Reg. 31400 5/13/80) are not placing reliance on human institutions for any longer than 100 years after repository closure, relying instead on engineering and environmental barriers. It should be noted that engineering and environmental barriers are more reliable not necessarily because of the time frame over which human institu-

tions can be relied on, but because the performance of the environmental/engineering barriers are more reliable than the performance of human institutions in maintaining the integrity of a site. Although the EPA reached a judgment that 100 years was an appropriate time limit for the reliability of human institutions, no background was given to support this judgment. There are several examples of institutions that have lasted well beyond 100 years (for example, the Catholic Church, Harvard University, the U.S. National Bank System, U.S. State Governments) and there doesn't seem to be any reason to believe that the likelihood of institutional breakdown increases significantly after 100 years. In fact, it was the consensus of the working group at the final EPA Workshop on Radioactive Waste that 100 years was too arbitrary a length of time, and that the 100-year limit should not become part of the EPA criteria (see ORP/CSD-78-2).

19. The inability to translate the language of the Minoans, Linear B, has been cited as an example of the difficulty future generations may have in deciphering the repository records. (<u>Supra</u>. n. 2, at 27). However, part of the difficulty in deciphering Linear B was due to the fact that it's discoverer jealously guarded the information for years and also was delayed because of the intervention of World Wars I and II. The primary difficulty was the lack of comparative texts. Storing the information on the repository in multiple locations should facilitate translation. <u>See</u> Chadwick, J., <u>The Decipherment of Linear B</u>.

20. Past history has shown that if you wish information to survive for many centuries it should be cut in stone or baked in clay. Paper products have not lasted and metal has been melted down for its resource value. Thorndike, Joseph J., <u>Discovery of Lost Worlds</u>, 302 (1979).

21. DOE, <u>Statement of Position</u>, NRC Waste Confidence Proceedings, PR-50, 51, p. II-196 (1980).

22. Stewart, William, "A Sociolinquistic Typology for Describing National Multilingualism," in Fishman, Joshua, <u>Readings in the Sociology of Language</u>. <u>See also</u> Fishman, Joshua, <u>Sociolinguistics</u>.

23. The DOE FEIS on the Management of Commercially Generated Radioactive Waste concluded that -

    (b) The drill crew may not be aware of radioactive material in the drilling mud as it is brought up; however, once samples are sent to their assay laboratory, the drillers would soon know of the radioactive nature of their exploratory effort. If the assay were crude they might conclude, in the case of drilling through a spent fuel element, that they had struck uranium, but very little sophistication in assay would be required to recognize that the radiation spectrum was not at all like that of natural uranium. The radiation characteristics of material brought up after passing through a solidified high-level waste canister would resemble natural ores even less.

However, Rochlin has pointed out that if our society depletes existing uranium ore beds, a future society could develop to a point where they are advanced enough to breach a repository but lack any knowledge of radioactivity. (Supra N. 4 and 27). However, this assumes depletion of the ore beds, disappearance of the mill tailings, and either selective loss of knowledge of radioactivity or a complete loss of knowledge.

24. Claiborne, H. C., Gena, Feruccio, Potential Containment Failure Mechanisms and Their Consequence at a Radioactive Waste Repository in Bedded Salt in New Mexico (ORNL-TM-4639).

PARAMETRIC ANALYSIS OF MINED GEOLOGIC DISPOSAL OF HIGH-LEVEL RADIOACTIVE WASTE

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ABSTRACT

A simplified mathematical model has been developed to screen potential mined geological repository sites taking into account the uncertainty in the input data. Initial input data that was assumed constant was inventory of radioactive wastes, number and size of cannisters, size of repository, and the ground water flow area. Though there is some uncertainty in these data, by far the greatest uncertainty pertained to leach rate of the waste form and cannister, ground water velocity, retardation rates of nuclides relative to ground water, distance to the biosphere and flow rate in the receiving waters in the biosphere. These were varied over realistic ranges from 1 to 4 orders of magnitude. The results showed that there are a wide variety of combinations of these parameters that allow a waste repository to be sited without exceeding the maximum permissible concentrations of isotopes in drinking water. It is concluded that for the artificially-created nuclides it is the intermediate time period, greater than 1,000 years and less than 1 million years, that poses the greatest problem.

INTRODUCTION

This conference is concerned with uncertainties associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste. Can we reduce or finesse these uncertainties sufficiently so that we will not be paralyzed by analyses but can move forward to actually putting high-level radioactive waste underground?

To do this we need to look at the system as a whole and not set arbitrary limits on how individual components behave. We need to be conservative and have some redundancy built into the system and utilize defense in depth principles but for the initial screening of sites we can afford to use very simple models to predict what the worst accidents will cause. We have been consistent admirers of Occam's principle which states that what can be done with fewer assumptions is done in vain with more. Therefore, we have used a brute force, simple model approach to show that, in the multiparameter system that represents a waste repository with the associated waste form, cannister, overpack, backfill and geology and hydrology, there are an infinite number of combinations that will result in concentrations released to the biosphere that will be less than some stated fraction of the maximum permissible concentrations. Possibly, more important, one can show that there are certain combinations of parameters that are not suitable for a waste repository system.

MODEL DEVELOPMENT

The model is unsophisticated in that it does not take into account explicitly dispersion or buildup of daughter products. Rather, it does calculate the mass flow rate of radioactive material leached from the waste form - cannister system in grams per day by multiplying the leach rate times the cannister area. The solubility limit for each of the isotopes of concern is used as the default value. The volumetric flow rate in cubic centimeters per day of each isotope is then calculated by multiplying the ground water velocity times the equivalent aquifer area and dividing by the retardation factor. The time of travel of each nuclide is determined by dividing the distance to the biosphere by the ground water velocity and multiplying it by the retardation factor. The steady state non-decayed concentration of radionuclides moving through the ground water is then equal to the mass flow rate divided by the volumetric flow rate or gm/cm^3. The activity of each nuclide is then calculated by taking into account its decay during its travel time to the biosphere. The concentration in the ground water is diluted by mixing with receiving stream and this instantaneous concentration is compared with the MPC for that isotope.

No reconcentration of the nuclides in the biosphere is assumed. For many isotopes such as H-3 this is correct, whereas for others at some trophic levels, magnification can be as much as 10^6. Though the results are presented as ratios of the permissible concentrations, it must be remembered that these are not necessarily the doses that will be received unless one takes his entire water intake untreated from this water course.

INPUT DATA

The parameters that were fixed in the analysis were the initial inventory of radioactive waste, the number and size of cannisters, the size of the repository, and the ground water flow area. The parameters that were varied were: leach rate of waste form and cannister of 10^{-1} to 10^{-5} in increments of 10 in gm/cm^2-day, ground water velocity 0.1, 10, and 1000 m/yr; retardation rates of 1, 100, 1000, and 10,000; distance to the biosphere of 500, 1000, 1500, 5000 m; and stream flows of 1, 10, 100, and 1000 m^3/sec. The retardation factor and the leach rate are subject to considerable uncertainty. The retardation factor has to be a function of the chemical form, the soil matrix, the pH and Eh of the transporting water, subject to the law of mass action and competing ions, and the exhaustion of sorption sites. The leach rate is a function of the exact waste form, the temperature, pH, Eh, and competing ions of the leach water. The difficulty in defining a procedure for leach rate is indicative of the many factors that determine the leach rate. For well defined, porous media the ground water travel time is relatively easily determined. However, for fracture flow, the determination is not so simple. The 21 isotopes of most concern were taken as the source terms and are listed in Table 1 with their activities in the year 2070 for the reprocessing cycle of the Final Environmental Impact Statement on Management of Commercially Generated Radioactive Wastes [1]. This case assumes that nuclear power capacity grows to 250 GW_e in the year 2000, that all plants operate for 40 years and that the last plant shuts down in 2040. This, then, would mean wastes from a total of 6400 GW_e years would need to be disposed.

RESULTS

The full results are too voluminous to be reported here. Therefore, only the results from three isotopes of different half-lives, permissible concentrations, and sorption characteristics are reported here:1) Strontium 90, relatively short half-life (relative to age when placed in a repository) 28.9 years, maximum hazard with a maximum permissible acceptable concentration of 10^{-5} microcuries per liter, and medium sorption of 10 ml/gm; 2) Plutonium 239, with long half-life 24,390 years of medium hazard with a maximum permissible concentration of 10^{-4} μCi/ml, and high sorption of 250 ml/gm; and 3) Iodine 129, of very long half-life of 17,000,000 years, of maximum hazard with a MPC_W of 10^{-5} and no sorption at 0 ml/gm [2].

The times to reach the biosphere at a distance of 1000 meters from the repository, ignoring decay, for a ground water velocity of 2.74×10^{-4} m/day and for retardation factors (time of travel of nuclide relative to time of travel of water) of 1, 100, 1000, and 10,000 are 1×10^4; 1×10^6; 1×10^7; and 1×10^8 years. For any other distance, ground water flow or retardation factor, the time of travel is linearly related. The calculations are made for release into the biosphere into a river with a flow of 1 m^3/sec. For other river flows, the concentrations and doses are linearly related.

The maximum permissible concentration factors used were those of the ICRP 2 and 6 [3, 4] prior to the publication of ICRP-30 [5]. The permissible concentrations of Sr 90, Pu 239, and I 129 are decreased by a factor of 24, increased by 2.7, and remained the same, respectively, in ICRP-30.

Strontium 90

Using a ground water velocity of 0.1 m/yr, the time of flow of the water to the nearest point in the biosphere, 500 m, is 5000 years. Since the half-life of Strontium 90 is approximately 30 years, then the percent not decayed at that time is 8.5×10^{-51}. Therefore, at these activity levels, the leach rate and the retardation factors are immaterial because the concentration of the nuclide in the biosphere is already greatly below the permissible concentrations, as shown in Figure 1. However, when the ground water velocity is increased to 10 m/yr, then leach rate and retardation factors do become important. As would be expected for a retardation factor of 100, the time for the Strontium 90 (not the water) to reach the biosphere is the same as for the water traveling at 0.1 m/yr. Consequently, as seen in Figure 2, only when the retardation factor is 1 are there any concentrations of Strontium 90 in the biosphere above the permissible limits.

While it might at first be thought that these calculations are superfluous in that everything should be linearly related, it should be noted that decay is a function of base 2 and that leach rates are solubility limited so that one can see, as in Figure 2, instances where two leach rates still have the same biosphere concentrations. This, of course, must be in the higher leach rates.

When the ground water velocity is increased another factor of 100, then to have the same concentrations as in the first scenario, the retardation factor must increase by 10^4. This is shown in Figure 3 where all concentrations with retardation factors of 10^4 are less than 10^{-2} MPC.

Plutonium 239

The same sort of profiles hold for Plutonium 239. However, because of the longer half-life of Pu 239, 24,390 years, at 500 meters and with a retardation factor of 1, the activity remaining is almost 90%. However, for higher retardation factors, say 100, the amount remaining is only 7.5×10^{-5} percent of the initial amount.

As can be seen in Figure 4, line AZ, a retardation factor of 1 and a leach rate of 0.1×10^{-4} gm/cm^2/day, is essentially horizontal. This, in effect, says that for long lived isotopes, the time of travel from points near to the repository to more distant points is short relative to the half-life. For example, the time to travel from the 500 meter point to the 5000 meter point is 450 years. In that time period, the activity of the Plutonium 239 has decreased about 1%.

Iodine 129

In Figure 5, where I-129 is shown at the highest ground water travel rate calculated, it can be seen that for the case of 1 m^3/sec the maximum concentrations are only 200 times the MPC values. For more realistic flows, the MPC would not be exceeded.

CONCLUSIONS

This simple model has shown its worth in that one can see the isotopes that are, in fact, of concern and the parameters that need to be reduced or increased to make the concentrations less than the permissible limits. In effect, it tells you what combinations of parameters will give you satisfactory sites. One can also see what range of uncertainty one can tolerate within these parameters and still have satisfactory sites.

In addition, one can see that, for long and very long-lived isotopes, short term delays in releases, say 1000 years, do not, in fact, have any appreciable effect on the dose rate or integrated doses. (For Plutonium 239, this reduces the activity and, therefore, the concentrations and integrated doses by less than 3 percent.) Consequently, the main purpose of 1000 year containment can only be to assure no problems with short half-lived isotopes and to retain the isotopes past the peak thermal periods. Yet, as shown in Figures 1-3, if one can show relatively slow travel times or moderate retardation coefficients, the same delay can be achieved.

It can also be seen in Figure 4 that, for rapid flow and high toxicity, the leach rate must be extremely low to make a difference in the dosages.

In Figure 5 it can be seen that the absolute value of the ratio of the concentration to the MPC is such that hazard will be low for any of the leach rates used, taking into account isotopic dilution and reasonable river water dilution. Therefore, one can see for the man-made isotopes that it is the intermediate time period that most likely must be most closely designed for, since it is likely not to be buffered sufficiently by the natural system nor of such low concentration that it can be simply diluted.

The other conclusions reached could have been postulated a priori, but are here stated for completeness and because the figures do verify them. For safer disposal sites, one would prefer:

1. Slow ground water velocity to high velocities;
2. Low leach rates to high leach rates;
3. High retardation ratios to low retardation ratios;
4. Longer distances to the biosphere to shorter distances;
5. Larger river flows to smaller river flows.

REFERENCES

1. U.S. Department of Energy, Final Environmental Impact Statement on Management of Commercially Generated Radioactive Wastes, (DOE/EIS-0046F) A41-A42 (October, 1980).

2. D. Isherwood, Geoscience Data Base Handbook for Modeling a Nuclear Waste Repository, Volume 1, NUREG/CR-0912 (UCRL-52719) 230 ff (January, 1981).

3. International Commission on Radiological Protection, Report of Committee II on Permissible Dose for Internal Radiation-1959, Pergamon Press (1960).

4. International Commission on Radiological Protection, Report of Committee II on Permissible Dose for Internal Radiation-Amended 1959 and Revised 1962, Pergamon Press (1964).

5. B. L. Cohen, Effects of ICRP Publication 30 and the 1980 BEIR Report on Hazard Assessments of High Level Wastes, (unpublished document 1981).

TABLE 1

RADIOACTIVITY INVENTORY - IN YEAR 2050

ISOTOPE	ACTIVITY, CURIES
H-3	2×10^{6}
C-14	2×10^{5}
Fe-55	2×10^{6}
Co-60	2×10^{7}
Sr-90	1×10^{10}
Tc-99	3×10^{6}
I-129	8×10^{3}
Cs-135	8×10^{4}
Cs-137	2×10^{10}
Th-232	2×10^{-5}
U-235	7×10^{1}
U-238	2×10^{3}
Np-237	2×10^{5}
Pu-238	7×10^{6}
Pu-239	7×10^{5}
Pu-240	2×10^{6}
Pu-241	7×10^{7}
Pu-242	5×10^{3}
Am-241	7×10^{8}
Am-243	1×10^{7}
Cm-242	4×10^{6}
Cm-244	3×10^{8}

* 250 MW_e in year 2000, phase-out; 1990 reprocessing start. 240,000 MTHM total

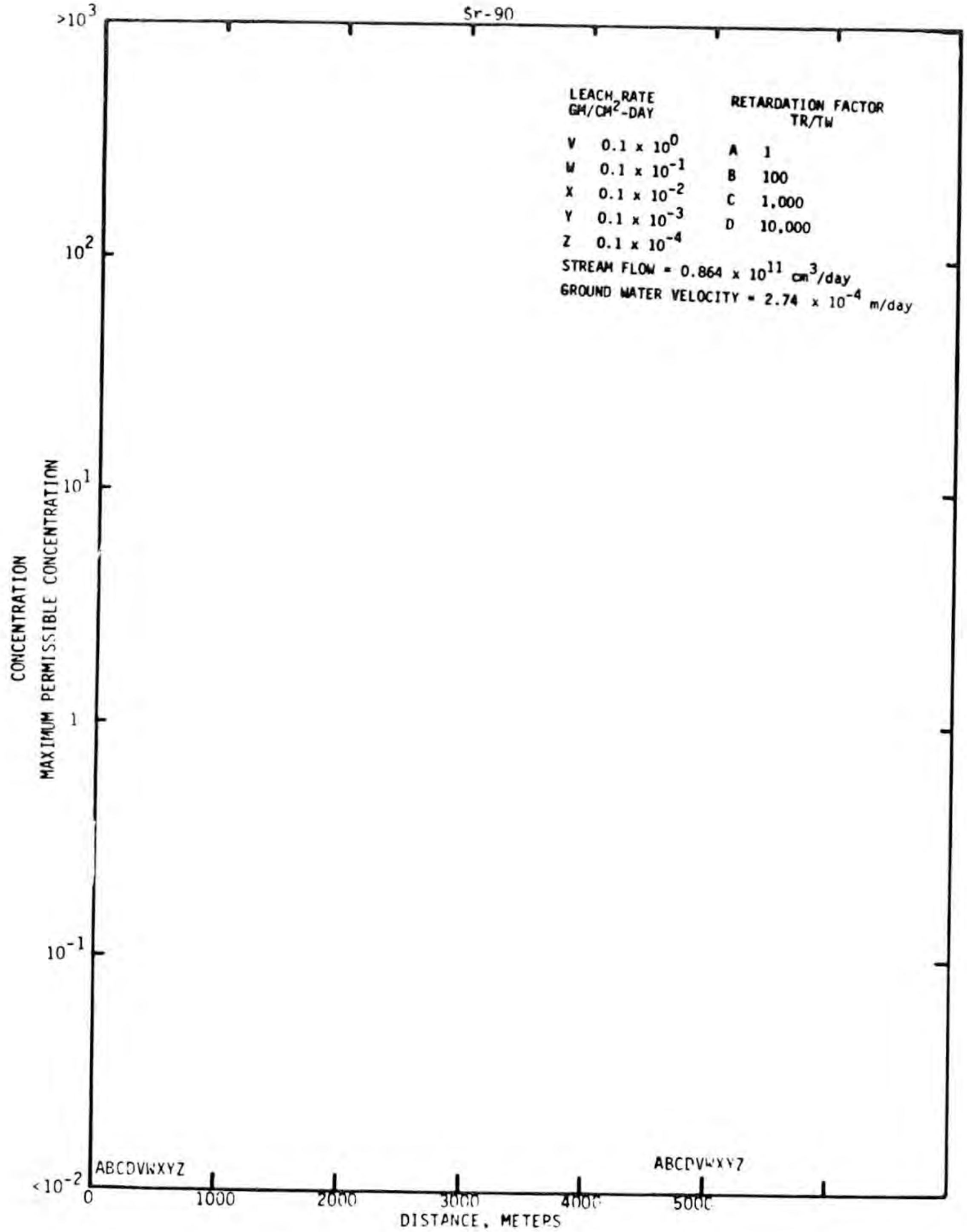

Fig. 1
Strontium-90 Concentration at Groundwater Velocity of 2.74 x 10^{-4} m/day

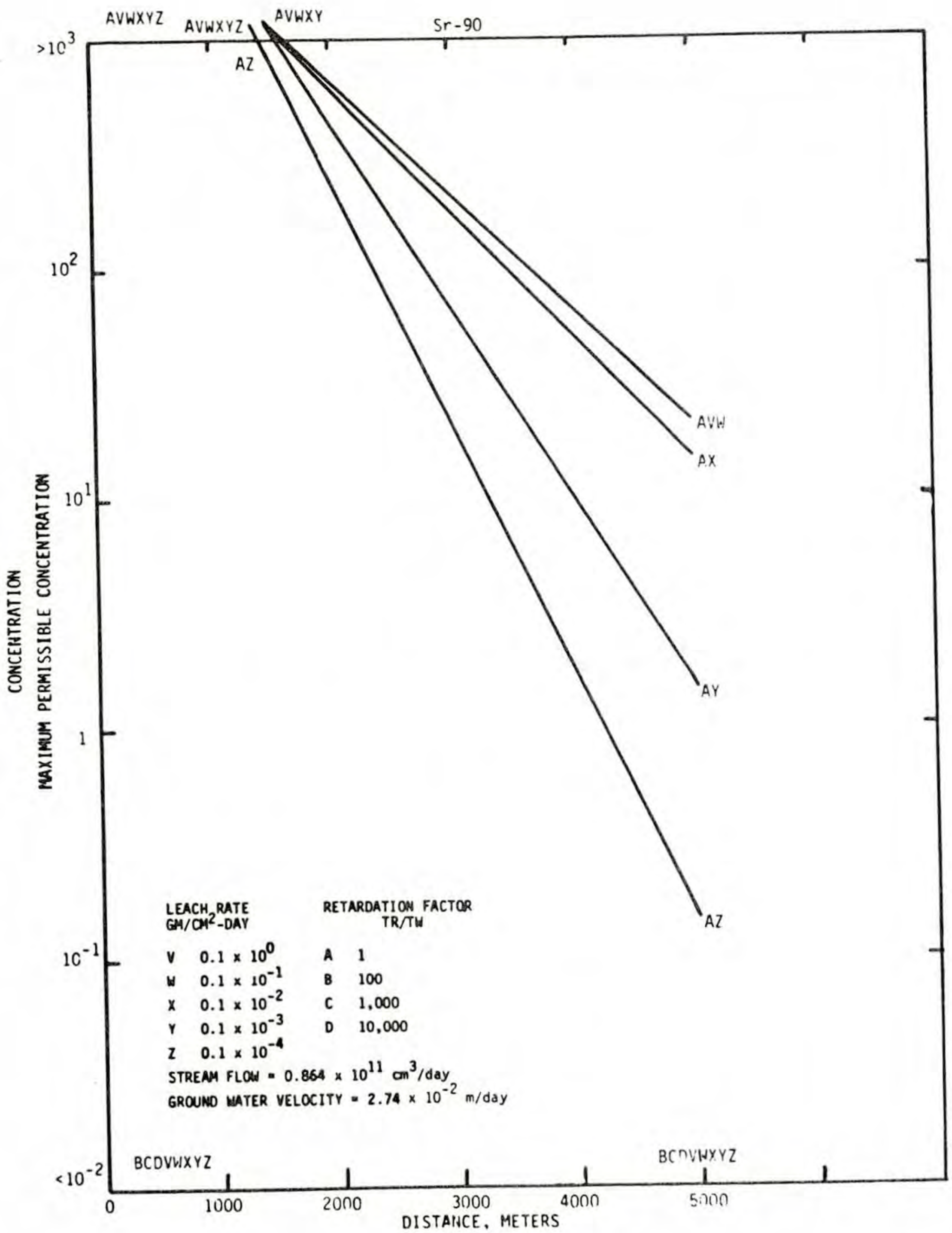

Fig. 2
Strontium-90 Concentration at Groundwater Velocity of 2.74×10^{-2} m/day

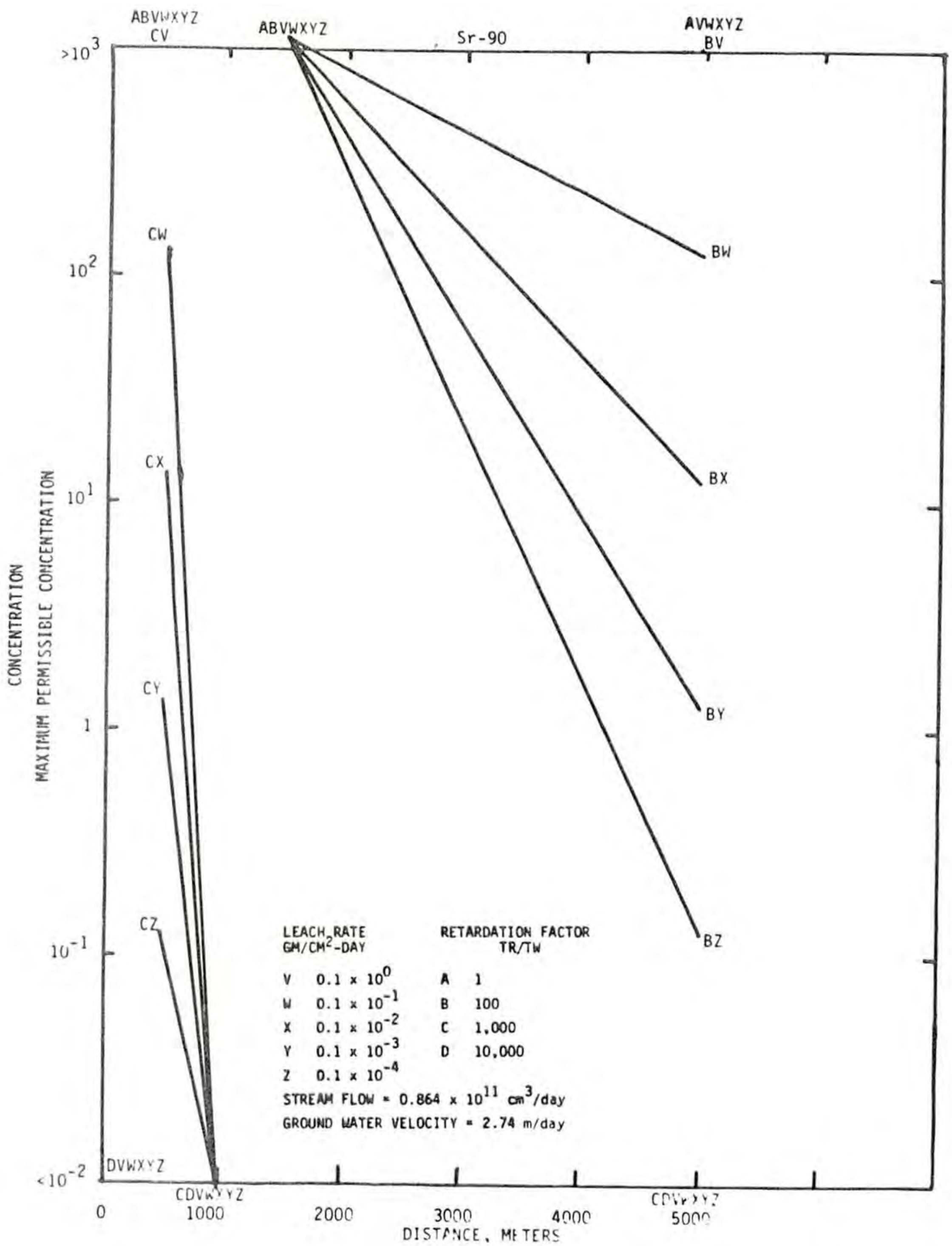

Fig. 3
Strontium-90 Concentration at Groundwater Velocity of 2.74 m/day

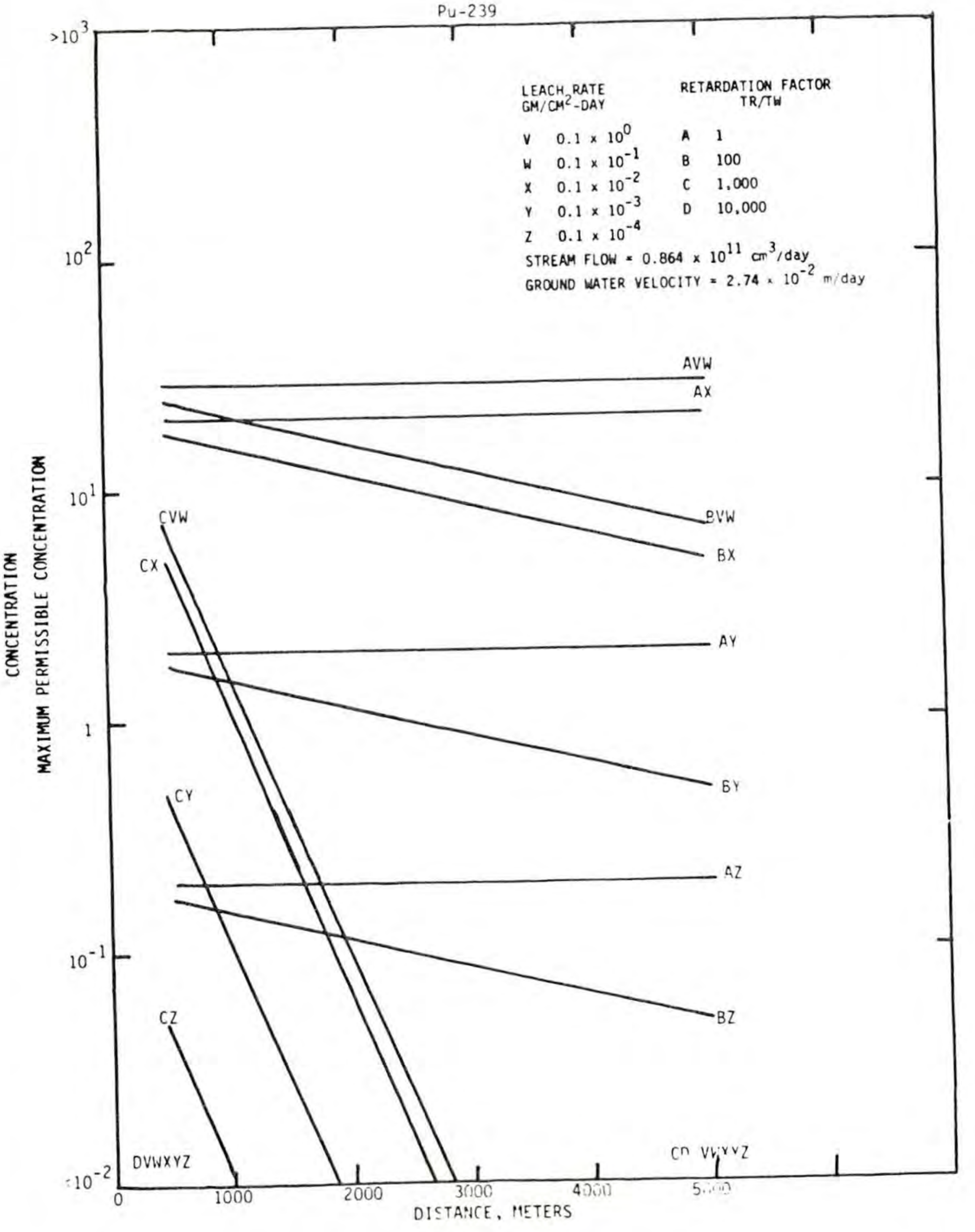

Fig. 4
Plutonium-239 Concentration at Groundwater Velocity of 2.74 x 10^{-2}m/day

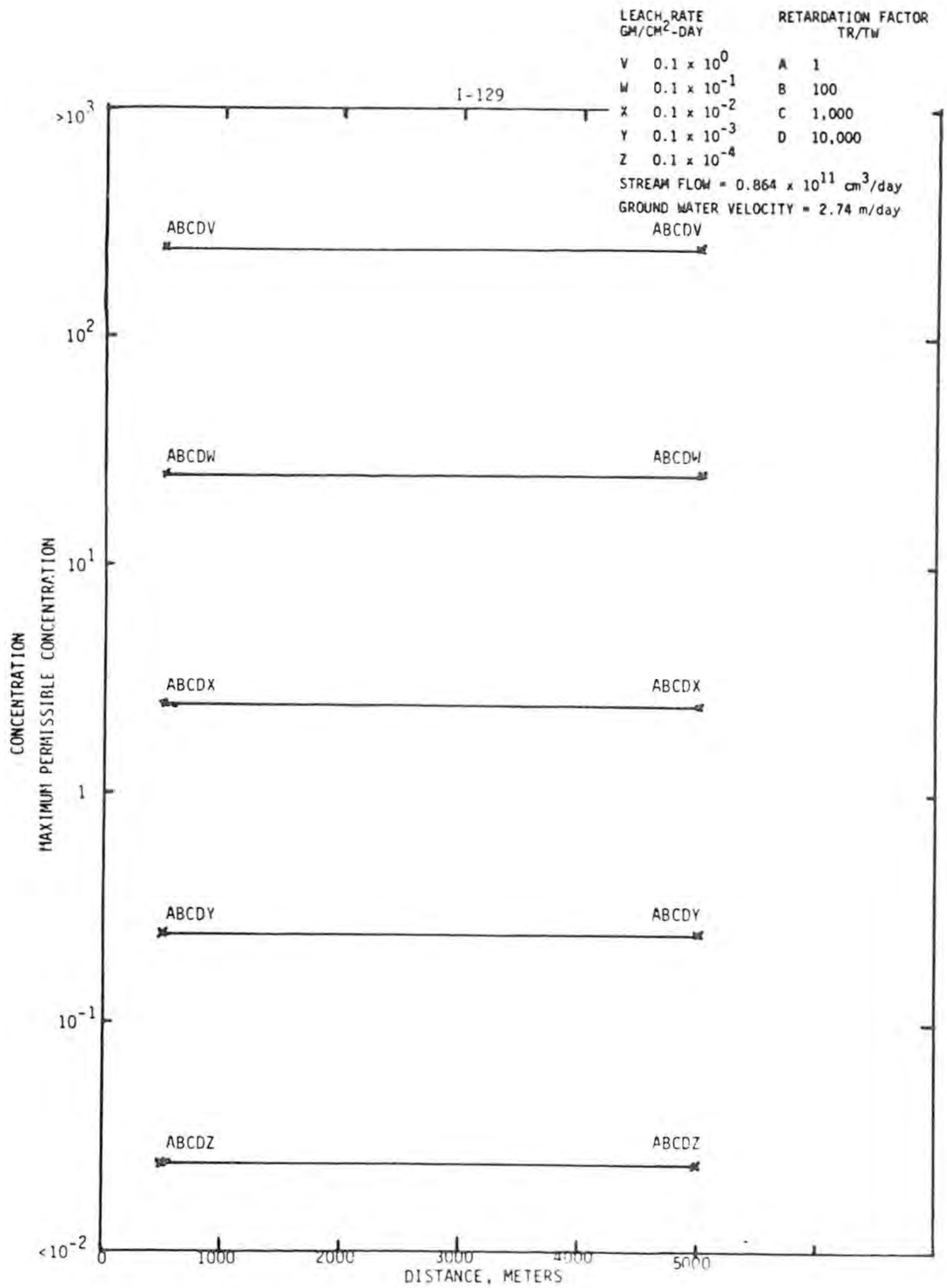

Fig. 5
Iodine-129 Concentration at Groundwater Velocity of 2.74 x 10^{-2} m/day

UNCERTAINTIES IN THE ASSESSMENT OF LONG-TERM COLLECTIVE DOSE AND HEALTH EFFECTS*

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ABSTRACT

This paper summarizes estimates of potential uncertainties in the separate components of a calculation of long-term population dose and health effects resulting from a known release of plutonium to a freshwater surface-water system. The components discussed include (1) radionuclide concentrations in the surface waters, (2) intake by an exposed individual per unit concentration in surface waters, (3) dose to an individual per unit intake, (4) size of the exposed population and its age distribution, and (5) the incremental cancer risk per unit population dose. For each component we discuss an uncertainty based on the range of possible values indicated by available data and an uncertainty based on an expected distribution of values about the mean for the exposed population. The analysis emphasizes significant uncertainties in the fraction of ingested plutonium absorbed into blood from the gastrointestinal tract and the risk factor for induction of bone cancer by alpha-particle irradiation.

INTRODUCTION

Estimates of potential uncertainties in predicting the long-term post-closure performance of a geologic repository for high-level radioactive waste involve consideration of (1) uncertainties in predicting the

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transport of radionuclides from a repository to the biosphere, e.g., via groundwater flow, and (2) uncertainties in predicting the population dose and number of health effects resulting from a release to the biosphere. The major thrust of this symposium has been directed toward the first of these considerations. In this paper, however, we concentrate on the second, and discuss the results of a preliminary study of potential uncertainties in estimates of long-term collective dose and health effects resulting from a known release of a long-lived radionuclide to the biosphere.

Estimation of the uncertainty in a calculation of long-term collective dose and health effects as a function of time after a release to the biosphere involves consideration of the uncertainty in each of the following: (1) the radionuclide concentrations in man's exposure environment as a function of location and time after the release; (2) the radionuclide intake by exposed individuals per unit concentration in the environment; (3) the dose to an exposed individual per unit intake; (4) the number of exposed individuals and their age distribution as a function of time; and (5) the risk of induction of fatal cancers per unit population dose. In this analysis we assume a known release of plutonium to a freshwater surface-water system and consider subsequent exposures via ingestion of contaminated drinking water.

The analysis emphasizes estimates of the potential uncertainty in each of the five separate components listed above for the purpose of identifying those components whose uncertainty contributes significantly to the overall uncertainty in the number of health effects per unit release. No attempt is made, however, to rigorously combine the separate uncertainties to obtain a proper estimate of the uncertainty in health effects. Rather, the largest uncertainties in the separate components are used to provide semi-quantitative estimates of potential uncertainties in population dose and health effects.

Two different types of uncertainties are discussed in this paper. For the first, the uncertainty is described by the range of possible parameter values obtained from available data. This type of uncertainty is appropriate if the primary concern is estimation of any potential effects which might be experienced by any exposed individual. It is important to recognize, however, that the range tends to emphasize extreme parameter values which may occur only with very low probability in an exposed population, particularly if the data on which the range is based are extensive. Therefore, we also consider a second type of uncertainty described by the distribution of values about the mean experienced by the exposed population, e.g., the standard deviation. This measure of uncertainty is appropriate if the primary concern is estimation of collective dose and health effects.

ESTIMATES OF UNCERTAINTIES IN SEPARATE COMPONENTS OF HEALTH EFFECTS CALCULATION

Radionuclide Concentrations

For a release of plutonium to surface waters, the relevant quantity for determining exposures due to ingestion of contaminated drinking water is the time-integrated concentration of plutonium in the receiving waters. We have studied the uncertainty in this quantity using a mixed-tank model given by Eqs. B-25 through B-33 of Ref. [1]. The predicted time-integrated concentration of plutonium in the receiving waters is sensitive primarily to variations in the plutonium distribution coefficient (Kd) and the rate of sedimentation [1]. The range of reported Kd values for freshwater systems is about a factor of 70 [2,3], whereas the sedimentation rate within a large system (the Great Lakes) varies only by a factor of three [1]. The predicted range of time-integrated plutonium concentrations is thus dominated by the range in Kd values and is about two orders of magnitude. For a particular freshwater system, however, the data indicate that the mean Kd value can be determined with a standard deviation of less than one order of magnitude [3]. Therefore, with site-specific data for the distribution coefficient and sedimentation rate, it should be possible to obtain a standard deviation of the time-integrated plutonium concentration for an exposed population of approximately one order of magnitude.

Individual Intake

Ingestion of contaminated drinking water is assumed to be the primary pathway for intake of plutonium by exposed individuals following a release to a freshwater system. The individual intake rate per unit concentration in the receiving waters is proportional to three factors – the total intake rate of water, the fraction of the total water intake from the contaminated receiving waters, and the fraction of plutonium in receiving waters which is transmitted by water treatment systems. An analysis of available data [4] indicates that the range in predicted values of the individual intake rate of plutonium could be as large as two orders of magnitude. For a particular site, however, it should be possible to determine the transmission of plutonium by water treatment systems with negligible uncertainty. Therefore, for collective dose assessments, we require only an estimate of the total intake rate of contaminated water by the exposed population. Available survey data on normal populations [5] indicate that this quantity could probably be determined with a standard deviation of less than one order of magnitude.

Dose per Unit Intake

The analysis of the uncertainty in the dose to an individual per unit intake of plutonium is based on the assumption that bone is the critical organ at risk. The dose to bone per unit intake is defined by the 50-year dose commitment [6],

$$DCF(50) = Sf_1f_2'(1-50\lambda_B)/\lambda_B \ ,$$

where DCF(50) is the 50-year dose-equivalent commitment per unit activity of intake, S is a dosimetric factor giving the dose equivalent per unit residence time of activity in bone, f_1 is the fraction of ingested plutonium absorbed into blood from the gastrointestinal (GI) tract, f_2' is the fraction of absorbed plutonium deposited in bone, and λ_B is the biological removal constant for plutonium in bone in units of 1/yr.

An analysis of available data [4] indicates that the ranges in values of the S factor and the quantity $(1-50\lambda_B)/\lambda_B$ are each only a factor of two, and the range in f_2' is only a factor of four. Furthermore, these ranges are very much less than the range of values for the GI-tract uptake fraction, so that the uncertainty in this parameter essentially determines the uncertainty in the dose per unit intake. In this paper, therefore, we examine in some detail data on the GI-tract uptake fraction for soluble plutonium.

Figure 1 shows selected data for GI-tract uptake of soluble plutonium nitrate in a variety of adult mammals. The nitrate form is expected to be reasonably characteristic of environmental plutonium [16]. The earlier value of 3×10^{-5} adopted for occupational exposures in ICRP Publication 19 [15] was based on the first chronic feeding experiments in the rat [8], whereas the value of 1×10^{-4} recently adopted in ICRP Publication 30 [16] was based on the chronic feeding experiments in the hamster [12].

The data in Fig. 1 show that the range in values of the GI-tract uptake fraction is three orders of magnitude. From the point of view of estimating an uncertainty in the GI-tract uptake fraction applicable to the mean value for an exposed population, the following results in Fig. 1 are significant: (1) consistently high uptake in the rat and guinea pig for plutonium biologically incorporated into or added to feedstuffs [9,11]; (2) an increase in uptake with decreasing mass administered for recent data in the rat [10]; (3) significantly greater uptake following chronic feeding of hamsters compared with acute feeding [12,13];* and (4) relatively low GI-tract uptake in the pig, the animal among those studied which has a digestive tract most similar to man's [8]. If we consider only the recent data in the rat, hamster, and guinea pig involving chronic feeding at low mass levels of intake and incorporation or addition of plutonium into a normal diet, conditions which should be most appropriate for human exposures to plutonium in the environment, it seems that the uncertainty in the GI-tract uptake fraction is about one order of magnitude. If, on the other hand, we also include the available data on the

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*These experiments also showed that uptake of plutonium in the valence state Pu(VI) is the same as uptake of Pu(IV) for both fasted and normally fed hamsters. This result suggests that the early measurement of $f_1 = 2 \times 10^{-2}$ for Pu(VI) in the rat [8] was probably due to extreme fasting and not to a dependence of uptake on the oxidation state of the administered plutonium.

pig because of its greater similarity to man compared with the other mammals, it appears that the uncertainty may be as large as three orders of magnitude. Thus, while the data on individual animal species under similar conditions of intake indicate that the variability in values of GI-tract uptake about the mean for an exposed human population is likely to be about an order of magnitude, the mean value appropriate to that population may be uncertain by three orders of magnitude.

Additional data on GI-tract uptake in the pig would be very useful in estimating a value which is applicable to human populations, especially in light of the apparent trend in Fig. 1 that measurements since 1978 give higher values than earlier results. Data on humans would also be extremely valuable since the applicability of data on other mammals to human experience is always open to question.

Exposed Population

The number of health effects resulting from a release of radionuclides to the environment depends on both the number of exposed individuals and their age distribution. Future projections of exposed populations are particularly speculative because the exposures occur primarily in the geographical region near the point of release of the plutonium to surface waters, and the size of the exposed population depends on migration as well as birth and death rates. An analysis in ref. [4] suggests that projections of exposed populations on a local and regional scale may be uncertain by as much as two orders of magnitude for time periods in the future beyond a few hundred years. The uncertainty in the number of health effects due to the uncertain age distribution of future populations was estimated to be about a factor of two. This uncertainty is negligible compared with the uncertainty in the number of exposed individuals.

Risk of Cancer Induction

In this paper we consider the risk of induction of bone cancers from alpha-particle irradiation using risk factors given in the recent BEIR report [17]. This type of cancer is of interest because many long-lived radionuclides in high-level waste, including plutonium, are alpha-emitting bone seekers.

The cumulative risk of bone sarcomas from alpha irradiation as a function of absorbed dose predicted by the two different models for the cumulative risk coefficient given in Table A-27 of ref. [17] are shown in Fig. 2. Both models are derived from fits to available data on humans, which occur only for high dose levels above 10 Gy (10^3 rads). At expected doses from environmental exposures, i.e., $<10^{-3}$ Gy (10^{-1} rads), however, the cumulative risk predicted by the two models differs by four orders of magnitude or more.

The BEIR report [17] emphasizes that the risk of cancer induction from exposures to radiation at environmental levels is largely unknown and that the estimates of risk such as those in Fig. 2 depend much more on the assumed mathematical form for the dose-response function than on the data themselves. If we assume that the linear risk factor is the appropriate model for estimating cancer risk from alpha irradiation at environment levels of exposure, then the uncertainty in the risk factor due only to the variability in available human data at high doses would be about one order of magnitude [17]. The linear model is perhaps the more reasonable choice because it provides the greater estimates of risk; furthermore, it is difficult on the basis of both animal data and theoretical considerations to rule out a linear component to the dose-effect relationship which should become dominant at low doses. The BEIR report [17] cautions, however, that the linear model does not necessarily provide overestimates of risk (and may, in fact, underestimate risk) because of evidence that the risk per unit dose of alpha radiation increases with decreasing dose rate. There are likely to be additional uncertainties involved in applying limited data on special human and animal populations to a more heterogeneous human population that would be exposed to environmental plutonium, but these uncertainties are difficult to quantify.

SUMMARY AND CONCLUSIONS

The analysis of uncertainties in the five separate components of a calculation of the number of health effects per unit release of plutonium to a freshwater surface-water system is summarized in Table 1. Order-of-magnitude estimates are given for both the range of expected values and an uncertainty in the mean of the distribution of values which would be experienced by an exposed population. The range is appropriate for considering potential effects on each exposed individual, e.g., for the purpose of determining compliance with applicable radiation protection standards, whereas the uncertainty in the mean is appropriate for considering collective dose and health effects.

The uncertainty in the mean in Table 1 has a somewhat different interpretation depending upon the particular parameter. For the environmental concentration and individual intake, the uncertainty in the mean represents a standard deviation which should be achievable on the basis of site-specific measurements. For the dose per unit intake, the smaller value also represents a potential standard deviation of the mean, but the larger value represents the current uncertainty in what the appropriate mean value for an exposed population should be. The uncertainty in the mean for the population is the same as the range of projected values; in this case, the parameter does not have statistical attributes. For the cancer risk the given lower limit is based on the assumption that a linear risk factor is the correct model at low doses. A possible upper limit assuming the linear model cannot be determined, and, furthermore, there is no compelling evidence that this model provides estimates of risk that would actually be experienced in a normal human population.

The analysis in this paper suggests that potential uncertainties in the dose per unit intake, the size of future exposed populations, and the cancer risk per unit dose are most important for determining an uncertainty in the mean for a predicted number of health effects in an exposed population. Uncertainties in environmental concentrations and individual intake appear to be potentially less important. Further experiments could perhaps reduce some of the uncertainties (e.g., measurements of GI-tract uptake of plutonium in man), but it seems unrealistic that the risk factor for cancer induction can be determined for environmental levels of exposure. For the risk factor, it may be more reasonable to use limiting values, which can be estimated from available data, in order to provide conservative estimates of the potential health detriment to man.

In this analysis we have not combined the estimated uncertainties in the separate components to obtain an estimate of the overall uncertainty in the number of excess health effects per unit release of plutonium to the biosphere. We caution that simply combining ranges of values for the separate components to obtain a range of possible effects tends to place misleading emphasis on extreme results which will occur only with a very low probability in any exposed population. Based only on the uncertainties in the second column of Table 1, we may crudely estimate an overall uncertainty in health effects of at least 2 to 3 orders of magnitude, exclusive of any additional unknown uncertainties in the cancer risk factor which could increase the overall uncertainty. A more quantitative and rigorous analysis of the overall uncertainty must be performed with care because of likely correlations among some of the different components, particularly correlations of the intake rate, dose, and cancer risk with age at exposure. Such an analysis should take into account estimates of probability distributions of the separate components over their assumed ranges in order to generate probability distributions for the expected number of health effects.

REFERENCES

1. Liquid Pathway Generic Study, NUREG-0440 (February, 1978).

2. W. C. Hanson (ed.), Transuranic Elements in the Environment, DOE/TIC-22800, U. S. Dept. of Energy (1980).

3. A. H. Seymour, A. Nevissi, W. R. Schell, and A. Sanchez, Distribution Coefficients for Radionuclides in Aquatic Environments. I. Development of Methods and Results for Plutonium in Fresh and Marine Water-Sediment Systems, NUREG/CR-0801 (October, 1979).

4. D. C. Kocher, R. W. Leggett, D. E. Dunning, Jr., M. T. Ryan, and K. F. Eckerman, Uncertainties in the Calculation of Long-Term Collective Dose and Health Effects – A Preliminary Assessment, NUREG/CR-1303 (ORNL/NUREG/TM-378) (June, 1980).

5. F. O. Hoffman and C. F. Baes III (eds.), A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides, NUREG/CR-1004 (ORNL/NUREG/TM-282) (October, 1979).

6. G. G. Killough, D. E. Dunning, Jr., S. R. Bernard, and J. C. Pleasant, Estimates of Internal Dose Equivalent for 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel Cycle Facilities, Vol. I, NUREG/CR-0150 (ORNL/NUREG/TM-190) (June, 1978).

7. J. Katz, H. A. Kornberg, and H. M. Parker, "Absorption of Plutonium Fed Chronically to Rats. I. Fraction Deposited in Skeleton and Soft Tissues Following Oral Administration of Solutions of Very Low Mass Concentration," Am. J. Roentgenol. Radium Therapy Nucl. Med. 73, 303 (1955).

8. M. H. Weeks, J. Katz, W. D. Oakley, J. E. Ballou, L. A. George, L. K. Bustad, R. C. Thompson, and H. A. Kornberg, "Further Studies on the Gastrointestinal Absorption of Plutonium," Radiat. Res. 4, 339 (1956).

9. J. E. Ballou, K. R. Price, R. A. Gies, and P. G. Doctor, "The Influence of DTPA on the Biological Availability of Transuranics," Health Phys. 34, 445 (1978).

10. M. F. Sullivan, "Absorption of Actinide Elements from the Gastrointestinal Tract of Rats, Guinea Pigs, and Dogs," Health Phys. 38, 159 (1980).

11. M. F. Sullivan, T. R. Garland, D. A. Cataldo, R. E. Wildung, and H. Drucker, "Absorption of Plutonium from the Gastrointestinal Tract of Rats and Guinea Pigs after Ingestion of Alfalfa Containing ^{238}Pu," Health Phys. 38, 215 (1980).

12. J. W. Stather, J. D. Harrison, P. Rodwell, and A. J. David, "The Gastrointestinal Absorption of Plutonium and Americium in the Hamster," Phys. Med. Biol. 24, 396 (1979).

13. J. W. Stather, J. D. Harrison, H. Smith, P. Rodwell, and A. J. David, "The Influence of Fasting and Valence on the Gastrointestinal Absorption of Plutonium in Hamsters and Rabbits," Health Phys. 39, 334 (1980).

14. L. K. Bustad, W. J. Clarke, L. A. George II, V. G. Horstman, R. O. McClellan, R. L. Persing, L. J. Seigneur, and J. L. Terry, "Preliminary Observations on Metabolism and Toxicity of Plutonium in Miniature Swine," Health Phys. 8, 615 (1962).

15. The Metabolism of Compounds of Plutonium and Other Actinides, International Commission on Radiological Protection Publication 19, Pergamon Press, Oxford (1972).

16. Limits for Intakes of Radionuclides by Workers, International Commission on Radiological Protection Publication 30, Part 1, Pergamon Press, Oxford (1978).

17. The Effects on Populations of Exposure to Low Levels of Ionizing Radiation, National Academy of Sciences – National Research Council, Washington, DC (1980).

Table 1. Estimated uncertainties in separate components of health effects calculation for plutonium

Component	Range (Orders-of-magnitude)	Uncertainty in mean (Orders-of-magnitude)
Environmental concentration	2	1
Individual intake	2	<1
Dose per unit intake	3	1-3
Population and age distribution	≤ 2	≤ 2
Cancer risk per unit dose	≥ 4	≥ 1

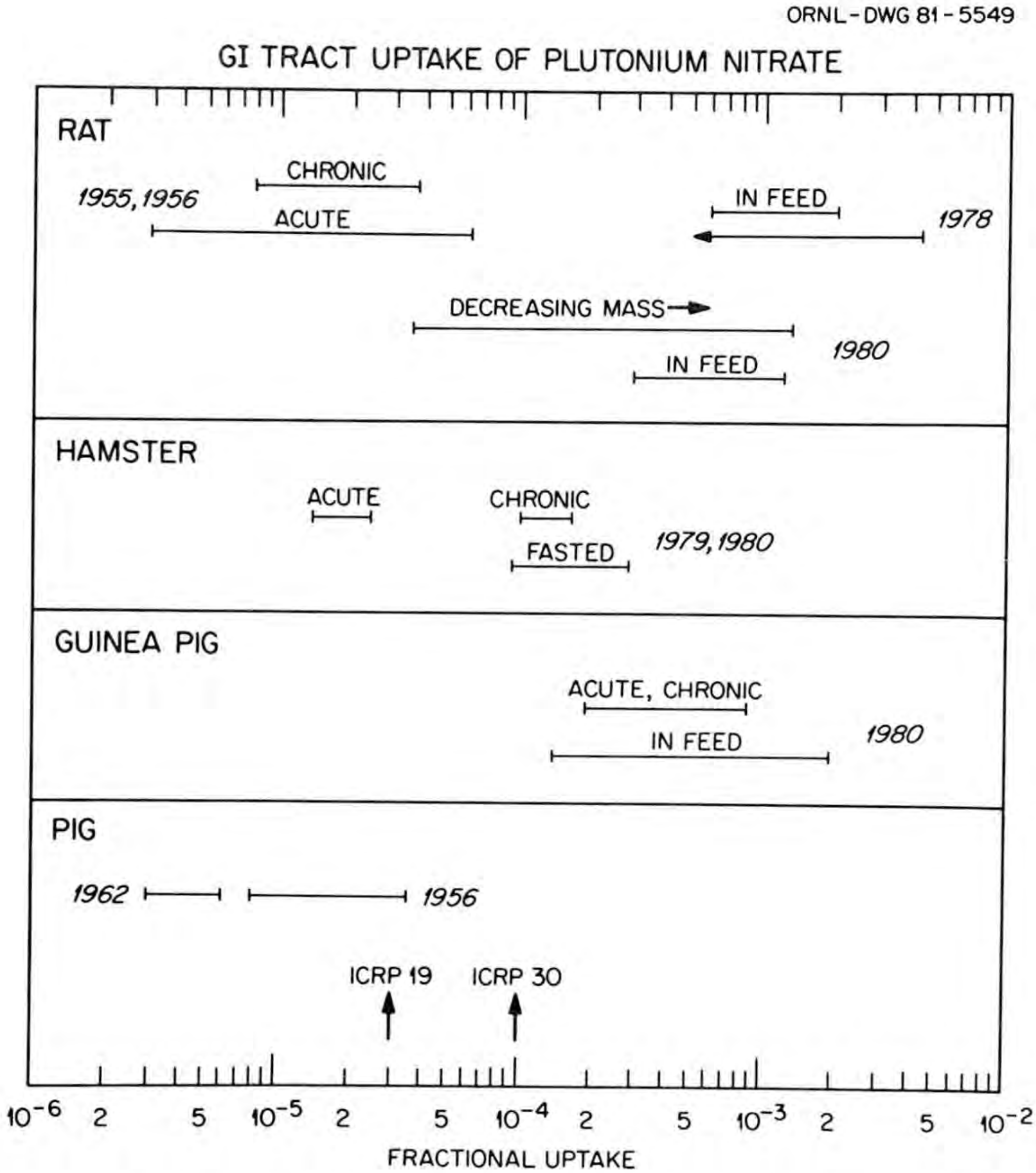

Fig. 1 - Selected data for GI-tract uptake of plutonium nitrate in mammals. The sources of the data are as follows: rat - ref. [7] (1955), ref. [8] (1956), ref. [9] (1978), and ref. [10,11] (1980); hamster - ref. [12,13] (1979, 1980); guinea pig - ref. [10,11] (1980); and pig - ref. [8] (1956) and ref. [14] (1962). The values adopted for occupational exposures in ICRP Publications 19 [15] and 30 [16] are indicated at the bottom of the figure.

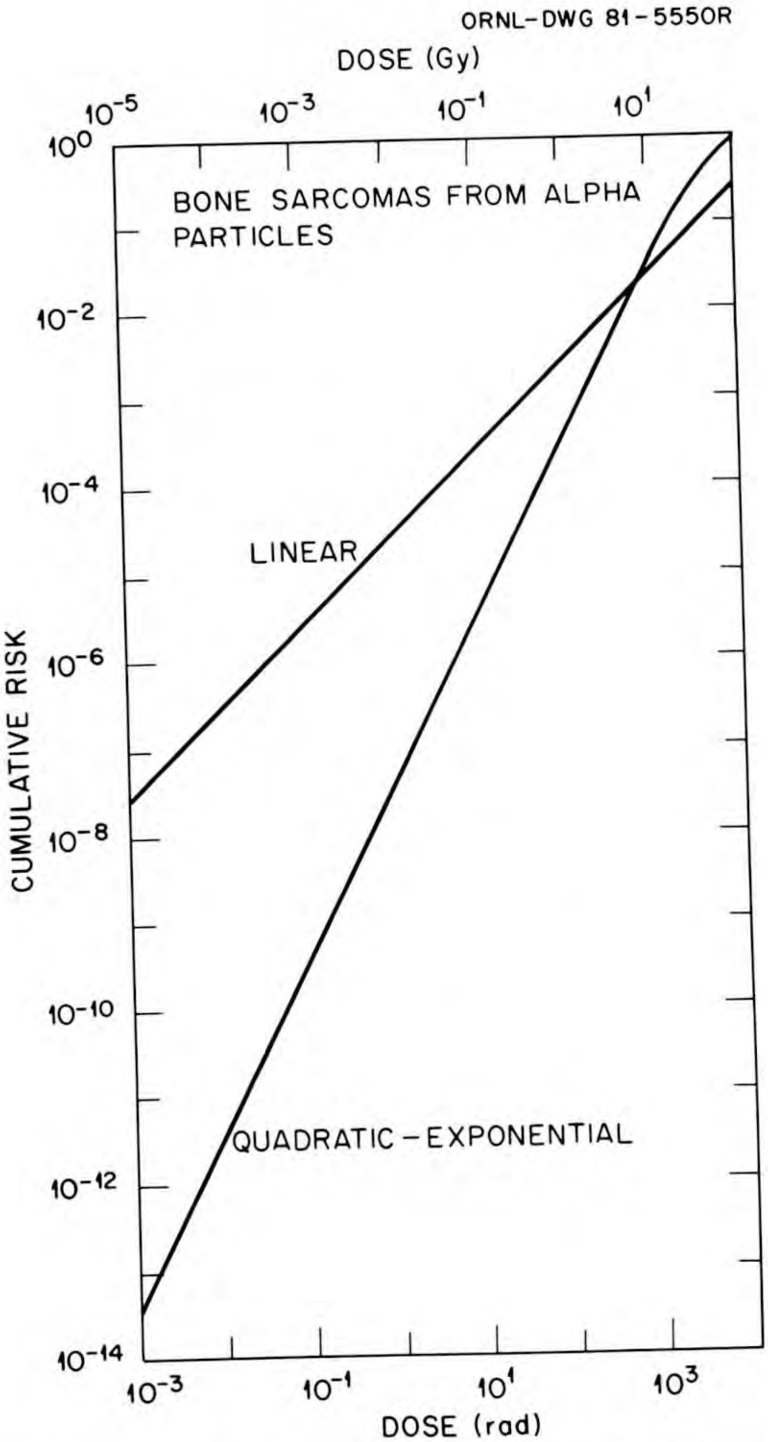

Fig. 2 - Cumulative risk of induction of bone sarcomas vs. absorbed dose of alpha radiation for two different models of the cumulative risk coefficient given in Table A-27 of ref. [17].

Session III:

SPECIAL PROBLEMS

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Sandia National Laboratories

MULTIPLE BARRIER WASTE PACKAGE PERFORMANCE ASSESSMENT

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ABSTRACT

A performance assessment model for multiple barrier packages containing unreprocessed spent fuel has been developed and applied to several package designs. The objective of this work was to develop input to programmatic decision making concerning engineered barrier package development. Package performance is determined in terms of time to initial release and period of time over which release occurs. The model contains a state-of-the-art corrosion rate data base which includes pitting, crack growth, and graphitization, as well as bulk corrosion. Corrosion rates for oxic and anoxic conditions at each of two temperature ranges (25-100 and 100-250^oC) are used. The model uses a rigorous treatment of radionuclide release after contact of the waste with water which includes resistance of damaged barriers and special backfills, temperature calculations that account for convection and radiation, a subroutine to calculate nuclear gamma radiation field at each barrier surface, and detailed stress calculations. The model was used to assess post-repository closure performance for several designs which were all variations of basic designs from the Spent Unreprocessed Fuel (SURF) program. Although the data base is limited, the calculation results suggest that waste packages can be designed which will not completely degrade for at least a few hundreds of years in salt, basalt, granite, and shale media. The calculation results suggest that delay times for radionuclide discharge from the package to the repository on the order to 10^5 to 10^7 years may be possible using a few inch thickness of intact sorption backfill.

INTRODUCTION

This paper discusses recent work to develop and apply a model for assessment of the waste isolation capabilities of multi-barrier engineered waste packages emplaced in deep geologic repositories. Uncertainties encountered and methods used for dealing with such uncertainties are highlighted. Further modifications and applications of the model are currently in progress.

Objectives

The objective of the work discussed was to develop an assessment model from state-of-the-art analytical models and data concerned with various near-field effects on the waste package. The results of the analysis are intended to be rough (order of magnitude) estimates of the expected release-free package life and nuclide release characteristics after package failure. The chief use of such estimates has been to carry out comparisons of different package designs. Such compositions have been useful in providing design and development program focus.

Scope

To date the work has been confined to studies of packages to contain a single PWR fuel bundle as described in the commercial waste EIS[1] which is 3.3 percent enriched with 33,000 MWD/MTU burnup and 6.5 year cooled. Time zero in the analysis is the time of repository closure. The analysis scenario is a flooded repository. The study was directed toward determining time at which repository water contacts the waste (leach begin time) and nuclide release rate for specific nuclides during the time thereafter.

Waste Package Design

The basic type of design in the study was a set of multiple cylindrical containers with a specialized backfill surrounding the unit. For the purpose of the study the "package" was considered to include the backfill. This is not necessarily consistent with some other documents. One of the multiple cylinders may actually be a sleeve placed in the emplacement borehole. Figure 1 depicts a half cross-section of a maximum package. A package concept may contain some or all of the elements depicted. The stabilizer is a filler placed around an unreprocessed, intact fuel bundle and may be a segmented solid, cast-in-place solid or a gas (e.g., air, nitrogen, argon, or helium). All designs considered in the study are variations on this basic concept.

MODEL DESCRIPTION

The BARIER model treats the package as a series of barrier elements. Each is successively attacked as repository water infiltrates the package. Once a particular barrier element is breached due to stress loads or is removed by bulk corrosion then attack of the next inner barrier proceeds. Once the water reaches the waste form a nuclide release model calculates the release profile.

A Barrier Element

Figure 2 is a schematic of a barrier element in the model. The figure shows a 1/2 longitudinal cross section through the cylinder of the element. All the possible components of a given element are shown. All or some of these components may be present in a particular element. The solid wall may be composed of two materials. Material #1 is the base material and is treated as a structural material. Material #2 is a cladding with no structural strength. Between the composite wall and the next element there may be either a gap filled with gas or a filler material. The presence or absence of the individual components of one element is conveyed by setting the diamter boundaries of each component. That is, if the outside diameter of material #1 is set equal to the outside diameter of material #2 then there is no cladding in that element. Each layer shown in Figure 1 (the maximum package) would be represented as an element in Figure 2. For example, the overpack and its adjacent filler to the outside.

The Performance Model

The performance model calculates the behavior in time of a package which is a composite of elements described above.

Assumptions and Ground Rules. The model assumes that all packages behave exactly the same and the one calculated is representative of all the packages in the repository. The model is therefore deterministic and the calculational methods are biased to provide the earliest failure time (therefore conservative). The repository is assumed to be sealed and flooded with characteristic ground water. There is little or no velocity of flow in the repository water. It is assumed that the package of concern has had a "normal" history. That is, it has suffered no damaging events and is manufactured to the the expected standards.

The model is concerned with no special events such as earthquakes, volcanism, direct human intrusion, etc. The performance behavior calculated is related to the sure, slow degradation of the package and resulting subsequent release of waste materials.

Model Procedure. A diagram of the program flow is shown in Figure 3. The basic procedure in the package degradation phase is a time step routine. At each time step a heat transfer model (conduction, convection, thermal radiation) calculates the temperature profile in the package. A nuclear radiation model then calculates the radiation field at the outside of the wall of current concern (this will later be coupled to the corrosion model). The corrosion model then calculates the amount of bulk or local corrosion for this time step. The next step is a calculation of the mechanical stresses to allow a check for barrier element integrity. The existing wall thickness is compared to the necessary wall thickness to withstand forces from repository lithostatic or from thermal pressure loads. It is then determined whether the barrier is intact or has failed. If the barrier is intact the

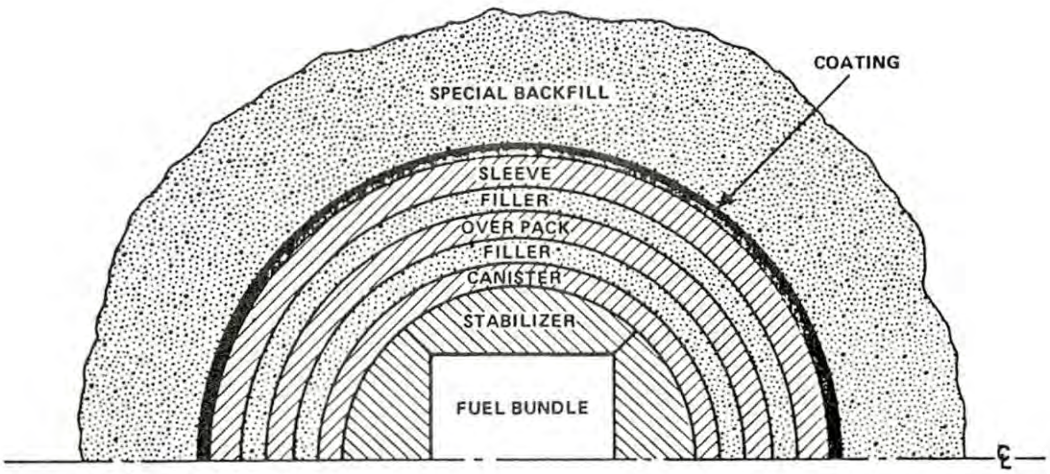

Figure 1. Maximum Barrier Package for Unreprocessed Spent Fuel.

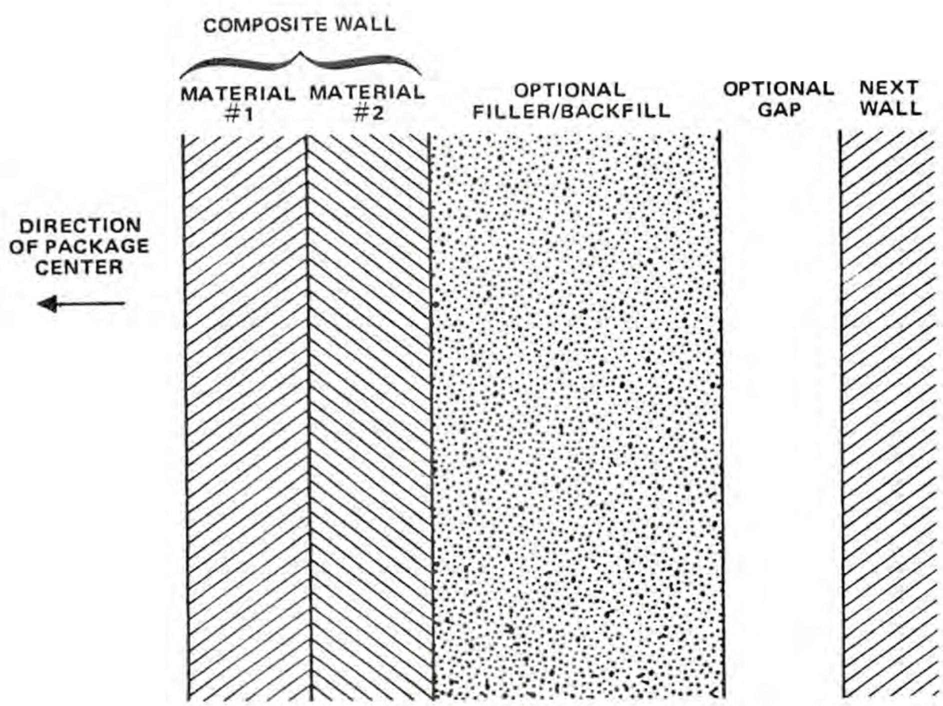

Figure 2. Barrier Element in the Performance Model.

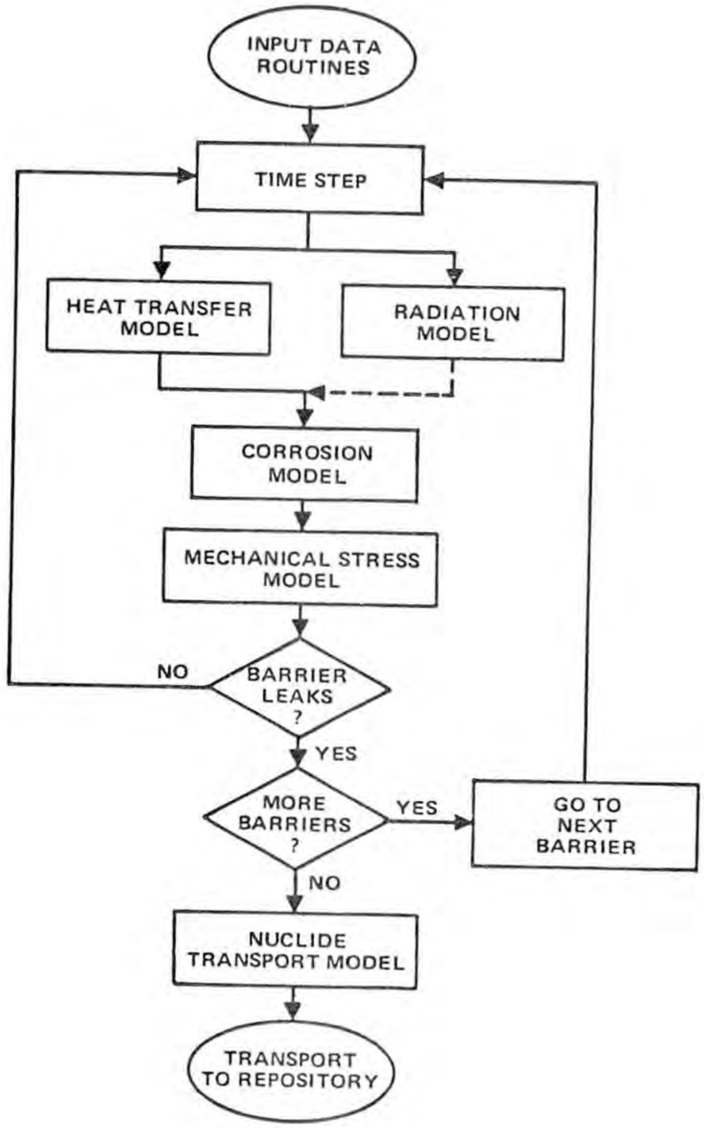

Figure 3. Schematic of Performance Model.

calculations are repeated for the next time step. If the barrier has failed and is leaking, then the next inner element is subjected to corrosion and the process is repeated. When the last element is penetrated the nuclide transport model then calculates the nuclide release profile.

DISCUSSION OF UNCERTAINTIES

The individual models in the procedure described above were developed to provide conservative best estimates and be compatible with the quality and quantities of data available to support the models. Therefore, the individual models are of varying complexity and uncertainty. This section contains discussion of the significant uncertainties encountered during the performance assessment work.

Corrosion Model. The corrosion model used for the study is a constant rate model. Temperature dependence is handled in discrete ranges. (Within a range of temperature one rate is used.) Different mechanisms are considered by choosing the largest of bulk rate, pitting rate, crack growth rate, or graphitization rate. The effect of radiation on the corrosion rate has not yet been incorporated.

The corrosion rates used involve significant uncertainties and are considered to be order-of-magnitude estimates at best. Much of the uncertainty lies in the lack of data at precisely the right temperature, pressure and water chemistry conditions of concern. Interpolations and occassionally extrapolation to the necessary set of conditions must be made. Of particular conern is the need for better localized corrosion data. A further uncertainty is due to the constant rate assumption. Most corrosion engineers observe long term corrosion to be a self-limiting phenomena. As time goes on the rate generally decreases significantly. Unfortunately the rate reduction occurs over very long times and is due to complex circumstances. The current data and phenomenological understanding are insufficient to allow more than a simple linear extrapolation in time. The results are likely to be very conservative showing loss rates of metals much higher than would actually occur over long periods. The model will tend to predict package failure much earlier than actual times due to this problem. The result is the tendency toward costly over design to account for large uncertainties.

Mechanical Stress Model. The mechanical stress model is an analytical, multi-layer equilibrium stress calculation. The model is a rigorous treatment and has been validated against recognized stress models such as the STEALTH code. Nevertheless it is simplified in its geometric treatment and assumes that the forces are non-directional (lithostatic). Effects of non-uniform shearing or localized stress are not accounted for. In that sense it may make less-than-conservative predictions.

Data on material properties (bulk modulus, shear modulus) for wall materials and especially backfill materials are lacking. This is particularly true for specific conditions of concern. We have used conservative numbers where estimates had to be made. This again tends to guide design activities toward costly overdesign.

Nuclide Transport Model. The major source of uncertainties in the nuclide transport model is the lack of data. Two key types of data needed are solubility data for specific nuclides and equilibrium sorption coefficients for specific nuclides in a particular backfill at condition of interest. Rough estimates have been used for these calculations with an effort to bias on the conservative side.

RESULTS

A large number of design variations in both creeping media (salt) and hardrock (basalt, granite) have been investigated. This section summarizes some results obtained for a sampling of the cases considered. Note that these results are not to be considered conclusive or necessarily proof that long-life packages can be provided. Rather the results are useful for comparison of alternatives for further development and to indicate that long-life packages are worth considering in comprehensive development programs.

Salt Repository

Simple cases for the salt repository are summarized in Table 1. These results are for oxic conditions in the repository. Except for the unclad stainless steel case the cast stabilizer packages are long lived because the waste form is not crushable and only bulk loss of containment breaches the package. Because of the aggressive environment the stainless steel can fails early due to local corrosion (pitting or cracking). This is remedied by the addition of a thin zircaloy cladding to protect the can. The case with a segmented steel (crushable) stabilizer, B1.20N, displays a long life time due to a heavy sleeve liner in the borehole which is also protected by zircaloy.

Figure 4 shows typical profiles for Pu-239 release from the package after failure some key values were also summarized in Table 1). These results, which show very long release times, are typical of most results obtained.

Hard Rock Repository

Sample results for the hard rock repository are shown in Table 2. The results are similar to those for salt. Packages are computed to last somewhat longer mainly due to the less aggressive environment encountered. Note that an unclad canister inside a simple cast iron sleeve looks promising (concept BE.34N). Nuclide release profiles are

Table 1. Sample Results for Packages in a Salt Repository, Oxic Conditions.

Package Number	PACKAGE DESCRIPTION			Time to Leach Begin (Yrs)	Pu-239 TRANSPORT		
	Component	Material	Thickness (CM)		Peak Discharge Time (Yrs)	Peak Discharge Rate (Ci/Yr)	Rate (Ci/Yr) at 10^5 Years
BE.1N	Stabilizer	Cast Lead	–	35	2.9×10^5	8.4×10^{-11}	7×10^{-14}
	Canister	304 SST	0.64				
	Sleeve	Iron	0.64				
	Backfill	Sand-Bentonite	42.				
BE.9N	Stabilizer	Cast Lead	–	2,500	2.9×10^5	5.8×10^{-10}	5×10^{-12}
	Canister	Zircaloy	0.64				
	Sleeve	Iron	9.0				
	Backfill	Sand-Bentonite	34.				
BE.6N	Stabilizer	Cast Lead	–	1,500	2.9×10^5	9.1×10^{-11}	8×10^{-14}
	Canister	304 SST	0.64				
	Sleeve	Iron	0.64				
	Sleeve Clad	Zircaloy	0.4				
	Backfill	Sand-Bentonite	42.				
BI.20N	Stabilizer	Steel Segments	–	1,500	2.8×10^5	6.1×10^{-10}	6×10^{-12}
	Canister	Steel	0.64				
	Sleeve	Iron	9.				
	Sleeve Clad	Zircaloy	0.4				
	Backfill	Sand-Bentonite	33.				
E.15N	Stabilizer	Cast Lead	–	2,500	2.9×10^5	1×10^{-9}	4×10^{-11}
	Canister	Zircaloy	0.64				
	Backfill	Sand-Bentonite	28.0				

Table 2. Sample Results for Packages in a Hard Rock Repository, Oxic Conditions.

Package Number	PACKAGE DESCRIPTION			Time to Leach Begin (Yrs)	Pu-239 TRANSPORT		
	Component	Material	Thickness (CM)		Peak Discharge Time (Yrs)	Peak Discharge Rate (Ci/Yr)	Rate (Ci/Yr) at 10^5 Years
BI.9N	Stabilizer	Steel Segments	–	12,000	2.9×10^5	5.5×10^{-10}	4×10^{-12}
	Canister	Steel	0.64				
	Sleeve	304 SST	9.				
	Sleeve Clad	Zircaloy	0.13				
	Backfill	Sand-Bentonite	33.				
BE.15N	Stabilizer	Cast Lead	–	1,300	2.9×10^5	8.2×10^{-11}	7×10^{-14}
	Canister	Inconel	0.64				
	Sleeve	Iron	0.64				
	Backfill	Sand-bentonite	42.0				
BE.34N	Stabilizer	Cast Lead	–	1,800	2.9×10^5	5.8×10^{-10}	5×10^{-12}
	Canister	304 SST	0.64				
	Sleeve	Iron	9.				
	Backfill	Sand-Bentonite	34.				

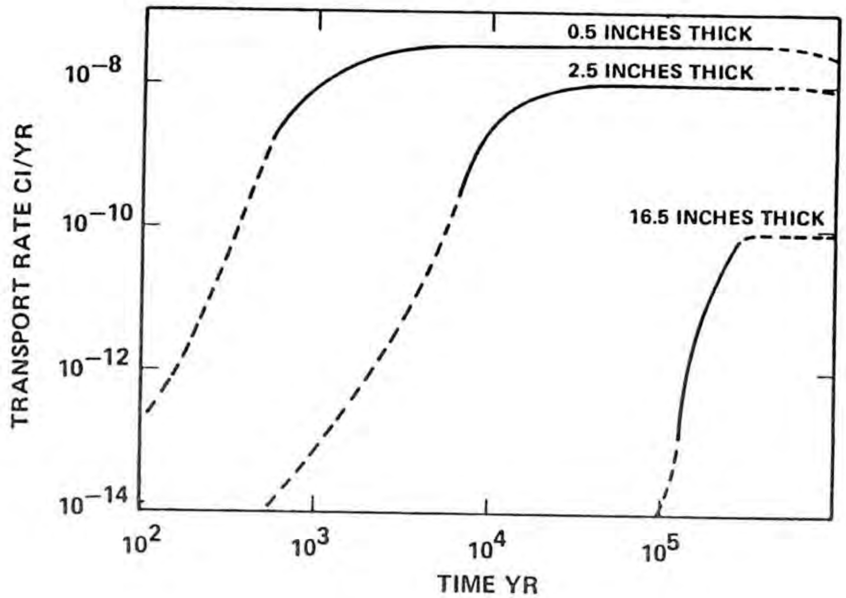

Figure 4. Pu-239 Transport from BE Concepts (Iron Sleeve, SST Can) with Various Backfill Thickness Spent Fuel Placed in Salt.

essentially the same for these packages as they were in the salt repository.

REFERENCES

1. U. S. Department of Energy, Final Environmental Impact Statement, Management of Commercially Generated Radioactive Waste, DOE/EIS-0046F, (Oct. 1980).

PREDICTING THE PERFORMANCE OF A BACKFILL BARRIER TO RADIONUCLIDE MIGRATION FOR A REPOSITORY IN SALT

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(Paper Not Submitted)

THE ROLE OF HYDRAULIC CONDUCTIVITY DATA IN REDUCING UNCERTAINTY IN RADIONUCLIDE TRANSPORT MODELING

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ABSTRACT

Difficulties in characterizing hydrogeological conditions at a potential repository site will create uncertainty in model predictions of radionuclide transport in groundwater flow systems. In this paper we carry out a series of stochastic simulations to investigate the reduction in that uncertainty when hydraulic conductivity data are available to characterize the heterogeneous nature of the porous medium. Results suggest that unless a considerable number of field measurements are available, transport predictions will be subject to large uncertainties.

INTRODUCTION

Model studies of the far-field hydrogeological environment will form an integral part of any site evaluation program for a high level radioactive waste repository. It is therefore important to develop an appreciation for the magnitude of uncertainties in model predictions of radionuclide transport in groundwater flow systems. Previous research has indicated that mass transport is very sensitive to the heterogeneity in hydraulic conductivity [1]. Accordingly, the hydraulic conductivity variations at a specific site are considered to be a realization of a stochastic process. The parameters describing this stochastic process include the mean and standard deviation of the lognormal probability density function for hydraulic conductivity [2], and terms describing the spatial autocorrelation between neighboring values of hydraulic conductivity [3]. A prime concern in waste management is to determine how the sampling program will influence the level of confidence which can be placed in predictions of radionuclide migration. In this paper we investigate the reduction in uncertainty when various quantities of hydraulic conductivity data are available to constrain the patterns of variability in hydraulic conductivity.

METHODOLOGY

The flow system investigated is shown in Figure 1. Radionuclides are assumed to enter the flow system as an instantaneous pulse at the start of the simulation. An initial requirement for this study is a two- dimensional cross section along which the variation in hydraulic conductivity is exactly defined. Because such information is not available for any real system, it is necessary to generate this field from a known stochastic process. This realization is then our analog of a real-world site. To construct the hypothetical field section, the porous medium is divided into a set of discrete blocks. Values of hydraulic conductivity are generated for each block using an algorithm that allows neighboring block values to be correlated [3]. The parameters describing the heterogeneity are independent of location within the flow domain (statistical homogeneity).

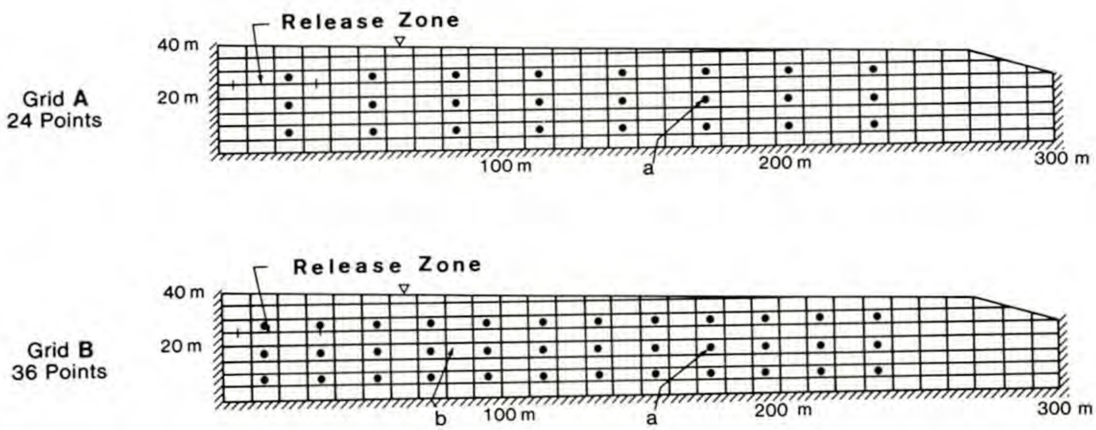

Fig. 1 Flow domain and sample grids used to select hydraulic conductivity values from hypothetical field section.

This study considers 2 different sample grids. Each dot shown on the grids in Figure 1 corresponds to a location where a hydraulic conductivity value has been obtained. Each column of the sample grid can be thought of as a borehole with measurements obtained at three different depths. To simplify the analysis, the possibility of measurement error is not considered.

Using Monte Carlo simulation, frequency histograms on model output are formed by the repetitive simulation of mass transport in a large number of realizations. Conditional simulation techniques [4] allow field measurements to be incorporated in each realization. Each realization can be thought of as one possible representation of the actual conditions at the repository site. Due to the spatial autocorrelation, known values of hydraulic conductivity exert an influence not just at the point where the value is fixed but over a surrounding neighborhood [5].

Transport in each realization is simulated using a hybrid deterministic-probabilistic technique [6]. A summary of simulation parameters follows. The parameters μ_y and σ_y define the mean and standard deviation of the log (base 10) hydraulic conductivity distribution. The estimates of these parameters formed from the measurements on the sample grids are denoted $\overline{Y_c}$ and S_{yc}; respectively. The integral scales $\overline{\lambda_x}$, $\overline{\lambda_z}$ are a measure of the average distance (in meters) over which neighboring values of hydraulic conductivity are correlated [3]. Porosity is assumed constant with the value μ_n. The number of realizations comprising the Monte Carlo simulation is denoted MC. Finally, the transport simulation is based on tracking NP reference particles using a timestep ΔT.

RESULTS

The hydraulic conductivity realization serving as the hypothetical field section is contoured in Figure 2. Simulation of transport yields a time of initial mass arrival at the water table of 28000 days. The maximum quantity of mass crosses the water table at 41500 days. The time of last arrival is 64000 days.

Figure 3 shows a series of nonnormalized frequency histograms on selected transport times for a reference case and conditional simulations of this field realization using sample grids A and B. The reference case represents a situation where no hydraulic conductivity values are fixed. In this case, the parameters of the hydraulic conductivity distribution are the same as those used to construct the hypothetical section.

Study of Figure 3 indicates that fixing the 24 values of hydraulic conductivity reduces the coefficient of variation (ν) in the time of initial arrival at the water table by 13.0% relative to the reference case. A decrease in the coefficient of variation can be taken as a measure of the reduction in uncertainty in prediction. Using sample grid B with 36 measurement points leads to a reduction of 31.3%. The coefficient of variation in the time at which the maximum quantity of mass crosses the water table shows no difference between the reference case and the simulation with 24 measurement points. Fixing the 36 hydraulic conductivity values leads to a reduction of 17.5%. For the time of last arrival at the water table, no significant difference in the coefficient of variation occurs when preserving the 24 data points; while a 15.9% reduction is observed for the 36 data points.

Hydraulic Conductivity Field (m/day)

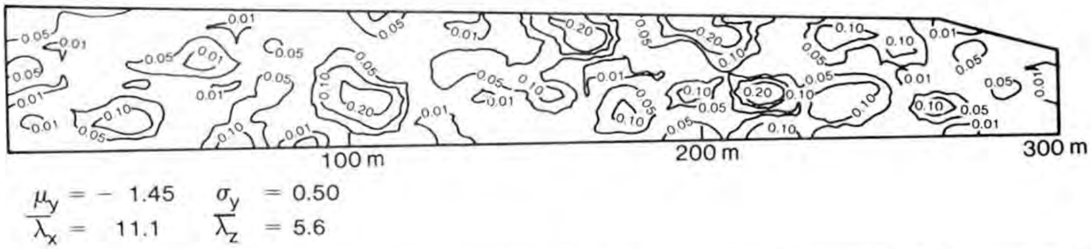

Fig. 2 Hydraulic conductivity variations representing conditions at field site.

Differences in Y_c from μ_y and S_{yc} from σ_y will contribute to the differences between these arrival time distributions and those in the reference case. If these differences are accounted for, it can be shown that the particular arrangement of the heterogeneities which develop as a consequence of the data set are contributing to an increase in the mean time of initial arrival. Note that for the sample grids considered, the mean arrival times can still differ significantly from the arrival times for the hypothetical field site.

One surprising result observed in Figure 3 is that the variability in the arrival time distributions remains relatively large, even though grid B represents a fairly dense sampling. This behavior can be explained by recognizing that the certainty with which the seepage velocities are determined strongly influences the ability to predict transport. To investigate the uncertainty in seepage velocities, probability distributions can be defined for V_x and V_z, the velocity in each of the coordinate directions, and θ, the direction of the velocity vector.

Figure 4 shows sets of frequency histograms on V_x, V_z, and θ at location a (Figure 1). This block has a known hydraulic conductivity value in both sample grids. For such locations the uncertainty in the velocity reflects the uncertainty in the hydraulic gradient. The hydraulic gradient is variable because of differences in the overall arrangement of the heterogeneous elements away from the measurement points in each realization.

The mean V_x decreases in comparison to the reference case because the hydraulic conductivity value preserved is lower than μ_y. Fixing the hydraulic conductivity values on sample grids A and B leads to more symmetrical distributions with a 46.8% and 53.1% reduction in the coefficient of variation; respectively, relative to the reference case.

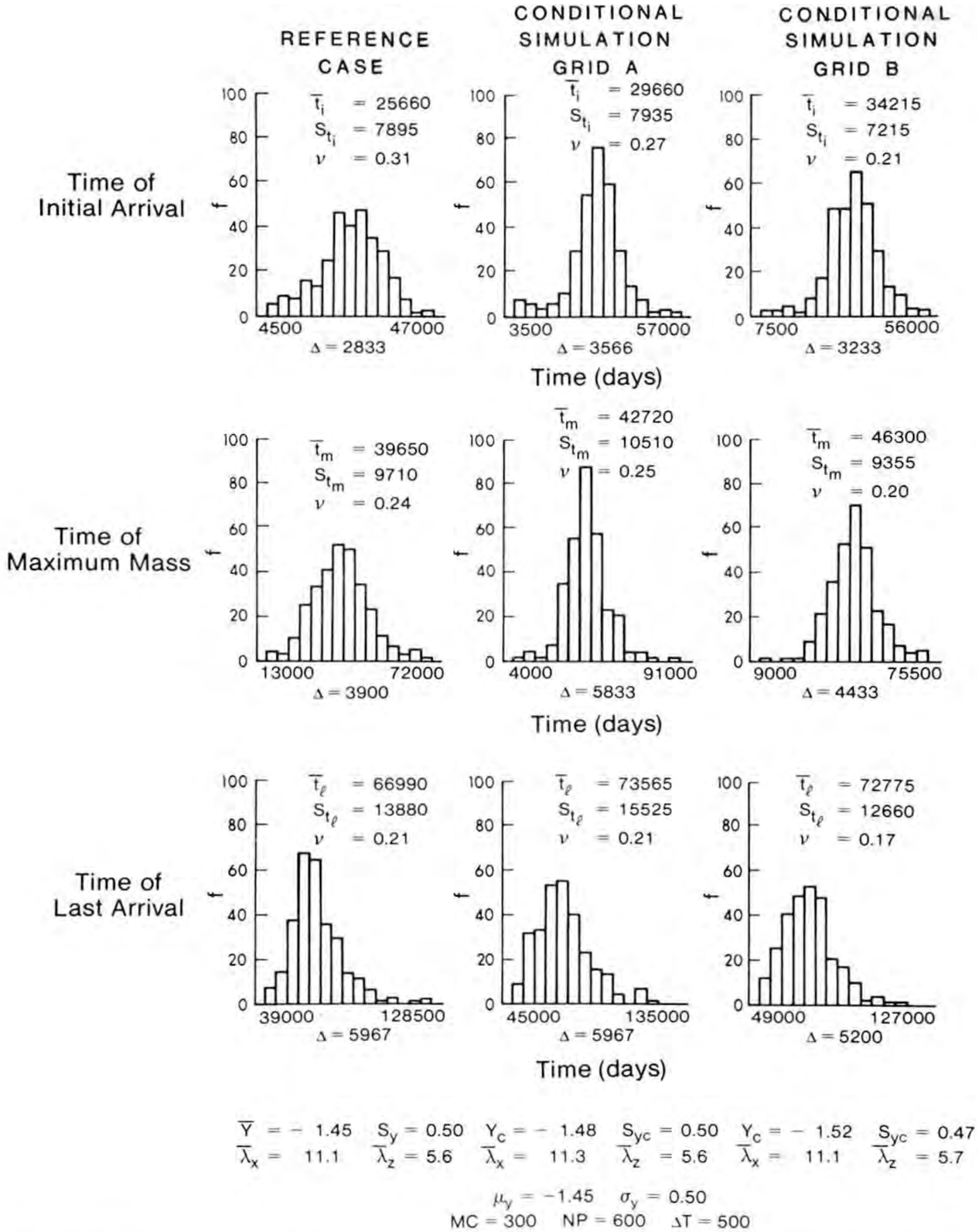

Fig. 3 Frequency histograms on the time of initial arrival at water table, the time the maximum quantity of mass crosses the water table, and the time of last arrival.

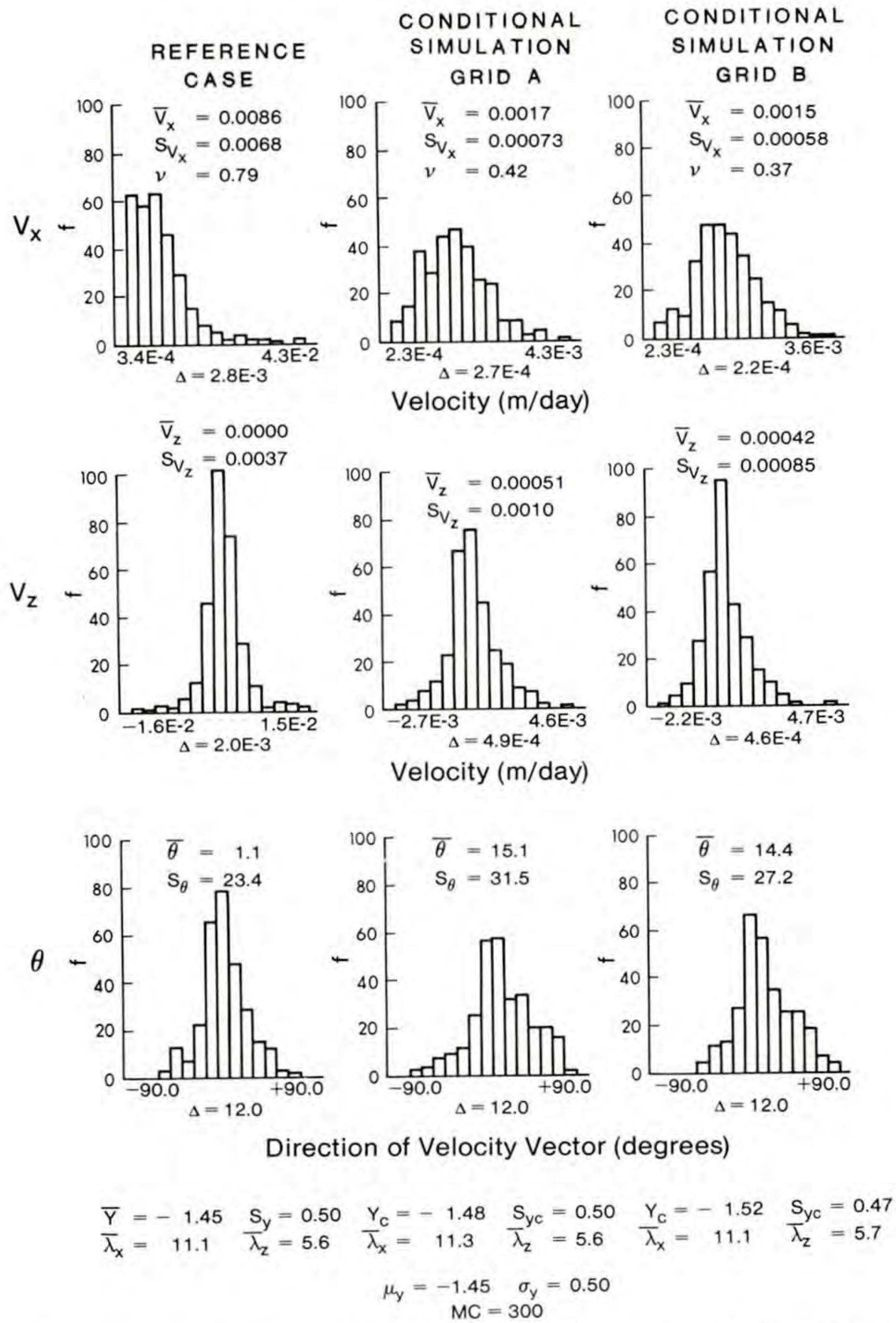

Fig. 4 Frequency histograms on V_x, V_z, and θ at location a (Fig. 1).

The middle set of histograms are formed on V_z. The positive mean for V_z in the conditional simulations indicates transport with a downward component across the low hydraulic conductivity block. The variability in V_z is reduced significantly in fixing the 24 data points with a smaller further reduction using the 36 points.

The lower set of histograms are formed on the direction of the velocity vector. The angle +90 with respect to the horizontal is vertically downward. The magnitude of S_θ reflects the variability in the direction of transport from a given location in the flow system. The distribution for θ shifts in the conditional simulations to a greater frequency of downward-directed velocity vectors. However, there is no reduction of the variability in the direction of transport from location *a* using either sample grids A or B.

The velocity distributions shown in Figure 4 are typical for blocks with known hydraulic conductivity values. The average reduction in the coefficient of variation for V_x in all 24 blocks located on sample grid A is 54.2%, relative to the reference case. For sample grid B, the average reduction in all 36 blocks is 57.9%. The average value for S_θ in those blocks located on grid B in the reference simulation is 24.0°. For the conditional simulation using sample grid A, the average of S_θ for the 24 blocks is 22.8°. Using sample grid B, the average of S_θ for the 36 blocks is 21.3°. Although preserving the hydraulic conductivity values reduces the variability in the magnitude of the velocity, the variability in the direction of transport from those blocks is only slightly constrained using sample grids A or B.

Now let us consider the reduction in uncertainty in the velocity for a block located between two blocks with known hydraulic conductivity values, such as location *b* in Figure 1. At such points, both the hydraulic conductivity and hydraulic gradient are unknown. For all those blocks located in similar positions to location *b* on grid B, the average reduction in the coefficient of variation is 10.1%, relative to the reference case. Farther away from the data points, the reduction is even less.

These simulations suggest that considerable hydraulic conductivity data may be necessary to obtain a reasonable degree of confidence in predictions of site behavior. For sample grids A and B, hydraulic conductivity data do not go far towards reducing the uncertainty in the seepage velocity. The data seem most effective in locally influencing the magnitude of the mean velocity and in reducing the uncertainty in the velocity at the measurement points.

CONCLUSIONS

Spatial variations in hydraulic conductivity play a critical role in controlling radionuclide transport in groundwater flow systems. They give rise to macroscopic dispersion and are a major component contributing to the uncertainty in predicting radionuclide migration. Results from our stochastic simulations suggest that unless a considerable number of field measurements are available to constrain the patterns of spatial variation in hydraulic conductivity, large uncertainties can be associated with the seepage velocities. As a consequence, transport predictions will be subject to large uncertainties.

Further research into the factors influencing the uncertainties in transport predictions is essential if groundwater transport models are to be utilized in a program of site evaluation. In particular, methods for reducing the magnitude of the uncertainties in prediction should be investigated. In this light we can include in situ velocity measurements, inverse simulation, and design of efficient sampling strategies. Detailed scenario evaluations involving future events disrupting the integrity of the groundwater system will be relatively unimportant if large uncertainties are present in predicting transport under existing conditions.

REFERENCES

1. L. Smith and F.W. Schwartz, "Mass Transport 2. Analysis of Uncertainty in Prediction," Water Resour. Res., 17(2), 351-369, (1981).

2. R.A. Freeze, "A Stochastic-Conceptual Analysis of One-Dimensional Groundwater Flow in Nonuniform Homogeneous Media," Water Resour. Res., 11(5), 725-741, (1975).

3. L. Smith and R.A. Freeze, "Stochastic Analysis of Steady State Groundwater Flow in a Bounded Domain 2. Two-Dimensional Simulations," Water Resour. Res., 15(6), 1543-1559, (1979).

4. J.P. Delhomme, "Spatial Variability and Uncertainty in Groundwater Flow Patterns," Water Resour. Res., 15(2), 269-280, (1979).

5. L. Smith and F.W. Schwartz, "Mass Transport 3. Role of Hydraulic Conductivity Data in Prediction," Water Resour. Res., (1981, in press).

6. L. Smith and F.W. Schwartz, "Mass Transport 1. A Stochastic Analysis of Macroscopic Dispersion," Water Resour. Res., 16(2), 303-313, (1980).

RELATIVE IMPACT OF DISSOLUTION TIME, TRANSIT TIME AND DISPERSION ON REPOSITORY PERFORMANCE

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ABSTRACT

Two studies are discussed in this report. The first is a deterministic study of the relative impact of dissolution time and transit time for two values of dispersivity per unit path length. The second is a probabilistic sensitivity study which analyzes the relative and absolute impact of input uncertainty on performance uncertainty.

INTRODUCTION

An important consideration in the promulgation of regulations, and the allocation of resources for research is the relative and absolute impact of the waste package and the geologic environment on overall repository performance. For many migration scenarios, the fluid motion is through cross-sectionally constrained pathways (e.g., boreholes, shafts, breccia pipes, etc.) and in such cases, two qualitative features stand out. The first is that there exist reasonably well defined discharge points from these pathways into aquifers and surface water systems, and the second is that the reduction in peak concentration due to transverse dispersion is small. It is then possible to characterize the waste migration by sets of two quantities; the average transit time (TT) of a short pulse and the temporal dispersion (S) the pulse experiences, as seen at the discharge point, one set for each pathway.

For each path and under the assumptions used in deriving the usual advection-dispersion equation, it may be shown [1] that the ratio of TT to S is independent of velocity and retardation. Specifically,

$$S/TT = (2D/L)^{1/2} \tag{1}$$

where D is the longitudinal dispersivity and L is the path length.

The waste package is characterized by two quantities: the corrosion (for breech) time (TC) of the canister, and the dissolution time (TD) of the waste form. The implicit assumptions are made that neither temporal nor canister variations of these parameters are great.

In this report, two types of studies are discussed. The first of a global deterministic sensitivity analysis of TT and TD, for two values of TT/S and fixed TC, displayed as contour plots.

The second is a probabilistic sensitivity analysis which investigates the effect two uncertain parameters TT and TD have on the uncertainty in the performance. It is performed for a variety of median values for TT and TD and dispersivity per unit length.

For both studies, 10 runs were made. Two fixed values for TT/S were used, one (TT/S = 10) corresponding to very little dispersion per unit length, and one (TT/S = 1) to very large. The latter is designed to mimic the effects of some transverse dispersion. TC was set to 500 yrs. Two types of waste were considered: spent fuel (SF) and reprocessed high level waste (HLW). The performance measure was the peak release rate (Ci/yr) for 10^6 MWE-yr repository.

The performance was evaluated separately for 3 groups of nuclides: Tc-99 and I-129 (Group 1); all other fission products (Group 2); and the actinides (Group 3). Within a group, the assumption was made that both dissolution times and retardation factors are the same for different nuclides.

Had ingestion rem/yr been chosen as a performance measure, then I-129 and Ra-226 would have taken on a significantly greater role in the performance of their respective groups.

Due to space limitations, not all of the results are reproduced. For a complete set of plots and supporting information, the reader is referred to Reference [2].

METHODS

Performance is computed by first calculating the flux (fraction/yr) assuming no decay. Then the potential hazard (Ci) as a function of time is multiplied by the fractional flux and the maximum over time of the product is calculated:

$$J_p^i = \max_t \; j(t)\, H^i(t) \tag{2}$$

where $H^i(t)$ = potential hazard (Ci) for group i
$j(t)$ = fractional flux
J_p^i = peak flux (Ci/yr) for group i.

j(t) is given by:

$$j(t) = \frac{1}{TD} \int_0^{TD} G(t-t')dt' \tag{3}$$

$$\text{with } G(t) = \frac{1}{(2\pi S^2 t^3/TT^3)^{1/2}} \exp-(t-TT)^2/(2S^2 t/TT) \tag{4}$$

The function G(t) is the Green's function for the one-dimensional advection-dispersion equation subject to semi-infinite boundary conditions [1]. The integral in (3) can be calculated analytically in terms of error functions, or numerically.

The evaluation of performance probability distributions is done by first discretizing the input probability distributions to sufficient accuracy and appropriately truncating them. Then the output is computed for all possible combinations of the inputs, while the associated probabilities are kept track of [2].

DETERMINISTIC SENSITIVITY STUDY

Figures 1 through 4 give the contour plots for 4 of the 10 deterministic runs.

From Fig. 1 it can be seen that so long as TT is less than the half-life (λ) of the main species present (Tc-99, $\lambda = 2 \times 10^5$ yrs), then there are distinct regions where the flux is independent of TD (TD << S = TT/10) or TT (TD >> S). When TT > 10^5 yrs, then the flux is always dependent on TT no matter how large TD is.

Comparing Fig. 2 with Fig. 1 indicates that when there is at least one major species for which significant decay is occurring, as in the actinides, then TT always has an impact on flux, but again, for TD << S, flux is independent of TD.

Comparing Fig. 3 with Fig. 2 indicates that variation in dispersion generally has little effect on flux. Specifically, to achieve the same performance for TT/S = 1 as seen from TT/S = 10 requires a change in TT of rarely more than a factor of 3.

Comparing Fig. 4 with Fig. 3 indicates that SF is substantially more hazardous than HLW, requiring factors of 3 to 5 for TT, or factors of 10 to 20 for TD, or some combination.

PROBABILITISTIC SENSITIVITY STUDY

In the second study, either TT or TD or both, were given lognormal distributions with a 2 order of magnitude spread between the 1st and 99th percentile, with the median values for each ranging from 10^3 to 10^6 years (except for the combination [10^6, 10^6], which was not performed due to program constraints). For each combination of median values, the uncertainty in flux is computed and displayed as a bar graph with 99, 75, 50, 25 and 1 percentile tick marks. The bar graphs are given in groups of 3, with the first representing the case where both variables are uncertain and the second and third where TT and TD are respectively set to their median values with certainty.

Fig. 5 reproduces results from one of the 10 runs. This case illustrates several features common to all cases. In particular (a bar over a variable denotes median value):

1) For $\overline{TD} >> \overline{S}$ and $\overline{TT}$ less than the half-life (λ) of any important species in the group, then performance uncertainty is a function only of TD.

2) For $\overline{TD} << \overline{S}$, then performance uncertainty is a function of TT only.

3) For $\overline{TT}$ greater than the half-life of any important species, then TT uncertainty is always important.

4) For $\overline{TD} \approx \overline{S}$, then the impact of TT uncertainty is greater than (for $\overline{TT/S} = 10$) or equal to (for $\overline{TT/S} = 1$) the impact of TD uncertainty, for TT less than λ.

   The first 3 conclusions essentially follow from the deterministic results of the preceeding section.

CONCLUSIONS

There are well defined areas, determined by the value of the ratios $\overline{TD/S}$ and $\overline{TT/\lambda}$, where both performance median value and its uncertainty are primarily a function of either TD or TT and their respective uncertainty spread. When $\overline{TT} >> \lambda$, then TT and its spread are always important in determining median performance and uncertainty. To equalize the impact of the actinides in SF and HLW requires either a factor of roughly 3 to 5 longer TT or a factor of 10 to 20 longer TD or some combination, for SF. Variations in TT/S from 1 to 10 (equivalent to variations in D from L/2 to L/200) rarely shifts equal performance contours by more than a factor of 3 for TT.

REFERENCES

1) L.E. Berman, D.A. Ensminger, M.S. Giuffre, C.M. Koplik, S.G. Oston, G.D. Pollak, B.I. Ross, Analysis of Some Nuclear Waste Management Options, The Analytic Sciences Corporation, Report No. UCRL 13917 (October 1978) (Available from NTIS).

2) W.G. Sutcliffe, K.G. Feller, N.K. Madsen, G.D. Pollak, Uncertainties and Sensitivities in the Performance of Geologic Nuclear Waste Isolation Systems, Report No. UCRL 53142 (Draft, 1981)

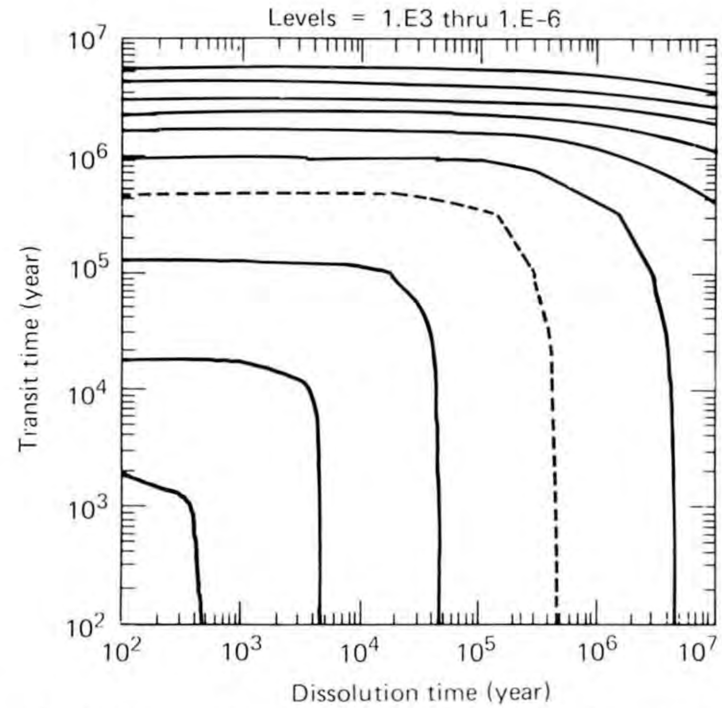

Fig. 1. Equal performance (Ci/yr) contours for Group 1 (Tc,I), HLW inventory, TT/S = 10. Dashed curve = 1 Ci/yr.

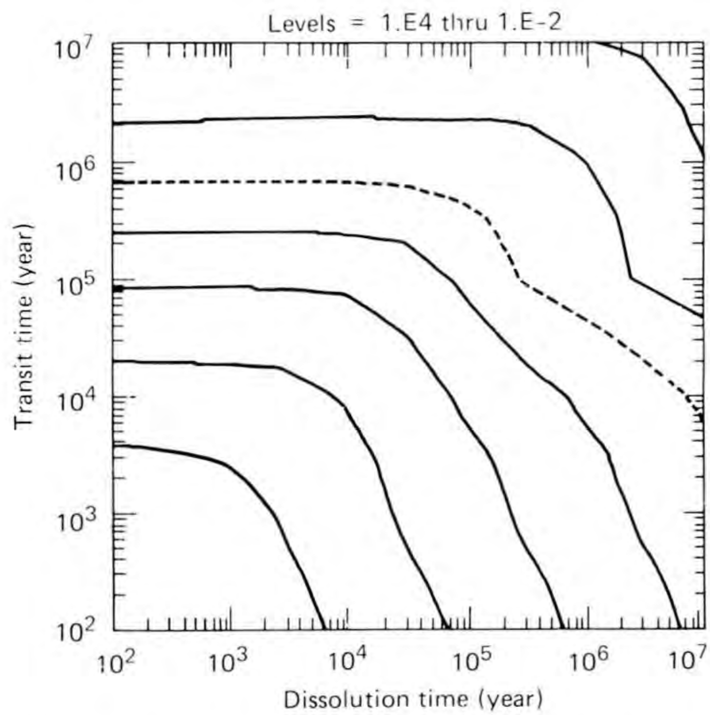

Fig. 2. Equal performance contours for Group 3 (Actinides), spent fuel, TT/S = 1. Dashed curve = 1 Ci/yr.

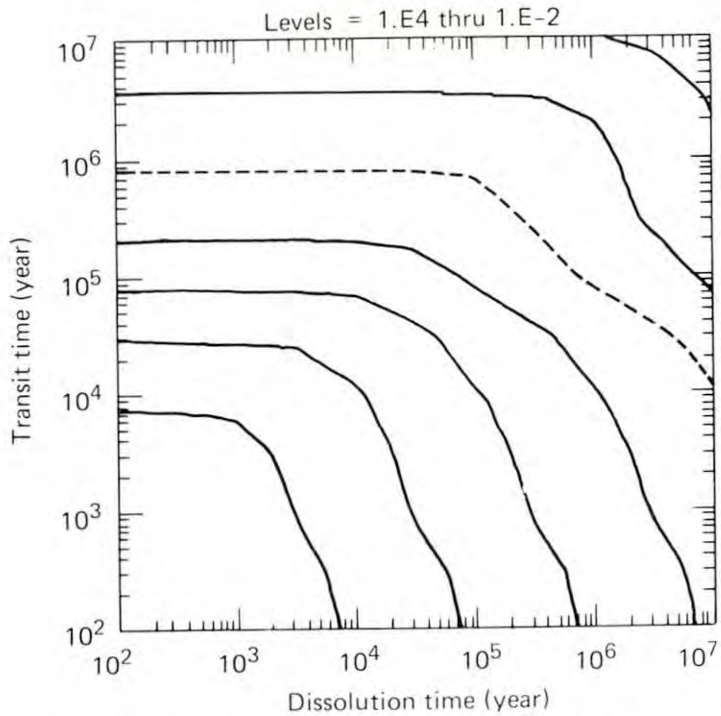

Fig. 3. Equal performance contours for Group 3 (Actinides), spent fuel, TT/S = 10. Dashed curve = 1 Ci/yr.

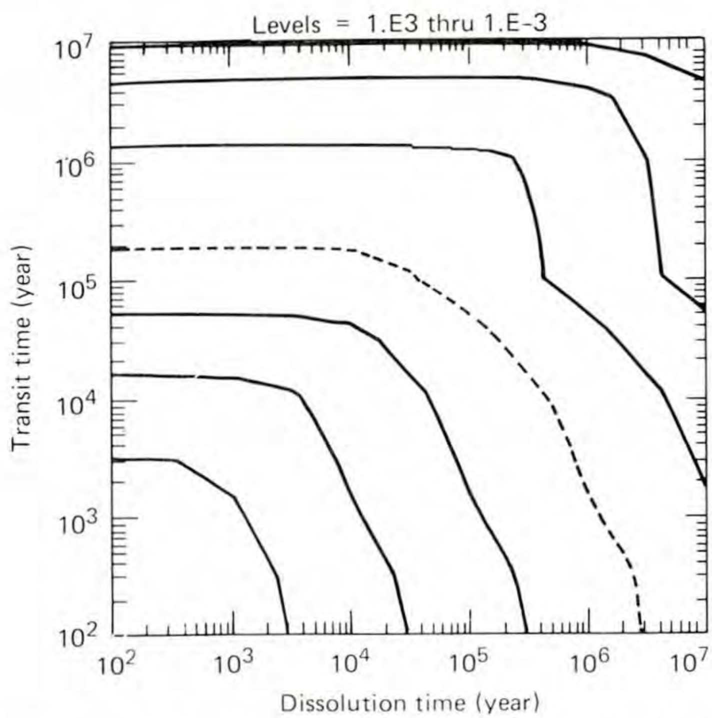

Fig. 4. Equal performance contours for Group 3 (Actinides), reprocessed HLW, TT/S = 10. Dashed curve = 1 Ci/yr.

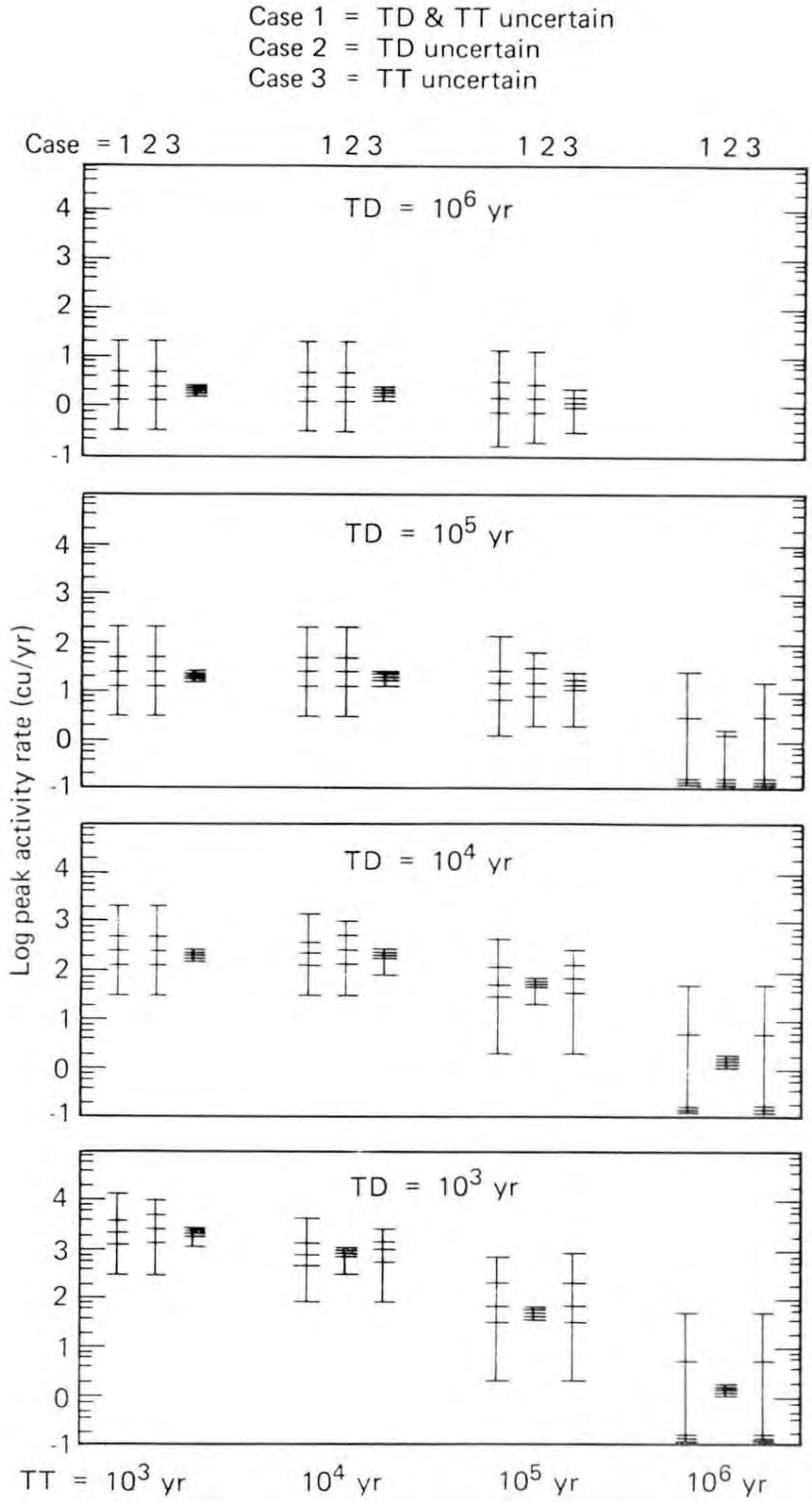

Fig. 5. Bar graph probability distributions for various combinations of median values for TT and TD. Tick marks correspond to various probability percentiles. Same run as in Fig. 1.

SOLUTE TRANSPORT IN POROUS MEDIA WITH IMMOBILE-WATER ZONES

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ABSTRACT

Three mathematical models describing one-dimensional flow with single-species solute transport and diffusion into an immobile-water zone are compared with reference to migration of radionuclides in two media that are candidates for a waste repository. The first model includes dispersion in the flowing-water zone and treats the concentration profile in the immobile-water zone explicitly. The second model has no spatial dimension in the immobile-water zone. The third model is based on the assumption of concentration equilibrium between the flowing-water and immobile-water zones. The first model requires numerical inversion of the Laplace transform solution. Analytical solutions are available for the second and third models (and some special cases of the first). Evaluation and comparison using parameter ranges from fractured granite and tuff show: a) the mechanism of diffusion into an immobile-water zone provides significant transport rate retardation and peak concentration attenuation; b) the simplifying assumption of an infinite immobile-water zone thickness is invalid; and c) the simpler models are adequate for high diffusional transport rates.

INTRODUCTION

A recent topic of interest in migration of radionuclides by flow in porous and fractured media is retardation and attenuation provided by diffusion into and out of immobile-water zones. Two conceptual geometries of the physical system will be considered.

The first geometry (fig. 1a) is of a rock, such as granite, containing a transmission fracture (with aperture 2b) with dead-end, side-branch fractures (of length ℓ). The proportion of the main fracture wall open to the dead-end fractures is designated α. The porosity of the rock matrix is assumed to be zero. The second geometry (fig. 1b) is of a rock with nonzero matrix porosity (ε_s) and very low hydraulic conductivity, so that fluid flow is through fractures of aperture 2b. Spacing between adjacent parallel fractures is 2ℓ. Fractured tuff or shale are examples. The transmissive fracture will be referred to as the "flowing-water zone" and the side-branch fractures (geometry 1) or the porous rock (geometry 2) as the "immobile-water zone", with reference to the presence or lack of water movement.

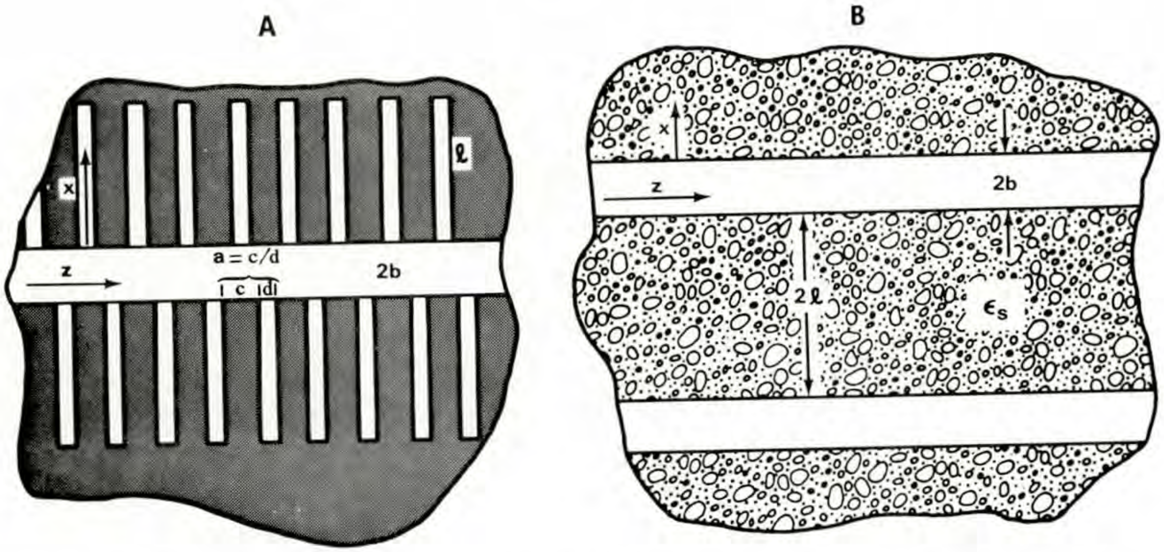

Fig. 1. Simplified Schematic Drawings of a Fractured Crystalline Rock (1A) and a Fractured Porous Rock (1B).

As a solute (radionuclide species) is transported by convection along the transmissive fracture, diffusive transport into and out of dead-end fractures or porous medium occurs, decreasing the net rate of transport and the peak concentration resulting from release of a finite slug of solute.

The purpose of this paper is to compare three mathematical models describing this physical system, indicate the conditions under which the simpler models are valid, and illustrate the potential of immobile-water zone diffusion to retard and attenuate radionuclide transport in two potential nuclear-waste repository media: namely granite and tuff. The first model (profile model) treats the concentration profile in the immobile-water zone explicitly and includes dispersion in the flowing-water zone; this is the most realistic model of the physical system and has not appeared previously in the literature with the source term and boundary conditions given here. The second model (mixing model) has no spatial dimension in the immobile-water zone; this implies physically a well-mixed immobile-water zone with a uniform concentration profile. Coats and Smith [1] were the first to present the solution of this model as their dead-end pore model. The third model (equilibrium model) is the most simplified being based on the assumption of concentration equilibrium between the flowing-water and immobile-water zones at all points along the transmissive fracture. It is the limiting case of the other two models for an infinite diffusional-transport rate.

Assumptions

The following assumptions have been made:

1. One-dimensional transport in the z-direction of a single solute species in the flowing-water zone,
2. One-dimensional transport in the x-direction by diffusion in the immobile-water zone,
3. No sorption or reaction of the solute species,
4. All parameters constant in space and time,
5. Species concentrations sufficiently low to not affect the flow field by density or viscosity variations,
6. Mass flux at the interface between the flowing-water and immobile-water zones distributed continuously along the transmissive fracture even though actually occurring at the entrances of discrete side-branch fractures,
7. Apertures of the side-branch fractures or pore sizes of the porous rock large enough to allow movement of solute molecules in the immobile-water zone.

TRANSPORT EQUATIONS

Dimensionless Groups

The transport equation may be written using the following dimensionless parameters:

$c = \frac{C}{C_o}$ dimensionless concentration in flowing-water zone,

$c_s = \frac{C_s}{C_o}$ dimensionless concentration in immobile-water zone,

$\eta = \frac{z}{L}$ dimensionless length down the flowing-water zone fracture,

$\xi = \frac{x}{\ell}$ dimensionless length into the immobile-water zone,

$\theta = \frac{tV}{L}$ dimensionless time,

$\gamma = \frac{D}{VL}$ dimensionless dispersion coefficient in the flowing-water zone,

$\alpha = \frac{b}{a\ell}$ or $\frac{b}{\varepsilon_s \ell}$ dimensionless volume ratio of flowing-water to immobile-water zone,

$\beta = \frac{D_m L}{\ell^2 V}$ dimensionless rate coefficient (ratio of diffusive transport rate to convective transport rate), and

$R = \frac{rL}{C_o V}$ dimensionless transport rate from the immobile-water zone to flowing-water zone,

where

C is the concentration in the flowing-water zone,

C_s is the concentration in the immobile-water zone,

L is the length from the entrance of the flowpath to the measurement point,

C_o is the source scaling concentration,

V is the uniform velocity in the flowing-water zone,

D is the dispersion coefficient,

D_m is the effective molecular diffusivity in the immobile-water zone,

r is the rate of species transport from the immobile-water zone to the flowing-water zone per unit volume of flowing-water zone, and

a, b, ℓ and ε_s are as previously defined.

Several of the parameters that comprise α and β are difficult to determine for an actual medium. This gives rise to some uncertainty in assessment of the importance of diffusion into the immobile-water zone for transport-rate retardation in relation to radionuclide waste disposal.

Transport in the Flowing-Water Zone

<u>Profile Model</u>. A dimensionless form of the transport equation in one dimension which describes transport in the flowing fluid (including convective, dispersive and source sink terms) is:

$$\frac{\partial c}{\partial \theta} = \gamma \frac{\partial^2 c}{\partial \eta^2} - \frac{\partial c}{\partial \eta} + R \qquad (1a)$$

Boundary Conditions: at $\eta = 0$ $\quad -\gamma \frac{\partial c}{\partial \eta} + c = F(\theta)$ (1b)

as $\eta \to \infty$ $\quad c \to 0$ (1c)

Initial Condition: at $\theta = 0$ $\quad c = 0$. (1d)

The source function for this evaluation is given by:

$$F(\theta) = H(\theta) - H(\theta - \theta_s) \qquad (2)$$

where H represents the unit step function, and θ_s the dimensionless time at which the source slug of solute ends.

Transport in the Immobile-Water Zone

The concentration profile in the immobile-water zone is governed by a diffusional transport mechanism. The transfer rate from the immobile water to the flowing-water zone is the diffusive flux which depends on the concentration gradient in the immobile-water zone at the interface:

$$\frac{\partial c_s}{\partial \theta} = \beta \frac{\partial^2 c_s}{\partial \xi^2} \tag{3a}$$

Boundary Conditions: at $\xi = 0$, $c_s = c$ (3b)

at $\xi = 1$, $\frac{\partial c_s}{\partial \xi} = 0$ (3c)

Initial Condition: at $\theta = 0$, $c_s = 0$. (3d)

The second boundary condition states that no solute leaves through the back end of the side-branch fractures or crosses the symmetry boundary midway between flowing-zone fractures in the porous medium. The transfer-rate expression is:

$$R = \frac{\beta}{\alpha} \frac{\partial c_s}{\partial \xi} \Big|_{\xi = 0} \, . \tag{4}$$

Mixing Model. This simpler model has no spatial dependence in the immobile-water zone; the solute concentration in the immobile-water zone is completely mixed to a uniform concentration throughout its thickness:

$$\frac{\partial c_s}{\partial \theta} = - \alpha R \tag{5a}$$

Initial Condition: at $\theta = 0$, $c_s = 0$ (5b)

Transfer Rate Expression: $R = - \beta(c - c_s)$. (6)

The transfer rate in the mixing model is simply proportional to the difference in concentration between the flowing-water and immobile-water zones. The transfer-rate coefficient, β, has been made the same as for the profile model by letting the transfer-rate constant be D_m/ℓ^2, which is a characteristic-rate scale factor for diffusion in the immobile-water zone. Combining equations 1a, 5a, and 6 gives the model equation formulation.

Equilibrium Model. The solute concentration in this simplest model is in equilibrium between flowing-water and immobile-water zones at each point along the transmissive fracture:

$$c_s = c \, . \tag{7}$$

The governing equation is obtained by combining equations 1a, 5a, and 7.

MATHEMATICAL SOLUTIONS

The solution for the general profile model requires numerical inversion of the Laplace transform. The method used, Crump [2], is based upon application of the trapezoidal rule to the Laplace inversion integral, which yields a Fourier-series representation of the function sought. Acceleration of the convergence of the infinite sum involved is obtained using the epsilon algorithm. The computation is done to a user-specified accuracy.

Solutions for the simpler models or for other boundary conditions have been presented by Barker [3], Skopp and Warrick [4], Rae and Lever [5], Coats and Smith [1], Lindstrom [6] and Lapidus and Amundson [7]. Lack of space prevents inclusion of any of these solutions here.

MODEL PARAMETERS FOR TWO MEDIA

Evaluation and comparison of these models of transport with an immobile-water zone were done using parameters estimated to characterize two media that are candidates for high-level nuclear waste disposal; a fractured granite and a fractured tuff. Parameter values came from a variety of sources including Rae and Lever [5], Lundstrom and Stille [8], Olkiewicz et al [9], and Grisak, Pickens and Cherry [10]. Value ranges, with corresponding dimensionless parameters, are given in Table 1.

It should be noted that the fractured granite was characterized by an effective porosity for the immobile-water zone, because no data were available on side-branch fracture apertures, lengths, or distribution down the flowing-water zone wall.

Table 1. Parameter Ranges for Granite and Tuff

	Granite		Tuff
b	0.05 - 0.5 mm		1 - 10 mm
ℓ	0.5 - 5 m		1 - 20 m
ε_s	0.005 - 0.03		0.30 - 0.45
D_m		0.0006 - 0.03 m^2/yr	
D		0.0004 - 5,000 m^2/yr	
V		0.0001 - 0.1 km/yr	
L		1 - 10 km/yr	
α	0.0003 - 0.2		0.0001 - 0.03
β	0.002 - 10,000		0.002 - 3,000
γ		0.0004 - 0.05	

SENSITIVITY ANALYSIS OF PROFILE MODEL

A sensitivity analysis was done for the profile model, because it is the most realistic representation. A base-case set of parameters was selected for granite: where b = 0.5 mm, ℓ = 1 m, ε_s = 0.005, D_m = 0.03 m^2/yr, V = 0.001 km/yr, and L = 1 km. The dispersion coefficient was taken to be zero for most of the analyses to emphasize the effects of the diffusional exchange with the immobile-water zone.

The base case (β = 30) and the results of varying the diffusion rate parameter (β) while holding the volume ratio parameter (α) constant are shown in figure 2. For very large β, the elution curve approaches the delayed but unattenuated profile characteristic of equilibrium sorption with no dispersion in the flowing-water zone. For very small β, the curve approaches that for a noninteracting solute with only convective transport and no peak attentuation. The transition through intermediate β values contains a curve with maximum-peak attenuation. It should be noted that a dimensionless-time unit is the time required for an impulse of solute to be transported to the measurement point by convection only.

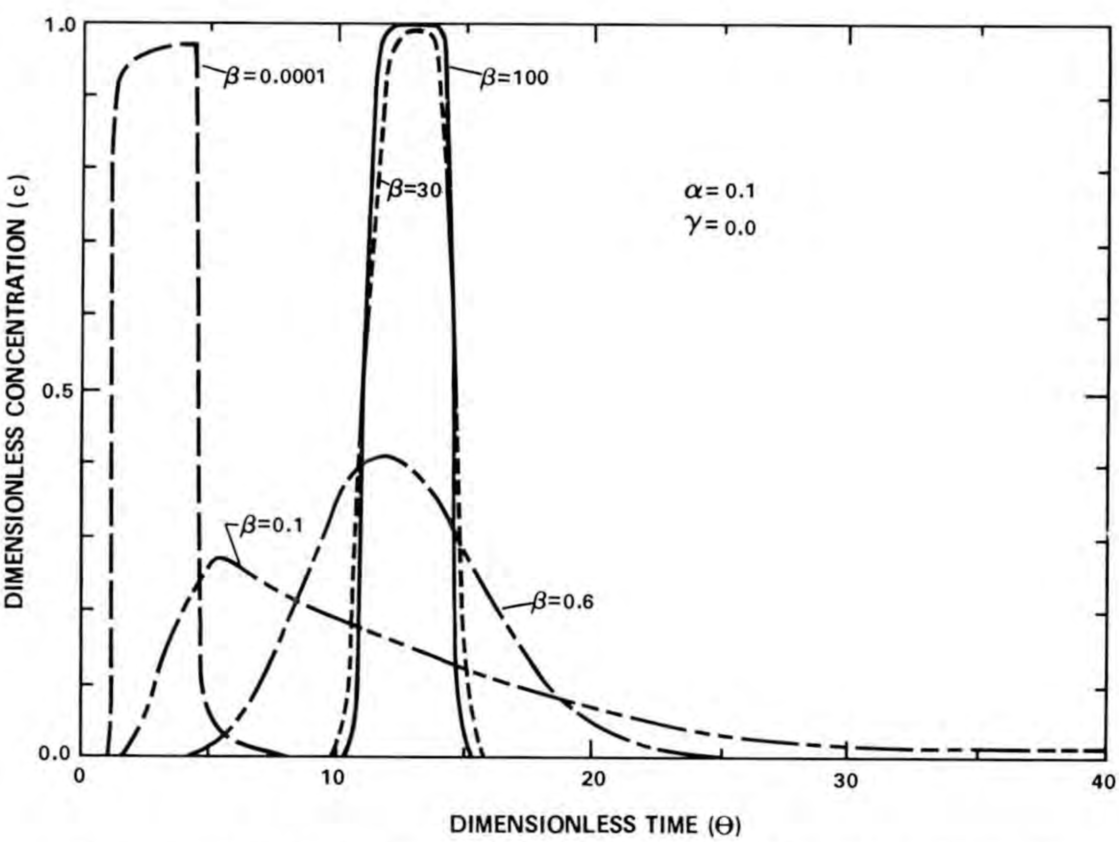

Fig. 2. Solute Elution Curves for the Profile Model with Variation in Diffusion Rate.

The results of decreasing both volume ratio and rate coefficient by increasing immobile-water zone thickness, ℓ, are shown in figure 3. Note that as $\ell \to \infty$, results underestimate peak concentration relative to finite thickness cases, unless α and β are much less than 1. This indicates that the Rae and Lever [5] solution overestimates the effects of the immobile-water zone of these two media on nuclide transport.

COMPARISON WITH SIMPLER MODELS

The profile-transport model was compared with the simpler mixing and equilibrium models for parameter values representative of granite and tuff. For the mixing-model comparison in figure 4, it is apparent that for large values of the rate coefficient, β, the transport rate in the immobile-water zone is fast enough for a uniform-concentration profile to be established rather quickly. Thus the mixing model, which assumes a uniform-concentration profile in the immobile-water zone, yields solute-effluent curves nearly equivalent to profile model curves.

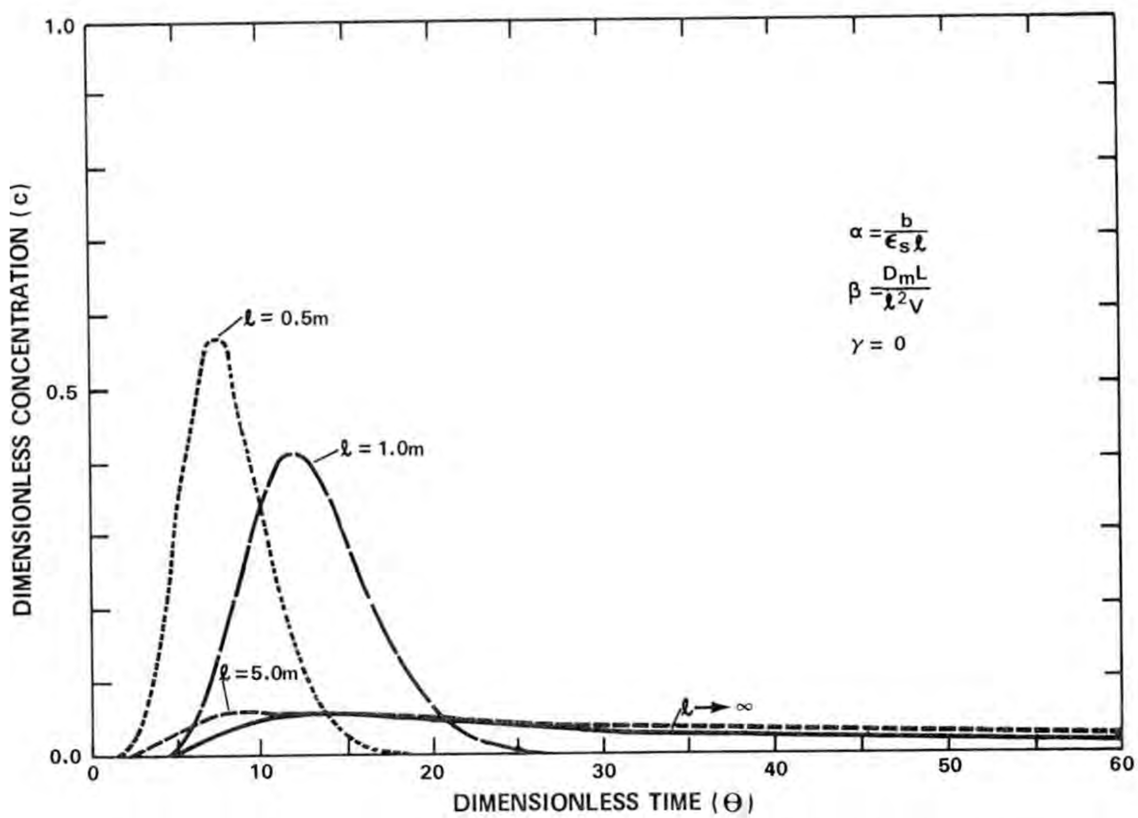

Fig. 3. Solute Elution Curves for the Profile Model with Variation in Immobile-Water Zone Thickness.

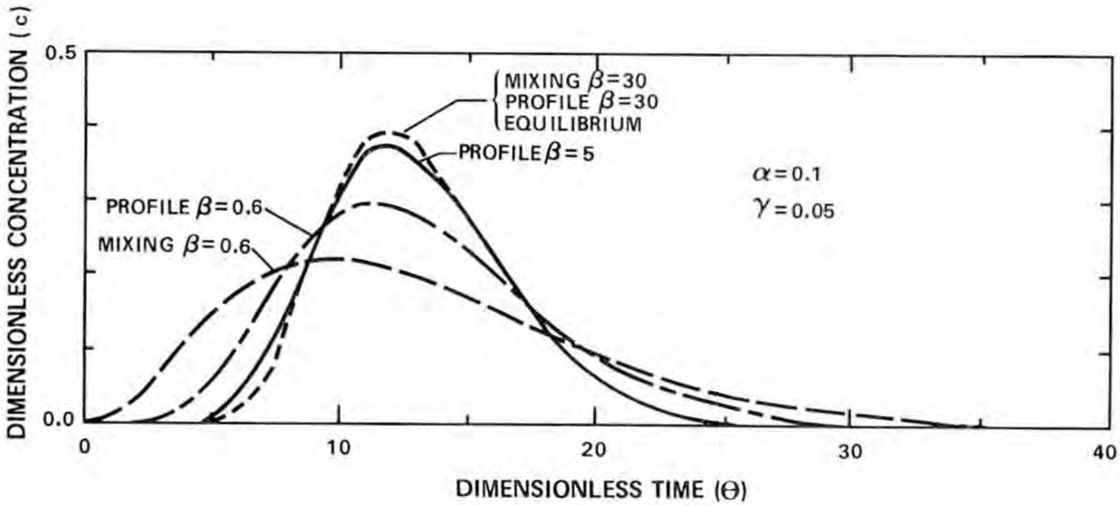

Fig. 4. Solute Elution Curves Comparing the Profile Model with the Mixing Model and Equilibrium Model.

It appears that the situation is adequately modeled by the simple equilibrium model for rate coefficients greater than about 5 (fig. 4). Large β values mean a rapid transfer rate between flowing-water and immobile-water zones which leads to a rapid establishment of equilibrium concentrations. Further evaluations showed that if the ratio of rate coefficient to volume ratio was greater than about 50, the equilibrium model was adequate. The equilibrium model characterizes the effects of diffusion into and out of the immobile-water zone as a retardation coefficient, $1 + \frac{1}{\alpha}$, which is analogous to equilibrium sorption retardation. It depends only on relative volume of immobile-water zone available for temporary storage of solute.

GRANITE VERSUS TUFF

Using a typical set of parameter values for fractured granite and fractured tuff, the profile model produced the curves in figure 5. Increased retardation and peak attenuation of the tuff is due to the larger relative volume of the immobile-water zone. It is the volume ratio that most strongly governs transport behavior in the system with the condition of 1,000 years convective traveltime, over the range of diffusion coefficient assumed. It is possible that the effective sizes of the solute species or complexes might be quite large relative to the aperture sizes of the side-branch fractures or porous immobile-water zone; this would make the effective diffusion coefficient, D_m, much

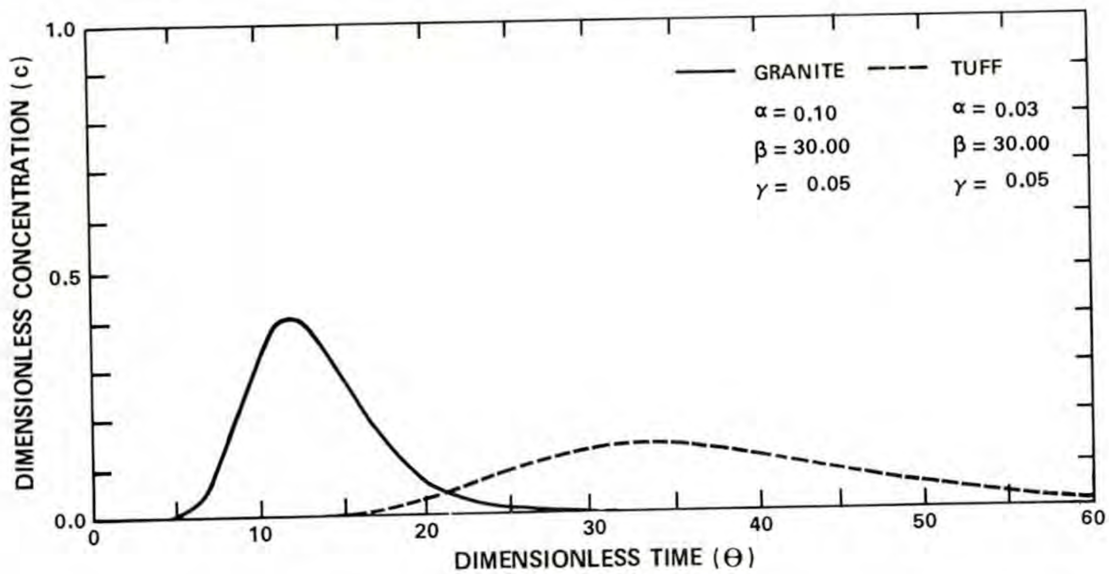

Fig. 5. Solute Elution Curves Comparing The Granite with the Tuff.

smaller. Under these conditions, the effect of the immobile-water zone on nuclide-transport rates could be greatly reduced.

CONCLUSIONS

1. It appears that nuclide-solute diffusion into and out of an immobile-water zone provides a mechanism for significant transport-rate retardation and peak-concentration attenuation in fractured granite and fractured tuff. Further work needs to be done in parameter measurement to confirm that the immobile-water zone is as accessible to a given solute species as these transport models assume.

2. The assumption of an infinitely thick immobile-water zone is not valid in that it leads to overestimation of retardation and attenuation effects.

3. The simpler mixing and equilibrium models for this system are adequate to simulate situations where the ratio of the rate coefficient to the volume ratio term is large (β/α greater than about 50). These situations have rapid diffusional-transport rates, relatively large flowing-water zone volumes, or both.

ACKNOWLEDGEMENTS

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REFERENCES

1. K. H. Coats and B. D. Smith, "Dead End Pore Volume and Dispersion in Porous Media", Soc. Pet. Eng. J., 4, 73-84 (1964).

2. K. S. Crump, "Numerical Inversion of Laplace Transforms Using a Fourier Series Approximation", Jour. ACM, 23, 1, 89-96 (1976).

3. J. Barker, "Laplace Transform Solutions for Solute Transport in Fissured Aquifers", Report No. WD/ST/80/11, Institute of Geological Sciences, London, (October 1980).

4. J. Skopp and A. W. Warrick, "A Two-Phase Model for the Miscible Displacement of Reactive Solutes in Soils", Soil Sci. Soc. Am. Proc., 38, 4, 545-550 (1974).

5. J. Rae and D. A. Lever, "Will Diffusive Retention Delay Radionuclide Movement Underground?", Report TP 853, AERE Harwell, U.K., (June 1980).

6. F. T. Lindstrom, "Pulsed Dispersion of Trace Chemical Concentrations in a Saturated Sorbing Porous Medium", Water Res. Res., 12, 2, 229-238 (1976).

7. L. Lapidus and N. R. Amundson, "Mathematics of Adsorption in Beds", J. Phys. Chem., 56, 984-995 (1952).

8. L. Lundstrom and H. Stille, Large Scale Permeability Test of the Granite in the Stipa Mine and Thermal Conductivity Test, Swedish-American Program on Radioactive Waste Storage in Mined Caverns in Crystalline Rock, Technical Project Report #2, LBL-7052, SAC-02, UC-70, TID-4500-R66, (July 1978).

9. A. Olkiewicz, J. E. Gale, R. Thorpe and B. Paulson, Geology and Fracture System at Stripa, Swedish-American Program on Radioactive Waste Storage in Mined Caverns in Crystalline Rock, Technical Report #21, LBL-8907, SAC-21, UC-70, (February 1979).

10. G. E. Grisak, J. F. Pickins and J. A. Cherry, "Solute Transport Through Fractured Media, 2, Column Study of Fractured Till", Water Res. Res., 16, 4, 731-739 (1980).

FRACTURES: CERTAINTIES AND UNCERTAINTIES

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ABSTRACT

For radioactive waste disposal, it is desirable that the integrity of the geological environment be preserved. Therefore, in order to minimize drilling, it is important to be able to predict the existence and characteristics of fractures beneath the surface.

Fractures include faults, and range in size from kilometers to microns. They are a major factor in the selection of a site for radioactive waste disposal primarily because of their potential as pathways. Furthermore, large fractures could act as foci for seismic activity which may impact on the disposal vault.

Some fracture characteristics may be regarded as being more or less predictable. They can be grouped, according to orientation, into sets and systems which are (1) evident at all scales, (2) identifiable from place to place over large areas, and (3) likely to occur at depth. Preliminary investigations show that where there is a high concentration of fractures at the surface, irrespective of orientation, there is, correspondingly, a high concentration in the subsurface; where there are few fractures at the surface, there tend to be few at depth.

Besides the aforementioned characteristics, there are several others, such as continuity, number of intersections, depth of penetration and frequency which commonly show variations, at all scales, from place to place. These variations make it difficult to predict, from surface observations, the fracture conditions that may exist between the vault and the biosphere. An attempt is being made to overcome these uncertainties by grouping fractures according to genesis and history rather than orientation. The initial results are encouraging.

EQUIVALENT PERMEABILITY OF FRACTURED ROCK: EFFECT OF FRACTURE SIZE AND DATA UNCERTAINTIES

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THE INFLUENCE OF GEOCHEMICAL VARIABLES ON LONG-LIVED RADIONUCLIDE MIGRATION AND RISK ASSESSMENT*

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ABSTRACT

The uncertainties associated with predicting long-lived radionuclide migration from geologic repositories are large because of the long times considered and the lack of a geochemical history for the artificial elements. The long-lived radioelements can be categorized according to their inherent chemical properties and geochemical interactions which contribute to varying levels of uncertainty in the predicted risk. The elements Am, Th, and I, although quite different in terms of mobility, have the least uncertainty associated with predictions of their long-term behavior once a reference case is determined. The chemically similar elements Np, Pu, and U represent a second case where the presence of mobile oxycations contributes to complicated solid-solution partitioning evaluations. A third category, created for technetium, considers an element for which one parameter (evaluated with difficulty), redox potential, contributes most to uncertainty, with a small difference in assumed geochemical environment resulting in either a highly retarded or highly mobile chemical form.

INTRODUCTION

Proposed technical criteria [1] for the disposal of high-level radioactive waste place major emphasis on the engineered portion of the geologic repository. In large measure, this is because substantial uncertainties exist with respect to the natural geologic barrier's capacity to retard or limit releases of radioactivity to levels below accepted standards [as defined by the U.S. Environmental Protection Agency]. Performance objectives for the geologic setting do exist, however, because after some time period following the intrusion of groundwater the integrity of the engineered barrier will degrade to such a degree that a release of radionuclides will occur. It is the role of the natural barrier to control such releases to below acceptable levels in the event that the release rate from the engineered portion exceeds acceptable levels.

In addition to the reality that predicting releases or transit concentrations of radionuclides is subject to large uncertainties imposed by the time frames considered (10^3 years and greater), it must also be recognized that any predictive model is also subject to mechanism and parameter estimation errors and therefore cannot be considered anything more than a mathematical formulation of scientific judgment. This statement follows largely from the fact that it is impossible to validate the geochemistry of the long-lived artificial elements under repository conditions. Under such constraints it is extremely difficult, if not impossible, to evaluate the degree to which bias has affected the accuracy of any predictions.

MIGRATION UNCERTAINTY AS A FUNCTION OF GEOCHEMICAL VARIABLES

The potential water-driven migration of radioactive elements out of a repository will be a function of:

radioelement species,
time (decay, equilibration),
nature of sorption surfaces, and
groundwater composition, flow rate.

The importance of each of these geochemical variables cannot be quantitatively evaluated under repository conditions because the exact combination of variables and their net influence is a site and time-specific function. However, the long-lived elements can be evaluated as to risk; that is, for which elements, because of their inherent chemical behavior, is there the largest risk that a migration assessment error would result in transport exceeding the established safety margin?

The risk question can be addressed by grouping the elements into categories according to the sensitivity of their migration to changing chemical conditions:

Category I. Th, Am(=Cm), I
II. Pu, Np, U
III. Tc

Category I elements are those whose mobility characteristics are impacted to the least extent by geochemical variations around those assumed for the repository. That is, although Am and Th are relatively immobile, equally important is the fact that groundwater conditions other than the reference case will not impact greatly on predicted mobility. For example, if the carbonate content is assumed to be x, then a 2x or 4x difference in actual composition will not likely contribute to highly different migration rates. The anion I^- is retarded to such a minor extent that we might assume a retardation factor of 1; that is, I^- will move at the same velocity as water. Again, variations in groundwater composition are not of major importance when this zero retardation assumption is made.

Category II elements are very similar chemically except for the redox stability of the various oxidation states. The oxidation states of importance are likely to be U(IV), U(VI), Np(IV), Np(V), Pu(IV), and Pu(V). Examined separately, the tetravalent state [as well as Pu(III)] could realistically fall into Category I. It is the potential presence of UO_2^{2+}, PuO_2^+, NpO_2^+, and their complexes, that warrants a separate class for this group. The redox environment may be the most important variable, followed by solute composition and sorption surfaces. This group is characterized by the fact that even if the redox composition of the groundwater is much different than the reference case, retardation is still operative. That is, if U(VI) or Np(V), instead of U(IV) and Np(IV), actually dominates the source term species, retardation will still occur. For U(VI), the key variables become carbonate concentration and sorption surfaces, while for Np(V), pH and sorption surfaces may be more important [2]. Inherently, this category contains the most complicated group.

Category III, created to accomodate Tc, considers the case where a single parameter, redox environment, dominates the factors contributing to uncertainty in migration rate. That is, under reducing conditions a very insoluble species like $Tc(OH)_4$ may exist [3], while only a small increase in O_2 content would favor a very mobile species, TcO_4^-. Thus Category III reflects characteristics found in I and II but is separately designated to emphasize the very small margin of "error" in the high retardation [$Tc(OH)_4$] vs. high mobility [TcO_4^-] prediction.

The appendix presents some illustrations of why these elements were grouped in this way. The classification scheme is largely a way of defining commonalities from which a risk evaluation can be discussed.

DISCUSSION

Category II, composed of U, Np, and Pu, is the most complicated in terms of migration uncertainty. In addition to the redox uncertainty, which is really a question of considering how confident one can be about the long-term reducing capacity of a groundwater flow path, variables such as sorption surfaces and complexing ligands are also important. For cationic species, regardless if they are anionic due to groundwater solutes [i.e., carbonate complexes of U(VI) - $UO_2(CO_3)_3^{4-}$, or organically-complexed Pu(IV)], competition for the metal will exist betwen the solid phase sorbant and the solution phase ligand. This interaction has been handled in the transport equation by a retardation factor and an equilibrium assumption. For uranium complexed by carbonate ions, however, it was concluded that equilibration was so slow in a sandstone aquifer that a removal rate model was better for calculating migration than the transport equation [4].

Experience indicates that the more hydrolytic or highly charged species [U(IV), Th(IV), Rare earth (III)] are much less mobile than

U(VI). Indeed, if the real world migration of U(VI) under repository conditions can be quantified, we could use comparative data such as in the appendix to establish ranges of likely Np(V) and Pu(V) migration.

In this discussion, the tetra- and trivalent oxidation states have been treated as being "insoluble." This is a qualitative judgement, but when all possible species are considered it is a fact that migration potential increases in the order IV $\leq$ III $<$ V $\leq$ VI $<$ VII, or specifically, Pu(IV) $\leq$ Am(III) $<$ Np(V) $\leq$ U(VI) $<$ Tc(VII). The oxyions, NpO_2^+, UO_2^{2+}, TcO_4^- are the greatest contributors to the mobile fraction. Organic complexes, although important [2], were not considered because at natural concentrations their role should be minor and should not affect the overall perspective discussed above.

The literature on migration of the artificial elements is inadequate to assign quantitative uncertainties to migration predictions. Those properties of groundwater, rocks, and the elements which interact to affect migration, especially near the low retardation spectrum where rates may be within a factor of 100 of the transit time of water, have been inadequately investigated. More attention needs to be given to ranges of possible migration rates given the variables listed earlier. Detailed mechanistic studies, although "good science," should be conducted in the overall context of which investigations offer the most realistic means of reducing knowledge uncertainties, as opposed to fine-tuning a numerical data set. For example, determining the stability constant for the Np(V)-carbonate interaction is far more important than refining the U(VI) constant. It can also be argued that sorption is a complex interrelationship between surface sites, solute ions, and chemical species, and therefore empirical data approximating projected repository conditions may be the most useful for assessment purposes. No amount of thermodynamic data will substitute for this empirical approach, provided that the empirical studies are not entirely "cookbook" number gathering.

SUMMARY

A quantitative evaluation of the uncertainty associated with the migration of long-lived radioelements is extremely difficult. Although a myriad of "K_d" - type numbers has been generated by waste management contracts, these data are laboratory numbers that only qualitatively bear any relationship to actual migration conditions. There has not been a study which attempts to determine the validity of these values under real-world conditions because such a study would be difficult to do. Therefore, any statistical uncertainty analysis based on these laboratory studies generates more numbers but will not contribute to our understanding of geochemical uncertainty. Because of the long time frames considered in repository evaluations, migration assessments can only be qualitative despite the large emphasis on mechanistic and simulation assessment methodology.

Based on the available laboratory and field data, the long-lived radioelements can be evaluated on a qualitative basis as to what geochemical variables contribute to uncertainty. Given the establishment of reference sets of retardation values, determined by site-specific research, it is suggested that the uncertainty in migration rates will be least for Category I elements (Th, Am, I) because of their simple chemical behavior. Uncertainty will increase, and is probably greatest, with Category II elements (U, Pu, Np), which may actually represent the most difficult group because redox environment, groundwater chemistry, and sorption surface characteristics interact to affect solid-solution partitioning. Category III, Tc, is subject to less uncertainty than Class II because redox is the dominant variable. On the other hand, the "margin of error" rests solely with the redox prediction.

ACKNOWLEDGMENTS

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REFERENCES

1. U.S. Nuclear Regulatory Commission, "Disposal of High-Level Radioactive Wastes in Geologic Repositories," 10 CFR Part 60, Federal Regist. 48 (130), 35280-35296 (July 8, 1981).

2. E. A. Bondietti, "Mobile Species of Pu, Am, Cm, Np, and Tc in the Environment," Proc., Int. Symp. on Migration in the Terrestrial Environment of Long-Lived Radionuclides from the Nuclear Fuel Cycle (1981), IAEA-SM-257/42, Vienna (in press).

3. E. A. Bondietti and C. W. Francis, "Geologic Migration Potentials of Tc-99 and Np-237," Science 203, 1337 (1979).

4. K. Frohlich and R. Gellermann, "On the Migration of Uranium Isotopes in Sandstone Aquifers," Proc., Int. Symp. on Migration in the Terrestrial Environment of Long-Lived Radionuclides from the Nuclear Fuel Cycle (1981), IAEA-SM-257/88P, Vienna (in press).

APPENDIX

Illustrations Using Laboratory Data

Tables A1 through A5 were abstracted from waste disposal research reports to serve as examples of how the various elements behave on a comparative basis. No credence should be placed on the absolute values of "K_d" (ratio of adsorbed to aqueous distributions, units of mL/g) nor should one table be numerically compared with another. Table A1 indicates that despite a wide range of solute compositions, Am(III) shows high adsorption to clay minerals, as does plutonium, although the final oxidation state distribution was not given [Pu(IV) was used initially]. Neptunium (V) showed poor adsorption and in absolute terms is more affected by solute composition than Am(III) or Pu(IV?). That is, the much lower sorption argues that the risk of migration is greater than Am or Pu and that less margin for error would exist because the sorption parameter value is closer to zero. Remember the absolute magnitude of these numbers is unimportant because they are laboratory-derived. Under field conditions, the exact form of the retardation parameter may not be expressed as "K_d," but it is certain to conclude that Np(V) will migrate faster than Am(III) and Pu(IV).

Table A2 also presents a comparison of two actinide species as a function of experimental solution composition. It can be noted that U(VI) is poorly sorbed when carbonate is present, while Am(III) is more strongly adsorbed when compared to the NaCl solution. This same effect on Am(III) behavior is also apparent by examining Table A1. Tables A1 and A2 demonstrate that although chloride and calcium affect Am(III) sorption, the absolute magnitude of the sorption remains higher (or at least as high) as for the oxycations, Np(V) and U(VI). Further discussions on chloride complexation and Ca(II) competition are beyond the scope of this paper.

Table A3 represents data for two sedimentary rock materials taken from the Rustler Formation, New Mexico. The aqueous phase represents deionized water equilibrated with the rocks until solute saturation was apparent. The dominant constituents of the aqueous phase do impact adsorption. In saturated $CaSO_4$, sorption of Np(V) and U(VI) is poor, but when the more alkaline, dolomite-saturated solution was used, Np(V) sorption increased more dramatically than U(VI) or Pu(IV). In any case, Np(V) and U(VI) present the more mobile case, and are the most sensitive to geochemical influences when a safety margin is anticipated.

Table A4 compares several rock types under very similar groundwater compositions. The purpose of this example is to illustrate how insensitive Am(III) is but how sensitive Np(V) and U(VI) remain (remember Table A3). Plutonium cannot be evaluated adequately because no oxidation state distribution was reported, although Pu(IV) was the initial oxidation state. The low K_d values suggest some Pu(V) formation.

Table A5 gives a more complete comparison of relative sorption under different solid-phase conditions. Note the poor U and Np sorption and the even poorer affinity of TcO_4^- and I^-.

A more realistic but still artificial example is presented in Table A6. Here, actual groundwater and shale from a site of migration of Pu, Am, Cm, and U were evaluated [6] to determine what happens to the adsorption of three oxidation state species when alkaline water from leaching of a disposal trench encounters both acidic (natural) shale and shale preneutralized to simulate 20 years of contact with alkaline water. What can be seen is that the relative behavior of Np(V) and U(VI) reverse, largely due to the effect of carbonate complexes on U(VI) and not Np(V). Again, we see the more pronounced effect that groundwater composition has on these two oxidation states when compared to Pu(IV).

APPENDIX REFERENCES

1. A. Mucciardi et al., " Statistical Investigation of the Mechanics Controlling Radionuclide Sorption, Part II," IN Proc., Task 4, Second Contractor Information Meeting, Vol. II, PNL-SA 7352 Vol. II, Pacific Northwest Laboratory (1978).

2. B. Erdal, Laboratory Studies of Radionuclide Distributions Between Selected Groundwaters and Geologic Media, LA-8088-PR, (Annual Report, October 1978 - September 1979), Los Alamos Scientific Laboratory.

3. C. W. Francis and E. A. Bondietti, "Sorption - Desorption of Long-Lived Radionuclide Species on Geologic Media," IN Proc., Task 4, Third Contractor Information Meeting, CONF-7910160 Vol. II, Pacific Northwest Laboratory (1979).

4. R. J. Silva, L. V. Benson, and J. A. Apps, "Studies of Actinide Sorption on Selected Geologic Materials" IN Radioactive Waste in Geologic Storage (S. Fried, ed.) Amer. Chem. Soc. Sym. Series 100 (1979).

5. B. Allard et al., " Disposal of Radioactive Waste in Granitic Bedrock," IN Radioactive Waste in Geologic Storage (S. Fried, ed.) Amer. Chem. Soc. Sym. Series 100 (1979).

6. E. A. Bondietti, "Mobile Species of Pu, Am, Cm, Np, and Tc in the Environment," Proc., Int. Symp. on Migration in the Terrestrial Environment of Long-Lived Radionuclides from the Nuclear Fuel Cycle (1981), IAEA-SM-257/42, Vienna (in press).

Table A1. Sorption of Np, Pu, and Am as a Function of Aqueous Composition[a]

Solid	Aqueous	K_d, mL/g		
		Np(V)	Pu(?)[b]	Am(III)
Vermiculite	88 m$\underline{M}$ NaCl	61	1600	180
	0.52 m$\underline{M}$ NaCl	33	2300	2900
	0.36 m$\underline{M}$ $NaHCO_3$	12	6800	12000
	0.40 m$\underline{M}$ $CaCl_2$	28	4600	412
Montmorillonite	88 m$\underline{M}$ NaCl	80	900	200
	0.52 m$\underline{M}$ NaCl	46	6800	4200
	0.36 m$\underline{M}$ $NaHCO_3$	9	7500	14000
	0.40 m$\underline{M}$ $CaCl_2$	18	5300	1100

[a]Information taken from reference [1] with some modification.

[b]Pu(IV) was initital oxidation state but final species not characterized.

Table A2. Sorption of U and Am as a Function of Aqueous Composition[a]

		K_d, mL/g	
Solid	Solution	U(VI)	Am(III)
"Sandstone"			
1	15 m$\underline{M}$ $NaHCO_3$	118 $\pm$ 34	1.1×10^5
	169 m$\underline{M}$ NaCl	9200	2.3×10^3
2	15 m$\underline{M}$ $NaHCO_3$	3.3 $\pm$ 1.4	2×10^4
	169 m$\underline{M}$ NaCl	580	3.3×10^2

[a]Table XXIII in reference [2].

Table A3. Comparative Sorption of Pu, U, and Np[a]

Solid (Rustler formation, New Mexico)	Aqueous	Np(V)	U(VI)	Pu(IV)
		K_d, mL/g		
magenta (gypsum)	15 m$\underline{M}$ $CaSO_4$	6.6	2.2	400
culebra (dolomitic)	0.4 m$\underline{M}$ Ca^{2+},	200	11	∿2000
	1.0 m$\underline{M}$ HCO_3^-			

[a]Reference [3].

Table A4. Comparisons of U, Np, Pu, and Am Sorption to Three Rocks from Similiar Aqueous Phases[a]

Solid	K_d, mL/g			
	U(VI)	Np(V)	Pu(?)	Am(III)
Quartz monzonite	0.3	0.5	225	1323
Basalt	2.8	0.5	60	1033
Shale	6.7	49	145	962

[a]Table VII in reference [4].

Table A5. Multi-element Behavior Towards Two Rock-Types[a]

Solid	K_d, mL/g							
	Ra	Th	Pu(?)	Np(V)	U(VI)	Am	TcO_4^-	I^-
Granite	500	4×10^3	320	63	6.3	3×10^5	0.5	0.8
Bentonite/quartz	158	316	160	13	16	500	0.32	0.32

[a]Table V in reference [6].

Table A6. Effect of Shale Acidity on the Sorption of Np, Pu, and U by Conasauga Shale[a]

Conasauga Shale Treatment	Final pH[b]	K_d, mL/g		
		U(VI)	Np(V)	Pu(IV)
Intact	5.8	41	5	51
$Ca(OH)_2$-treated	7.9	14	70	53

[a]Reference [2] in main text section.

[b]The initial pH of the actual groundwater after tagging with Pu, Np, and U was 9.1.

GEOCHEMISTRY OF A CONTACT METAMORPHOSED ZONE: IMPLICATIONS FOR RADWASTE DISPOSAL IN CRYSTALLINE ROCKS

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ABSTRACT

The Eldora stock (Colorado) intruded the 1.5 billion year old Idaho Springs Formation 58 million years ago. Radiogenic ^{40}Ar was lost from, and ^{87}Sr redistributed in, recrystallized minerals as a function of distance from the contract. Previous studies (1) have shown that the elemental and isotopic systematics for K, Ar, Rb, Sr, U, and Pb can be explained by heating of the intruded rocks as a function of distance from the contact. The Eldora stock is one of the best documented intrusives in terms of geochronologic, mineralogic, geochemical and heat flow interpretations (1). New studies have been undertaken to address the problem of elemental migration from the high SiO_2 stock into the more basic Idaho Springs rocks; as this allows the stock to be treated as analogue for high temperature radwaste without benefit of canister or overpack. Our data show no movement of elements from the stock into the intruded rocks except perhaps within 2-3 m. from the contact where magma infiltration is noted. Lanthanide and actinide variations in the intruded rocks are due to pre-stock emplacement events, and Rb-Sr whole rock and stable O isotopic data argue for closed system behavior of the Idaho Springs Formation. The intruded rocks are thus considered favorable for consideration for radwaste storage.

INTRODUCTION

In cases involving igneous rock intrusion, the intrusive body is treated as a heat engine chemically different (in cases examined) from the intruded rocks. In the case of a high temperature intrusive which took some 10^5-to-10^6 years to crystallize, this represents a type of worst case scenario for possible elemental transfer from intrusive into intruded rocks. Study of U, Th, the lanthanides, Rb-Sr geochronology, stable O distribution, and various other elements in the intruded rocks as a function of distance from the contact allows the problem of elemental transfer to be made. We have chosen to study new samples obtained from the Eldora stock, Colorado as previous geochronologic, mineralogic, geochemical, and heat flow model studies are among the best documented (1). These earlier studies were oriented toward behavoir of individual mineral isotopic systematics, however, and while the diffusion and heat flow models provide valuable information, the problem of elemental and/or isotopic perturbations of whole rocks systems have not been previously addressed in detail and no attempt has been made to use the Eldora stock for purposes of a radwaste analogue. Samples were collected in Summer 1980 from the stock and from the Idaho Springs Formation at the contact to 5,000 m. removed from the contact.

The fresh stock rocks range in composition from monzonite to granodiorite. The rocks of the Idaho Springs Formation consist predominantly of alternating felsic- and mafic-rich plagioclase--hornblende gneiss with occasional secretion pegmatites. The age of the Idaho Springs Formation is approximately 1.5 - 1.6 billion years and the rocks were regionally metamorphosed approximately 1.4 billion years ago (1) at which time the pegmatites were injected. Because of their more complex history, elemental variations due to pre-stock emplacement thermal events must be carefully deciphered before attempting to interpret the stock-induced effects at 58 million years ago.

METHODS

Samples from the stock, and of felsic, mafic and pegmatites of the Idaho Springs Formation were taken along a traverse essentially identical to that used by previous investigators (1). A map of the traverse was made (2) and fresh samples were obtained from the contact outward to a distance of 5,000 m. Samples from another site near Antelope Creek, Colorado of both stock and Idaho Springs Formation were also taken for comparative purposes. Splits of the samples were used for thin section preparation, for fission track study, for Rb-Sr and K-Ar geochronologic study, for neutron activation analysis, and for stable O isotopic analysis. The petrography was carried out at both the University of New Mexico (UNM) and Lawrence Berkeley Laboratories, the Rb-Sr work at UNM, the stable O analyses at the University of California at Riverside, and the NAA analyses at the Los Alamos National Laboratory. The techniques employed have been described elsewhere (3).

Discussion of the earlier work (1)

The previously reported Rb-Sr, K-Ar and U, Th-Pb radiometric ages and other data are of value in allowing essential heat models for the Eldora stock to be calculated. In brief, the stock is known to be steep-to-vertical on the sides, and the heat distribution about the stock fairly uniform based on radiogenic Ar, Sr and Pb behavior in minerals and the distance from the contact of the orthoclase-microcline transition ($400^{\circ}C$ at distances from 300-800 m. from the contact). The model advocated by earlier workers (1) is that for a dike with large vertical extent downwards, thickness equal to the average EW dimension, and NS dimension also equal to the average of the stock, and non-convective cooling is assumed. This model has been shown to best explain the isotopic variations as a function of distance from the contact (1). Unfortunately, most of these studies were aimed at behavior of specific minerals and not whole rock samples for elements other than a few Rb-Sr data (e.g., a whole rock isochron of 1.4 to 1.6 billion years is given, but scatter is severe). Of interest, though, is that radiogenic Pb from the stock was shown to be confined to within 3 m. from the contact; and in this zone there is petrographic evidence for infiltration of the Idaho Springs Formation by fluids from the stock.

Discussion of the New Results

The loss of radiogenic ^{40}Ar from minerals as a function of distance from the contact is well known (1); thus we determined the approximate distance from the contact of the current samples by determining their K-Ar ages. In addition, we have carried out Rb-Sr whole rock age determinations on samples both at, near and removed from the contact. The whole rock isochron (4; in press) age is 1.5 ± 0.1 billion years, in exact agreement with earlier work. Minerals from various whole rock samples are more reset near the contact than removed from it, a fact pointed out earlier (1). However the fact that radiogenic ^{87}Sr was redistributed within, and not removed from, the whole rock samples is significant. Further, there is no evidence for gain of Rb or Sr in the Idaho Springs Formation from the Eldora stock except possibly within the 1-to-3 m. immediate contact zone where some mixing of magmatic fluids with the metamorphic rocks occurred.

To further investigate the matter of closed versus open system conditions, as well as to address the problem of conductive versus convective colling, trace element and stable isotopic analyses of the samples used for Rb-Sr whole rock geochronology were carried out. The rare earth chondrite ratio versus rare earth atomic number distribution plot is shown in Fig. 1. Samples 1a, 1b are from fresh stock material and samples 2, 3 from stock-plus-meatmorphic rocks in the 1-to-3 m. contact zone. These four curves are all similar and show pronounced enrichment of the light rare earths (LREE) a fact noted by separate studies (5). Curves 5a, 6a are from mafic units of the Idaho Springs Formation whereas curves 5b, 6b-1 and 6b-2 are from felsic units. The LREE are depleted in the mafic units but enriched in the felsic units; this behavoir is noted not only for these samples taken 20-to-25 m. from the contact (nos. 5, 6) but for samples taken 2,500 m. from the contact where resetting of K-Ar mineral systematics due to heat from the contact is not noted. Our interpretation of these data is that some REE partitioning between mafic and felsic rocks took places during the 1.2 - 1.4 billion year regional metamorphism mentioned earlier. Open systems were evidenced for that time by the formation of secretion pegmatites and K-Ar and Rb-Sr mineral ages; and some pegmatites are enriched in Eu whereas others are depleted, especially when adjacent to mafic units.

The stable oxygen isotopic data are shown in Fig. 2 with del ^{18}O plotted versus the (Fe/Sc) x 10^{-3} ratio which is good indicator of behavior due to oxidation as Fe^{2+}/Sc ratios in many rocks are commonly different than Fe^{3+}/Sc; hence any change in the Fe^{2+}/Fe^{3+} ratio will,

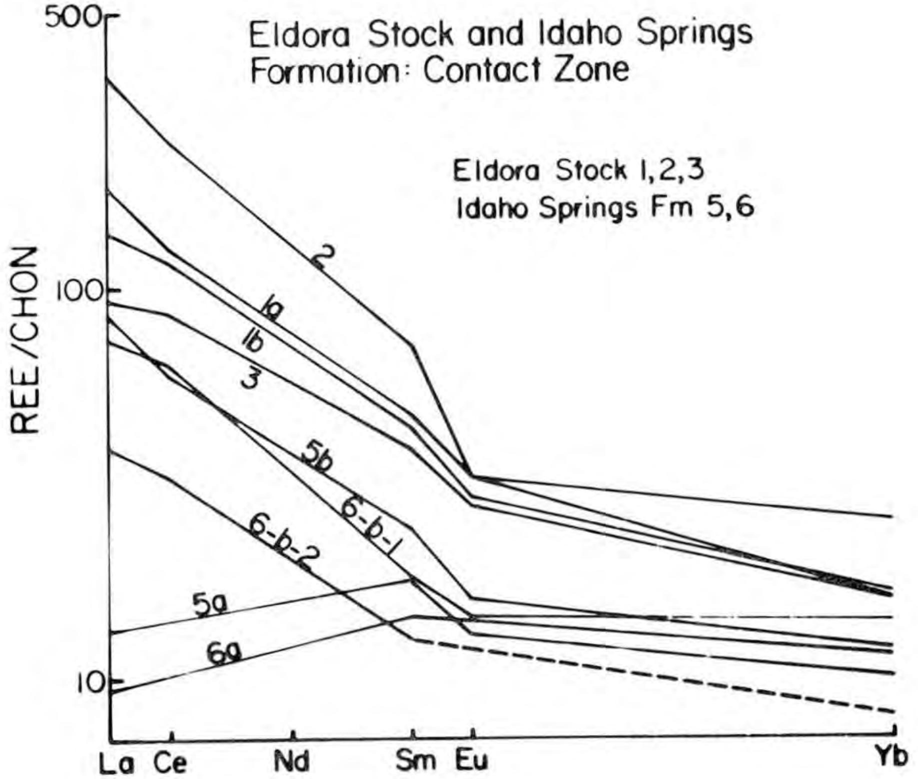

Figure 1. Rare earth element distribution plots for Eldora Stock (nos. 1a, 1b), Idaho Springs Formation (5a, 5b, 6a, 6b), and mixed samples (2, 3).

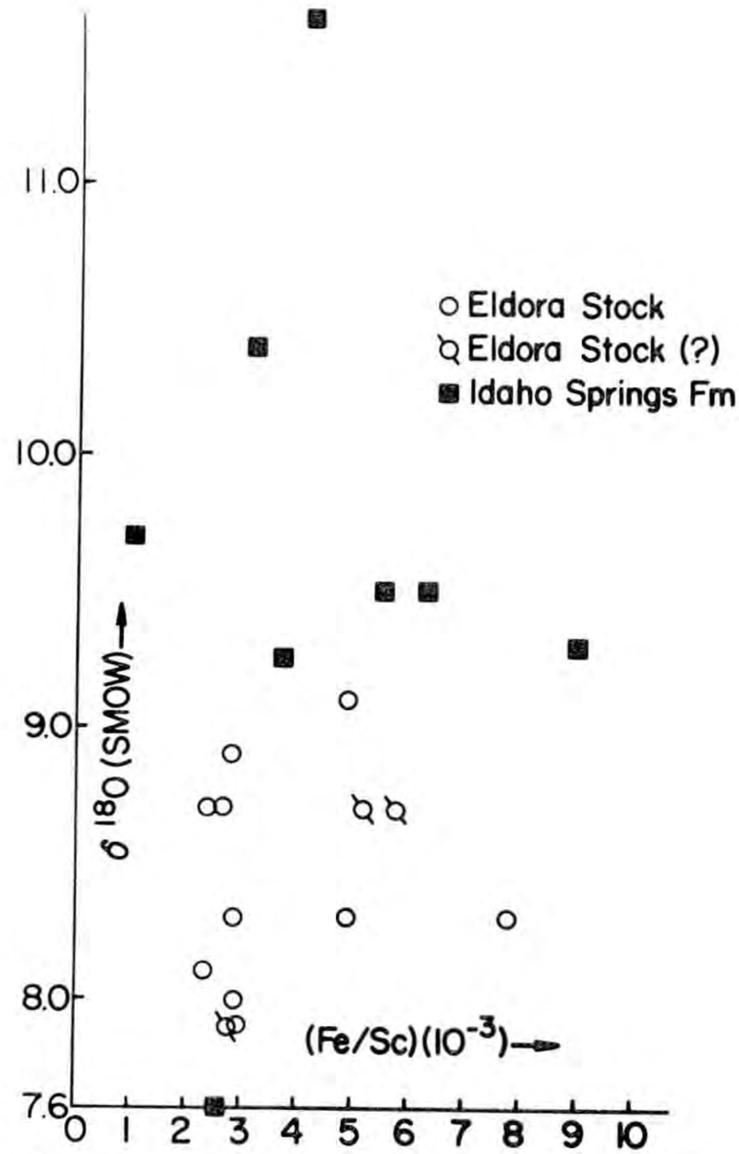

Figure 2. Del ^{18}O vs (Fe/Sc) x 10^{-3} for Eldora stock (open circles), Idaho Springs Formation (Closed boxes) and mixed samples (open-circles-slash).

with open systems for O isotopic exchange, commonly show a strong correlation of increasing del ^{18}O with increasing Fe/Sc. Our data, however, show a narrow range in del ^{18}O (8-to-9 o/oo) for Eldora stock and stock-metamorphic mixes (in the contact zone) and fairly narrow range in Fe/Sc. The Idaho Springs Formation samples show a significantly greater range in del ^{18}O and a wider range in Fe/Sc, but there is no correlation with ^{18}O vs. Fe/Sc as a function of distance from the contact. This indicates essentially closed system conditions for oxygen in the intruded rocks and no exchange of oxygen from the stock and the Idaho Springs Formation. In turn, this implies a lack of convective cooling (usually accompanied by hydrothermal solutions; 6) and colling by conduction appears more likely. Other chemical data support these conclusions (5).

The rocks of the Idaho Springs Formation possess low porosity and permeability, and the numerous fracture fillings and veinlets appear to be pre-stock in age. Except in the 0-3 m. zone of mixing of some fluids from the crystallizing stock due to infiltration into cracked and deformed parts of the Idaho Springs Formation accompanying emplacement, closed system conditions for the 10^5-to-10^6 years for stock crystallization of the Idaho Springs Formation is demonstrated by the geochemical data. If rocks of the Idaho Springs (or equivalent) Formation were to be used for the storage of radwaste, especially when one considers the very small volume of canisters and their lower temperatures, breaching of radwaste canisters would result in only local (cm. ?) migration of radionuclides. We thus consider these rocks suitable for possible storage of radwaste.

REFERENCES

1. S. R. Hart, G. L. Davis, R. H. Steiger, and G. R. Tilton, "A Comparison of the Isotopic Mineral Age Variations and Petrologic Changes Induced by Contact Metamorphism" in Radiometric Dating for Geologists, Intersci. Pubs., NY, 73-110 (1968).
2. H. A. Wollenber, Written Communication, (1980).
3. D. G. Brookins, Uranium Deposits of the Grants, New Mexico Mineral Belt (II), USDOE-BFEC 76-029E (1979).
4. E. C. Simmons and C. E. Hedge, "Minor-Element and Sr-Isotope Geochemistry of Tertiary Stocks, Colorado Mineral Belt", Contr. Mineral. Petrol. 67, 379-396 (1978).
5. D. G. Brookins, L. H. Cohen, H. A. Wollenberg, and M. S. Abashian, "Radwaste Storage in Crystalline Rocks: A Natural Analog", Inter Atomic Energy Agency Sym. on Migration in the Terrestrial Environment of Long-lived Radionuclides from the Nuclear Fuel Cycle (in press; for July 1981).
6. E. M. Parmentier and A. Schedl, "Thermal Aureoles of Igneous Intrusions: Some Possible Indications of Hydrothermal Convective Cooling, Jour. Geol. 82, 1-22 (1981).

POST-CLOSURE ASSESSMENT FOR A CANADIAN NUCLEAR FUEL WASTE DISPOSAL CONCEPT

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ABSTRACT

The Canadian concept for nuclear fuel waste disposal involves emplacement in a crystalline rock formation in the Canadian Shield. The concept is now being assessed generically. Processes within the vault, the geologic formation and the biosphere are being analyzed by field and laboratory research, and predictive mathematical models. Information from these analyses is being incorporated into a system simulation employing lumped-parameter models of the processes, coupled to yield an estimate of risk for the overall concept in the post-closure phase. The uncertainties in parameters are accounted for by performing a Monte Carlo calculation with the coupled models to give a probability/consequence risk analysis. Results demonstrate that the approach adopted provides a suitable framework for establishing the acceptability of a nuclear waste disposal concept, including a systematic treatment of the uncertainties.

INTRODUCTION

The Canadian Concept for nuclear fuel waste disposal involves immobilization and emplacement in a crystalline rock formation in the Canadian Shield. Currently, the program is in a research phase to acquire the knowledge necessary to determine the acceptability of the concept and to develop procedures to compare potential disposal sites, once acceptability in principle is demonstrated (1).

During the research phase, post-closure assessment studies are being done to predict how radioactive material might escape from a disposal vault and migrate through the geosphere and biosphere to cause a radiation dose to man. The purpose of the assessment is to assist in establishing research priorities, to assist in the design of the disposal system and to form a basis for submission to the public, the scientific community and regulatory agencies.

In constructing a methodology to assess the risk associated with the disposal of radioactive waste in the Canadian Shield, several important factors must be considered. First, processes governing the movement of radioactive contaminants through the manufactured and natural materials in the disposal system and through the biosphere are very complex and, in many cases, are not well understood. Consequently, models to analyze these processes are not fully developed, and appropriate idealizations that would allow mathematical analysis have not been agreed upon by researchers.

Second, characterization of crystalline rock, particularly at vault depths, and of physical and chemical conditions that would exist after construction of a nuclear waste disposal vault has only recently become the subject of intensive investigation. This is reflected in large uncertainties in the input data used in the analytical models. Also, in the present phase of the study, the risk assessment is generic. Data are taken from many sources, including several field research areas in the Canadian Shield, and this results in a large variability in data values. Thus, analytical results obtained using single data values become virtually meaningless. Instead, the risk assessment methodology must deal quantitatively with the variability in results.

Third, the disposal system will consist of many components operating together to reduce the potential for exposure to radiation. The effectiveness of the disposal system should be assessed by considering the operation of the system as a whole, rather than by dealing with individual components. The latter course would lead to an exaggerated idea of the importance of the components. In the best of circumstances, it could lead to an over-designed and, therefore, overly expensive system. In the worst case, it could lead to poor design by misdirecting limited resources to research in areas that are not optimum in the sense of reducing the overall risk of waste disposal. Therefore, in determining the dependence of radiation exposure on the many parameters that specify system behaviour, parameter values should be varied simultaneously and the results interpreted statistically (2,3).

With these considerations in mind, the SYVAC (Systems Variability Analysis Code) computer program has been developed to perform generic assessments, taking into account the variability and uncertainty in the state of the disposal system and its surroundings (4,5). A short description of the code and a recent assessment study illustrating its use follow.

THE SYVAC COMPUTER PROGRAM

SYVAC is an executive program that combines responses from several components to simulate overall system response. The transport of radionuclides through a system component, such as a rock mass, is represented by a submodel that depends on system parameters. Submodels representing all components of the system are executed in sequence under the control of SYVAC to provide an estimate of the consequence associated with a given set of parameter values.

A Monte Carlo analysis is performed, i.e., many consequence estimates are made, each time using a set of parameter values obtained by random sampling from specified distributions. The end result is a histogram of consequence estimate versus frequency of that estimate. Intermediate detailed data are saved for subsequent analysis.

It is important to note that SYVAC is designed to allow different system structures (i.e., linking of subsystems) to be defined at execution time, and to allow alternative submodels representing subsystem behaviour to be easily inserted. Also, it allows submodels of arbitrary complexity to be used. No physical assumptions are made by SYVAC concerning the behaviour of a subsystem. For example, the response of a subsystem may be highly nonlinear and may depend on the state of other subsystems.

CURRENT ASSESSMENT

SYVAC has been used with simple models for the vault, geosphere, and biosphere. The vault submodel assumes that water flows in a vertical direction through the rooms and pillars of the mined region. The flow through a room is determined by the ratio of permeability of the room to that of the pillars. The release of radionuclides from containers of used fuel deposited in the vault is assumed to be controlled by the solubility of UO_2 in granite groundwater, except for small fractions of some radionuclides (Cs, I), which migrate to the fuel-sheath gaps during irradiation and which are assumed to be released instantaneously upon container failure. Mass transport away from a container is calculated as one-dimensional transport across material surrounding the container. Containers fail according to a time-dependent function that depends on system parameters.

The radionuclide source produced by the vault submodel is used by the geosphere submodel to predict, as a function of time, quantities of radionuclides leaving the rock and entering the biosphere. The prediction is made by assuming one-dimensional convection of radionuclides without hydrodynamic dispersion. The holdup of radionuclides by chemical interaction with the rock is accounted for by using the retarded-velocity method.

Nuclides entering the biosphere are assumed to enter one of two compartments, which represent soil or a lake. In the soil compartment, radionuclides are distributed between the solid and liquid (groundwater) phases assuming the partitioning mechanism is ion exchange, that equilibrium is attained and that the ratio of concentrations in the solid and liquid phases is independent of concentration. In the lake perfect mixing is assumed. The loss of radionuclides from both compartments, due to runoff, is taken into account. The fifty-year committed dose equivalent to an individual in a group receiving maximum exposure is estimated from soil and lake concentrations using dose/concentration ratios computed by the FOOD-II (6) and NEPTUN (7) programs.

Tables 1, 2 and 3 list the parameters for the vault, geosphere and biosphere submodels respectively, along with the values, or distributions of values, used.

Figure 1 shows the results of 1000 estimates of maximum dose equivalent to an individual in the most exposed group. On this histogram, vertical lines are drawn at the natural background dose, at 1% of natural background and at 1% of the regulatory limit for members of the public. Approximately 700 other cases, in which no dose results before one million years (due to an estimated transit time to the surface exceeding a million years), were excluded from the histogram. This time cutoff was arbitrarily chosen, but was justified on the basis that doses beyond this time are due mostly to ^{238}U and its daughters, which are already present in nature.

An alternative way of plotting these results is shown in Figure 2 where the complementary cumulative probability is plotted against the annual dose equivalent estimate. Again, possible measures of acceptability are indicated and the probability of exceeding those measures can be read from the curve. For example, the probability of a consequence estimate exceeding 1% of natural background can be read as 0.005, or 0.5%.

CONCLUSIONS

The method being adopted for risk assessment for the Canadian nuclear fuel waste management program is based on an analysis of the behaviour of the overall system rather than individual components. It is recognized that there will always be uncertainties in the present and future states of the system, and that the risk assessment must take these uncertainties into account. We believe the system variability analysis approach includes a systematic treatment that provides a framework for establishing the acceptability of any nuclear waste disposal concept including a systematic treatment of the uncertainties.

REFERENCES

1. J. Boulton, Ed., Second Annual Report of the Canadian Nuclear Fuel Waste Management Program, AECL-6804 (1980).

2. R.L. Iman, J.C. Helton and J.E. Campbell, Risk Methodology for Geologic Disposal of Radioactive Waste: Sensitivity Analysis Techniques, NUREG/CR-0394 (1978).

3. B. Ross, C.M. Koplik, M.S. Giuffre, S.P. Hodgin, J.J. Duffy and J.Y. Nalbandian, User's Guide to NUTRAN: A Computer Analysis System for Long-Term Repository Safety, unpublished work, TR-121 (1980). Available from Atomic Energy of Canada Research Company, Chalk River, Ontario K0J 1J0.

4. K.W. Dormuth and R.D. Quick, "Accounting for Parameter Variability in Risk Assessment for a Canadian Nuclear Fuel Waste Disposal Vault", Int. J. Energy Systems 1, 125 (1981).

5. K.W. Dormuth and G.R. Sherman, SYVAC - A Computer Program for Assessment of Nuclear Fuel Waste Management Systems, Incorporating Parameter Variability, AECL-6814 (to be published).

6. R. Zach, FOOD II: An Interactive Code for Calculating Concentrations of Radionuclides in Food Products, AECL-6305 (1978).

7. R. Zach, NEPTUN: An Interactive Code for Calculating Doses to Man Due to Radionuclides in Aquatic Food Chains, AECL-6450 (1980).

Table 1. Probability Distributions of Vault Parameters

Parameters	Values	Distribution
Amount of fuel in vault	350 000 Mg	Constant
Number of containers in vault	246 000	Constant
Temperature of vault	100°C	Constant
Effective area of a container for diffusion	9.3 m^2	Constant
Effective area of a container for flow	4.9 m^2	Constant
Thickness of buffer	1 m	Constant
Amount of Cs and I in gaps	0.7%	Constant
Hydraulic conductivity of rock	10^{-12} to 10^{-7} m/s	Uniform in log
Hydraulic gradient	10^{-5} to 10^{-3}	Uniform
Hydraulic conductivity of buffer	10^{-14} to 10^{-8} m/s	Uniform in log
Diffusion coefficient in water at 100°C	0.1 - 1.0 m^2/a	Uniform
Solubility of UO_2	mean value = 3×10^{-7} mol/m^3	Log normal
Mean time for container failure	1000 - 5000 a	Uniform
Container failure rate		Gaussian
Effective porosity of buffer for diffusion and flow	10^{-2} to 10^{-1}	Uniform in log
Radionuclide distribution coefficients in buffer	depends upon nuclide	Uniform

Table 2. Probability Distributions of Geosphere Parameters

Parameters	Values	Distribution
Path length	1 - 40 km and 40 - 1000 km, probability of the first range is twice that in the second range	Uniform in each range
Porosity	10^{-6} to 10^{-3}	Uniform in log
Permeability	10^{-19} to 10^{-14} m^2	Uniform in log
Hydraulic gradient	10^{-5} to 10^{-3}	Uniform
Surface sorption coefficient	Depends upon radionuclide	Constant

Table 3. Probability Distributions of Biosphere Parameters

Parameters	Values	Distribution
Fraction of precipitation less evaporation penetrating soil	0.45	Constant
Volume fraction of water in unsaturated soil	0.2	Constant
Soil bulk density	1500 kg/m^3	Constant
Soil compartment depth	1.2 m	Constant
Discharge lake depth	15 m	Constant
Dose/Concentration Ratios	Depends upon radionuclide	Constant
Discharge soil area	20 to 200 km^2	Uniform
Soil distribution coefficient, K_d	Depends upon radionuclide	Uniform
Discharge lake area	1 to 1000 km^2	Uniform
Precipitation (less evaporation)	0.3 to 0.7 m/a	Uniform
Catchment area/discharge lake area	1 to 10	Uniform

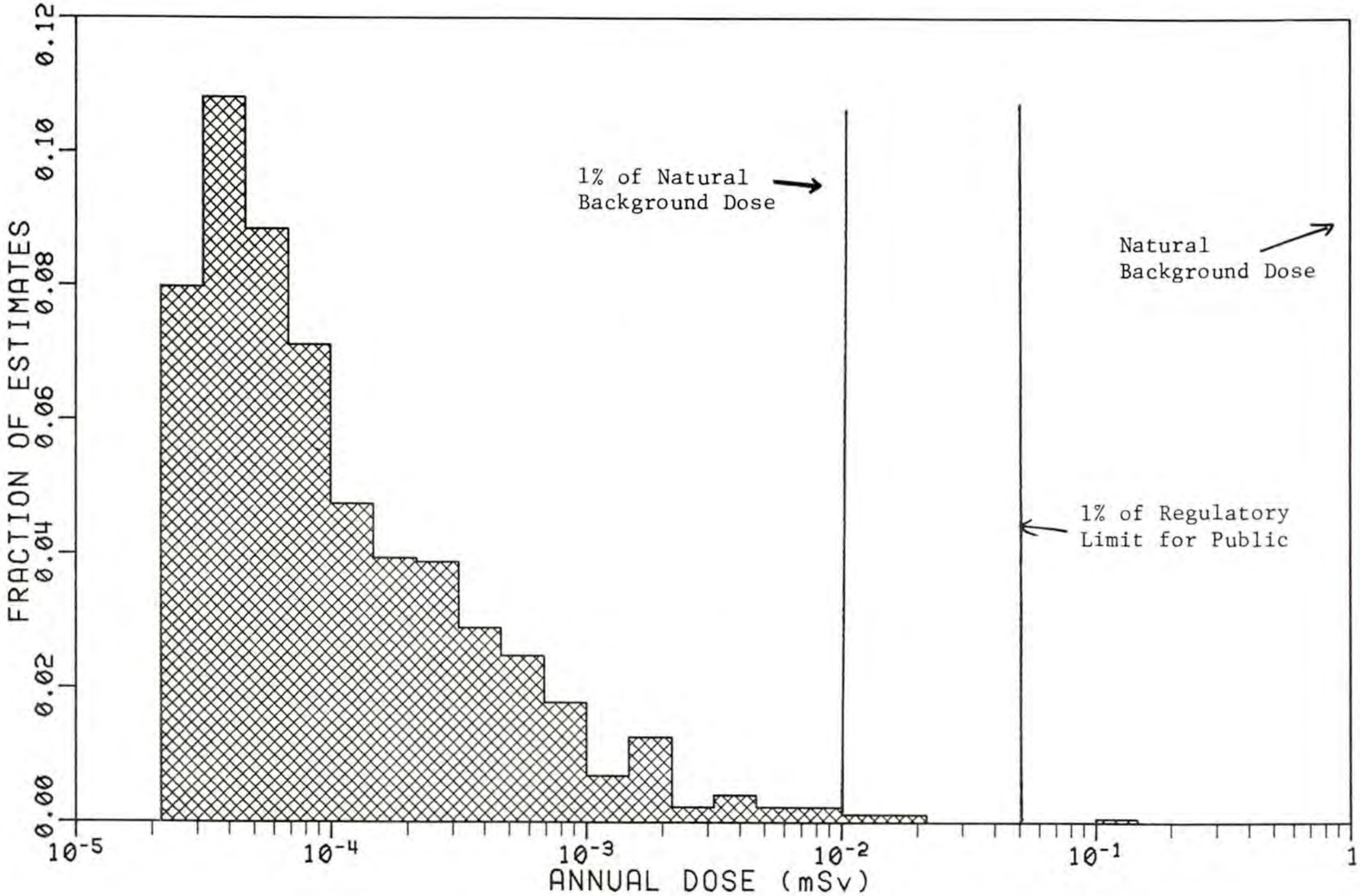

Fig. 1 Maximum dose equivalent to the most exposed individual.

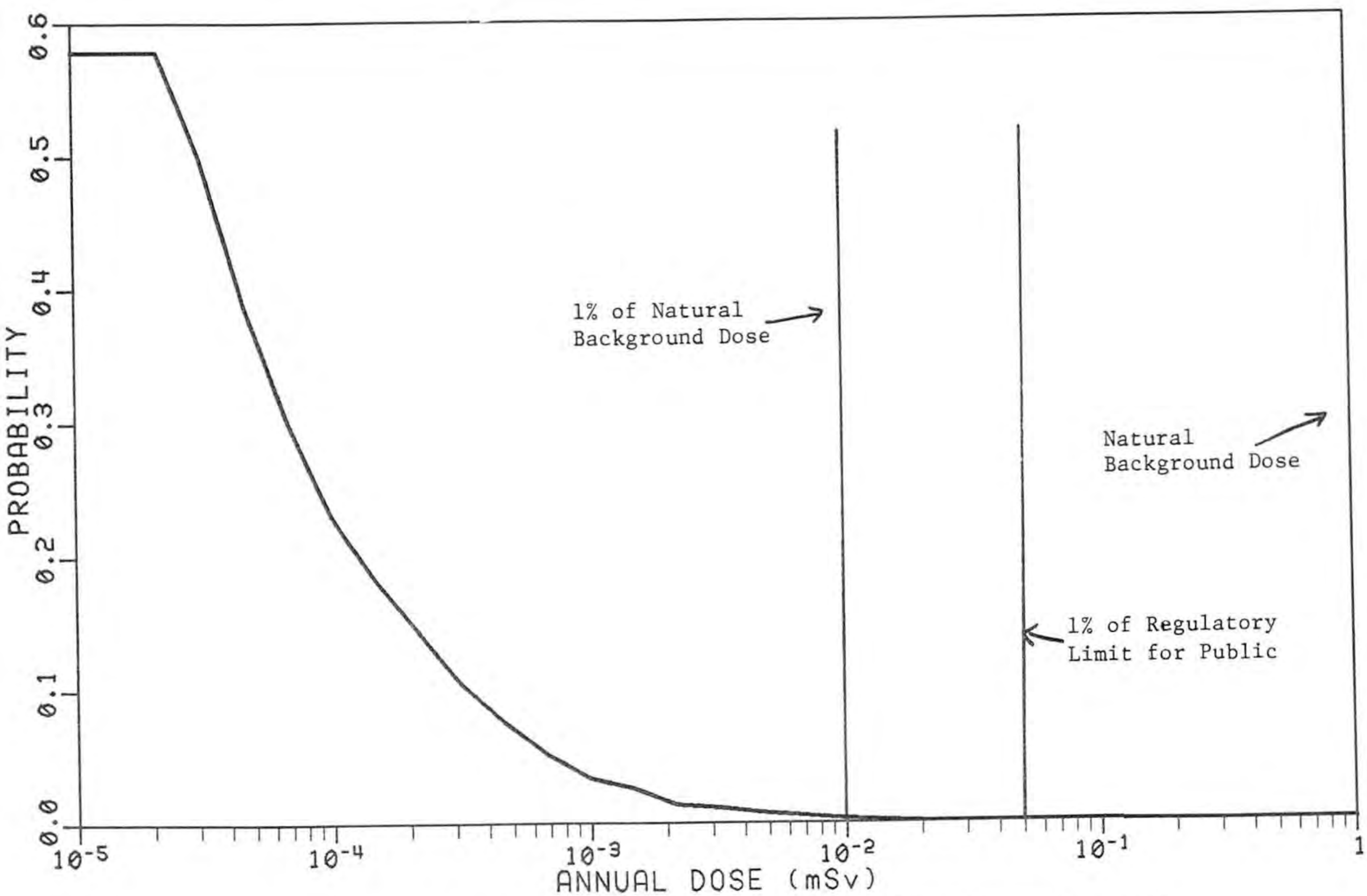

Fig. 2 Downward cumulative probability for maximum dose equivalent to most exposed individual.

THE EFFECT OF HYDROLOGIC PARAMETER VARIATION ON RADIONUCLIDE TRANSPORT IN THE RUSTLER AQUIFER (SOUTHEASTERN NEW MEXICO)

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ABSTRACT

A simple mathematical model was used to evaluate the effect of varying hydrologic transport parameters on the movement of ^{239}Pu from the proposed WIPP radioactive waste repository through the Rustler aquifers to the Pecos River. The evaluation concluded that the parameters with the most significant effects in the dose calculations are the hydraulic conductivity (K) and the distribution coefficient (K_d). This is mostly due to the uncertainty associated with bounding these values in a system where radionuclides may exist in various chemical forms and flow through aquifers dominated by fracture flow. However, in all plausible cases calculated concentrations were less than permissible levels for drinking water. The evaluation was a useful first approximation to uncertainties analysis. A better understanding of the bounding values of the K_d and K parameters in fracture flow system is needed.

INTRODUCTION

The mission of the Environmental Evaluation Group (EEG) is to provide a technical evaluation of the radiological aspects of the proposed Waste Isolation Pilot Plant (WIPP Project) for the State of New Mexico. A significant portion of this evaluation involves the geologic and hydrologic suitability of the site. Since the time table for the WIPP Project is ahead of that for the ONWI program, EEG has been forced to make evaluations in some cases prior to the existence of a proposed methodology. The case discussed in this paper involves a simple parametric analysis of a hydrologic transport evaluation.

The proposed WIPP Site is located in Southeastern New Mexico, 40 km east of Carlsbad, and 24 km northeast of the Pecos River at the closest point. The repository horizon would be in the Salado formation (predominately halite) about 650 m below the surface. The Magenta and

Culebra aquifers are locatd in dolomite layers in the Rustler formation about 460 m above the repository. Flow in these aquifers is believed to be primarily due to fracture permeability. The transport of nuclides from the repository through these aquifers to the Pecos River has been considered to be one of the most significant pathways.

In the Draft Environmental Impact Statement on WIPP the Department of Energy (DOE), using a geosphere-transport model developed by Intera, modeled the transport of radionuclides from several breach scenarios to the Pecos River through the Magenta and Culebra aquifers, [1]. Only single values were used for the various aquifer parameters. Recently another modeling study by D'Appolonia has incorporated parameter variation [2]. Greenfield obtained reasonable agreement with the results in the Draft EIS by using a simplified hand calculation in which dispersion was neglected [3]. This calculation was useful because it suggested that the Intera Model contained no significant conceptual or mathematical errors and also emphasized the inter-relationships of the key parameters.

However, this calculation did not answer the question of whether this model was truly conservative or whether plausible changes in one of the parameters might lead to significantly higher concentrations in the Pecos River. The sensitivity of the various transport parameters on the arrival time and concentration of ^{239}Pu in the Pecos River was evaluated by Wofsy in order to answer this question [4].

METHODOLOGY

Site specific data and ranges in values found in the literature for field or laboratory studies were used to estimate the plausible changes that might occur in the various parameters of the transport equation. In the final analysis the values considered plausible were chosen subjectively.

This is a simplified approach with a data base that does not permit the probability of occurrence to be computed. The principal value of the procedure is to determine which parameters have plausible values that may lead to non-negligible radionuclide concentrations in the accessible environment. Only these critical parameters need to be characterized more precisely.

This analysis uses the same release rate of ^{239}Pu into the Rustler aquifer that was used in Communication Event 1 in the WIPP Safety Analysis Report (SAR)[5]. No uncertainty was assumed in this release rate. The analysis was limited to ^{239}Pu because it comprises about three-fourths of the curies in the repository during the period from 1,000 to 100,000 years and would be the dominant radionuclide in the Pecos River if it is sufficiently mobile.

Transport Equations

The transport equations used are shown in Table I. These equations assume one-dimensional, homogeneous, and darcian type flow. The average nuclide velocity is less than the average water velocity in all cases where the distribution coefficient is greater than zero. Time of travel of a nuclide can be determined by the bottom expression which can also be used to obtain the sensitivity of varying the individual parameters.

Due to radioactive decay the calculated concentration of a nuclide reaching the biosphere will decrease exponentially with time. This is shown graphically on Figure 1 where the ^{239}Pu concentration is seen to be especially sensitive to arrival time between 1,000 and 100,000 years. Consequently, the expected arrival time is the dominant value in the evaluation. The parameters chosen by DOE led to an estimated arrival time for ^{239}Pu at the Pecos River of 140 million years. Since ^{239}Pu arrival times of much less than one million years are of most significance, the parameters evaluated were those that might change by orders-of-magnitude.

Changes in effective porosity were not considered because values usually do not vary by an order-of-magnitude and the effect on radionuclide travel time is significant only in cases where the distribution coefficient (K_d) is less than about one mℓ/g. Also, aquifer density changes were not considered since they vary by much less than an order of magnitude.

Table 1. Key Transport Expressions

Average linear velocity of water,	$\bar{v} = \frac{K}{\theta}\left(\frac{\Delta h}{\Delta \ell}\right)$	ft/year
Average nuclide velocity,	$r = \dfrac{\bar{v}}{1 + \frac{\rho}{\theta} K_d}$	ft/year
Time of travel of nuclide,	$T = \dfrac{d(\theta + \rho K_d)}{K\left(\frac{\Delta h}{\Delta \ell}\right)}$	years

Where:
- K = hydraulic conductivity, ft/y
- θ = aquifer porosity
- $\frac{\Delta h}{\Delta \ell}$ = hydraulic gradient
- ρ = aquifer density, g/mℓ
- d = distance, ft.
- K_d = distribution coefficient, mℓ/g

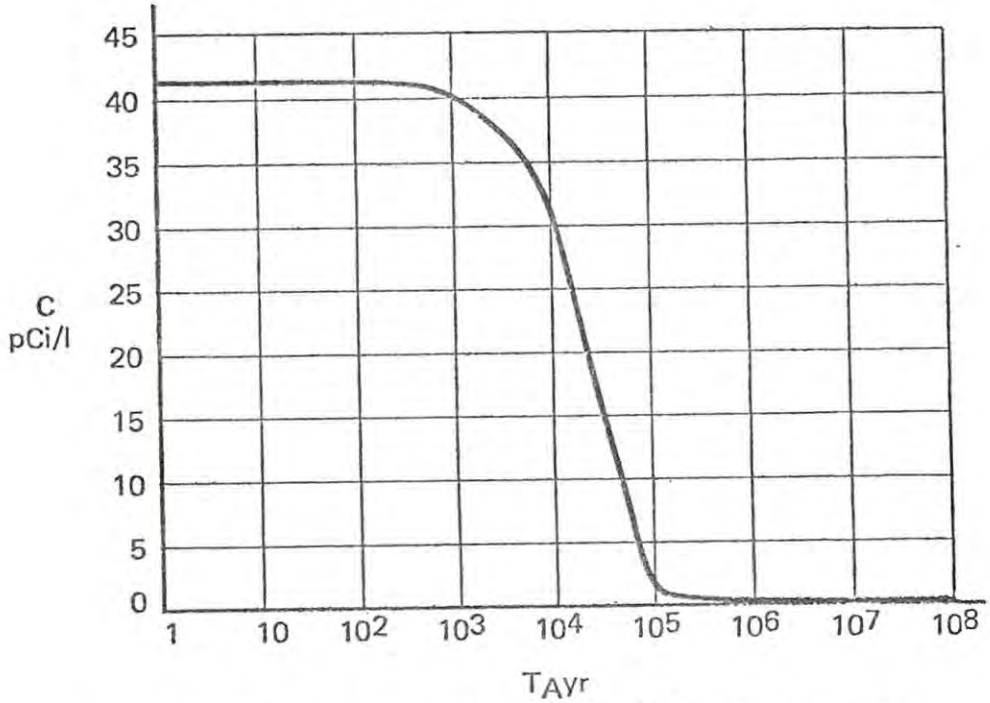

Fig 1. Peak Concentration (C) of ^{239}Pu in the Pecos River as a Function of Arrival Time (TA).

Hydraulic Gradient

Hydraulic gradients have the potential of changing significantly in some locations if hydrologic conditions change drastically; which might occur over a several thousand year period. However, Davis has concluded that the average gradient in the Rustler aquifer would increase by no more than 3 1/2 times even if the probable recharge area were saturated all the way to the surface [6]. Consequently, variation in the hydraulic gradient was not included in the analyses.

Hydraulic Conductivity

Flow in the Rustler aquifers is believed to be governed by fracture flow and for this reason it is difficult to know what areal average hydraulic conductivity (K) value is appropriate to use. The DOE used 1 foot per day for the first 5 miles and 4 feet per day for the next 10 miles. An average K value of 10 feet per day for the entire path length was chosen as the worst plausible case. While this choice is probably conservative, higher values are certainly possible, and there is no way of knowing at present if this is truly an upper limit for this parameter.

Distribution Coefficient

The problems with determining K_d values are well known to workers in this field. Variations occur between laboratories; with changes in water solution; with the chemical state of the nuclide; and with loading effects. Variations in observed values of several orders-of-magnitude have been reported for some radionuclides. An average Plutonium K_d value of 10 mℓ/g was chosen as the worse plausible, although calculations were also performed for values of 1 and 0 mℓ/g.

Heterogeneity Effects

The discussion up to this point has been limited to plausible variations in average transport parameters. Use of average values is reasonable if the aquifer and the nuclide form are relatively homogeneous. However, with radionuclides that are present in all sorts of physical forms and matrices and with an aquifer where fracture flow is believed to be dominant it may not be conservative to assume that all nuclides have average behavior.

Radioactive decay of a nuclide also affects the possible significance of a small, mobile subpopulation of nuclides. For example, 1% of the ^{239}Pu nuclides arriving at a time of 2,800 years would result in almost 3 times the concentration (and radiation dose) caused by the remaining 99% arriving at 200,000 years.

Heteorogeneity in K_d values could be expected if there is a variation in the chemical state of a portion of the plutonium nuclides. This is possible due to their variable form at time of storage and the presence of chelating agents in the repository. Also, column infiltration studies often show that small populations of transuranics will move either with the speed of water of much more rapidly than predicted by their K_d values. These analyses considered that 1% of the plutonium moving with the speed of water or 10% moving at 0.1 times the speed of water were plausible conditions.

No calculations were made using heterogeneous hydraulic conductivity values. It is apparent that in an aquifer where fracture flow is dominant some particles of water will move, at least for a time, at a velocity significantly more rapid than the average velocity. The extent that this might influence early arrival times at a distance of 15 miles is unknown. Another uncertainty is the possible relationship between velocity heterogeneity and K_d heterogeneity since water particles moving more rapidly through fractures probably have lower than average effective K_d values.

FINDINGS

The calculated concentrations of ^{239}Pu in the Pecos River for the various modifications are shown in Table 2. The highest plausible value is 1.7 pCi/ℓ. This value results from transit times of 28,000 and 113,000 years. This should be compared to 15 pCi/ℓ permitted in the EPA drinking water standards for finished drinking water.

These calculations are believed to bound the concentrations of ^{239}Pu in the Pecos River. However, an exception is the possible effect of heterogeneity on hydraulic conductivity values and perhaps a combined K and K_d heterogeneity effect.

The limited scope of this evaluation needs to be kept in mind. Uncertainty in the source term was not considered and concentrations of other radionuclides were not estimated.

Table 2. Peak Plutonium-239 Concentrations in the Pecos River

Modification	Pu-239 Concentration (pCi/ℓ)
Base Case[a]	<(1-50)[b]
K_d = 10 mℓ/g	(2.6-6)
K = 10 ft/d, K_d = 10 mℓ/g	1.7
K_d = 1 mℓ/g[c]	7.0
K_d = 0 mℓ/g[c]	37.
1% Pu-239 moves at 1.0v	0.37
10% Pu-239 moves at 1.0v[c]	3.7
10% Pu-239 moves at 0.1v	1.7

(a) Base case conditions (from WIPP Safety Analysis Report) are:
- hydraulic gradient = (3.8-3);
- distribution coefficient (K_d) = (2.4 + 3) mℓ/g;
- hydraulic conductivity (K) = 1 ft/day;
- distance = (2.6 + 4) ft;
- porosity = 0.1;
- aquifer density = 2.0 g/mℓ

(b) (1-50) = 1×10^{-50}

(c) These modifications are not considered plausible.

CONCLUSIONS

This simplified parametric analysis was useful because it indicated the considerable range in plausible nuclide arrival times in the biosphere (Pecos River). However the range of values used, while they appear reasonable, were obtained subjectively with no attempt to make probability and uncertainty determinations.

The results suggest that uncertainty in the average values used by DOE could be large enough to warrant a more sophisticated analysis where probabilities could be expressed in quantifiable terms. Efforts should also be made to better understand the low probability "tail" of both the distribution coefficient and hydraulic conductivity in fracture-flow systems. For example, can the log-normal distribution that appears to fit K values in most media be extrapolated to the 1% level in fracture systems?

A methodology that is able to accurately assess the uncertainty associated with radionuclide breach and transport will probably not be available within the next few years. Yet there is a need for workers in this field to try to reach early agreement on an approximate methodology that can be used on an interim basis. Several potential advantages could come from the existence of such methodology:

(1) more precision in designing field sampling programs to optimize the usefulness of collected data;

(2) a better ability to estimate "realistic" bounding cases;

(3) a means by which relative uncertainty between alternative designs and projects could be estimated.

REFERENCES

1. U. S. Department of Energy, Draft Environmental Impact Statement, Waste Isolation Pilot Plant, DOE/EIS-0026-D, (April 1979).

2. D'Appolonia Consulting Engineers, Inc. Modeling Verification Studies Long-Term Waste Isolation Assessment-Waste Isolation Pilot Plant (WIPP) Project Southeastern New Mexico, (January 1981).

3. Greenfield, Moses A., Appendix VI of Radiological Health Review of the Draft Environmental Impact Statement Waste Isolation Pilot Plant, DOE, EEG-3, (August 1979).

4. Wofsy, Carla, The Significance of Certain Rustler Aquifer Parameters for Predicting Long-Term Radiation Doses from WIPP, EEG-8, (September 1980).

5. U.S. Department of Energy, Waste Isolation Pilot Plant Safety Analysis Report, (January 1980).

6. Davis,S. N., Hydrogeologic Effects of Natural Disruptive Events on Nuclear Waste Repositories, PNL-2858, (June 1980).

Session IV:

RISK ASSESSMENT — THE ONWI/PNL PERSPECTIVE

Chairman

Rowena O. Chester

Oak Ridge National Laboratory

COMMENTS ON THE NRC APPROACH TO RADIOACTIVE WASTE DISPOSAL STANDARDS AND THE TECHNICAL APPROACH TO PERFORMANCE ASSESSMENT IN THE NWTS PROGRAM

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INTRODUCTION

The uncertainties associated with the disposal of radioactive wastes is an important topic and one that has been given insufficient technical attention. The Nuclear Regulatory Commission and its contractor, the Oak Ridge National Laboratory, should therefore be commended for sponsoring this meeting.

This paper is divided into two parts. The first part is a commentary about what was presented and what was not presented in the first two and one-half days of this meeting concerning the Nuclear Regulatory Commission's approach to setting radioactive waste disposal standards. The second part is a description of the technical approach to performance assessment in the National Waste Terminal Storage Program.

The comments concerning the NRC approach to waste disposal standards are made as a member of the "technical public"; the description of the NWTS technical approach to performance assessment is made as a member of a DOE contractor organization.

NRC APPROACH TO RADIOACTIVE WASTE DISPOSAL STANDARDS

A draft of the Nuclear Regulatory Commission's proposed radioactive waste disposal standards (10 CFR 60) appeared in the Federal Register during 1980. Key passages from this draft are the following:

> "In considering whether there should be other barriers, a key question which needs to be answered is whether it is prudent, in view of the nature of the problems and uncertainties involved, to rely on the geologic setting alone to accomplish the functions stated above."
>
> Federal Register/Vol. 45, No. 94/p. 31396
>
> "... it is desirable to specify technical criteria associated with the regulatable elements in such a manner as ... to predicate their technical justification on the

results of quantitative modeling ... in those instances where quantitative modeling can contribute to their technical justification."

Federal Register/Vol. 45, No. 94/p. 31398

"The Department shall design waste packages so that there is reasonable assurance that radionuclides ... will be contained for at least the first 1,000 years afters decommissioning ... starting 1,000 years after decommissioning ..., the radionuclides ... will be released from the underground facility at an annual rate that ... is in no case greater than ... one part in one hundred thousand of the total activity ..."

Federal Register/Vol. 45, No. 94/p. 31400

Summarizing, the first quote says that the key issue associated with geologic disposal is the degree of reliance on geologic barriers and that the uncertainty in site-related models and data is the reason for the concern. The second quote says that it is desirable to use models in providing a technical basis for regulatory standards, and the third quote provides an example of the numerical component performance standards in the regulation.

What has so far been missing from the standards is a quantitative analysis of the uncertainty in site-related models and data and a quantitative analysis which relates the component performance standards as a set to the overall system performance standard. Since these analyses are desirable from the draft text of 10 CFR 60, their presentation by the NRC at this meeting for review by the participants would have been welcomed. Because such a basis has not been provided, the proposed standard cannot be subjected by others to a technical review for correctness. Hence the standard may be forcing isolation system development activities in directions that do not lead to improved performance or justified increases in the confidence that it can be achieved.

From the presentation of the NRC rationale, the developers of the "package lifetime" standard apparently are interested in designing the components of the isolation system. Therefore, NRC should start its contractors applying performance assessment and uncertainty analysis technologies to provide the quantitative technical basis for a proper set of component performance standards. This technical basis needs to be part of the standarads so that a meaningful review of the standards is possible and so that future implementers of the standards will understand the situations for which the standards are applicable.

This is a serious matter. There are billions of dollars of future government expenditures riding on these standards, and it is important to ensure that the benefits to future generations and the peace of mind of the present generation are commensurate with those costs.

The Department of Energy strongly supports the principle that nuclear wastes should be isolated in a conservatively designed system of multiple barriers which act in concert to protect present and future humans and their environments from the waste. The Department has not endorsed the concept of numerical standards on isolation system components but instead has stated the following in its "Confidence Rulemaking" document:

> "Waste containment within the immediate vicinity of initial placement should be virtually complete during the period when radiation and thermal output are dominated by fission product decay. Any loss of containment should be a gradual process which results in very small fractional waste inventory release rates extending over very long release times, i.e., catastrophic losses of containment should not occur."

TECHNICAL APPROACH TO PERFORMANCE ASSESSMENT IN THE NWTS PROGRAM

The performance assessment activities in the NWTS program can be conceptually understood by examining Figure 1. In order to accomplish the disposal of commercial radioactive wastes in the United States, the NWTS Program must characterize and select sites, design and construct underground repositories, design and manufacture waste packages, obtain licenses to operate the facilities, and develop the technology necessary to accomplish the previous four items. All five of these activities require decisions for which technical bases are required. The role of NWTS Performance Assessment is to provide those bases through use of mathematical modeling technology and relevant data. These modeling applications in turn require an isolation system performance model consisting of submodels for the three major subsystems, the waste package, the repository, and the site, and these submodels in turn require models for individual isolation system phenomena. Finally, the individual models require laboratory, in situ, and field information both to develop the models and to provide values for the parameters necessary for their application.

NWTS Performance Assessment operates on the general principle that future isolation system performance should be predicted as realistically as resonably possible and the uncertainties in those best-estimate predictions should be quantified to the extend practical. This principle places some stringent demands on the application of performance assessment modeling technology and those demands in turn place other demands on the development of the technology. As a consequence, the NWTS Performance Assessment capability will contain two levels of models, an integrated system of simple models and an unintegrated system of complex models. In applying the technology, a large number of simulations of system futures will be made using the integrated system of simple models. These simulations will include deterministic modeling of processes using distributions for the event occurrence probabilities. The performance measure values from these multiple simulations will be combined to provide a best-estimate prediction of performance and confidence limits on that best-estimate prediction. Then a small number of these simulations will be checked using the unintegrated set of complex models. This approach attempts to strike a balance between efficiency and accuracy in performance assessment applications. The approach is efficient because many calculations are made with simple models and accurate because a portion of the results from the simple models is checked against the results from the complex models.

In understanding the NWTS performance assessment technology application approach, it is helpful to define what the approach is not as well as what it is. Approaches not being used in NWTS Performance Assessment include traditional risk assessment (fault/event tree analysis) and worst-case/maximum credible scenarios/design basis scenarios. The former is not being used because nuclear waste isolation systems are passive, nonoperating systems composed of components which degrade rather than fail, and isolation system performance is dominated by continuous processes rather than discrete events. The latter is not being used because it is especially difficult to define the boundary between the credible and the incredible for this problem and the use of these concepts improperly focuses activities on the definition of the credible/incredible boundary rather than the understanding of isolation system behavior. The worst-case/maximum credible scenario/design basis scenario approach also distorts the design of the system toward the prevention or mitigation of extreme and unlikely conditions when system performance is usually dominated by less extreme, more likely conditions.

The NWTS performance asssessment technology development approach follows a formal and logical sequence of activities. First, the requirements for the modeling capability are defined. This involves the definition of performance specifications for both the integrated system of simple models and the unintegrated set of complex models. The performance specifications define "what" equations or relationships the various models describe and "what" model interrelationships are needed for particular applications. Second, existing models are compared with the performance specifications to determine what models are already available for use, what models can be modified for use, and what models need to be developed. Next, the models requiring modification and the new model developments are provided by a process that includes individual model design specifications, model coding, model verification, model validation, and model limitations definition. The individual model design specifications define "how" the equations or relationships defined the individual model performance specifications are solved in the computer programs. The model vertifications check to determine if the computer programs, as coded, meet the design specifications (i.e., that they do the calculations correctly) through use of analytical benchmarks and code-to-code comparisons. The model validations check to determine if the computer programs meet the performance specifications (i.e., that they predict the truth to an acceptable degree) through comparisons with laboratory, in situ, and field data including the results from natural analog studies. The model limitations definitions use the results from the validation activity to determine the conditions under which the models are not valid. Finally, the models are integrated by a process that includes systems model design specifications, model coding, and code checking. The systems model design specifications define "how" the model interrelationships defined in the systems model performance specification are accomplished in the systems code (or codes). The systems model cannot be verified because there will neither be an analytical benchmark nor another systems model to which it can be compared. The verification of the systems model must therefore rely on careful checking of the code. Likewise, the systems model cannot be independently validated because it is not possible to conduct tests of nuclear waste isolation systems over time periods of interest to performance assessment. Thus, the verification and validation of the systems performance model must be inferred from the verification and validation of the individual models whose interactions it manages.

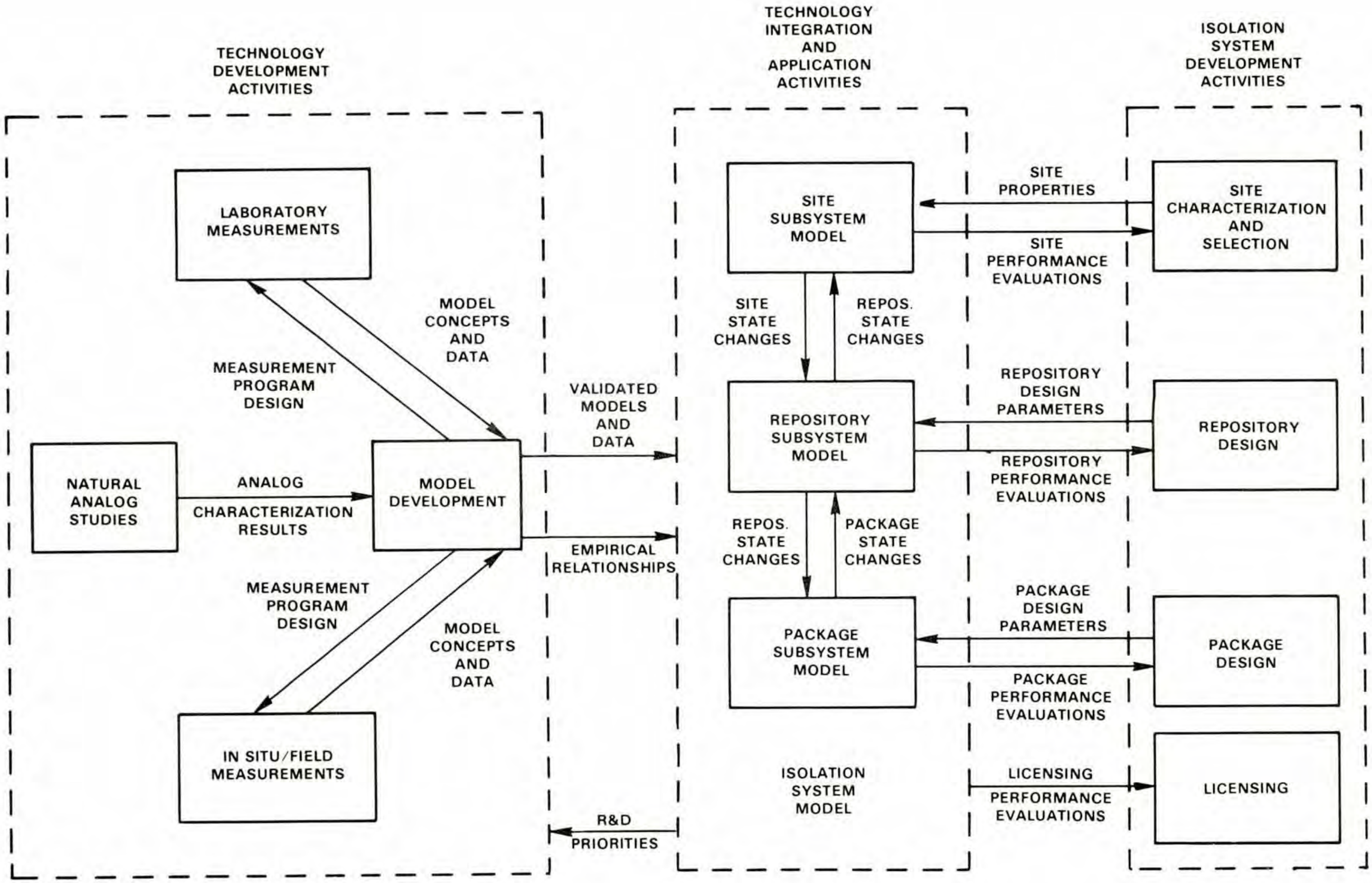

FIGURE 1. NWTS PERFORMANCE ASSESSMENT RELATIONSHIP TO OTHER ACTIVITIES

UNCERTAINTIES IN PREDICTING LONG-TERM ENVIRONMENTAL IMPACTS FROM GEOLOGIC REPOSITORIES

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ABSTRACT

The assessment of environmental impacts of construction and operation of a deep geologic repository for high-level radioactive waste can be carried out using methods well-established for conventional radioactive waste management and production facilities. Uncertainties of assessments in impacts of repositories increase with time after closure and are related to predicting long-term conditions of the accessible human environment (e.g., population trends and distributions, water use, land use, diets, and habits), and long-term changes in ecosystems, and geologic and hydrologic regimes surrounding and above the repository.

Primary concerns are related to the thermal and fission product decay period which is in the order of several hundred years. Thermal changes in the immediate vicinity of the repository could have a direct effect on root systems and indirect effects on ground water regimes, possibly contributing to long-term changes in ecosystems.

Releases of radionuclides to the accessible environment are not expected to occur under predicted conditions and would not occur for thousands to tens of thousands of years under extraordinary circumstances. The characteristics of the human environment at that time cannot readily be predicted and therefore uncertainties in consequence results may be large.

BACKGROUND

As part of the National Waste Terminal Storage (NWTS) licensing program, a number of topical reports are being developed to provide an approach to the documents that will be required for a license application to the Nuclear Regulatory Commission (NRC) for a nuclear waste repository. These reports are both general in nature, such as the Preliminary Information Report (PIR), and the NWTS Licensing Plan, and specific in nature, such as Licensing Topical Reports on defined issues. The PIR consists of two parts: Part 1, which is similar in scope to a safety analysis report, and Part II, which is similar to an environmental report. The purpose of the PIR is to provide the NWTS program with information on how well the technical program is meeting developing licensing requirements, to identify important licensing issues, and to provide a working document for the U.S. Department of Energy (DOE) and the NRC to facilitate discussion of the issues and requirements.

The environmental analysis, Part II, identifies the data needs and methodologies for evaluating potential environmental effects. It also provides a preliminary analysis of potential impacts resulting from repository construction, operations, decommissioning, and long-term isolation. This report [1] provides most of the basis for this paper.

To provide an assessment of potential impacts, an environmental setting was defined. A reference site in a salt dome in the Gulf Interior Region (GIR) was chosen for this analysis. Baseline conditions were developed from information in published environmental reports on nuclear power plants and other projects in the GIR. Thus, the reference site was representative of realistic conditions, both favorable and unfavorable. This site did not comply with certain NWTS program siting criteria for a nuclear waste repository and is not among the sites under investigation by DOE. With this site as the baseline and the repository description developed for Part 1 of the PIR, potential environmental impacts were identified for the two major phases of the project: short-term (construction, operation, and decommissioning) and long-term (isolation).

SHORT-TERM ENVIRONMENTAL EFFECTS

Analyses indicate that the short-term environmental impacts of a repository are similar to those of any large industrial project located in a rural area:

Primary impacts during construction are:

- o Commitment of resources including land uses, water, ecological habitat, subsurface mineral resources, construction materials, energy, and labor supply,
- o Wind-blown particulates, air emissions, erosion, runoff from disturbed surface soils, and excavated salt, and

- o Socioeconomic effects resulting from the in-migration of a large labor force into a rural area.

Primary impacts during operation are:

- o Mined rock handling, storage, and disposal,
- o Commitment of resources including land uses, water, subsurface mineral resources, process materials, energy, labor supply, and spent fuel, and
- o Effluents consisting of air emissions from the coal-fired steam facilities, windblown salt, leachate runoff from the excavated salt stockpile, and trace radiological emissions.

Because the operations labor force is approximately the same size as the construction labor force, the transition from construction to operation should not result in increased socioeconomic impacts.

Primary impacts during decommissioning will be similar to those during construction, with two exceptions:

- o Potential occupational exposures to radiation during decontamination, and
- o Socioeconomic impacts to local communities from the loss of employment and the out-migration of repository workers.

Repository accidents would be caused by either natural phenomena or facility malfunction and could result in a failure of environmental control and mitigation systems. Accidents can be classified as repository accidents involving radioactivity, not involving radioactivity, and transportation accidents involving radioactivity. Transportation accidents not involving radioactivity are a function of increased rail or highway usage.

LONG-TERM ENVIRONMENTAL EFFECTS

Long-term environmental effects are those resulting from permanent emplacement of nuclear waste in a deep geologic repository. Environmental effects of isolation under normal conditions, as well as those caused by phenomena that result in release of radionuclides to the biosphere, must be considered. These phenomena may be induced by natural events, human interference, the waste itself, and/or the repository. The time period of primary concern is 10,000 years after repository closure because this is the period of time required to reduce the radiological risk of fission products in the spent fuel.

Evaluation of potential environmental impacts during the isolation phase present a unique challenge, primarily because of the long time frame involved. Traditional methodologies for impact analysis require definition of baseline conditions. However, environmental baseline conditions cannot be defined for extended time periods (beyond 40 to 50 years) because of the unpredictability of human sociological, political, economic, and technical development. Although theorization and conjecture may be fascinating exercises, the range of conditions is almost boundless. Therefore, performance assessment methods have been developed to analyze the disposal system after the repository has been sealed. These methods analyze the combined effects of phenomena that might affect the disposal system. Three kinds of future conditions can be identified:

- Conditions under which radionuclide releases from the package would not occur,
- Conditions under which radionuclide releases from the package occur but radiation doses are not received by people, and
- Conditions under which radionuclides are released and a resultant radiation dose is received by people.

Mathematical models have been developed and applied to gain confidence that long-term performance of the disposal system will be acceptable. These studies show that, under normal conditions, the effects for several thousand years will be limited to thermal expansion and subsidence in the "near-field" (areas near the repository). These studies have also predicted the consequences of releases of radionuclides from the repository in the far future. The vast majority of possible disposal-system conditions would not deliver any measurable doses to people. Some possible, but unlikely phenomena, such as ground-water flow directly through the repository, could deliver radiation doses that would be a fraction of the doses delivered by natural background radiation [2].

If the disposal system performs as expected, the radioactive waste emplaced in it will remain isolated from the biosphere for hundreds of thousands to millions of years. This long-term containment and isolation will be provided by the multiple barriers of the waste package, the repository structure, and the site. In the absence of human interference and other unexpected phenomena, the repository will perform its containment and isolation function while the radioactivity of the waste is diminished by decay. Potential environmental effects of this normal condition will be limited to the effects of heat emanating from the emplaced waste and subsidence resulting from compaction of the backfill material surrounding the waste canisters.

The assessment of long-term performance requires analysis of the phenomena, or potential events, that are important to the future integrity of the system. This analysis results in the identification of phenomena for assessment of potential environmental effects. These phenomena can be classified as either natural or human-induced. All of the natural phenomena are incredible (probability less than 1 x 10-7). The human-induced phenomena have greater probability of occurrence (although such probabilities have not been calculated), but would result in relatively low releases of radioactivity to biological pathways.

Impacts of the repository during isolation on cultural resources, and vice versa, are indirect, speculative, and complex. The issues involved with interactive human uses are, however, cultural in nature. Given that the repository must remain isolated and unbreached for an extremely long time period, some social/cultural measures must be taken to prevent human penetration of the area. A short-term measure involves some sort of institutional control of the site, such as placing the area under the jurisdiction of the U.S. Government or a regulatory agency.

The fact remains, however, that the waste materials in the repository must be isolated for a longer time period than human beings have yet planned for. It is apparent that 250,000 years of institutional control is far beyond the scope of our cultural capabilities and that radioactive-waste isolation in a continentally located geologic repository cannot be dependent upon institutional stability and must take into account radical cultural change.

For the most part, successful isolation will depend upon stable geologic conditions and permanent sealing of all shafts. Even under these conditions, the possibility exists for human interference with the site. Essentially, once the repository has been abandoned, it becomes a cultural resource, an archaeological site. Our responsibility to unknown future generations requires us to make some attempt at establishing a very long-term communication resource. When all other records have been lost, the archaeological record will remain. Present-day archaeological investigations document a history of habitation reaching back at least 10,000 years, which gives us some confidence in attaining our objective.

UNCERTAINTIES IN ENVIRONMENTAL PREDICTIONS

Remaining uncertainties in predicting short-term environmental impacts of repositories are primarily related to socioeconomic issues and include:

- Potential methods (tax incentives, protection of property equity, and participation in direct community services) to mitigate adverse socioeconomic impacts induced by the influx of a large construction and operating work force and by the severe reduction in labor force when the facility is decommissioned, and

- Public concern over transportation of waste.

Uncertainties in predicting long-term environmental impacts of repositories can be categorized as:

- o Ecological
  - Lack of understanding of the biologic response to small changes in soil, and ground and surface water temperatures.
- o Physical
  - Ability to predict performance of fractured and, perhaps, water-bearing rock masses above and around the repository.
- o Socioinstitutional
  - Ability to provide permanent markers and/or means of reducing the potential of inadvertently contacting the repository, and
  - Ability to predict the conditions of the accessible environment and the characteristics of the human culture inhabiting it.

These issues are currently being studied by the Office of Nuclear Waste Isolation (ONWI) as part of the ongoing NWTS program. Alternative measures to mitigate the adverse socioeconomic effects of repository development also are being studied. The Waste Isolation Performance Assessment Program (WIPAP) and other performance-assessment programs, including in situ testing, are addressing the issue of predicting long-term rock mass response. A licensing group task force is studying the issue of human intrusion. Other issues, such as research into the effects of temperature increases on soils and biota and the issue of public concern about waste transportation, require further consideration.

REGULATORY REQUIREMENTS

Preliminary assessment indicates that environmental impacts are within an acceptable range of effects provided the design of systems, such as the coal-fired steam generator, mined rock handling facilities, and waste-water treatment, incorporates mitigation of environmental effects. The reference repository design used in the evaluation would comply with existing air, water, and solid waste regulatory requirements. However, such regulations are continuously being updated, either through additional legislation or interpretation by the courts, and will require constant surveillance to assure compliance.

Compliance with National Environmental Policy Act (NEPA) and Council on Environmental Quality (CEQ) requirements will be reviewed by the NRC. Such requirements are reflected in draft environmental report (ER) guidelines for a geologic repository [3]. The PIR, Part II, was based on these guidelines as well as existing ER guidelines for other nuclear facilities licensed by the NRC. In response to new CEQ guidelines issued in November of 1978 [4], the ER guidelines for all nuclear facilities may be revised.

SUMMARY

In summary, by focusing attention on critical environmental issues, ONWI documents are providing a basis for assessment of the environmental aspects of the licensing program. Many of the environmental issues identified in this analysis are being addressed and others are being considered for inclusion in the NWTS program. Continued surveillance of environmental regulations is required to assure compliance with applicable regulatory requirements. The present policy of maintaining communication with the NRC should provide for compliance with the NRC's licensing environmental guidelines.

REFERENCES

1. Office of Nuclear Waste Isolation, Battelle Project Management Division, Preliminary Information Report for a Conceptual Reference Repository in a Deep Geologic Formation, Draft ONWI-121, February (1981).

2. U.S. Department of Energy, In the Matter of Proposed Rulemaking on the Storage and Disposal of Nuclear Waste (Waste Confidence Rulemaking), Statement of Position of the United States Department of Energy, PR-50, 51, 44 FR 61372, April 15 (1980).

3. U.S. Nuclear Regulatory Commission, Format and Content of Environmental Reports for Conventional Geologic High-Level Waste Repositories, Waste Management Staff Guide, High-Level and Transuranic Waste Branch, Office of Nuclear Materials Safety and Safeguards, Draft, November (1978).

4. Council on Environmental Quality, National Environmental Policy Act - Regulations; Implementation of Procedural Provisions; Final Regulations, November 29 (1978).

THE ROLE OF DETERMINISTIC AND PROBABILISTIC MODELING IN LONG-TERM WASTE ISOLATION SAFETY ASSESSMENTS

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ABSTRACT

The Department of Energy, as part of the National Waste Terminal Storage (NWTS) program, is developing and testing a comprehensive methodology for assessing the postclosure performance and safety of nuclear waste isolation systems. Different analysis methods will be used for components of this methodology in order to best utilize the current state of the art and to minimize the uncertainties inherent in long-term predictions. This approach will make it possible to realistically define the potential range of future system states and to quantify the margins of safety provided by individual system components and by the overall system.

INTRODUCTION

The assessment of the postclosure performance of nuclear waste isolation requires the analysis of future changes in the site, the repository, and the waste packages[1]. Consequently, it requires techniques to analyze natural phenomena (climatic, hydrologic, geologic, and geochemical changes), repository- and waste-induced phenomena (thermal, mechanical, chemical, and radiation effects), and to evaluate potentially interfering future human activities. The analysis methodology must include all important phenomena, their interactions, and their implications with respect to the health and safety of future generations. The analysis methods must recognize differences in the complexity of the phenomena, in their importance with respect to component and overall system performance, and in the source and magnitude of uncertainties associated with each phenomenon. Following is a description of the rationale for the selection of appropriate analysis techniques and of the approach planned for their application to postclosure safety assessments.

RATIONALE

Different analysis techniques have to be selected for different phenomena to account for differences in their importance to overall system safety, in their complexity (including space and time variability), and in the source and magnitude of uncertainties. Following are relevant definitions and the rationale for selecting appropriate techniques.

Categories of Phenomena

The phenomena to be analyzed have been classified into events and processes. Events occur suddenly, instantaneously, or, in relation to geologic time frames, over short periods. Processes occur gradually, over long time periods. Examples of events are earthquakes, floods, and meteorite impacts. Examples of processes are long-term climatic changes, ground-water flow, and waste package degradation.

Categories of Techniques

The mathematical techniques have been classified as deterministic, stochastic, and probabilistic. Deterministic models assume that the variables follow definite laws of certainty. They ignore the chance of occurrence of the variables and also the uncertainties in the variates. Stochastic models assume that the chance of occurrence of the variables and/or the uncertainties in the variates follow definite probability distributions. They define the sequence or the spatial distribution of the occurrence of the variates probabilistically. The models may be purely random if the variates are considered to be independent among themselves, or non-pure random if some dependency among the variates is considered. Probabilistic models consist of deterministic and stochastic components. The deterministic components assume that the variables follow definite laws of certainty; the stochastic components assume that the chance of occurrence of the variables and/or the uncertainties in the variates follow definite probability distributions. Deterministic, stochastic, and probabilistic models or model components may be time-dependent or time-independent.

Uncertainties

The choice of the appropriate method or combination of methods for modeling a particular phenomenon depends on the source and magnitude of the uncertainties inherent in simulating that phenomenon. Uncertainties in the predictions arise from three major causes: (1) limitations in our understanding of the basic phenomena, (2) limitations in the capability of current mathematical analysis techniques, and (3) limitations in our ability to characterize or adequately measure the analysis parameters (i.e., due to a limited number of measurements and due to inaccuracies in individual measurements). Some of these uncertainties are significant for modeling the current system; they become compounded by the long time frame of concern.

Uncertainties vary greatly between phenomena to be analyzed. For some engineered system components, for instance, many years of experience exist, including theoretical research and practical operating experience. Therefore, engineered system performance can be predicted with a fair amount of certainty provided the external forces acting on the engineered system can be predicted. On the other hand, quantitive understanding of several natural phenomena, including geologic and hydrologic processes, is quite limited. Different approaches are therefore necessary to analyze different phenomena.

CHOICE OF APPROPRIATE METHOD

Deterministic methods will generally be used to simulate cause-effect relationships, including time-dependencies of processes. Stochastic methods will be used to define (1) occurrence probabilities of events that cannot be modeled deterministically, (2) uncertainties in cause-effect relationships, and (3) uncertainties in the model parameters. Since there are uncertainties in most, if not all, phenomena we need to model, probabilistic methods would be the more general approach. Whether probabilistic methods will be used, however, and how strong the stochastic versus the deterministic component will be in a particular model, will depend on the relative magnitude and importance of the three sources of uncertainties mentioned earlier. It becomes obvious, therefore, that a combination of deterministic and stochastic methods is necessary to define the system uncertainties while keeping the data needs and computational efforts within realistic bounds. This essentially will be the approach for modeling events and processes to be used in the NWTS program for long-term waste isolation safety assessments.

Modeling Approach for Events

The time of occurrence and magnitude of events will generally be modeled by stochastic methods. For example, the time of occurrence of an earthquake or meteorite impact will be defined stochastically; probability distributions will define their magnitudes (e.g., the intensity of an earthquake or the size of a meteorite). The effects of an event of a given magnitude occurring at a given time, however, will be modeled deterministically (e.g., the effects of a magnitude-5 earthquake 1000 years after repository closure on rock stresses, hydraulic conductivities, etc.).

Modeling Approach for Processes

The cause-effect relationships of processes, including their changes with time, will generally be simulated by deterministic models. Most current ground-water flow and radionuclide transport models, for instance, are deterministic. Efforts are in progress, however, to introduce stochastic aspects to consider uncertainties in the input data. For example, ground-water flow models being developed will compute the mean and the variance in the flow field (velocity and direction) on the basis of the mean and the variance in the hydraulic conductivity, porosity, and gradient. The radionuclide transport models can then compute the mean and variance in the radionuclide concentrations on the basis of the mean and variance in the flow field instead of using an empirical dispersion coefficient. Until these models are fully operational, however, parametric studies and sensitivity analyses will be performed to quantify the uncertainties.

A similar, although not identical, approach is being developed to consider the uncertainties in simulating long-term climatologic, hydrologic, and geologic processes, such as climatic and tectonic changes. The input parameters can be defined by probability density functions. Repeated simulations with parameter sets selected at random from these functions will result in a series of sequences of future system states within realistic ranges and will define the associated uncertainties.

Integrated Systems Analysis Approach for Events and Processes

Probabilistic methods will then be used to analyze the synergistic effects of events and processes. These methods will involve deterministic modeling of the cause-effect relationships of long-term processes and the stochastic superposition of randomly occurring events on these processes. The effects of this superposition on the system components can be analyzed deterministically. At the same time, uncertainties in the input parameters for both events and processes can be defined by probability distributions. As a result, not only will the range of potential future system states be defined, but the associated uncertainties can also be quantified. Not all of these capabilities are fully operational at this time, but sufficient progress is being made to indicate the feasibility of this approach.

STEPWISE APPROACH TO SAFETY ASSESSMENT

Theoretically, the stochastic superposition of events on processes will result in an infinite number of combinations of events and processes. It is, therefore, impossible to predict a single time series of system states. Consequently, multiple analyses will be required to determine the range of all possible future system states. It is not practical to apply this superposition to detailed modeling of the entire waste-isolation system due to the complexity of the system and the component processes. The range of possible future system states, however, can be determined in a stepwise approach.

First, only the effects of natural hydrogeologic system changes will be evaluated by multiple analyses with simple models which will define the range of potential future hydrogeologic system states. Next, the effects of potential human interference will be evaluated by multiple analyses which will define the range of potential future states of the major system components directly affected by the interference. From these analyses, a more limited number of analyses will be selected for more detailed evaluations which can be expected to define possible future system states at specified levels of confidence or with specified margins of safety. The assessment will demonstrate that the selected analyses actually bound potential system states, and define the related uncertainties.

Care will be taken that the selected analyses remain realistic, that is, that unrealistic analyses will be avoided. Consequently, the analysis of unrealistic and so-called nonmechanistic scenarios is neither necessary nor advisable, as sometimes suggested as proofs of repository safety. Advocates of that approach believe that if it can be shown that the repository is safe for these unrealistic, more than worst-case scenarios, then it will also be safe for more realistic scenarios with considerable margins of safety. This approach is not valid for the following four reasons:

1. The unrealistic analyses are not necessary since it can be shown how safe the system will be on the basis of realistic analyses which also define the levels of confidence.

2. There is no assurance that the unrealistic scenarios actually bound potential system performance; that is, there may be realistic cases that are worse.

3. The unrealistic approach by itself (without other realistic analyses) does not provide a quantitative measure of the margin of safety.

4. The unrealistic approach may increase system costs needlessly to provide unnecessary and ill-defined margins of safety.

CONCLUSION

In summary, the long-term safety assessment will be based on numerical analyses, including mathematical modeling and more qualitative, subjective judgements. Neither a strictly stochastic nor a strictly deterministic approach can be expected to be adequate, because the former does not recognize the cause-effect relationships of the phenomena and the latter does not recognize the uncertainties in the analyses. This applies to the overall approach rather than just to modeling, since deterministic and stochastic evaluations will be performed not only by modeling, but also by other quantitative and qualitative methods. Consequently, a judicious combination of these approaches is needed to account for the cause-effect relationships, the time-dependence, the spatial variability, and the uncertainties in the system performance. Important information about the system safety will not be apparent without the application of all of these techniques.

As explained before, the selection of the appropriate approach for analyzing a particular phenomenon depends on the type of phenomenon, its importance to overall system safety, and the source and magnitude of the uncertainties. These approaches will change with time because improvements can be expected in our understanding of the basic phenomena, in mathematical analysis techniques, and in our ability to measure the analysis parameters.

The uncertainties in predicting waste isolation system performance over thousands of years are significant. However, our current knowledge of the important phenomena, the state of the art of mathematical modeling in support of qualitative and simpler quantitative evaluations, and our ability to measure the system parameters are adequate to prove the long-term safety of repositories. Our analysis capabilities will be sufficient to analyze the important events and processes realistically and to define the related uncertainties. In addition, we will be able to show that our analyses are founded on a sound scientific basis.

Other papers in this symposium provide additional justification for using a combination of analysis methods. They also give details on the current NWTS approaches for modeling particular phenomena, for quantifying uncertainties, and for assembling the individual techniques into a comprehensive analysis system.

REFERENCE

1. H. C. Burkholder, The Technical Approach to Uncertainty Analysis in the National Waste Terminal Storage Program, Proceedings, Symposium on Uncertainties Associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste, Gatlinburg, Tennessee (March 9-13, 1981).

AEGIS METHODOLOGY AND A PERSPECTIVE FROM AEGIS METHODOLOGY DEMONSTRATIONS

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ABSTRACT

Performance assessment of long-term waste isolation involves a number of distinct analytical steps. The Assessment of Effectiveness of Geologic Isolation Systems (AEGIS) Program currently has the methodology for performing these analytical steps. The methodology is applicable at various levels of sophistication, depending on the analytical need and the amount of information available.

Demonstrations of AEGIS methodology began in 1978. Experience gained from these demonstrations has been useful in directing further methodology development and has illuminated the need for stochastic approaches to the interpretation of data. Applications of AEGIS methodology has shown that the geosphere can provide an effective barrier to the migration of most radionuclides even when released by hypothetical, catastrophic processes.

INTRODUCTION

The Assessment of Effectiveness of Geologic Isolation Systems (AEGIS) Program has been developing and applying methodology to evaluate the post-closure performance of a geologic repository for the deep disposal of nuclear waste. Methodology development began at the Pacific Northwest Laboratory (PNL) in 1972 with studies of geologic isolation systems performed under the Advanced Waste Management and Waste Management Systems Studies Programs sponsored by the Atomic Energy Commission (AEC) and the Energy Research and Development Administration (ERDA). On 1 October 1976, the Waste Isolation Safety Assessment Program (WISAP) was established. WISAP was supported by the Office of Waste Isolation (OWI), managed for the Department of Energy (DOE) by the Union Carbide Nuclear Corporation. Since 1 July 1978, OWI was replaced by the Office of Nuclear Waste Isolation (ONWI), managed for DOE by the Project Management Division of the Battelle Memorial Institute. On 1 October 1979, WISAP was replaced by two separate programs--the Waste/Rock Interaction Technology (WRIT) Program and the AEGIS Program.

The objectives of the AEGIS Program are to: 1) develop the capabilities needed to assess the post-closure safety of waste isolation in geologic formations, 2) demonstrate the assessment capabilities by performing analyses of reference sites, 3) apply the assessment methodology to assist the National Waste Terminal Storage (NWTS) Program in site selection, waste package and repository design, and 4) perform repository site analyses responsive to the time schedule and to the level of sophistication required to meet the licensing needs of the NWTS Program.

The first methodology demonstration occurred in 1978 as an application of consequence analysis to a hypothetical repository release in bedded salt (Paradox Basin). Since that time, AEGIS methodology has been used in several, site-specific applications of increasing complexity. AEGIS hydrologic, contaminant transport, and dose codes will be used to support in situ experiments at the Nevada Test Site and to evaluate the basis for EPA draft standards. These applications represent an awareness that AEGIS methodology, which has been primarily developed for repository licensing activities, has immediate usefulness in other technical areas of the NWTS Program. The usefulness of the methodology as a whole or in part is expected to increase in future years.

This paper summarizes the AEGIS methodology, the experience gained from methodology demonstrations, and in particular, provides an overview for the following subject areas:

- estimating the response of a repository to perturbing geologic and hydrologic processes that could compromise the integrity of the repository and result in releases of radioactivity to the surrounding environment
- estimating the transport of radionuclides from a repository to man
- assessing the sources and magnitudes of uncertainties associated with the models and methods used for future projections.

AEGIS METHODOLOGY

Federal legislation requires that geologic repositories be licensed by the Nuclear Regulatory Commission (NRC) and meet radiation protection standards being established by the Environmental Protection Agency (EPA). Research programs are in progress to determine acceptable waste forms, canister designs to contain the waste, backfill materials to fill the void spaces in the mined repository, and geological formations that isolate the repository from the accessible environment. These components compose a system for disposal of high-level waste. While specific performance critera may be imposed on individual components, the ultimate test of the repository will involve an evaluation of the total geologic isolation system as a unit.

The performance assessment of long-term waste isolation involves a number of distinct analytical steps. Initially, the specific nature of the engineered components in the repository and of the surrounding geologic and hydrologic systems must be adequately understood. Because natural geologic processes and future human activities may alter these systems over time, an evaluation must also be made to determine if there are any quantifiable processes or a series of events in geologic time, for affecting the integrity of the repository. If such a process is identified, the transport of radionuclides from the repository to the environment and the environmental impacts can be estimated. AEGIS currently has the methods for performing these analytical steps.

The AEGIS approach is applicable at various levels of sophistication, depending on the analytical need and the amount of information available. For example, in the site selection and evaluation phase, preliminary assessments based on minimal data will probably be adequate. However, in later phases of site qualification and licensing, more detailed information will be provided, and AEGIS personnel will be required to perform a thorough performance assessment with increased accuracy and reduced uncertainty. The analytical methods are being continuously improved to eliminate analytical deficiencies identified during methodology demonstrations as the site selection and licensing process develops. The AEGIS approach for evaluating the effectiveness of a geologic repository has been discussed by Silviera et al. [1]. An appreciation for the AEGIS methodology can be obtained from the schematic diagram of the AEGIS analysis followed in the basalt reference site initial analysis (RSIA) as shown in Figure 1. The dotted lines indicate existing capability not used in the basalt analysis or new capability to be added in the near future.

Conceptual Model Development

The analysis begins with a search and compilation of all the available geological, hydrological, and geochemical information for the region under study. Much of this information is obtained from sources such as federal and state publications, DOE-sponsored programs, Landsat imagery [2], DOE Geologic Project Managers, other government agencies (e.g., USGS), and general scientific literature. Because ground water is the most probable transport mechanism for radionuclides released from deep underground repositories, the local and regional geology and hydrology must be evaluated. This involve identifying the various aquifer systems, their hydrologic characteristics, and their recharge and discharge locations. Extensive data tabulations will require the use of a computer storage and retrieval system such as the Comprehensive Information Retrieval and Model Input Sequence [3].

The geologic history of the region is determined through studies of the various rock types and their distribution in the area, and of the regional and local structural features. This geologic understanding and consideration of natural geologic processes of the region contributes to the preparation of a conceptual model. Additional information involving the influence of geochemical effects on the integrity of the repository includes a chemical analysis of the ground water and the mineral types that may contact the ground water, as described by Deutsch [4]. This additional information can provide insight into the origin and source of ground water, residence time, and chemical evolution of the ground-water system.

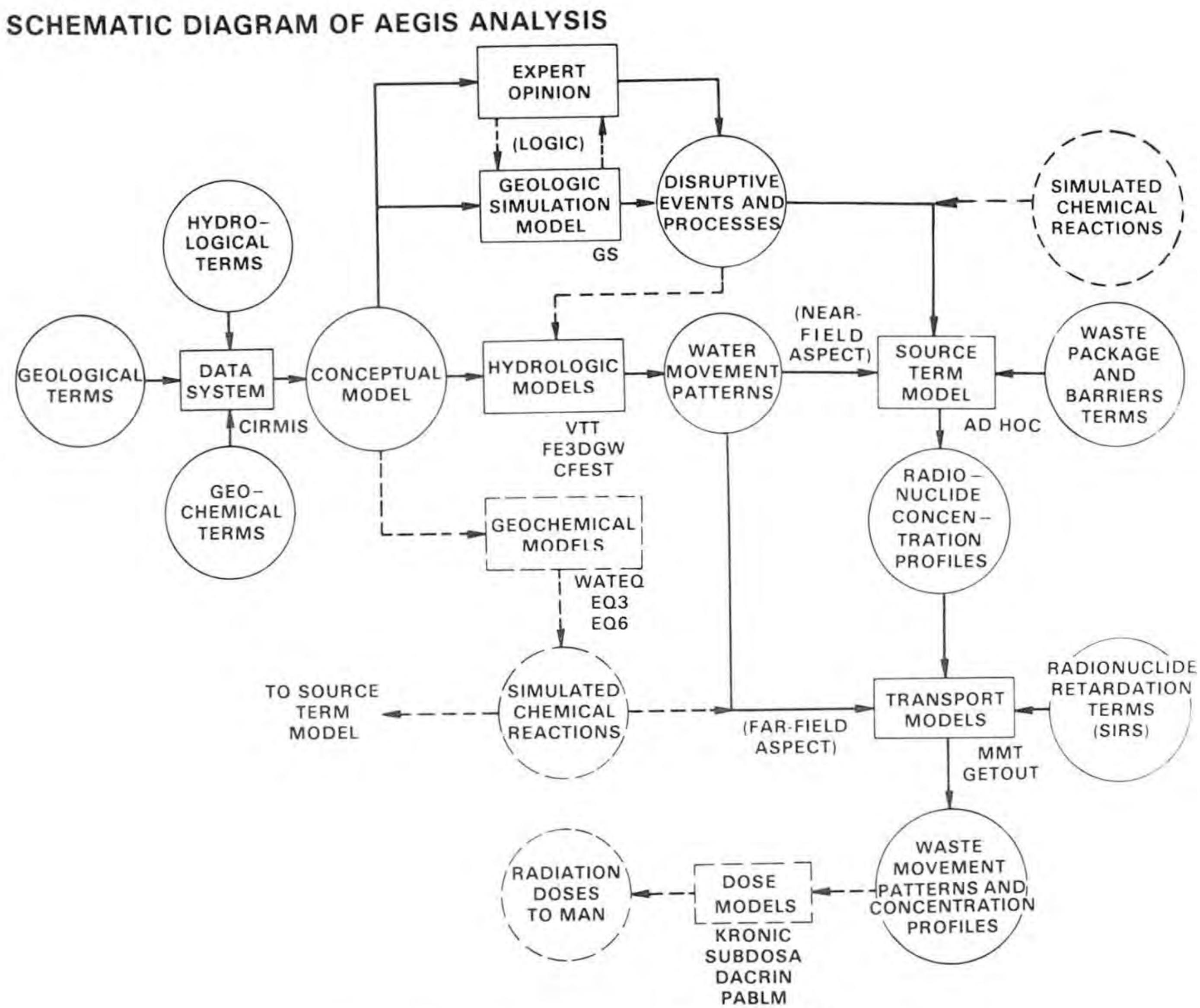

Fig. 1. Schematic diagram of AEGIS analysis

The preparation of a conceptual model of the hydrologic system is performed with the cooperation of Geologic Project Managers who are actively involved with identifying and characterizing the geological and hydrological properties of a particular site. The result is a composite description of the region, which should include potentiometric maps, structure maps for the various geologic and hydrologic units, pumping and recharge distributions, and boundary conditions. Often this information must be interpolated from drilling logs, static well measurements from the same aquifer, topographic maps, meteorological measurements, and stream flow records. Specific information that occasionally must be interpreted from incomplete field data includes: 1) the locations and amounts of ground-water recharge and discharge at formation outcrops and perimeters, and 2) the hydraulic connections between deep confined aquifers, shallow unconfined aquifers, and surface-water systems. The conceptual model forms the common basis necessary to obtain compatibility between the hydrologic systems simulation and the geologic process simulation.

Hydrologic Systems Simulation

After the conceptual model of the hypothetical repository site and region has been developed, computer codes are used to simulate flow patterns within the hydrologic system. The hydrologic codes used as part of the AEGIS methodology include the Variable Thickness Transient [5], and the Finite-Element, Three-Dimensional Ground-Water code [6]. Information obtained from the hydrologic models is then used for input into the source term model and the radionuclide transport model.

An important characteristic of the AEGIS computer code usage is the interactive computer system at PNL. Interactive computer systems allow the user to receive immediate response from the computer, greatly enhancing code and model development. Because the AEGIS computer codes must be adapted to model the conceptual description of the particular repository region, the interactive computer system makes that process much more efficient and, more importantly, substantially increases the involvement of hydrologists and geoscientists in the AEGIS computer operations. Thus, the output from these codes can be readily used to evaluate and refine the conceptual model of the regional and local hydrogeologic systems.

Geologic Process Simulation

The geology surrounding a repository will continue to change over geologic time. Projections of the possible evolutionary paths can be made from knowledge of the geologic history of the region, the existing states of the geologic and hydrologic systems, and the scientific understanding of the geologic processes in effect. AEGIS staff use the conceptual description of the existing hydrogeology as a focus for evaluating future states of the system. A team of consultants is employed to help characterize the natural processes that could affect the repository. A computer model is used to assist the AEGIS staff and consultants in the analysis and in keeping track of the potentially large numbers of natural processes and interactions.

Based on these considerations, AEGIS methodology identifies and quantifies a set of processes that could lead to a loss of repository integrity and that need to be further analyzed. AEGIS scientists seek to provide an auditable method for the derivation and quantification of these objectionable states, with estimates of the plausibility and time of occurrence. In this process, AEGIS scientists can also provide the justification for discarding other objectionable states as being implausible.

If the Geologic Simulation Model [7] identifies a process or series of events that would cause a loss of repository integrity, the input data for the hydrologic models are modified to reflect the boundary conditions surrounding that objectionable state. The information obtained from the Geologic Simulation Model is further used as input into the development of a source term model.

Source Term Modeling

Analysis of ground-water transport of radionuclides from a nuclear waste repository is dependent upon an appropriate definition of a "source term"--a projection of the rate of release of radionuclides from the repository [8]. Development of the source term is directly influenced by the loss of repository integrity, aquifer interconnections, and the near-field flow regime. In addition, local ground-water chemistry, canister and backfill materials, and the temperature profile from the heat generated by the nuclear waste must be considered. Ground water that hypothetically contacts the waste may be an oxidizing or a reducing agent and affects the waste leach rate and solubility limit. The WRIT Program at PNL supports AEGIS applications by defining a source term; however, the development of a useful model requires a multidisciplinary approach incorporated into AEGIS methodology demonstrations on an ad hoc basis. The source term model provides radionucide concentration profiles to the transport model.

Transport Modeling

The codes used for radionuclide transport, like the ground-water codes, are multidimensional and interactive. These codes include the Multicomponent Mass Transport code [9] and the GETOUT code [10]. Radionuclide migration as affected by the geologic medium is simulated by a distribution coefficient (Kd). The Kd is a measure of the ability of a geologic medium to retard a given radionuclide. A storage and retrieval system for experimental data on sorption/desorption analyses for a wide variety of radionuclides, ground-water compositions, and rocks and minerals is the Sorption Information Retrieval System [11,12]. In addition to distribution coefficients, other code input includes:

- ground-water flow velocity
- longitudinal dispersivity
- of ground-water flow path length
- dimensions of the stream tube
- nuclide half-lives
- a matrix describing radionuclide chain decay
- porosity
- leach rate information.

After the radionuclide concentrations are mathematically introduced into the ground-water movement patterns, waste movement patterns are outlined over elapsed time. Contaminant transport results are summarized by both the point-source concentration versus time profile and by concentration versus arrival distribution curves for significant radionuclides that are hypothetically released to the biosphere. When appropriate, dose models are used to calculate radiation doses to man directly or to man through his food chain.

Dose Modeling

When radionuclides are released to the atmosphere or to surface waters, they may disperse or they may accumulate in the environment. Even short-term (acute) releases can lead to long-term environmental contamination, which in turn leads to long-term irradiation of individuals and populations. Pathways of human exposure to these radionuclides include direct radiation from contaminated air, water, sediment, and soil; ingestion of contaminated drinking water, aquatic food products, terrestrial farm crops, and farm animal products; and inhalation of airborne materials. These exposure pathways are evaluated by AEGIS staff using computer codes such as KRONIC, SUBDOSE, DACRIN, and PABLM [13,14,15,16]. Site-specific information about demography, local crop production practices, eating habits, and recreational activities are required as input for these codes.

A PERSPECTIVE FROM AEGIS METHODOLOGY DEMONSTRATIONS

Demonstrations of AEGIS methodology began in the summer of 1978 with a release consequence analysis of a hypothetical repository in bedded salt [17]. Professionals from Bechtel National, Inc. provided technical support, and the analysis was promptly completed using VTT and MMT codes in a test of WISAP consequence methodology. A chronological listing of methodology demonstrations, beginning with this bedded salt test case, is shown in Figure 2. A comparison of numerical results from hydrologic models developed by INTERA and PNL for the Waste Isolation Pilot Plant (WIPP) site in the Delaware Basin of New Mexico followed in the winter of 1978. With equivalent input data and equivalent data interpretations, the hydrologic model developed from the SWIFT code and the hydrologic model developed from the VTT code produced the same results [18].

In 1979, release consequence portions of the WISAP methodology were applied to analyze hypothetical sites in Swedish bedded salt and in Swedish granite for the International Nuclear Fuel Cycle Evaluation (INFCE) Program. The FE3DGW code was used to characterize the multilayered hydrogeologic system of bedded salt and the multilayered hydrogeologic system of fractured granite [19]. These applications represented a definite increase in modeling complexity and an opportunity to apply a porous-flow model (FE3DGW) to a regional, three-dimensional, fracture-flow simulation. A more complete demonstration of WISAP methodology was included in the release scenario and consequence analysis applied to the Hainesville Salt Dome in the first reference site initial assessment [20]. The release

FY-78/79	PARADOX BEDDED SALT	VTT AND MMT TEST CASE CONSEQUENCE AND SENSITIVITY ANALYSIS
FY-79	PERMIAN BEDDED SALT	SWIFT/VTT COMPARISON AND MMT APPLICATION FOR WIPP SITE
FY-79	SWEDISH BEDDED SALT	FE3DGW AND MMT APPLICATION FOR INFCE
FY-79/80	SWEDISH GRANITE	FE3DGW AND GETOUT APPLICATION FOR INFCE
FY-79/80	GULF COAST INTERIOR SALT DOME	WISAP METHODOLOGY DEMONSTRATION (WITHOUT GEOLOGIC SIMULATION MODEL)
FY-80/81	COLUMBIA PLATEAU BASALT	AEGIS METHODOLOGY DEMONSTRATION
FY-81	PARADOX BEDDED SALT	GEOLOGIC SIMULATION MODEL DEMONSTRATION
FY-82	TERTIARY TUFF	AEGIS INTEGRATED METHODOLOGY DEMONSTRATION NEVADA TEST SITE

Fig. 2. WISAP/AEGIS Methodology Demonstrations

scenario was developed using a team of consultants and PNL staff, and the concensus of professional opinion endorsed the stability of the salt dome with respect to natural disruptive mechanisms. However, the technical group considered a human-intrusion scenario involving solution mining to be of sufficient significance to justify a release consequence analysis. The FE3DGW, VTT, MMT, and PABLM codes were used in the consequence analysis, and a near-field assessment of the solution cavity and surrounding hydrogeologic media was required to obtain a radionuclide source term. The FE3DGW code was used for these near-dome simulations, and the point source of contamination was introduced into the regional hydrologic and transport analysis [19]. Because of the difficulty in projecting demographic patterns, food-consumption habits, and exposure pathways, dose calculations using the PABLM code were limited to the operational aspects of the solution mining scenario at 100 yr and 1,000 yr after repository closure.

In the fall of 1979, the second reference site initial assessment for a hypothetical repository in the Columbia Plateau Basalt was initiated. This analysis benefited from knowledge obtained through earlier developments of the Geologic Simulation Model for basalt [21]. The Basalt Waste Isolation Program (BWIP) of Rockwell Hanford Operations has supplied extensive data and useful criticisms of technical phases in the AEGIS analysis. The basalt assessment is the most comprehensive demonstration of the AEGIS methodology attempted to date, with completion scheduled for April 1981. The third reference site initial assessment is scheduled to begin in the fall of 1981 with application of the AEGIS integrated methodology to a hypothetical repository at the Nevada Test Site.

General Uncertainty Categories

Sources of uncertainties associated with the models and methods used in AEGIS methodology demonstrations can be organized into four general categories:

1. time considerations
   - short sample time
   - long projection time

2. accuracy considerations
   - code selection
   - data interpretation

3. precision considerations
   - field measurement tolerances
   - model calculational tolerances

4. human considerations
   - political stability
   - public perception.

The inability to quantify many of these sources of uncertainty leads to the use of bounding calculations, conservative estimates, and professional concensus.

Estimating Repository Response

Experience gained from AEGIS methodology demonstrations has been helpful in directing AEGIS methodology development. The discussion of the Geologic Simulation Model by Foley and Petrie [22] will address some of the above uncertainties and their incorporation into the AEGIS methodology development. A useful perception of the uncertainties involved in regional data base limitations and how they can be accommodated in a practical sense is discussed by Schalla and Leonhart [23]. Variations in data for the Columbia Plateau Basalt establish the background for a case example in the determination of a significance criterion defining the loss of repository integrity, as discussed by Zellmer and Lindberg [24]. Preliminary modeling results using the Geologic Simulation Model in the Columbia Plateau Basalts are summarized by Petrie and Foley [25].

Estimating Radionuclide Transport

Experience gained from AEGIS methodology demonstrations estimating the transport of radionuclides from a repository to man has illuminated the need for stochastic approaches to the interpretation of data variations [26,27]. Investigations toward modification of existing hydrologic and transport codes to incorporate time, accuracy, and precision considerations of uncertainty are in progress. To date, uncertainty associated with modeling of contaminant transport has received more attention by AEGIS staff than that associated with hydrologic flow modeling. Geostatistical modeling of pore velocity is discussed by Devary and Doctor [28] as an example. Experience and insight is combined in the basic development of contaminant arrival distribution concepts known as "Geohydrologic Response Functions" [29]. These response functions are proposed as a means of providing simple summary results to those in or contributing to the decision-making process in nuclear waste management.

AEGIS Perception

In summary, three conclusions are highlighted by the AEGIS methodology demonstrations:

- Performance assessment of the repository site will always be data limited.
- Useful modeling results for performance assessment can be obtained from the best technology and data available at the time of analysis.
- The geosphere can provide an effective barrier to migration of most radionuclides even when released by hypothetical, catastrophic processes and events.

REFERENCES

1. D. J. Silviera, et al., A Short Description of the AEGIS Approach, Pacific Northwest Laboratory, PNL-3398 (September, 1980).
2. J. G. Stephan, Well Location and Land-Use Mapping in the Columbia Plateau Area, Rockwell Hanford Operations, RHO-BWI-C-61 (October, 1979).
3. D. R. Friedrichs, CIRMIS Data System (4 volumes), Pacific Northwest Laboratory, PNL-3161 (March, 1980.
4. W. J. Deutsch, Geochemical Modeling of the Nuclear-Waste Repository System--A Status Report, Pacific Northwest Laboratory, PNL-3518 (December, 1980).
5. A. E. Reisenauer, Variable Thickness Transient Groundwater Hydrologic Model (3 volumes), Pacific Northwest Laboratory, PNL-3160 (April, 1980).
6. S. K. Gupta, et al., Finite-Element Three-Dimensional Groundwater (FE3DGW) Flow Model--Formulation, Program Listings and Users' Manual, Pacific Northwest Laboratory, PNL-2939 (January, 1980).
7. G. M. Petrie, et al., Geologic Simulation Model for a Hypothetical Site in Basalts of the Columbia Plateau, Pacific Northwest Laboratory, PNL-3542 (April, 1981).

8. W. L. Kuhn, Development of a Source Term for Radionuclide Release from Spent Fuel in a Basalt Repository, Pacific Northwest Laboratory, PNL-SA-8935 (December, 1980).

9. J. F. Washburn, et al., Multicomponent Mass Transport Model: A Model for Simulating Migration of Radionuclides in Ground Water, Pacific Northwest Laboratory, PNL-3179 (July, 1980).

10. W. V. DeMier, et al., GETOUT--A Computer Program for Predicting Radionuclide Decay Chain Transport Through Geologic Media, Pacific Northwest Laboratory, PNL-2970 (August, 1979).

11. D. D. Hostetler, et al., Status of Sorption Information Retrieval System, Pacific Northwest Laboratory, PNL-3139 (September, 1979).

12. D. D. Hostetler, et al., Status Report on SIRS: Sorption Information Retrieval System, Pacific Northwest Laboratory, PNL-3528 (November, 1980).

13. D. L. Strenge and E. C. Watson, KRONIC--A Computer Program for Calculating Annual Average External Doses from Chronic Atmospheric Releases of Radionuclides, Pacific Northwest Laboratory, BNWL-B-264 (June, 1973).

14. D. L. Strenge, et al., DACRIN--A Computer Program for Calculating Organ Dose from Acute and Chronic Radionuclide Inhalation, Pacific Northwest Laboratory, BNWL-B-389 (December, 1974).

15. D. L. Strenge, et al., SUBDOSA--A Computer Program for Calculating External Doses from Accidental Atmospheric Releases of Radionuclides. Pacific Northwest Laboratory, BNWL-B-351 (June, 1975).

16. B. A. Napier, et al., PABLM--A Computer Program for Calculating Accumulated Radiation Dose from Radionuclides in the Environment, Pacific Northwest Laboratory, PNL-3209 (March, 1980).

17. J. R. Raymond, Test Case Release Consequence Analysis for a Spent Fuel Repository in Bedded Salt, Pacific Northwest Laboratory, PNL-2782 (January, 1980).

18. C. R. Cole and F. W. Bond, Comparison of INTERA and WISAP Consequence Model Application, Pacific Northwest Laboratory, PNL-3070 (January 1980).

19. C. R. Cole and F. W. Bond, Summary of Four Release Consequence Analyses for Hypothetical Nuclear Waste Repositories in Salt and Granite, Pacific Northwest Laboratory, PNL-3548 (December, 1980).

20. M. A. Harwell, Reference Site Initial Assessment for a Salt Dome Repository, Pacific Northwest Laboratory, PNL-2955 (July, 1981).

21. J. A. Stottlemyre, et al., A Conceptual Simulation Model for Release Scenario Analysis of a Hypothetical Site in Columbia Plateau Basalts, Pacific Northwest Laboratory, PNL-2892 (January, 1981).

22. M. G. Foley and G. M. Petrie, The AEGIS Geologic Simulation Model, Pacific Northwest Laboratory, PNL-SA-9111 (March, 1981).

23. R. Schalla and L. S. Leonhart, Dealing with Regional Hydrologic Data Base Limitations, Case Example: The Columbia River Basalt, Pacific Northwest Laboratory, PNL-SA-9094 (March, 1981).

24. J. T. Zellmer and J. W. Lindberg, A Study of Present and Perturbed Ground-Water Flow Rates Through a Hypothetical Basalt Nuclear Waste Repository to Establish a Significance Criterion, Pacific Northwest Laboratory, PNL-SA-9112 (March, 1981).

25. G. M. Petrie and M. G. Foley, Preliminary Results of Monte Carlo Simulations for the Pasco Basin, Washington, Using the AEGIS Geologic Simulation Model, Pacific Northwest Laboratory, PNL-SA-9114 (March, 1981).

26. P. G. Doctor and R. W. Nelson, Geostatistical Estimation of Parameters for Transport Modeling, Pacific Northwest Laboratory, PNL-SA-8482 (August, 1980).

27. C. S. Simmons, Relationships of Dispersive Mass Transport and Stochastic Convective Flow Through Hydrologic Systems, Pacific Northwest Laboratory, PNL-3302 (January, 1981).

28. J. L. Devary and P. G. Doctor, Geostatistical Modeling of Pore Velocity, Pacific Northwest Laboratory, PNL-SA-9096 (March, 1981).

29. R. W. Nelson and F. H. Dove, Evaluators for Deep Nuclear Waste Repositories--Geohydrologic Response Functions, Pacific Northwest Laboratory, PNL-SA-9092 (March, 1981).

THE AEGIS GEOLOGIC SIMULATION MODEL

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ABSTRACT

Assessment of the post-closure performance of a nuclear waste repository has two basic components: the identification and analysis of potentially disruptive sequences and the pattern of geologic events and processes causing each sequence, and the identification and analysis of the environmental consequences of radionuclide transport and interaction subsequent to disruption of a reposiitory. The AEGIS Scenario Analysis Task is charged with identifying and analyzing potentially disruptive sequences of geologic events and processes. The Geologic Simulation Model (GSM) was developed to evaluate the geologic/hydrologic system surrounding an underground repository, and to describe the phenomena that alone, or in concert, could perturb the system and possibly cause a loss of repository integrity.

The AEGIS approach has been to use an unintegrated series of models for repository performance analysis; the GSM for a low-resolution, long-term, comprehensive evaluation of the geologic/hydrologic system; followed by more detailed hydrogeologic, radionuclide-transport, and dose models to more accurately assess the consequences of disruptive sequences selected from the GSM analyses. This approach can be used to estimate the liklihoods of potentially disruptive evolutionary developments within the geologic/hydrologic system. The more costly consequence models can then be focused on a few disruptive sequences chosen for their representativeness and effective probabilities.

GEOLOGIC SIMULATION MODEL

The AEGIS Geologic Simulation Model is at present specifically designed for analysis of a hypothetical repository in the Columbia River Basalts of the Pasco Basin, Washington, although adaptation to other geologic terranes is in progress. It is designed to meet a wide range of performance criteria. Some of these criteria are: 1) auditability, 2) ability to accommodate objective and subjective input, 3) facilitation of parametric and sensitivity studies, 4) facilitation or assistance in describing disruptive sequences and their probabilities, 5) establishment of limits or initial conditions for input into the consequence analysis models, and 6) flexibility to accommodate an increasing data base.

An interactive, computerized approach with a quasi-deterministic, process-response methodology was adopted for the following reasons:

- Significant interactions exist among disruptive phenomena. For example, climate, precipitation, glaciation, sedimentation, and ground-water recharge are all synergistic. This requires modeling of each process in as deterministic a manner as possible to identify and allow potential interactions.

- Most phenomena are time-dependent, and initiation and consequences cannot always be considered simultaneously. A time-integrated, process-response model is necessary to accommodate gradual changes and to incorporate their effects into the geologic/hydrologic system.

- Uncertainties in data, in the way some processes should be modeled, and in the understanding of some processes that appear inherently stochastic dictate reliance on probability distributions rather than on discrete numbers for the values of most input variables. Further, the sensitivity of the model output to variations in the input must be analyzed for cost-effective studies. The data-handling and bookkeeping requirements imposed by these criteria make computerized analysis a necessity.

- The methodology must explicitly reveal the conceptualization of the geologic/hydrologic system and its implementation to geoscientists to provide for peer review and auditability of the study. A user-interactive computer model is necessary to allow review by knowledgeable scientists who are not familiar with computer usage. This capability also allows specialists a comprehensive overview of the entire model and encourages interaction among specialties.

The GSM simulates geologic and hydrologic system response, as a result of ongoing processes and the passage of time, for a period of a million years. This time frame is long enough to accommodate requirements for performance analyses for thousands, tens of thousands, or even hundreds of thousands of years. It is not, however, so long that significant changes in the details of global tectonic activities and their rates must be considered. Figure 1 shows the model time frame in perspective with present experience in design, which emphasizes the need for care and innovation in developing the assessment methodology. Figure 2 shows the time intervals used in the process-response model: 100 years for the first 20,000 years, 1000 years for the next 180,000 years, and 10,000 years for the remaining 800,000 years. This has the advantage of saving computer time while concentrating analysis on the geologic near term (1,000 to ure 2 also shows that the selection of time intervals for integration, hence the resolution of the analysis, mirrors the availability and reliability of geologic data from the past 20,000, 200,000, and 1 million years.

The GSM is composed of a main program, which incorporates the system conceptual model, and peripheral packages to control data input and output and to perform statistical analyses of output data for summarization and interpretation. The main model consists of eleven submodels, each of which

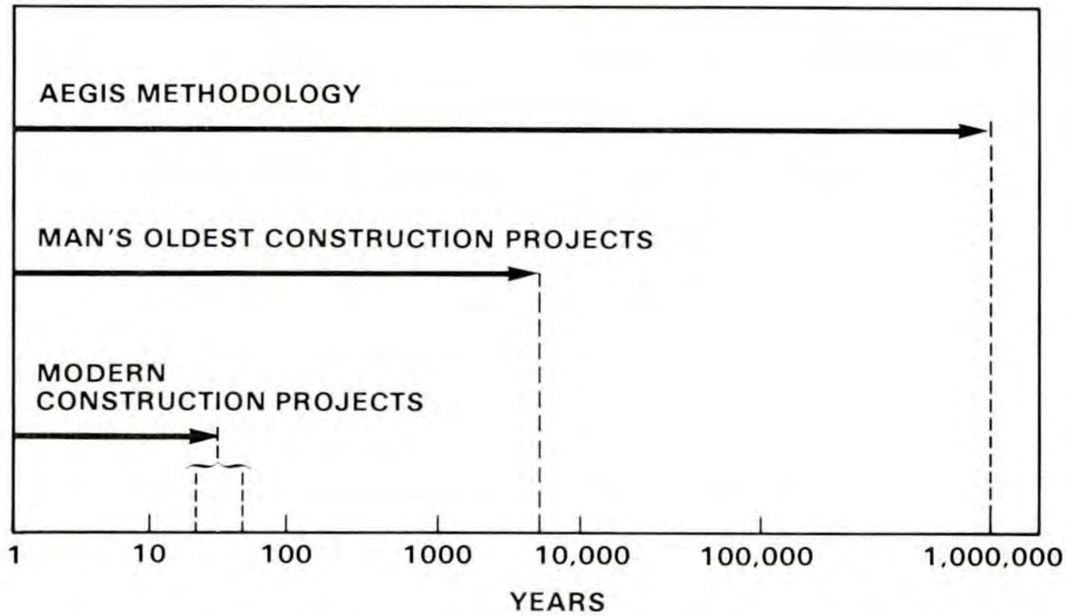

Fig. 1. A perspective on the length of time for which the GSM must be applicable, compared with historic and prehistoric constructions.

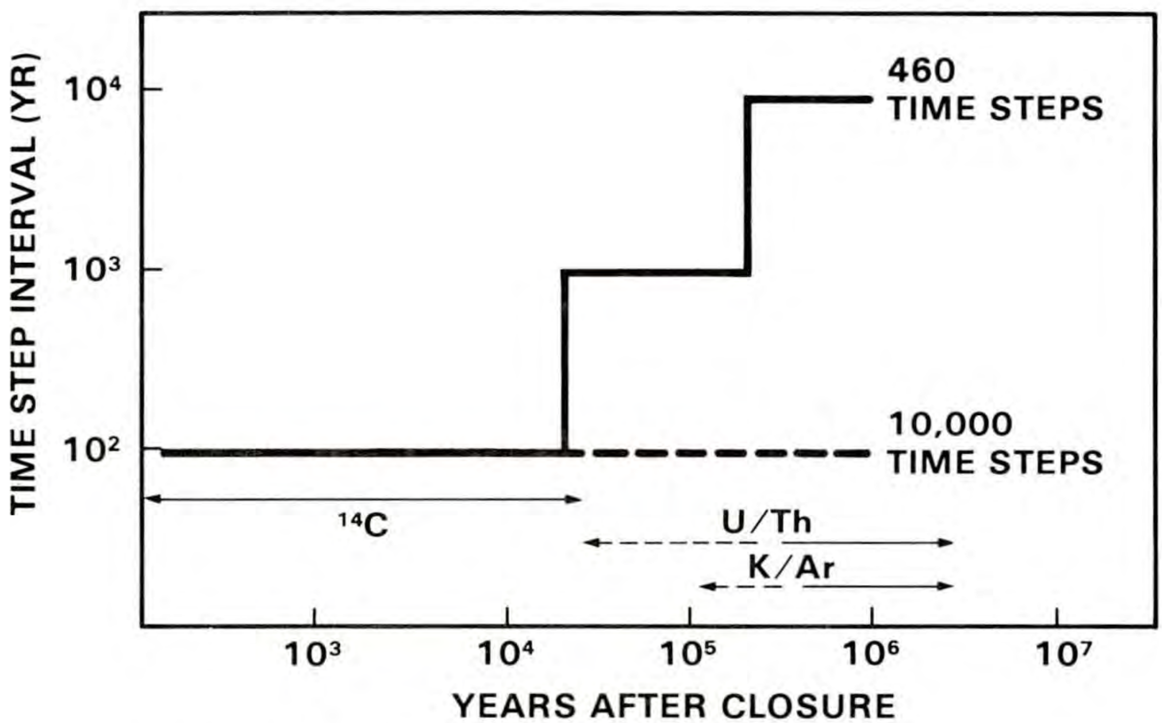

Fig. 2. Progressively increasing time steps used in the GSM. Analysis is concentrated in the first 20,000 years, based on the more abundant and accurately determined ^{14}C dated process rates of the past 40,000 years.

addresses a class of potentially disruptive phenomena: 1) climate, 2) continental glaciation, 3) deformation and faulting, 4) geomorphic events, 5) hydrology, 6) magmatic activity, 7) meteorite impact, 8) sea-level fluctuations, 9) shaft seal failure, 10) sub-basalt basement faulting, and 11) features not detected during construction of the repository. These submodels were developed in cooperation with specialists in the relevant geoscience disciplines (Figure 3). Each submodel is as physically based and sophisticated as computer-system and cost limitations allow. Each is necessarily site specific to maximize the knowledge of relevant geologic processes and rates.

Details of the GSM are described in Petrie et al. [1]. However, Figures 4 and 5 show the simplification of the three-dimensional confined ground-water system of the modeled area to a one-dimensional one, which was necessary to fit into the GSM. This abstraction is adequate for the purpose of simulation modeling as long as its behavior is compatible with that of the complex models used in later consequence analyses.

Clearly, many of the processes included in the GSM are subject to uncertainties that may result from lack of adequate field data, or lack of understanding of a phenomenon or of a process that is inherently stochastic at our present level of understanding. For this reason, input data are in the form of probability density functions (PDF), or scalars or polynomial functions with associated PDF uncertainty terms. Use of PDFs gives the GSM the flexibility to incorporate objective and subjective input data from a variety of geoscience disciplines.

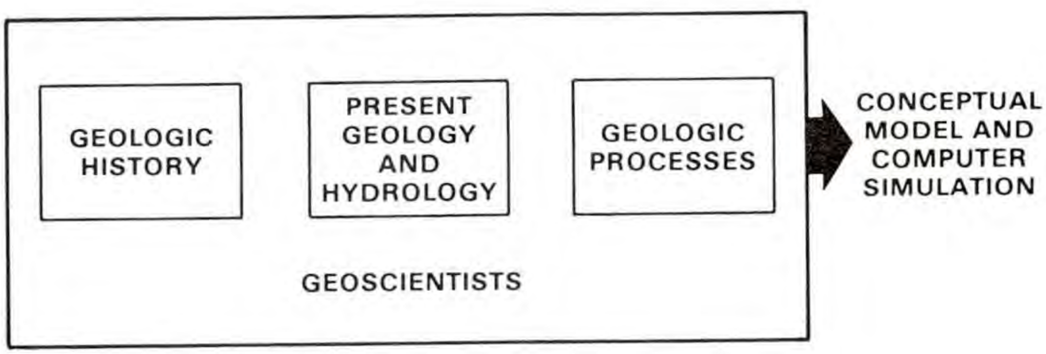

Fig. 3. The types of data used for site-specific modeling.

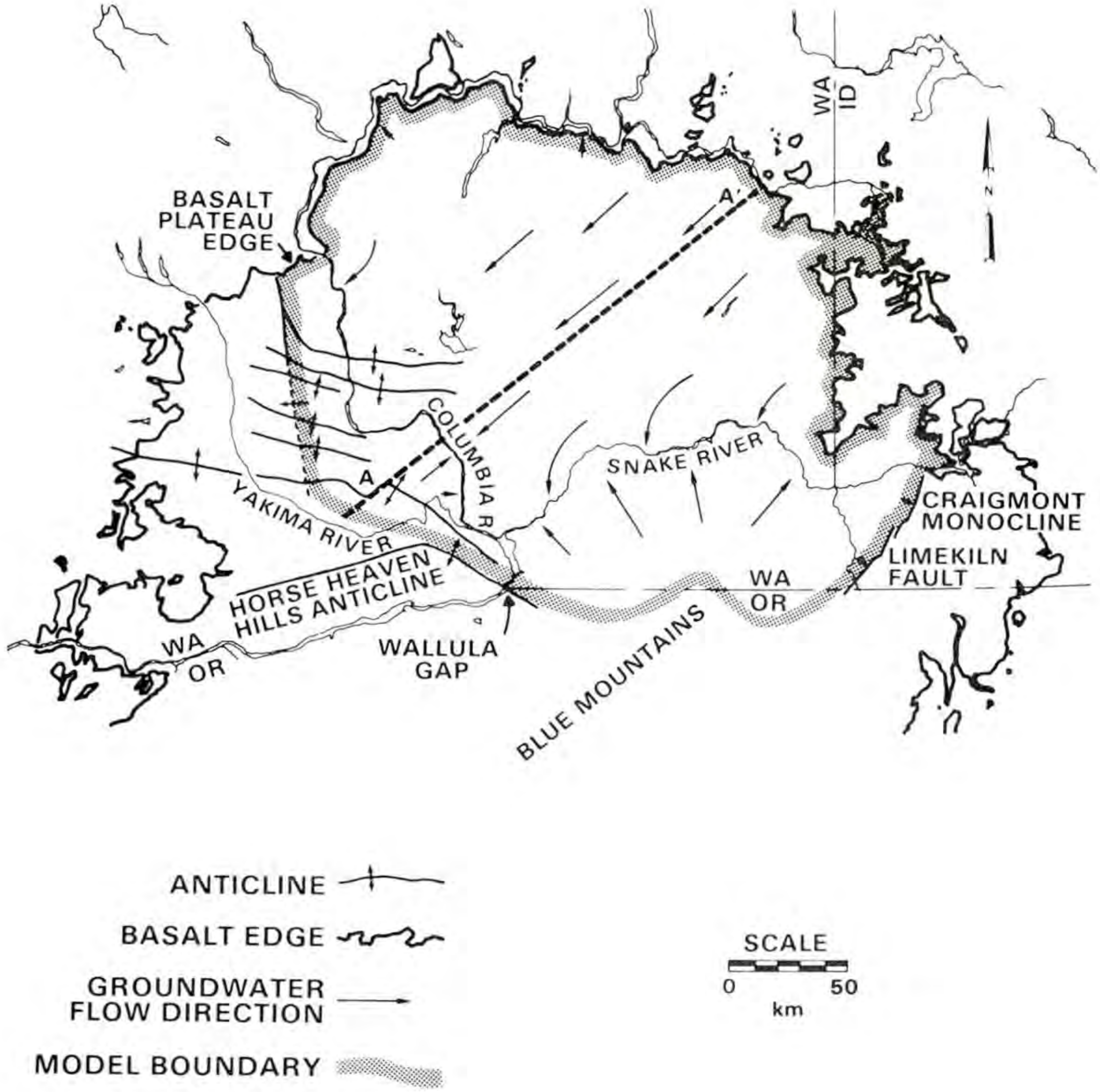

Fig. 4. Index map of the Columbia Plateau showing the regional confined-aquifer flow patterns and their relationship to the model cross-section (AA').

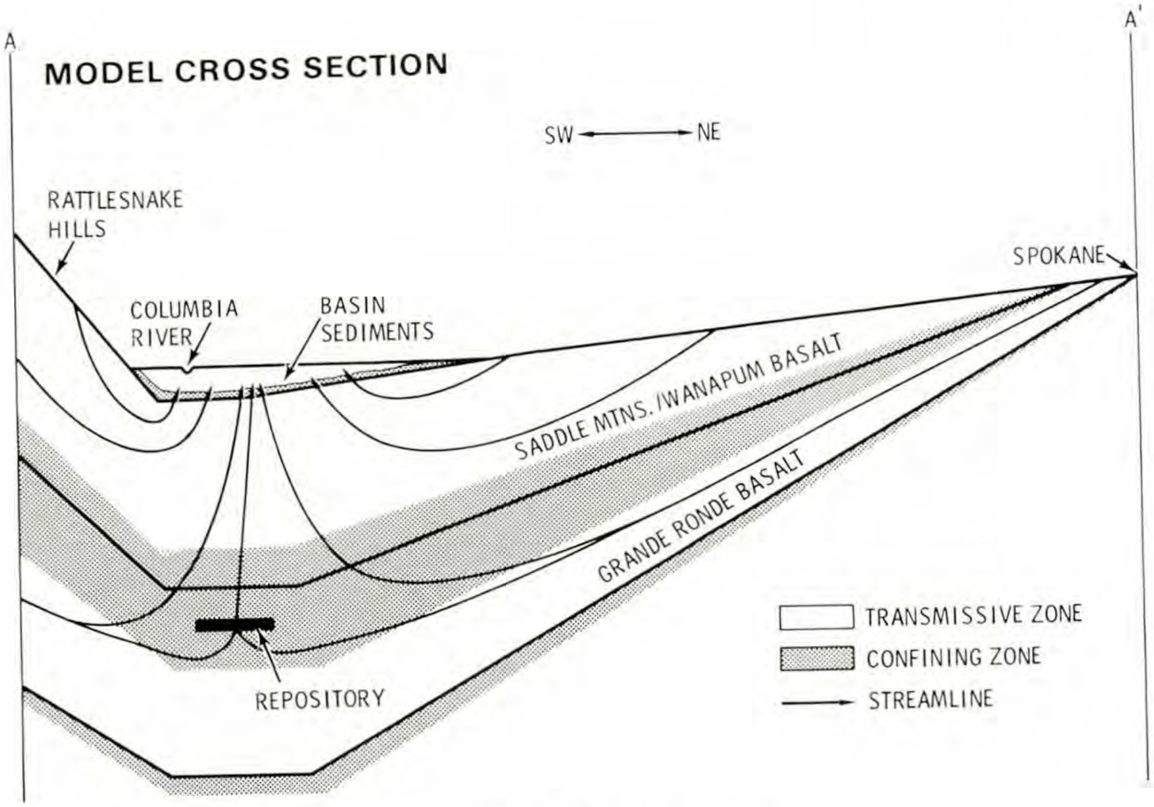

Fig. 5. The GSM model cross-section.

The GSM can be operated in single-run and Monte Carlo modes. The single-run mode is interactive and provides for user selection of desired disruptive events and associated rates or magnitudes. This allows an expert in the relevant discipline to evaluate the output of the GSM for sensitivity studies or to determine its reasonableness, and to "fine tune" any submodels that do not appear to behave realistically.

The Monte Carlo mode uses the pre-established PDFs for inputs of individual phenomenon occurrences, rates, magnitudes, and phase relations. Hundreds or thousands of individual million-year simulations in a Monte Carlo run generate a large number of disruptive event sequences that may then be analyzed by geoscientists for plausibility (Figure 6). The ones found plausible are assessed for liklihood by a statistical package, and those that exceed appropriate regulatory or other (see Zellmer and Lindberg [2]) standards are chosen for more detailed consequence analyses.

Input from geoscientists is critical at all stages of the disruptive sequence analysis, scenario selection, and conseouence analysis. The GSM is designed to complement and focus these efforts by providing broad outlines of long-term geologic and hydrologic changes that might occur in the vicinity of a repository, and by placing bounds on their magnitudes and probabilities. A later paper in this sequence [2] will describe some preliminary results from the basalt GSM to illustrate this process.

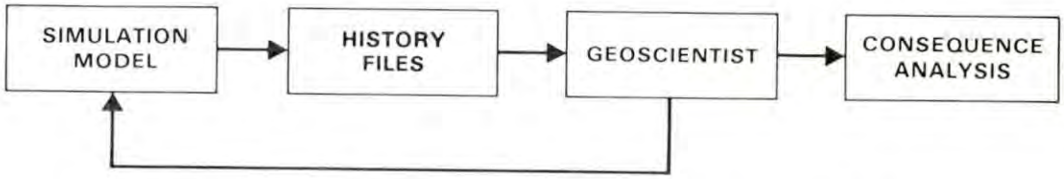

Fig. 6. The logic used in analyzing GSM data.

REFERENCES

1. G. M. Petrie, J. T. Zellmer, J. W. Lindberg and M. G. Foley, Geologic Simulation Model for a Hypothetical Site in Basalts of the Columbia Plateau, PNL-3542, Pacific Northwest Laboratory, Richland, Washington, 1981.

2. J. T. Zellmer and J. W. Lindberg, A Study of Present and Perturbed Ground-Water Flow Rates through a Hypothetical Basalt Nuclear Waste Repository to Establish a Significance Criterion, PNL-SA-9112, Pacific Northwest Laboratory, Richland, Washington, 1981.

3. G. M. Petrie and M. G. Foley, Preliminary Results of Monte Carlo Simulations for the Pasco Basin, Washington, Using the AEGIS Geologic Simulation Model, PNL-SA-9114, Pacific Northwest Laboratory, Richland, Washington, 1981.

DEALING WITH REGIONAL HYDROLOGIC DATA-BASE LIMITATIONS CASE EXAMPLE: THE COLUMBIA RIVER BASALTS

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ABSTRACT

Limitations are encountered in assembling hydrologic data for a broad geographic region, such as the Columbia Plateau in the northwestern U.S., into a conceptual model of the hydrologic system. These limitations may become resonant in subsequent numerical simulations of hydrologic system behavior. Included among such data limitations are irregular spatial distributions of data, decreases in information with increasing depth from the land surface, uncertainties about the reliability of reported hydrologic data, disparities in time-dependent parameters, and lack of field verification of data. The preparation of a regional hydrologic system description, therefore, first involves a comprehensive data evaluation, wherein the data are classified and ranked in terms of their utility to the study. The results of this evaluation are essential in planning future data acquisition activities, as well as in selecting and developing models. In turn, iterative use of modeling, data refinement, and data acquisition is considered to be highly effective. The case example of preparing a hydrologic system description for the Columbia Plateau, as required for repository siting, illustrates methods of determining the accuracy of certain data, compensating for data limitations, evaluating the need for acquiring additional data, and refining data through iterative techniques. Emphasis is placed on professional subjectivity, which has proven to be essential in data base evaluation and refinement.

INTRODUCTION

When modeling natural systems on a regional scale, data limitations are inevitable. This paper describes the type and magnitude of data uncertainties that affect conceptual and numerical modeling of the Columbia Plateau regional hydrologic systems. Specific limitations include:

- irregular spatial distributions of data
- decreases in information with increasing depth from the land surface
- uncertainties about the reliability of reported hydrologic data

- disparities in time-dependent parameters
- and lack of field verification of data.

Systematic evaluation of the data produced by field studies before they become the building blocks of conceptual and computer models is necessary. The accuracy of field measurements can influence a modeler's treatment of the data and his subsequent conclusions. Figure 1 depicts the iterative relationship among data acquisition, data evaluation, and modeling.

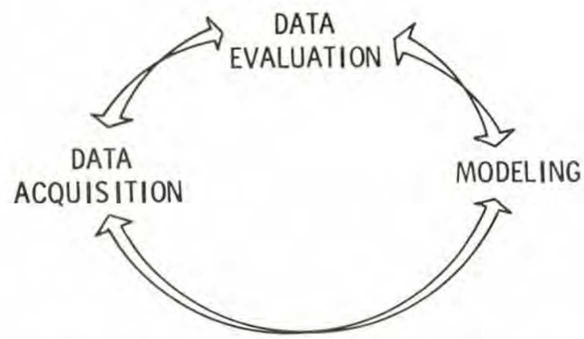

Fig. 1. The iterative process used in systems characterization and simulation

In this paper we briefly discuss the Columbia Plateau area and its hydrologic framework. A discussion of data evaluation, compensation, refinement, and level-of-confidence methods follows. Examples are presented to indicate the importance of ranking data sources according to reliability.

SETTING AND BACKGROUND

The lavas of the Columbia Plateau (Figure 2) comprise a tholeiitic flood-basalt province of moderate size (approximately 77,000 square miles) and moderate volume (estimated at 77,000 cubic miles [1]). The Columbia River basalts are unique in that the horizons high in primary porosity (e.g., flow contact zones and some interbedded sediments) are often separated by thick sections of the dense flow interiors. Characteristically, laterally transmissive horizons occur within "interflow zones" (along flow contacts) and are confined between thick, extensive layers of dense (columnar) basalt. These sections of the dense flow interiors result when large volumes of lava extrude in a short time. The net effect is a hydrogeologic system that resembles a classical, confined, multiple aquifer complex.

Data generated through irrigation well reconnaissance studies and waste-management programs at Hanford over the past two decades has drastically improved our Columbia Plateau ground-water information [2]. Our ability to use subsurface data to differentiate between individual flows is a relatively recent development. Any evaluation of data for the Columbia Plateau must account for the sequence of development of these events.

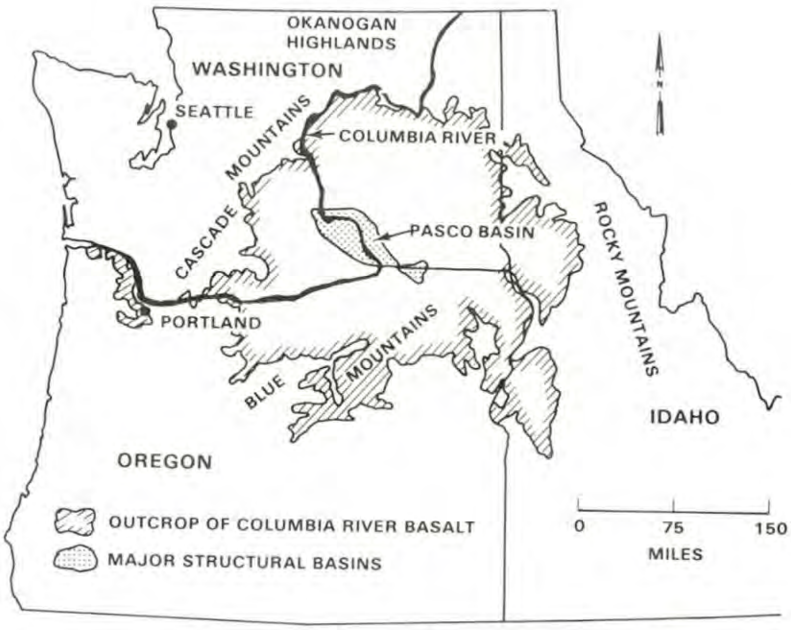

Fig 2. The Pasco Basin within the Columbia Plateau

EVALUATION OF DATA

The precept that all data can be of some value to a study begs several questions, the first of which concerns the value of erroneous data. If we can recognize, evaluate, and understand the nature of an error, we can avoid errors associated with the use of such erroneous data. Because all data has a degree of error, a determination of what constitutes a tolerable error must be made preceding data evaluation. Also, data evaluation should be a continuing and iterative component of every study. The effectiveness of an evaluation is controlled by the degree of communication among the investigators, who must systematically classify and rank data (in terms of reliability and importance). A specified methodology should be applied consistently throughout the study. Whenever professional subjectivity is a feature of the methodology, the basis for such judgments should be appropriately documented.

The following paragraphs discuss methods of evaluating hydrologic data (in view of the Columbia Plateau regional study). We will consider the role of geologic data in hydrologic evaluations, methods of compensation and refinement, and the degree of confidence to be placed in the data.

The Role of Geologic Data in Hydrologic Evaluations

Regional hydrologic studies must initially draw information from geologic studies, maps, and reports. Stratigraphic data, for example, often proves useful to initially delineate hydrostratigraphic units and geohydrologic structures for the Columbia Plateau.

Not all geologic data can be used to interpret hydrologic patterns. The available geologic data may not have been collected with hydrologic interpretation in mind, but rather for geotechnical studies, facility siting, economic evaluations, background environmental studies, or academic research. Each of these types of studies may employ different collection techniques and investigative methods. Consequently, problems with data integration may exist at the outset.

Methods of Compensation and Refinement

Data may be interpolated, extrapolated, corroborated, and verified by model trials, depending upon their original quality and the needs, objectives, and structure of a study. Methods of compensation and refinement also vary according to the scale of the study. Regional hydrologic studies generally require large volumes of data, but tend to require a lesser degree of accuracy for individual parameters. A variance in data sources that may be unacceptable for a site-specific study may be within acceptable limits for a regional study. Interpolation and extrapolation properly employed can extend the utility of certain data. In the performance of these techniques the premise regarding the values of each datum becomes important. For example, data that are not considered good control points could possibly be used in a supplementary or supportive capacity to enhance control by interpolation, extrapolation, or corroboration.

Assigning Confidence Levels to Data

As mentioned earlier, if we are to assume that every datum is of some value to a study, we must systematically evaluate the worth of all the available data within the context of the study's objectives. Thus, we can assign a confidence level to each datum or data group. Decisions concerning the reliability of data are based on the subjective judgments of professionals familiar with data collection techniques. Some criteria typically used to evaluate data are reflected in the answers to the following questions:

1. <u>How</u> were the data collected?
   - What methods or techniques were used and was quality assurance reviewed?
   - What equipment was used and who collected the data?
2. <u>What</u> exactly do the data represent?
   - Do the data support old, or infer new, information?
   - Do the data supplant or conflict with other data?
3. <u>When</u> were the data collected?
   - Are the data useful as historical or current records?
   - How do the time frames of individual data points compare?
4. <u>Where</u> were the data collected?
   - Is the spatial and stratigraphic distribution sufficient?
5. <u>Why</u> were the data collected:
   - Were the data assembled for research?
   - Were the data assembled for regulatory compliance or enforcement?

The above list is only an example; the questions and ranking system necessarily differ from one study to the next. Tailoring criteria to a specific study is the province of experienced and knowledgeable professionals. Their conclusions, in turn, should become a part of the data record.

EXAMPLE 1--UNCERTAINITES CAUSED BY DECREASE IN INFORMATION WITH INCREASING STRATIGRAPHIC DEPTH

In dealing with a multi-layered, confined ground-water system such as present in the Columbia Plateau, it is necessary to identify discrete zones of lateral ground-water transmission or to delineate several such zones with common hydraulic characteristics. As demonstrated in Figure 3, the number of wells penetrating to successively greater depths below the land surface decreases significantly. Of 295 selected wells in the Pasco Basin for which drillers' logs are available, more than 90% bottom out within the upper 600 ft of the Columbia River Basalt. This may be compared with the minimum thickness of basalt within the Pasco Basin, which is on the order of 5000 ft.

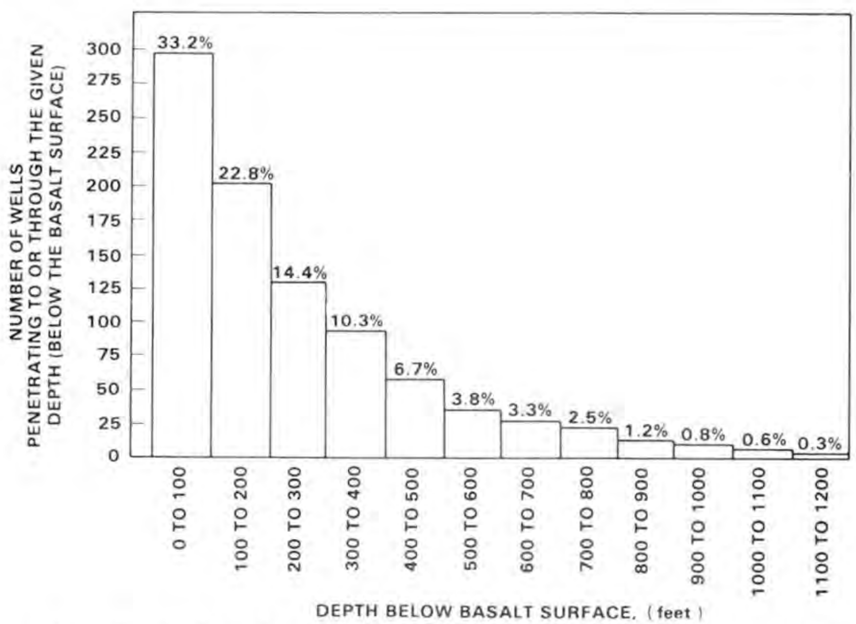

Fig. 3. Distribution with depth of Pasco Basin wells for which drillers' logs were available [3]

Although deep wells are relatively rare, available data for such wells tend to be comparatively more complete than the data on shallow wells. This condition is primarily accounted for by Federal and state agency-sponsored investigations of deep aquifers.

Compensation for deficiencies in the quality of data on shallow sources and deeper hydrologic units is commonly made by borehole correlation. Using correlation methods, we can take advantage of the enhanced quality of data for deep wells to interpret data for shallow wells. Also, shallower wells can be used to interpret the probable stratigraphy within and below the depths penetrated by surrounding "shallow" wells, using the assumption that deeper strata mimic the proportions and configurations of shallower formations. Values predicted by this technique for the stratigraphic surfaces in the Pasco Basin were within one percent of subsequently field-measured values.

EXAMPLE 2--UNCERTAINTIES CONCERNING THE RELIABILITY OF DATA SOURCES, TEMPORAL EFFECTS, AND THE NEED FOR FIELD VERIFICATION

The Washington State Department of Ecology receives information on well locations through water-well reports. If these reports are inaccurate, serious errors in hydrologic interpretation may result. Frequently, during field checks of the location of some of the large production wells, the WSDOE staff members find that water-well drillers have not accurately reported well positions. Summers and Weber [3] note, for example, that 8 of 59 wells reportedly sampled in the Pasco Basin outside the boundaries of the Hanford Site were not found during their field survey. This result suggests that the location of the wells was erroneously charted. It is not unreasonable to assume that a substantial fraction--perhaps as high as one-third--of the well locations shown on well reports is incorrect.

Reports filed with WSDOE by well drillers, however, do contain useful information about the well's location by township, range, section and subsection; a well construction schedule; and data on the elevation of the land surface, depth to water, well lithology, total depth, casing, and pumping. To test the reliability of this data source, we selected two types of information for scrutiny: 1) the ground-surface elevation estimate by the driller, and 2) the depth to water (or static water-level elevation). Errors in estimating these parameters can be more serious than errors related to temporal changes in the potentiometric surface. For purposes of this paper, seasonal variations in a water table, long-term water level changes, the effects of short term well pumping, and interference from nearby wells are classified as temporal changes.

We selected a sampling of 56 wells from a network of 135 wells from an area of approximately 120 square miles south of Spokane, Washington. All but 2 of the 56 wells represent discrete aquifers in the Wanapum Basalt Formation; the exceptions represent a composite of two aquifers. Simple criteria were used to select the 56 target wells for this study. The vertical error for survey closure could not exceed ± 1 ft and the well drillers' report had to include estimates of the ground-surface elevation and depth to the water level.

Estimates made by the five busiest drillers were examined. The drillers are arbitrarily referred to as A, B, C, D, and E. Error distributions for ground-surface-elevation estimates were calculated and plotted according to the individual driller (Figure 4). The results are highly variable. Essentially, Driller C's data tend to underestimate depths and overestimate outlier features by upwards of 100 ft. The data of Drillers

A and B vary less than C's, but Driller A also has a tendency to underestimate. Driller D committed only small or modest errors, but consistently underestimated the actual surface elevations. In comparison, Driller E overestimated slightly on only two occasions.

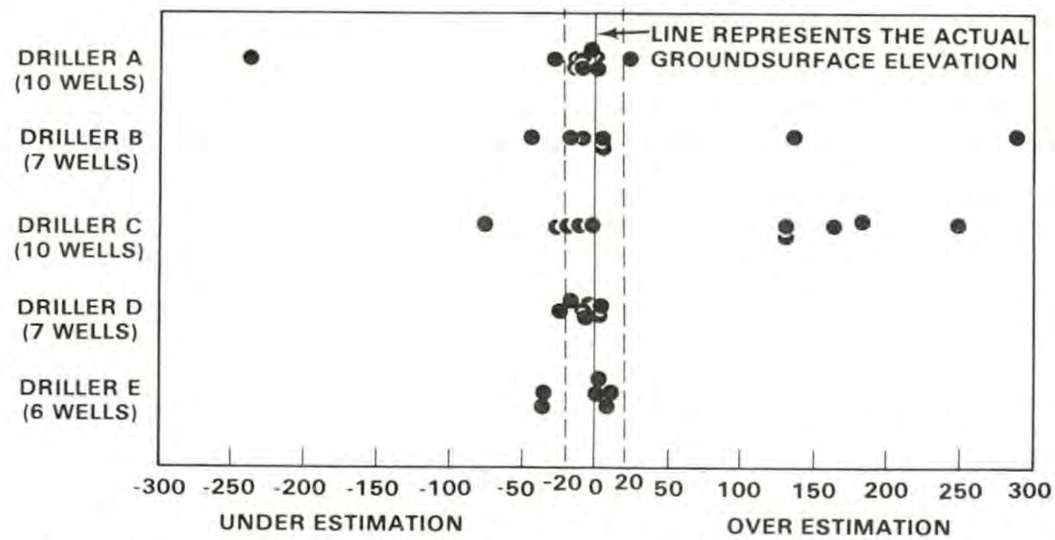

Fig. 4. A comparison of the accuracy of the ground-surface elevation estimates by the five most prolific drillers

Figure 5 provides even more insight into the uncertainty that surrounds potentiometric information provided by drillers. Once again, Drillers A and B are seen to commit modest to severe errors, and Driller C shows the greatest scatter. Before temporal corrections, Driller D easily surpassed the apparent accuracy of Driller E. The perceived performance of all of the drillers improved when we made all of the measurements time-equivalent and adjusted for errors caused by cascading water, well interference, and short-term water-well impacts. The reliability of estimates by Drillers C and D improved very little under this treatment. (D's estimates were good to begin with.) To put these water-level residuals into perspective, we should note that the static water levels in these wells typically range between 50 and 200 ft in depth. Also, all of these wells were drilled between 1971 and 1980 (most of them in the last four years). If the water levels at the time of drilling had been measured using appropriate methods, they would have been measured by steel tape or electric sounder and probably would be accurate to at least the nearest tenth of a foot. Although such accuracy may not be necessary for the modeling of an entire region or a sub-basin several thousand square miles large, it emphasizes how varied the data recorded by different drillers can be. This variability not only applies to drillers' data, but also to data recorded by state agencies, researchers, consultants, and other sources whose records we have reviewed and compared. When such field-verified samplings are made, each source providing a particular parameter can be ranked according to reliability in terms of the sensitivity of the numerical model.

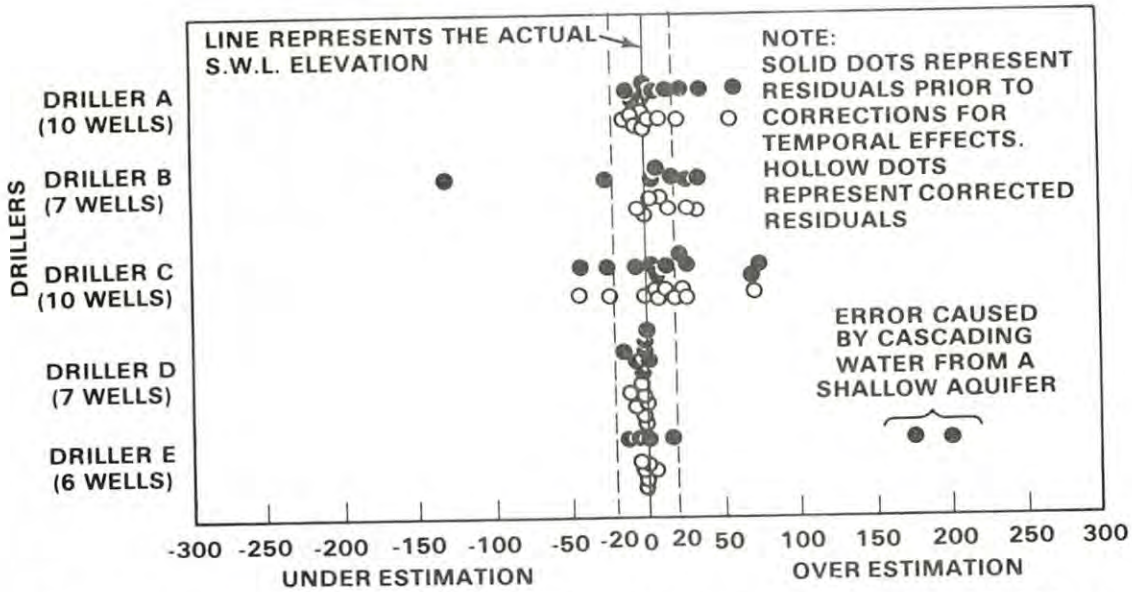

Fig. 5. Comparison of water-level residuals by individual drillers

CONCLUSIONS

With few exceptions, data limitations must be considered in every study. Studies of regional magnitude that involve characterization of natural systems (such as the hydrologic system) may contain the problems presented in this paper. Researchers and modelers must be cognizant of the accuracy of information sources. The reliability of data sources should, thus, be determined before initiating data input and construction of model parameter distribution maps (i.e., potentiometric surface maps). Data evaluation techniques (e.g., field verification) make these determinations through systematic classifications and rankings according to their use to the study and to the reliability of the information sources. Upon completion of a preliminary assessment, numerical simulations can be employed to provide feedback. Model feedback also can be used to plan future data acquisition and to revise computer codes. This iterative loop should be considered a continuing and essential component of the regional study.

REFERENCES

1. C. W. Myers and S. M. Price (Principal Authors) et al. Geologic Studies of the Columbia Plateau: A Status Report, Rockwell Hanford Operations, RHO-BWI-ST-4, Richland, Washington, 1979.

2. Hydrology Staff, Hydrologic Studies within the Columbia Plateau, Washington, An Integration of Current Knowledge, Rockwell Hanford Operations, RHO-BWI-ST-5, Richland, Washington, 1979.

3. W. K. Summers and P. A. Weber, Data for Wells Penetrating Basalt in the Pasco Basin Area, Washington, Rockwell Hanford Operations, RHO-BWI-C-19, Richland, Washington, 1978.

A STUDY OF PRESENT AND PERTURBED GROUND-WATER FLOW RATES THROUGH A HYPOTHETICAL BASALT NUCLEAR-WASTE REPOSITORY TO ESTABLISH A SIGNIFICANCE CRITERION

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ABSTRACT

Characterizing potential perturbations to the hydrogeologic system is a critical factor in computer modeling of the geology and hydrology of deep-underground, nuclear waste repositories. In this paper, calculated ground-water flow rates for various natural perturbations are used to define the threshold value of flow that signifies loss of repository integrity caused by a gradual increase of ground-water flow rate through the repository over geologic time. Our work suggests that an increase by one order of magnitude in the ground-water flow rate through a hypothetical basalt repository is a useful threshold value for preliminary assessments. An improved threshold value can be obtained by using output from the Geologic Simulation Model (GSM) for basalt rather than the simple, single-valued flow rate calculations.

INTRODUCTION

Release scenarios for underground nuclear waste repositories generally assume that, following a breach, ground water transports the radionuclides away from the repository. Often not addressed, however, is the definition of "breach." For use in the Geologic Simulation Model (GSM), described in the previous paper by Foley and Petrie [1], we propose that "breach" can be defined as ground-water movement through the repository exceeding a certain threshold value, or volumetric flow rate. A method of determining this threshold value for release scenario modeling is to compare the natural, unperturbed flow rate or "base case" with that associated with the proposed release scenarios. The threshold value determined by this method should equal or exceed the natural flow rate, but should be much lower than that associated with the release scenarios to allow for uncertainties inherent in the calculations.

The need for a threshold value is obvious when one considers the amount of data output by the GSM. Without having this value for use as a filter, the data analysts would be required to examine each GSM run to determine if the results warrant further detailed study. By providing for automatic machine sorting of runs with flow rates exceeding the threshold value, the data analysts can immediately focus on those runs that are of greatest interest, thus saving a great amount of time, energy and money.

In general, threshold values and release scenarios dealing with complex and variable natural systems cannot be generic. They must, depending on scale, be either site- or region-specific. If region-specific, the region should not encompass more than a single geologic or hydrologic province or subprovince. Ideally, one should focus on a region and then gradually increase concentration on a specific subregion or site as data becomes available and as the understanding of the local geologic and hydrologic systems increases. For example, the GSM models the Pasco Basin region of the Columbia Plateau province in southeastern Washington state (Figure 1). The Hanford Site, located at the center of the Pasco Basin, is being investigated as a possible repository location. The base case flow rate, release scenarios and threshold valves discussed in this paper were developed from a conservative interpretation of available data specifically for the Pasco Basin region.

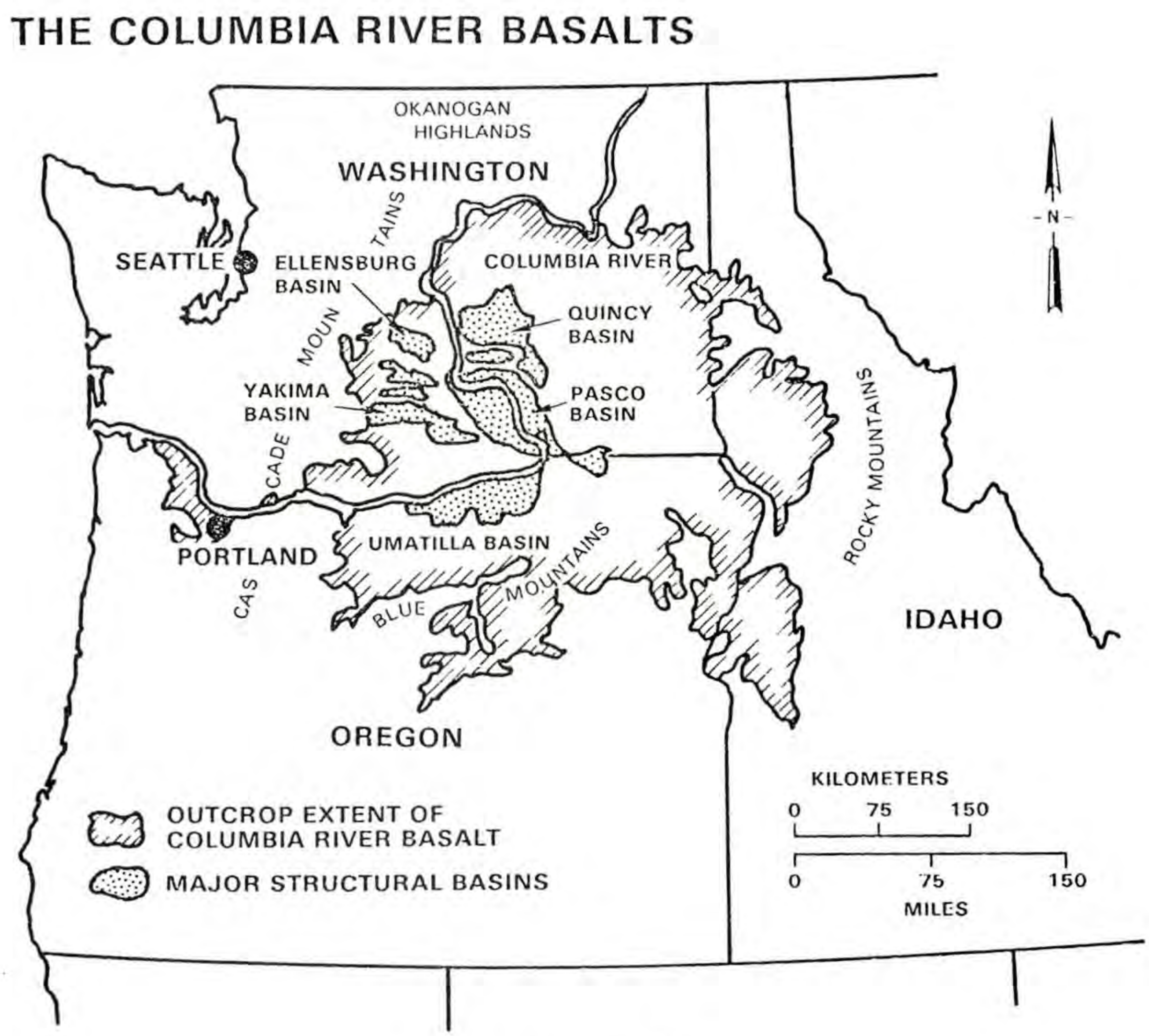

Fig. 1. Location Map

Base Case Flow Rate

The base case flow rate is the estimated rate at which ground water could flow through a repository in the Pasco Basin under the present hydrologic conditions. The GSM assumes that much of the water discharging into the Pasco Basin originates in the topographically higher region to the northeast where the deep aquifers of the basin are recharged (Figure 2). The volume of ground water that could flow from these aquifers and through the repository is probably quite low, but can be estimated using the Darcy equation,

$$Q = KA \frac{\Delta H}{\Delta L}$$

where:

Q = flow rate
K = permeability
A = cross sectional area
ΔH = hydraulic head difference
ΔL = flow path length.

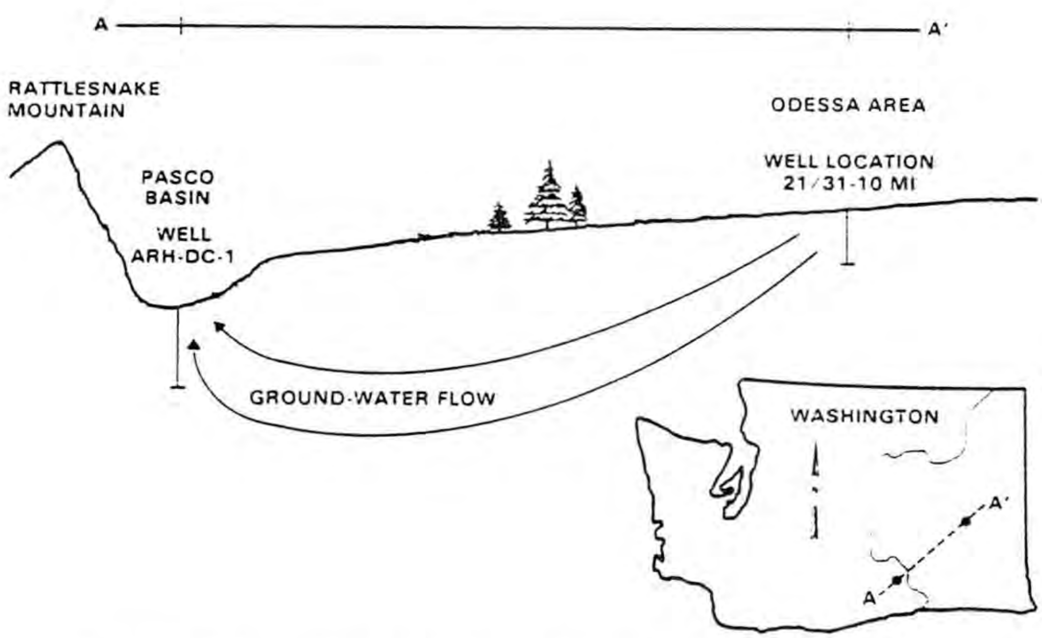

Fig. 2. Schematic Southwest to Northeast Cross-Section of the Ground-Water Model Area Showing Ground-Water Flow Direction

The hydrologic data used in the calculation of the base case flow rate is given in Table 1. These data were obtained from three boreholes that penetrate the Umtanum unit (proposed repository horizon) of the Grande Ronde basalt at the Hanford Site. Using the ARHCO [2] data for DC-1, the calculation results in an upward flow of 0.007 m^3/day/km^2 through the repository. Downward flow volumes of 0.020 and 0.019 m^3/day/km^2 are obtained by using the data from DC-2 and DC-6. Of these values, the upward flow appears to have the greatest potential for transporting radionuclides to the accessible environment. Consequently, for analytical conservatism, we assume for the base case that ground-water flow through the repository will be in an upward direction due to a 2 m hydraulihead differential across the Umtanum unit and yield a flow volume of 0.007 m^3/day/km^2 of repository area. Note, however, that ground-water investigations are still continuing and that recent data may suggest more horizontal than vertical flow.

Table 1. Base Case Vertical Ground-Water Flow Through the Repository Site

Well	Reference	ΔH^a (m)	Q (m^3/day/km^2)
DC-1	LaSala and Doty [3]	0	0
DC-1	ARHCO [2]	2	0.007 (upward flow)
DC-2	Apps and Others [4]	5.49	0.020 (downward flow)
DC-6	Apps and Others [4]	5.18	0.019 (downward flow)

$$A = KA \frac{\Delta H}{\Delta L}$$

$K = 3.048 \times 10^{-7}$ m/day (Deju and Fecht [5])

$A = 1\ km^2$

$\Delta L = 82$ m (ARHCO [2])

a Change in hydraulic head across Umtanum unit

Disruptive Phenomena and Release Scenarios

In developing release scenarios for a nuclear waste repository in the Pasco Basin several potential disruptive phenomena were analysed. These were: climatic change, glaciation, magmatic activity, folding, faulting, shaft seal failure, undetected features, geomorphic processes, meteorite impact, alteration of the hydrologic system and man. As a result of the analyses four release scenarios were developed: 1) climatically-induced increase in recharge and ground-water flow through the repository, 2) fault rupture of the repository, 3) fracturing of the repository host rock by folding, and 4) borehole penetration of the repository. No attempt was made to quantify the probability of occurrence for these scenarios, but the probabilities are thought to be quite low. The potential effects on ground-water flow through the repository associated with each of these scenarios will be discussed briefly.

Increased Recharge Scenario

The increased recharge scenario describes an increase in ground-water flow through the repository that results from climatic change at the recharge areas for the deep aquifers of the Pasco Basin--the Wanapum and Grande Ronde Basalts (Figure 3). Increased recharge could result from increased precipitation, glacial ice or other climatically-related phenomena. This scenario relies on the several assumptions listed below.

1. Present hydraulic heads immediately below and above the Umtanum unit (reference repository host horizon) are similar to those at borehole ARH-DC-1 as reported in ARHCO [2]).

2. Present hydraulic heads of the aquifers in the Odessa area are similar to those reported by Luzier and Burt [6].

3. Hydraulic conductivity of the Umtanum unit is 3.048×10^{-7} m/day (Deju and Fecht [5]). Vertical and horizontal hydraulic conductivities are assumed to be similar.

4. The thickness of the Umtanum unit at the reference site is 82 m (ARHCO [2]).

5. Potentiometric surfaces of the aquifers in the Odessa area rise to the ground surface due to increased recharge. (Figure 4).

6. Potentiometric surfaces of the basalt aquifers in the Pasco Basin rise also (due to the increased recharge), and the percent hydraulic head loss from the recharge area at Odessa is held to the same percentage as present in both aquifers (Figure 4). As a first order approximation we assume that discharge to the Columbia River has no significant effect.

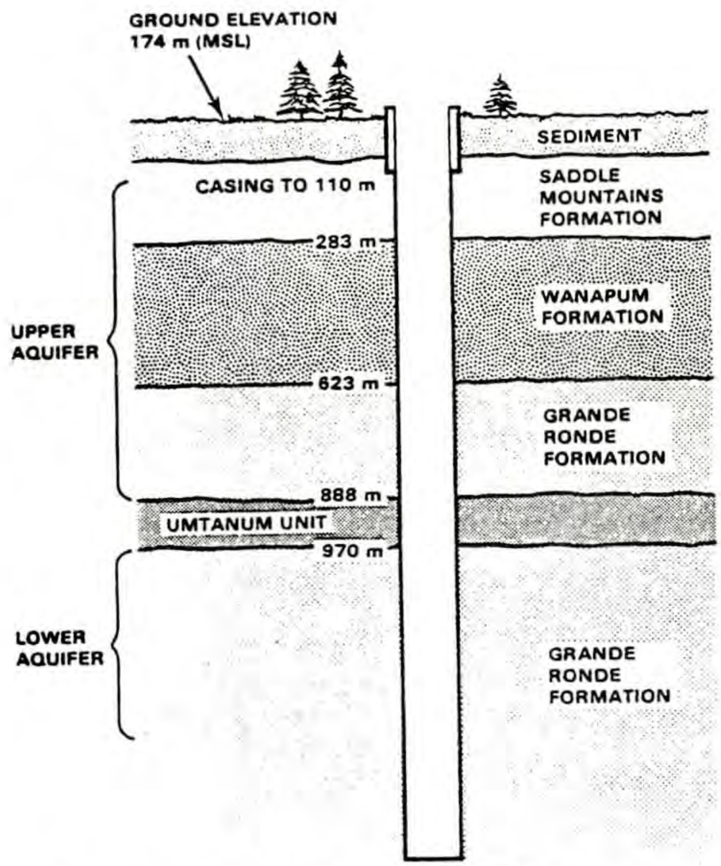

Fig. 3. Schematic Drawing of Well ARHC-DC-1 Structures, Geology and Hydrology (Sources: ARHCO [2]; LaSala and Doty [3] Gephart and others [9]).

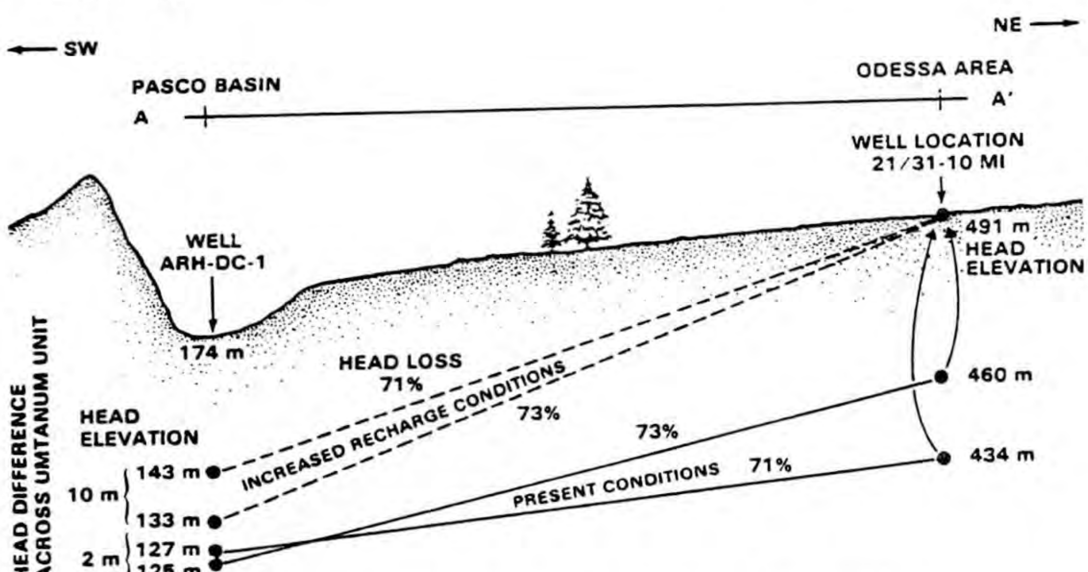

Fig. 4. Schematic Southwest to Northeast Cross-Section of the Regional Ground-Water Model Area Showing Increase of Head Difference in the Pasco Basin Due to Increased Recharge in the Odessa Area (Source for present head data: Luzier and Burt [6]).

As a result of the increased recharge in the Odessa area the head differences between the upper and lower aquifers (Figure 3) in the Pasco Basin increase from the conservative base case value of 2 m to 10 m. By substituting the 10 m head difference into the Darcy equation and holding the other parameters constant an upward flow volume of 0.037 $m^3/day/km^2$ through the repository results. Therefore, the hydraulic head changes proposed in this scenario could increase ground-water flow through the repository by a factor of nearly 5 over the base case situation.

For the fault intersection scenario, the repository conceptual model shown in Figure 5 was used. The following assumptions were made: 1) the fault zone intersects the repository and allows hydraulic interconnection of the upper and lower aquifer systems, 2) the hydraulic conductivity of the fault zone is 0.02 m/day, 3) the effective width of the fault zone is 2 m, and the length is 1 km, 4) the repository horizon is 82 m thick and 5) the hydraulic head in the lower aquifer system ranges from 2.0 to 10.0 m higher than in the upper aquifer system. Because there appears to be no data concerning the hydraulic conductivities of young fault zones in the Columbia Plateau basalts, the value given by LaSala and Doty [3] for fractured, brecciated or weathered Grande Ronde basalt was used. This value is of the same order of magnitude as that used by Baca [7] for fractured basalt in his preliminary repository modeling effort. The hydraulic head differential value of 2 m was taken from a report on hydraulic testing of Well DC-1 by ARHCO [2]. This data source was used because of the long period of time over which testing occurred and scarcity of data concerning the hydrologic properties of the Grande Ronde basalt. The 2 m value represents the conservative base case conditions and the 10 m value, obtained from the increased recharge scenario, represents the maximum expected value.

Using the above assumptions, the total volume of ground water flowing through the fault zone from the lower aquifer system, through the repository, and into the upper aquifer would be 0.448 $m^3/day/km$ of fault length for a 2 m head differential and 2.44 $m^3/day/km$ for a 10 m head differential.

The folding scenario assumes that the repository will be located in a low permeability horizon and that at some time after closure tectonic stresses will cause an increase in the permeability of the repository host horizon. The increased permeability acting in conjunction with a hydraulic head differential between the upper and lower aquifer systems will cause ground water to flow through the repository and result in intermixing of the upper and lower aquifer systems.

The rock fracturing scenario assumes the following: 1) the hydraulic conductivity of the disrupted zone is 0.007 m/day, 2) hydraulic head differential between upper and lower aquifer systems ranges from 2 to 10 m, 3) the repository horizon is 82 m thick, and 4) the repository is 1 km^2 in area. The hydraulic conductivity value is a weighted average of values reported by LaSala and Doty [3] for the Grande Ronde Basalt. The 2 m hydraulic head differential between the upper and lower aquifer systems is a conservative

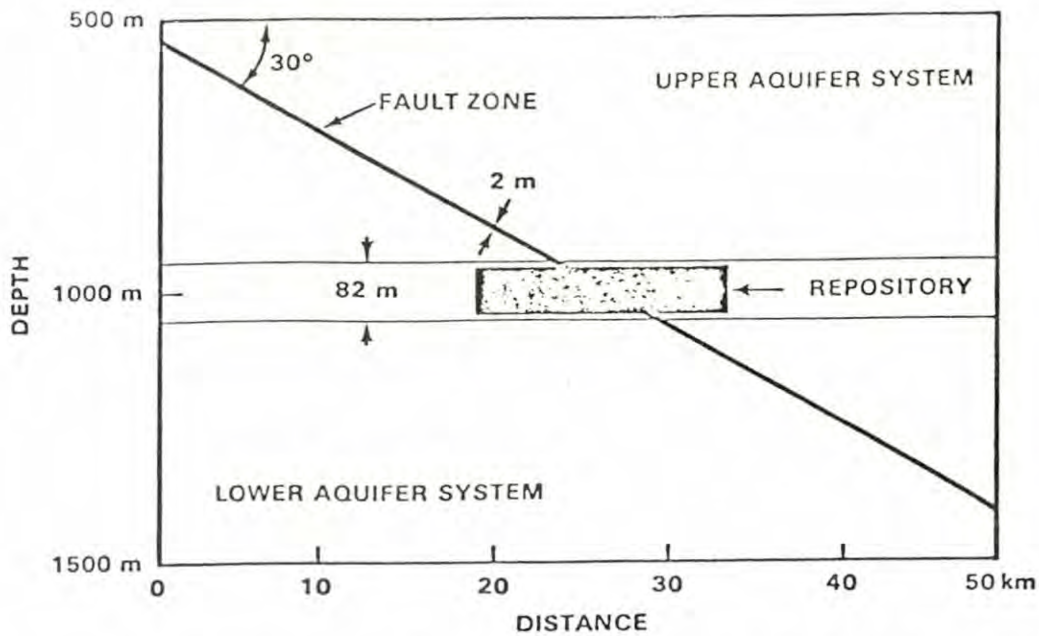

Fig. 5. Section View of Conceptual Model for Fault Intersection of Repository

value based on a hydraulic testing report of drill hole DC-1 [2]. This value was used because of the duration of testing and the scarcity of hydrologic data for the Grande Ronde Basalt. The 10 m value, taken from the increased recharge scenario, represents the maximum expected head differential.

Using the above assumptions, the maximum amount of ground water that could flow through the repository is 171 m^3/day/km^2 for a 2 m head differential and 854 m^3/day/km^2 for a 10 m head differential.

Borehold Penetration Scenario

Because his activities cannot be predicted with any certainty during the next 10^6 years, we assume that man will cause a repository breach. The scenario assumes that after the repository is sealed and administrative control is lost, an exploratory borehole is drilled through the repository. No attempt was made to calculate the probability of occurrence for the release scenario. This scenario provides an estimate of the amount of ground-water that could flow through a typical borehole. The borehole is assumed to be uncased below 110 m because: 1) this may allow a higher, more conservative flowrate, and 2) if a casing was installed it would degrade in scaling ability with time.

The borehole is assumed to be similar to borehole ARH-DC-1 on the Hanford Site. The geology, hydrology, and well structures of that borehole were used in developing the scenario. The equation used to calculate the flow rate through the borehole was derived from the well-known Thiem equation; geology and well structures were taken from LaSala and Doty [3], Gephart and others [8], and ARHCO [2]; hydraulic head data were taken from ARHCO [2] and Gephart and others [8]; radius of influence was taken from Gephart and others [9]; and transmissivity data were calculated from Gephart and others [8], LaSala and Doty [3] and ARHCO [2].

Well ARH-DC-1 was drilled to a depth below 1480 m, and except for casing to a depth of 110 m, the borehole is open. The well penetrates the Umtanum unit (the repository horizon) between about 888 m and 970 m below the ground surface. Because the conservative choice of head difference across the Umtanum unit is approximately 2 m (under present conditions with the lower aquifer having the higher hydraulic head), water would flow up the well if the well was not obstructed. The flow rate up the well (and through the Umtanum unit) would be approximately 18.3 m^3/day under present hydrologic conditions and 91.5 m^3/day under conditions of the increased recharge scenario.

Threshold Flow Rate Value

As stated earlier, one method of defining a threshold value is by comparison of the base case and release scenario flow rates. Using the information given in this paper one finds that, except for the increased recharge scenario, the potential estimated flow rates associated with the release scenarios exceed the base case flow rate by orders of magnitude. Thus, for conservatism, the threshold flow rate value constituting a breach in the GSM will be defined as any flow rate exceeding the base case flow rate by one order of magnitude. For the data discussed in this paper, the threshold value would be 0.07 m^3/day.

One must recognize, however, that the accuracy of this threshold value may be questionable because of the many uncertainties associated with its determination. The base case and release scneario flow rates were calculated using single value parameters. In reality, these parameters will have ranges of values. For example, reported values of permeability for the Columbia River Basalts range over seven orders of magnitude. Thus, permeability cannot be defined as a single value. Rather, the entire range of values as well as uncertainties associated with these values must somehow be factored into the analysis.

One method of addressing this problem is to use the results of the GSM runs to help define the threshold value. Because the GSM uses density curves for much of its input, the calculated flow rates will also be in the form of ranges of values. The calculated flow rates for several hundred or perhaps several thousand simulation runs may have a distribution such as is shown in Figure 6. With this type of information one can choose a threshold value with much greater confidence than is possible with single-value calculations.

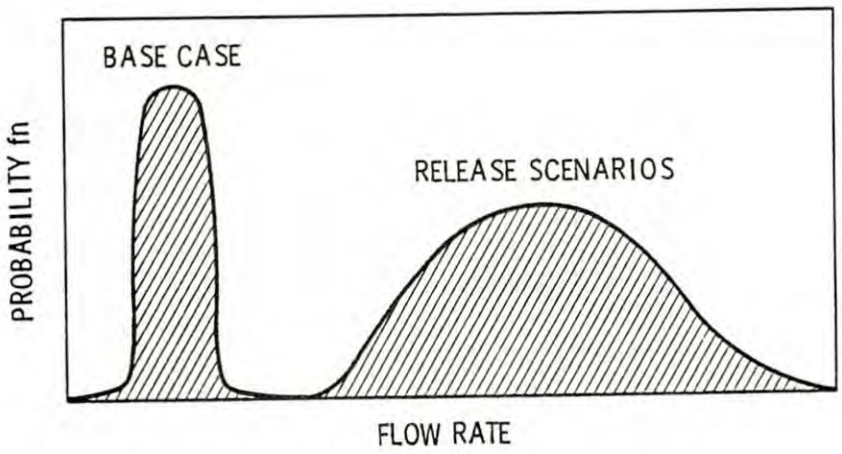

Fig. 6. G.S.M. Output Showing Comparison of "Base Case" and Release Scenario Flow Rates

Conclusions

The use of threshold values to determine if a breach has occurred during GMS runs can greatly reduce the amount of effort that would otherwise be associated with data analyses. However, if the threshold value is to have any validity it must account for the inherent variations and uncertainties associated with the data and method used to determine it. In general, if single-valued parameters are used the results may be suspect. The problem can be partially alleviated by using the results of GMS simulations to help define a more accurate threshold value that is based on distributions of values rather than on single values.

REFERENCES

1. Foley, M. G. and G. M. Petrie. 1981. The AEGIS Geological Simulation Model, PNL-SA-9111 A, Pacific Northwest Laboratory, Richland, WA.

2. ARHCO. 1976. Preliminary Feasibility Study on Storage of Radioactive Wastes in Columbia River Basalts. ARH-ST137, V. 1, Atlantic Richfield Hanford Company, Richland, WA.

3. LaSala, A. M., Jr. and Doty, G. C. 1971. Preliminary Evaluation of Hydrologic Factors Related to Radioactive Waste Storage in Basaltic Rocks at the Hanford Reservation, Washington. Open-File Report No. 71-180, U.S. Geological Survey.

4. Apps, J., T. Doe, B. Doty, S. Doty, R. Galbraith, A. Kearns, B. Kohrt, J. Long, A. Monroe, T. M. Narasimhan, P. L. Nelson, C. R. Wilson and P. A. Witherspoon. 1979. Geohydrological Studies for Nuclear Waste Isolation at the Hanford Reservation. LBL-8764, Lawrence Berkeley Laboratory, Berkeley, CA.

5. Deju, R. A. and K. R. Fecht. 1979. Preliminary Description of Hydrologic Characteristics and Contaminant Transport Potential of Rocks in the Pasco Basin, South-Central Washington. RHO-BWI-LD-20, Rockwell International.

6. Luzier, J. E. and R. J. Burt. 1974. "Hydrology of Basalt Aquifers and Depletion of Ground Water in East-Central Washington." Washington Department of Ecology Water-Supply Bull. 33, 53 p.

7. Baca, R. G. 1979. Basalt Waste Isolation Project Annual Report - Fiscal Year 1979. RHO-BWI-79-100, Rockwell Hanford Operations, Richland, WA.

8. Gephart, R. E., R. C. Arnett, R. G. Baca, L. S. Leonhart and F. A. Spane, Jr. 1979 A. Hydrologic Studies Within the Columbia Plateau, Washington: An Integration of Current Knowledge. RHO-BWI-ST-5, Rockwell Hanford Operations, Richland, WA.

9. Gephart, R. E., P. A. Eddy and R. A. Deju. 1979 B. Geophysical Logging and Hydrologic Testing of Deep Basalt Flows in the Rattlesnake Hills Well Number One. RHO-BWI-ST-1, Rockwell Hanford Operations, Richland, WA.

PRELIMINARY RESULTS OF MONTE CARLO SIMULATIONS FOR THE PASCO BASIN, WASINGTON, USING THE AEGIS GEOLOGIC SIMULATION MODEL

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ABSTRACT

Examples of statistical output and specific results from the AEGIS Geologic Simulation Model are presented in this paper. One goal of the statistical analysis is to disclose and demonstrate interrelations among variables. The two devices used for the AEGIS model for this analysis are contingency tables and correlation matrixes. Another goal is to provide a degree of characterization of significant parameters. This goal is met initially through use of cumulative distribution functions and density curves. Examples are provided of uses of the statistical information derived by the above methods.

INTRODUCTION

This paper presents some tentative results from the AEGIS Geologic Simulation Model. A summary of the modeling theory can be found in Foley and Petrie [1], with a complete characterization given in Petrie et al. [2]. Particular emphasis will be placed on those results with application outside the Hanford site. The observations are tentative because the input data are being refined, the model logic is under review, and ongoing field work may provide new information that could cause changes to the basic conceptualization of the system.

To provide adequate background some typical examples of statistical output from the model are considered before presenting specific results. This output is used for preliminary analysis [3] and is designed to be relatively simple to use and present because:

- The large amount of data generated by the model must be considered in a timely manner.
- The uncertainty inherent in the data may not justify extremely complex analysis.

- It is easier to avoid what has been called the Cassandra effect (i.e., being correct but not believed) because some people believe that complex statistical analysis serves to confuse the issue.

The system does allow for complex analysis; however, an exploratory data analysis approach is used first.

STATISTICAL ANALYSIS

One goal of the statistical analysis is to disclose and demonstrate interrelations among variables. To this end, two devices are used in the AEGIS model: contingency tables and correlation matrixes. Table 1 shows an example of a contingency table. If a relation exists between the two variables, a pattern in the numbers will be found. A correlation matrix (Table 2) serves the same purpose as the contingency table: it highlights relations between variables. A value of '1' indicates a perfect positive relation between corresponding variables; a '-1' is a perfect negative relation. A zero indicates no relation between corresponding variables. These numbers prove nothing in themselves; it is the process of finding the geologic explanation for a given value that is worthwhile.

Another goal of the statistical package is to provide a degree of characterization of significant parameters. This goal is met initially through the use of cumulative distribution functions and density curves. Figure 1 shows an example of a probability density function (PDF) for the depth to the repository at 20,000 years into the future. Figure 2 shows the corresponding cumulative probability curve. Figure 3 shows the same curve but with an uncertainty term on the axis (95% confidence). Thus, one component of uncertainty, the precision, is defined. Accuracy, the other component of uncertainty, cannot be so easily measured. Given that the model is built with a "worst case bias," any inaccuracy should be conservative. However, two important warnings on biasing the model toward the worst case should be made. First, too much worst case bias is unrealistic, and may identify most events as potentially harmful. This is undesirable because significant events do not get the study they need when study is diverted to unimportant ones. Second, with all the interactions and time lags among the various events and processes, it is not always obvious what the worst case may be. For example, Table 3 shows the current best estimate for changes in hydraulic conductivity caused by faulting events. Incorporating this information into the model becomes significant only if faulting events within 1 km of the site are considered. Doing so would be "worst case" because this would introduce a bias toward higher hydraulic conductivity values. However, this would introduce a maximum bias on the order of only 4%.

Table 1. Contingency Table

		Y						
		1	2	3	4	5	6	Total
	1	6	2	0	0	0	0	8
	2	1	3	1	1	0	0	6
	3	0	2	4	1	0	0	7
X	4	1	0	2[a]	4	1	0	8
	5	0	0	0	3	1	0	4
	6	0	0	0	0	1	0	1
	Total	8	7	7	9	3	0	34

[a]Number of points in cell (4,3)

Table 2. Correlation Matrix Highlighting Relations Among Variables

	HCNEGR[a]	HCSWGR[b]	HCNEWS[c]	HCSWWS[d]	RDEPTH[e]
HCNEGR[a]	1.000	-0.857	0.999	-0.857	0.044
HCSWGR[b]	-0.857	1.000	-0.856	1.000	0.016
HCNEWS[c]	0.999	-0.856	1.000	-0.856	0.044
HCSWWS[d]	-0.857	1.000	-0.856	1.000	0.016
RDEPTH[e]	-0.044	0.016	0.044	0.016	1.000

[a]Head value at the start of the N.E. Grande Ronde ground-water flow system.

[b]Head value at the start of the S.W. Grande Ronde ground-water flow system.

[c]Head value at the start of the N.E. Wanapum - Saddle Mountain ground-water flow system.

[d]Head value at the start of the S.W. Wanapum - Saddle Mountain ground-water flow system.

[e]Depth to the repository.

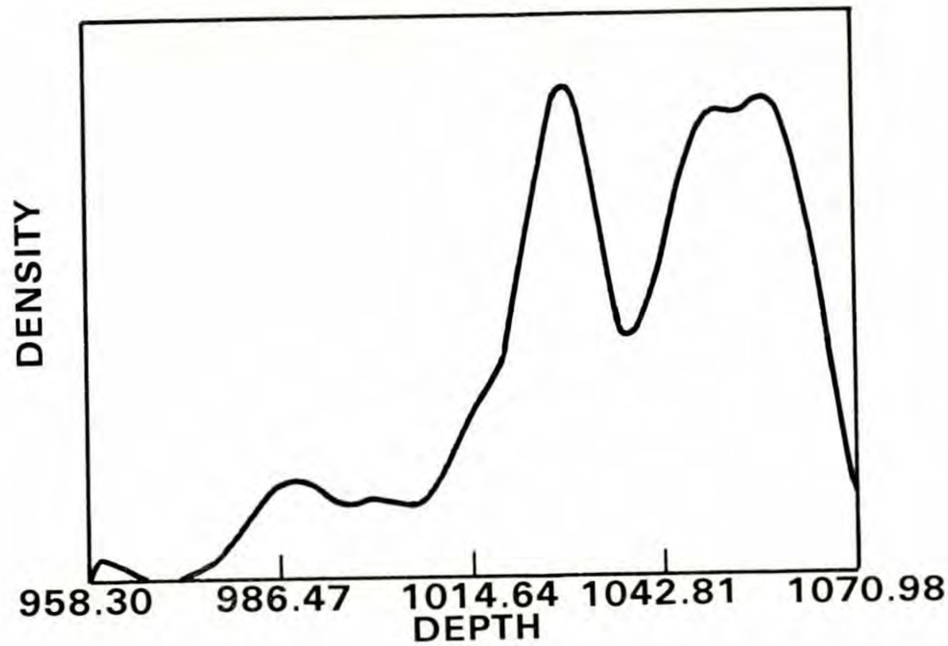

Fig. 1. Probability density function for the depth to the repository.

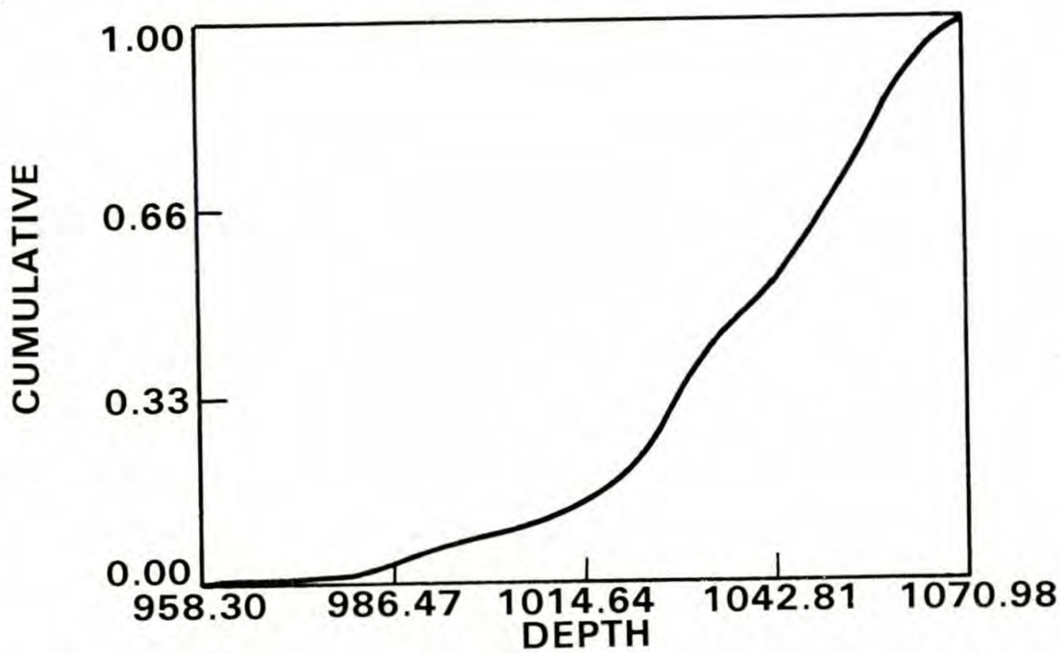

Fig. 2. Cumulative probability curve for depth to the respository.

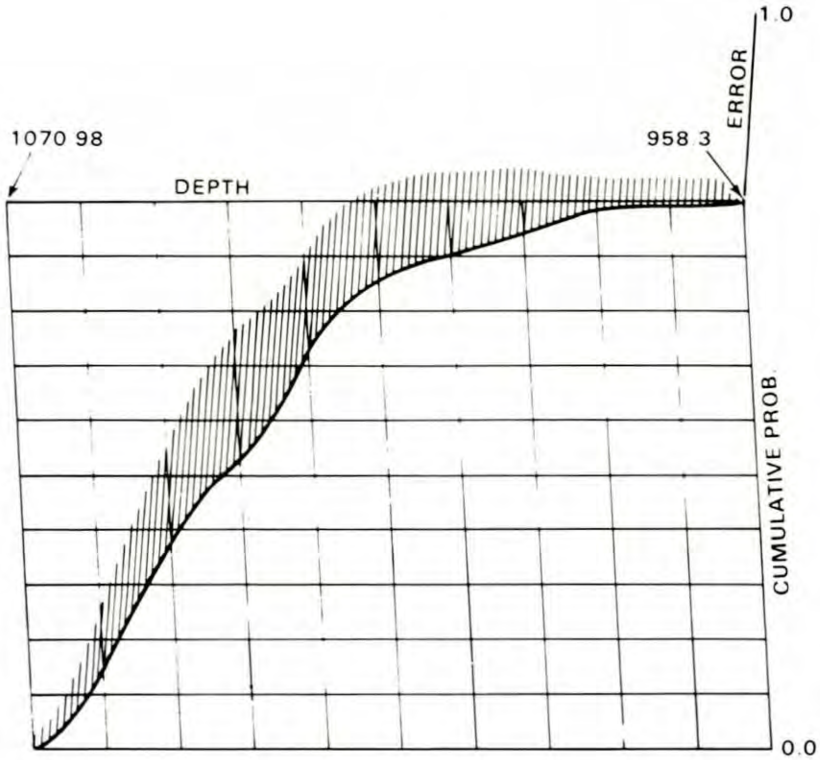

Fig. 3. Cumulative probability curve with an uncertainty term on the Z axis.

Table 3. Changes in Hydraulic Conductivity with Faulting Events[a]

Distance from Epicenter (km)	Percent Reduction in Hydraulic Conductivity
0.0 - 0.1	50%
0.1 - 0.5	25%
1.0 - 2.0	15%
2.0 - 5.0	4%
5.0 - 10.0	2%
10.0 - 25.0	1%

[a] This table is valid only for the Pasco Basin. Furthermore, while it is based on the best data currently available (Crosby unpublished Consultant Report, 1980), it is tentative and subject to change with new information.

Several uses have been identified for the information shown in Figure 3. For instance, it can be used to quantify the amount of uncertainty in the amount of erosion that will take place during the first 20,000 years. If this value is low enough, an argument can be made that river erosional factors need no longer be studied. The cumulative distribution curve also can reduce the uncertainty in the analysis because the depth to the repository affects the probability of drilling into the repository. In addition, the curve is used in a detailed analysis of a meteorite impact case because a greater number of meteorites can potentially affect the repository if erosion has removed part of the protective cover.

A question raised during the modeling effort was whether the uncertainty in understanding the current system state invalidates any attempt to extrapolate into the future. To explore this question, the model was stripped down to two components, one that set the starting conditions of the model and one that evaluated these conditions in terms of ground-water flow. Any elements that allowed for changes with time were removed. Figure 4 shows an example of one PDF (of several) used to set the hydraulic conductivity of part of the Model Cross Section [1]. This stripped-down version of the model was run in the Monte Carlo mode. From this experiment, the ground-water flow for the southwest ground-water system PDF was found as shown in Figure 5. While the scatter was large, it was smaller than expected. The hydraulic conductivity at the tightest part of the flow path seemed to be controlling (i.e., forming a bottleneck) the ground-water flow. To test this idea, the hydraulic conductivity of the repository was set to an <u>extremely</u> low value. Figure 6 shows the result of this experiment. Figure 7 shows the result of setting the value of the hydraulic conductivity at the repository at a low but reasonable value. Both of these trials caused a significant reduction in the flow velocity. These results suggest that: 1) if a critical controlling point in a system can be identified, it may be necessary only to reduce the uncertainty at that point to an acceptable level rather than to rework the whole system; and 2) undue pessimism with regard to system performance may cause an obvious solution to a problem to be overlooked.

Results from the AEGIS Geologic Simulation Model emphasize this latter point when considering "disruptive" processes that may affect a repository containment system. For instance, arguments have been made in proposed repository-siting regulations that no site should be located where it could be affected by a continental glacier. However, the modeling effort has identified several benefits from being close to a large ice sheet:

- lowered head gradients (thus slowing down the ground water flow)
- reduced ground-water recharge area
- increased distance between the waste and the biosphere
- reduced uncertainty in man-caused effects.

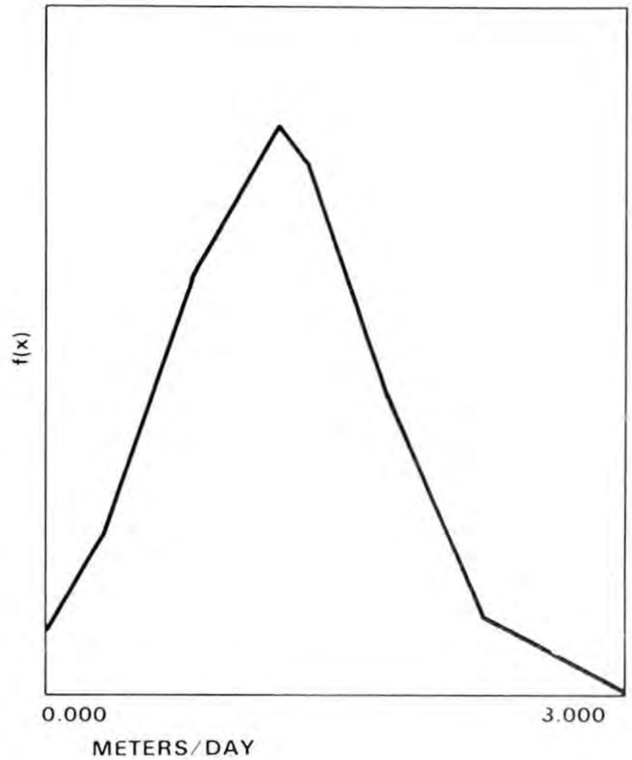

Fig. 4. Probability density function for the hydraulic conductivity of the Wanapum - Saddle Mountain basalt of the Pasco Basin.

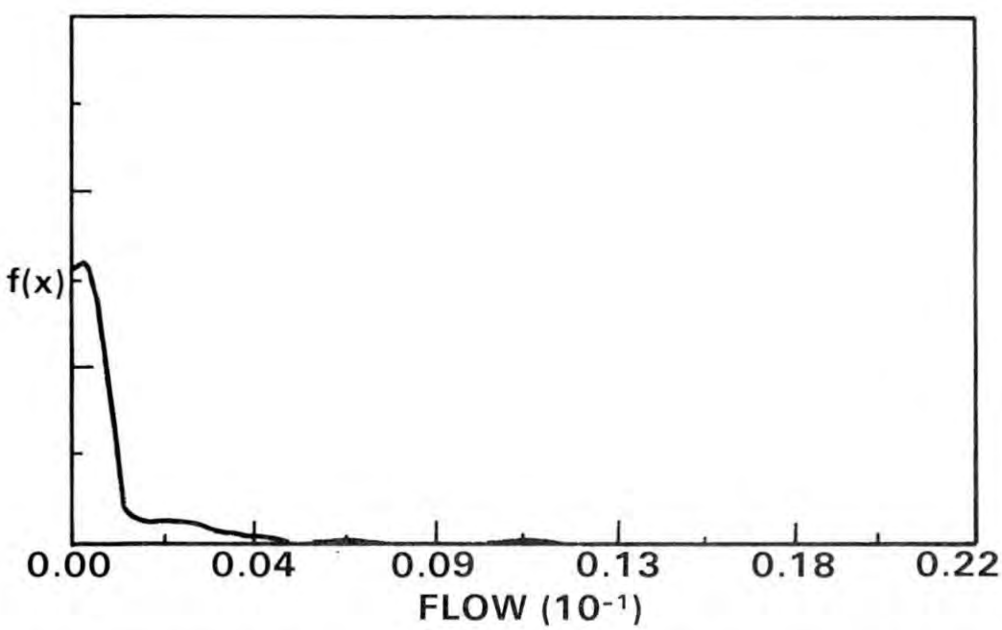

Fig. 5. Probability density function for the southwest Grande Ronde ground-water flow path.

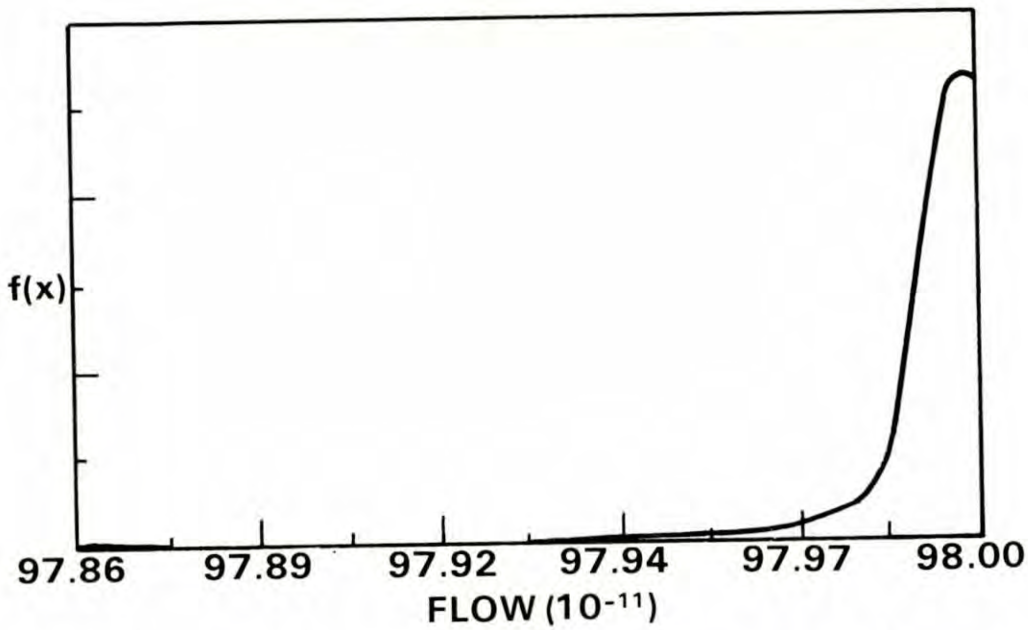

Fig. 6. Probability density function with the hydraulic conductivity of the repository set at an extremely low value.

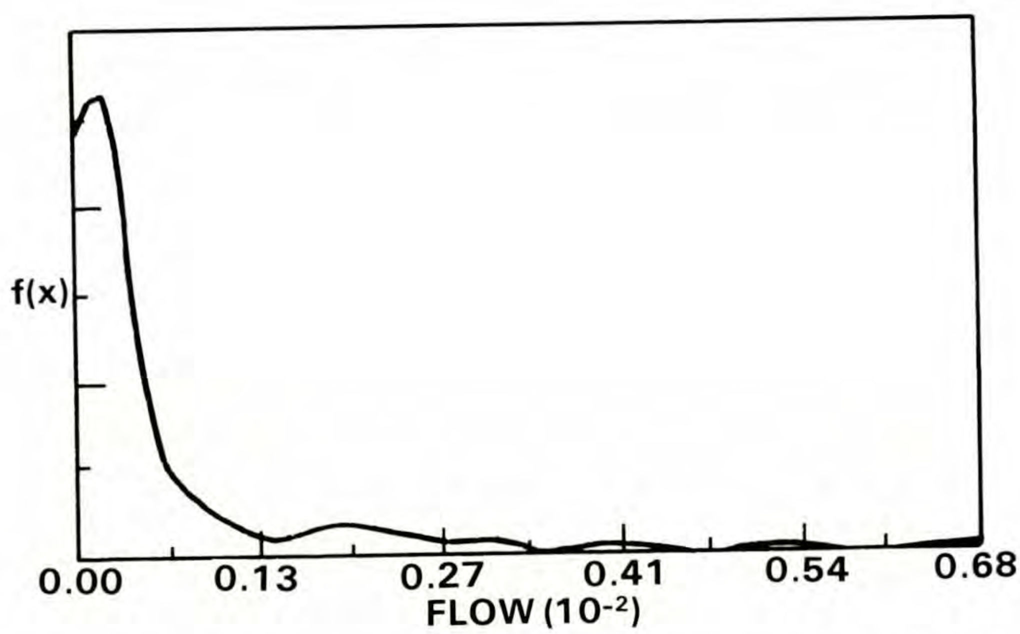

Fig. 7. Probability density function with the hydraulic conductivity of the repository set at a low, but reasonable, value.

This is not to say that continential glaciers or other potentially disruptive processes do not pose hazards. Rather, beneficial and harmful effects have to be considered on a site-by-site basis. The use of sweeping generalizations in siting regulations could result in the removal of several good sites from consideration.

REFERENCES

1. M. G. Foley, and G. M. Petrie, The AEGIS Geologic Simulation Model PNL-SA-9111, Pacific Northwest Laboratory, Richland, Washington, (1981).

2. G. M. Petrie, J. T. Zellmer, J. W. Lindberg, and M. G. Foley, Geologic Simulation Model for a Hypothetical Site in Basalts of the Columbia Plateau, PNL-3542, Pacific Northwest Laboratory, Richland, Washington, (1981).

3. B. Kleiner, and T. E. Graedel, Exploratory Data Analysis in the Geophysical Sciences: Rev. of Geophysics and Space Physics, V. 18, p. 699-717, (1980).

GEOSTATISTICAL MODELING OF PORE VELOCITY

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ABSTRACT

The characterization of ground-water pore velocity, including statistical uncertainties, is necessary to estimate convective or dispersive contaminant transport in geologic nuclear waste repositories. Pore velocity is a function of media transmissivity, media porosity, and ground-water potential. Geostatistical modeling techniques (kriging) are applied to Hanford Reservation field data to estimate porosity, transmissivity, and potential surfaces and to determine the associated estimation uncertainties. These quantities are combined statistically using first order expansion methods to stochastically characterize the pore velocity.

INTRODUCTION

Hydrologic modeling, involving the numeric solution of partial differential equations, provides a means for predicting the concentrations and transit times of contaminants in the ground water should a nuclear waste repository be breached. Because hydrologic data are subject to spatial variability as well as measurement error, hydrologists realize that the models are also stochastic [1,2,3,4,5,6,7]. This paper discusses the application of geostatistical techniques to the stochastical modeling of spatially varying hydrologic field parameters.

The basis of the traditional transport modeling is to provide a conservation of mass statement with convection to yield the expression:

$$\frac{\partial}{\partial t} c(x,t) + \nabla \cdot v(x)\, c(x,t) = s \tag{1}$$

where $c(x,t)$ represents the contaminant concentration at time t for spatial location $x = (x^1,x^2)$ in the plane, $v(x) = (v_1(x), v_2(x))$ is the ground-water pore velocity vector at location x and is assumed to be time invariant, and s is a source term. The deterministic approach assumes that the velocities are known perfectly at every location x. This is not true. The $v(x)$ are random quantities that must be estimated from discrete measurements of various combinations of the following data: hydraulic conductivity, effective porosity, and ground-water potential (level). Thus, the pore velocity may be considered as the realization of a stochastic process whose index set is contained in the plane. Equation (1) may be considered as a partial differential equation with stochastic coefficients. To solve for $c(x,t)$ (also a stochastic quantity) it is necessary to have an estimate of $v(x)$, denoted $v^*(x)$, as well as a characterization of the estimation error, $v(x) - v^*(x)$. Specifically, the solution of (1) requires at least $v^*(x)$, the bias of $v^*(x)$, and the covariance structure of $v(x) - v^*(x)$.

The pore velocity is related to the hydraulic conductivity (k), effective porosity (p), and ground-water potential (Φ) through Darcy's law:

$$v(x) = (v_1(x),\ v_2(x)) = -(k(x)/p(x))\ \nabla\Phi(x) \tag{2}$$

We shall use geostatistical techniques, e.g., kriging, to estimate conductivity, porosity, and potential gradient surfaces from hydrologic field data. We shall then statistically combine these quantities to produce unbiased estimates of $v(x)$ and to stochastically characterize the estimation error, $v(x) - v^*(x)$. These techniques will be demonstrated using hydrologic field data from the Hanford Reservation.

DESCRIPTION OF HANFORD FIELD DATA

The Hanford nuclear reservation is located within the Pasco Basin, in the northcentral portion of the physiographic province known as the Columbia River Plateau. Some 2200 wells drilled within the reservation boundaries are collectively known as the Hanford wells. The majority of the 1700 functioning wells are used for ground-water hydrological data collection or basalt stratigraphic characterization.

For this study we selected a 20,000-ft x 20,000-ft region within the Hanford reservation known to have a tremendous range of hydraulic conductivity values. This type of hydrologic data set was needed to determine the applicability of geostatistical techniques to ground-water pore velocity modeling. Hydraulic conductivity and ground-water potential measurements were made at the 2000-ft x 2000-ft square grid points of the region. Unfortunately effective porosity values were not known at these grid points and a constant porosity value of 0.10 had to be used throughout the region; this value was supplied by PNL hydrologists.

STATISTICAL PRELIMINARIES

Geostatistics and kriging are statistical techniques that can be used to estimate a surface from spatially distributed data. They were developed in the early 1960's, primarily by the French mathematician Georges Matheron, to solve mining estimation problems [8,9,10,11,12,13]. Geostatistics is the more general term, but kriging, which refers to the estimation method itself, is often used in a more general sense.

Let $Z(x)$ be the value of a continuous surface at location $x = (x^1,x^2)$ in the plane. $Z(x)$ is assumed to be the realization of a stochastic process. Given that Z is observed at discrete locations $x_1, \ldots, x_n$, the kriging estimate of a random variable W is the linear combination of the observed data

$$W^* = \sum_i \lambda_i Z(x_i)$$

such that:

(i) $E(W - W^*) = 0$

(ii) $E(W - W^*)^2$ is minimal

(E denotes the probabilistic expectation operator.) Typically W is one of the following random variables:

(i) $Z(x_o)$ - punctual value

(ii) $\nabla Z(x_o) = (Z_{x^1}(x_o), Z_{x^2}(x_o))$ - gradient value

(iii) $\int_A Z(x)dx$ - integrated value over a set A in the plane

(iv) $\nabla^2 Z(x_o) = Z_{x^1x^1}(x_o) + Z_{x^2x^2}(x_o)$ - Laplacian value

We shall be concerned only with kriging punctual or gradient values for Z surfaces.

Traditional least squares regression analysis is not appropriate for estimating W because the observed Z data is not necessarily statistically independent. Least squares regression estimates ignore any stochastic continuity (or correlation) present in the Z surface.

The estimation of pore velocity involves the estimation of the hydrologic parameter surfaces

(i) $k(x)$ - hydraulic conductivity
(ii) $p(x)$ - effective porosity
(iii) $\nabla\Phi(x)$ - gradient of ground-water potential

which are related by Darcy's law

$$v(x) = (v_1(x), v_2(x)) = -(k(x)/p(x))\ \nabla\Phi(x)$$

Assuming that $k(x)$, $(1/p(x))$, and $\nabla\Phi(x)$ are realizations of stochastic processes and unbiased estimators $k^*(x)$, $(1/p(x))^*$, and $\nabla^*\Phi$ exist for these hydrologic parameters (e.g. from kriging), then we must consider the estimator

$$v^*(x) = -\ k^*(x)\ (1/p(x))^*\ \nabla^*\Phi(x)$$

Specifically, we must examine the bias and the estimation error of $v^*(x)$.

The unavailability of porosity data determined that $(1/p(x))$ was modeled as a constant (non-stochastic) value. Furthermore, the spatial correlation between the K and Φ surface values was entirely attributed to the spatial correlation between the mean values $E[K(x)]$ and $E[\phi(x)]$. This implies that $K(x)$ and $\phi(x)$ may be modeled as independent stochastic processes. The estimation bias,

$$E[v^*(x) - v(x)] = 0$$

since $k^*(x)$, $\nabla^*\Phi(x)$, and $(1/p(x))^*$ are unbiased, mutually independent estimators of $k(x)$, $\nabla\Phi(x)$, and $(1/p(x))$. Using a first order expansion, the error variance is given by

$$\mathrm{Var}[v_i^*(x) - v_i(x)] = (1/p(x))^2 \cdot \{m^2_k(x)\ \mathrm{Var}(\Phi^*_{x^i}(x) - \Phi_{x^i}(x)) + m^2_{\Phi_{x^i}}(x)\ \mathrm{Var}(k^*(x) - k(x))\} \tag{3}$$

FIELD DATA CONDUCTIVITY ANALYSIS

The conductivity data were measured at the 2000-ft x 2000-ft square grid points of the 20,000-ft x 20,000-ft region of the Hanford Reservation. Figure 1 displays these data values. The extreme spatial variability of the conductivity data (range greater than 100,000 ft/day) precluded kriging k estimates directly. Taking natural logarithms permitted the following stochastic model to be made for the conductivity data:

$$\mathrm{Log}\ k(x) = k'(x) = m_{k'}(x) + e(x)$$

where $m_{k'}(x) = E[k'(x)]$ is a cubic polynomial function of $x = (x^1, x^2)$ in the plane and $e(x)$ is the realization of Gaussian stationary process. This implies $k(x)$ is the realization of a lognormal process.

A least squares linear regression fit estimated the mean function $m_k'(x)$ and permitted the estimation of $e(x)$ at observed grid points; the estimated value of $e(x)$ is simply the residual value at grid point x. The covariance function of $e(x)$,

$$E[e(x)e(x + h)] = c(x, x+h) = c(h)$$

was estimated from

$$\frac{1}{N} \sum r(x)\ r(x + h) \tag{4}$$

where $r(x)$ is the residual value at grid point x, $h = (h^1, h^2)$ is a two-dimensional increment, and the sum is taken over all grid point pairs (cardinality = N) whose difference equals h. A nonlinear regression fit to the covariance data yielded

$$c(h) = 0.151 \exp(-0.565\ (h/2000)^2) \tag{5}$$

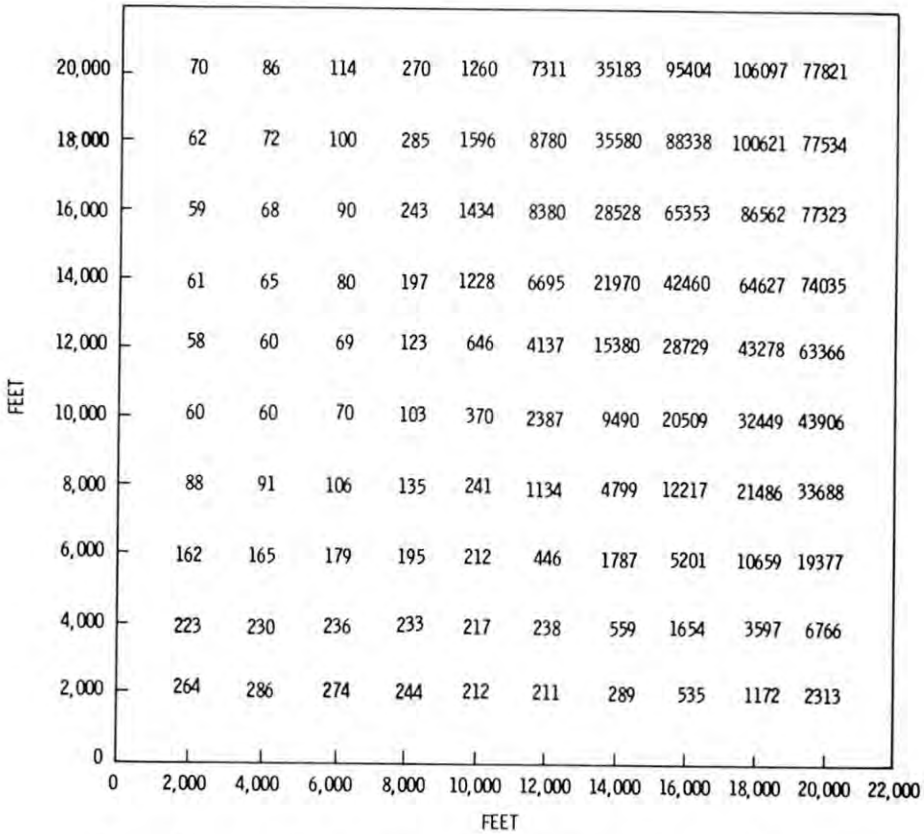

Fig. 1. Media conductivity data values (ft/day)

The covariance function of Equation (5) was input to the PNL kriging routine to estimate k'(x) on the [4000-ft, 16,000-ft] x [4000-ft, 16,000-ft] region.

Examination of the kriging error cross covariance function permitted the following approximation to be made.

$$\mathrm{Cov}(k'(x)-k'^*(x),\ k'(y)-k'^*(y)) = \frac{0.00283}{1 + 7.04\ (d/2000)^2} \tag{6}$$

where d is the distance between x and y.

Since k(x) is lognormally distributed

$$k^*(x) = 1.005\ \exp(k'(x)) \tag{7}$$

is an unbiased estimator of k(x). The estimation error variance is given by

$$\mathrm{Var}(k(x)-k^*(x)) = 0.0033(k^*(x))^2 \tag{8}$$

FIELD DATA POTENTIAL GRADIENT ANALYSIS

The potential (ground-water level) data were measured at the same grid positions within the Hanford Reservation as the conductivity data. Figure 2 displays these data values. The following stochastic model was used to describe the potential data:

$$\Phi(x) = m_{\Phi}(x) + e(x)$$

where $m_{\Phi}(x) = E[\Phi(x)]$ is a quadratic polynomial function of $x = (x^1, x^2)$ and $e(x)$ is the realization of a Gaussian stationary process.

A least squares linear regression fit estimated the mean function $m_{\Phi}(x)$ and permitted the estimation of $e(x)$ at the observed grid points. The covariance of $e(x)$ was estimated from the residuals using Equation (4) and a nonlinear regression fit as in the previous analysis of the conductivity data. The covariance of $e(x)$ is given by:

$$c(h) = 22.27 \exp(-0.45\ (h/2000)^2) \tag{9}$$

Because the covariance function of Equation (9) satisfies the differentiability conditions previously described and by definition the potential surface is smooth, we were able to krige estimates of the potential gradient. Figure 3 displays the kriged values of $\Phi_{x^1}(x)$ for the [4000-ft, 16,000-ft] x [4000-ft, 16,000-ft] region.

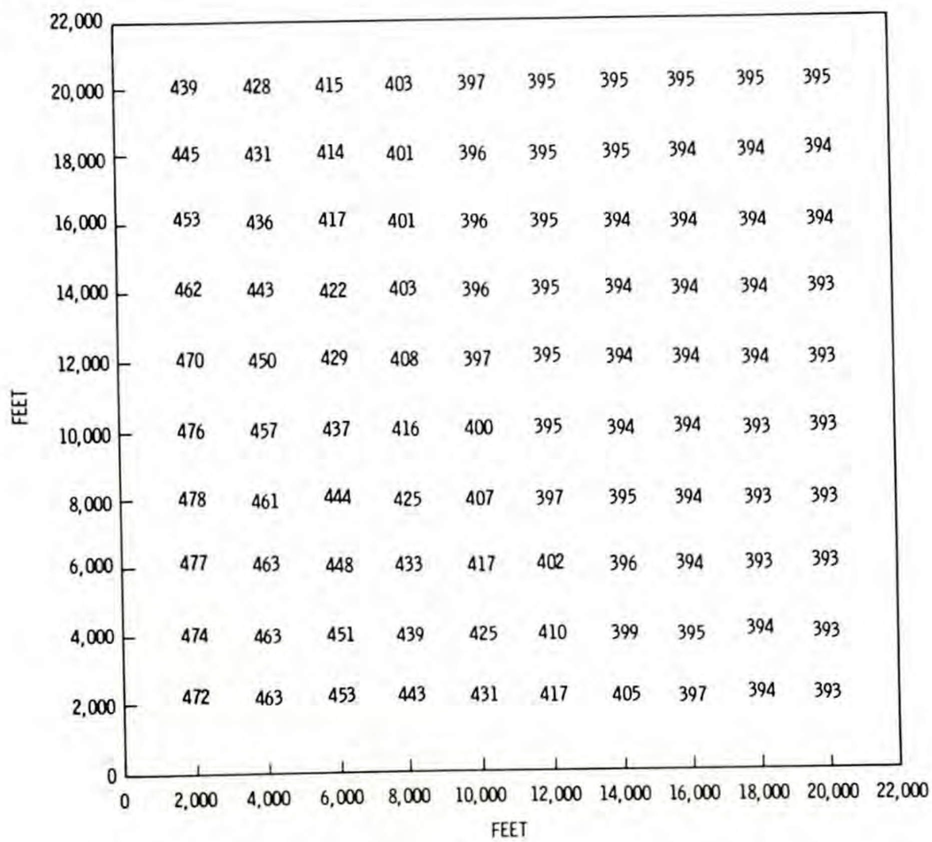

Fig. 2. Potential data values

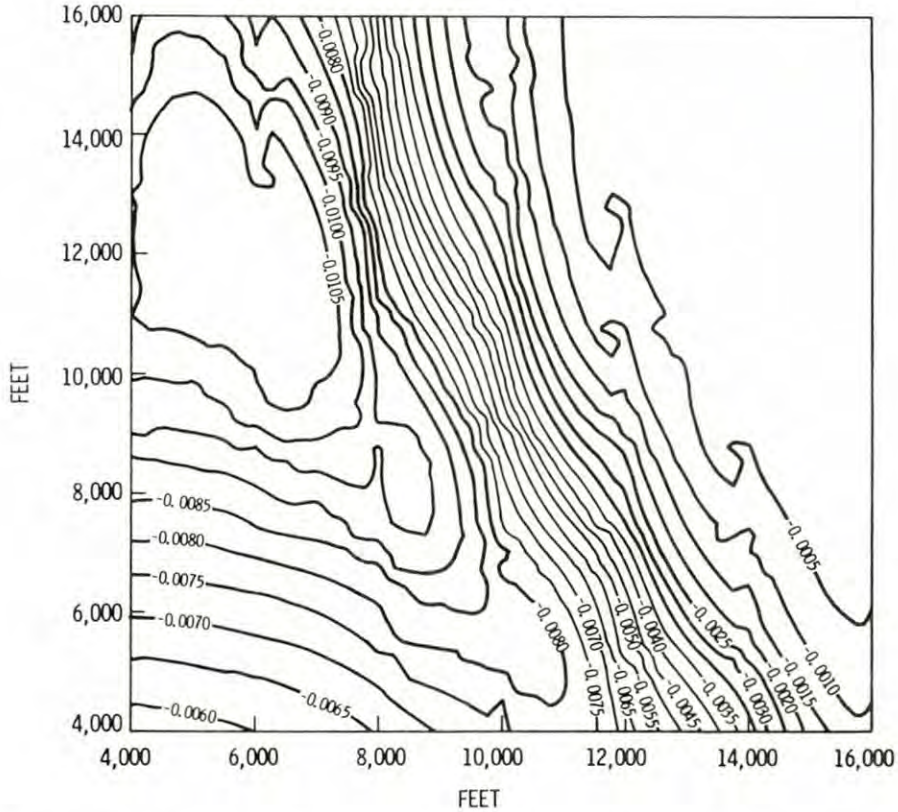

Fig. 3. Contour plot of $\Phi^*_{x_1}$, (east direction gradient)

Examination of the kriging error cross-covariance function allowed the following approximation to be made:

$$\mathrm{Cov}\ (\Phi_{x_i}(x)-\Phi^*_{x_i}(x),\Phi_{x_i}(y)-\Phi^*_{x_i}(y)) = \frac{1.39 \times 10^{-7}}{1 + 7.61\ (d/2000)^2} \qquad (10)$$

where $i = 1$ or 2 and d is the distance between x and y.

PORE VELOCITY CALCULATIONS

Darcy's law gives the pore velocity as a function of hydraulic conductivity, effective porosity, and ground-water potential:

$$v(x) = (v_1(x),v_1(x)) = -(K(x)/p(x))\ \nabla\Phi(x)$$

Since $K(x)$ and $\Phi(x)$ are uncorrelated stochastic processes and $p(x)$ equals the constant value 0.1, the unbiased estimate of $v(x)$ is given by

$$v^*(x) = -10K^*(x)\ \nabla\Phi^*(x)$$

where $k^*(x)$ is given by Equation (7) and $\nabla\Phi^*(x)$ is the kriged value of the potential gradient. Figure 4 displays the pore velocity estimates in the

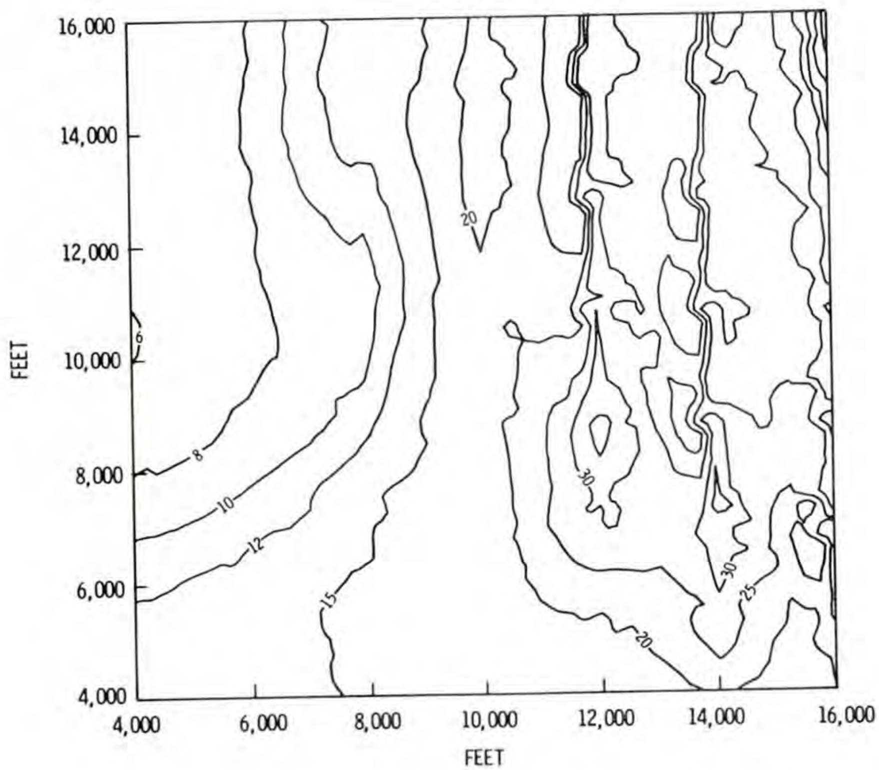

Fig. 4. Contour plot of $v_1*(x)$ (east direction)

x^1(east) direction in the [4000-ft, 16,000-ft] x [4000-ft, 16,000-ft] region. Notice that in the region where the conductivity values are largest (upper right quadrant), extremely small potential gradients give rise to significant pore velocities. The complexity of the level curves in this quadrant may be attributed to the high uncertainties (standard deviation equal 3.73 x 10^{-4}) associated with the kriged estimates of the potential gradient surfaces.

Using Equation (9) the variance of the pore velocity estimation error for either coordinate (north or east) is given by:

$$\mathrm{Var}(v_i(x)-v_i*(x))$$
$$= 100(k*(x))^2\ (1.784 \times 10^{-7} + 3.287 \times 10^{-3}\ (\Phi*_{x^i}(x))^2),\ i = 1 \text{ or } 2.$$

Figure 5 displays the standard deviation of the pore velocity estimation error for the [4000-ft, 16,000-ft] x [4000-ft, 16,000-ft] region. As expected the pore velocity estimation uncertainty is largest where the gradient relative uncertainty was largest (upper right quadrant).

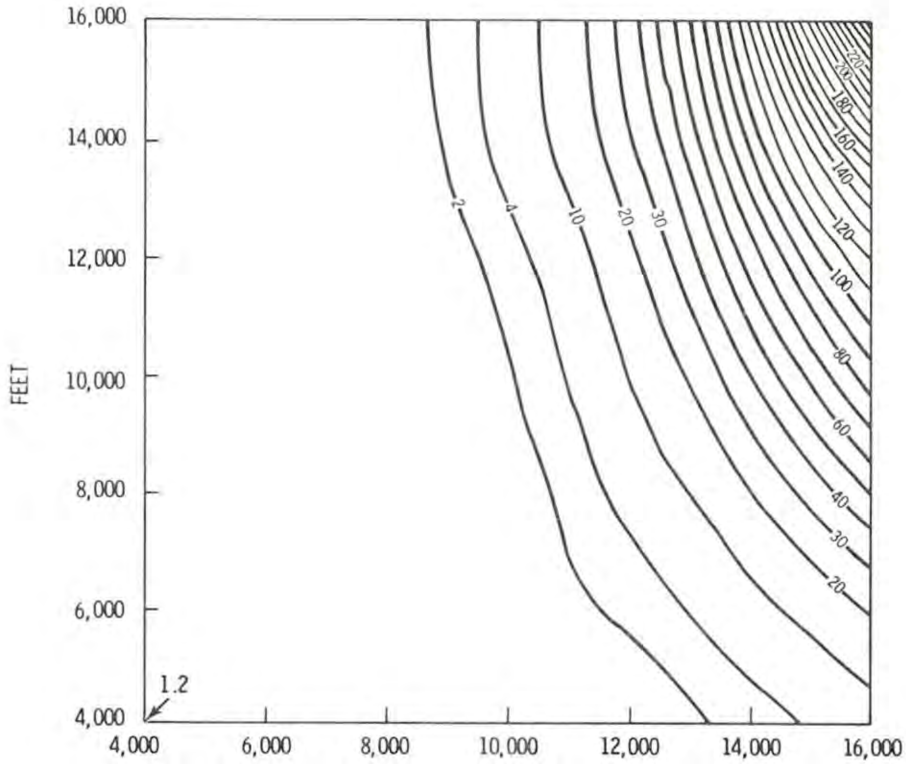

Fig. 5. Contour plot of the estimation error standard deviation for $v_1^*(v)$

CONCLUSIONS

In this paper we have discussed the application of geostatistical data analysis techniques (e.g. kriging) to the modeling of spatially varying hydrologic field parameters. The minimum variance property of kriging algorithms ensures that the maximal amount of information from the expensive well-data is utilized in determining the ground-water flow in a potential repository site. Traditional least squares regression analysis techniques ignore all spatial correlation and continuity present in the field data and add unnecessary uncertainties to the design process.

Kriging estimation techniques were applied to Hanford reservation data to accurately calculate hydraulic conductivities, ground-water potential gradients, and pore velocities. A first order expansion was used to statistically combine hydraulic conductivity and ground-water potential gradient uncertainties (and porosity uncertainties if data exists) to characterize the pore velocity uncertainty. This technique permits the estimation of pore velocity uncertainties even when direct pore velocity measurements do not exist. The product error propogation technique worked well except in the hydrologic region where the ground-water potential gradient was not accurately estimated ($\nabla\Phi \simeq 0.0$).

REFERENCES

1. A. A. Bakr, L. W. Gelhar, A. L. Gutjahr, and J. R. MacMillian. "Stochastic Analysis of Spatial Variability in Subsurface Flows 1. Comparison of One- and Three-Dimensional Flows." Water Resour. Res. 14(2):263-271. (1978).

2. J. P. Delhomme. "Kriging in the Hydrosciences." Advances in Water Resources. 1(5):251-266. (1978).

3. J. P. Delhomme. "Spatial Variability and Uncertainty in Groundwater Flow Parameters: A Geostatistical Approach." Water Resour. Res. 1 15(2):269-280. (1979).

4. G. Gambolati and V. Giampiero. "Groundwater Contour Mapping in Venice by Stochastic Interpolators 1. Theory." Water Resour. Res. 15(2):281-290. (1979).

5. L. W. Gelhar, A. L. Gutjahr, and R. L. Naff. "Stochastic Analysis of Macro-Dispersion in a Stratified Aquifer." Water Resour. Res. 15(6): 1387-1397. (1979).

6. V. Giampiero and G. Gambolati. "Groundwater Contour Mapping in Venice by Stochastic Interpolators 2. Results." Water Resour. Res. 15(2): 291-297. (1979).

7. A. L. Gutjahr, L. W. Gelhar, A. A. Bahr, and J. R. MacMillan. "Stochastic Analysis of Spatial Variability in Subsurface Flows 2. Evaluation and Application." Water Resour. Res. 14(5):953-959. (1978).

8. J. P. Chiles. "How to Adapt Kriging to Non-Classical Problems: Three Case Studies." Advanced Geostatistics in the Mining Industry. (1975).

9. P. Delfiner and J. P. Delhomme. "Optimum Interpolation by Kriging." Display and Analysis of Spatial Data. Eds. J. C. Davis and M. J. McCullagh, John Wiley and Sons, Inc., NY. (1975).

10. D. K. Krige. "Two-Dimensional Weighted Moving Average Trend Surfaces for Ore Valuation." Journal of the South African Institute of Mining and Metallurgy. pp. 13-79. (1966).

11. G. Matheron. "Principles of Geostatistics." Econ. Geol. 58:1246-1266. (1963).

12. G. Matheron. The Theory of Regionalized Variables and Its Applications. Vol. 5. Centre for Mathematical Morphology, Paris School of Mines, Fontainebleau, France. (1971).

13. R. A. Olea. "Optimal Contour Mapping Using Universal Kriging." J. Geophys. Res. 79(5):695-702. (1974).

EFFECTIVE SUMMARY EVALUATORS FOR DEEP NUCLEAR WASTE REPOSITORIES - GEOHYDROLOGIC RESPONSE FUNCTIONS

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ABSTRACT

Insight has been gained by the Assessment of Effectiveness of Geologic Isolation Systems (AEGIS) Program methodology demonstrations as hydrologic system modeling has been applied to evaluate hypothetical waste-repository sites in various geologic media. Among the results obtained from the hydrologic modeling are the Geohydrologic Response Functions, which summarize key geohydrologic effects that are important in site selection and repositiory evaluation. The response functions efficiently interrelate the three vital factors needed in the decision-making process: the quantity, arrival time, and location of contaminants reaching the biosphere. Geohydrologic Response Functions, presented in two sets of complete and easily used contaminant arrival curves, facilitate communication between technical staff and decision-makers and allow sharper definition and qualification of the requirements for realistic repository site selection.

INTRODUCTION

Over the past four years hydrologic system modeling has been applied to evaluate hypothetical waste-repository sites in various geologic media [1,2,3,4]. The purpose of these analyses has been to gain experience in using the AEGIS model sequence and to identify and define analytical deficiencies in the AEGIS methodology. Research and development activities directed toward improvement of the performance assessment methods have followed.

Of central importance to the AEGIS methdology is hydrologic analysis early in the site-selection process, which is actually the hyrologic and geologic design phase for the repository. Hydrologic modeling preceding the more complete geological evalution reduces the number of potential sites, and thus reduces costs of field data collection. Early use of hydrologic evaluation adds insignificant costs because that same model is always required at a later licensing stage to provide boundary conditions for a more detailed model of the site. This paper describes the Geohydrologic Response Functions and their use in transferring extensive results of technical analyses into simple summary relationships and in helping the public and decision makers to evaluate the adequacy of a repository design.

BACKGROUND FOR GEOHYDROLOGIC RESPONSE FUNCTIONS

The response functions can be developed either by starting from their theoretical origin and logically progressing through the analytical steps to their ultimate use, or by conversely beginning with the needs of the decision-making process and showing how the response functions satisfy those needs. It is convenient to focus on the ultimate use of the response functions by considering the pertinent factors in an evaluation of the contamination potential of a repository. The major analytical need is to estimate the extent, if any, of contamination that will reach the biosphere from a loss of repository integrity. Specifically, three factors are perceived to be of major significance to the decision-making process:

- quantities of contaminants reaching the biosphere
- times of contaminant arrival at the biosphere interface
- locations of contaminant emergence.

The quantity of contaminant reaching the biosphere (expressed as contaminant mass, concentration, or activity) must be known to effectively evaluate the seriousness to the environment and man. Small amounts of contaminant may be negligible, while larger quantities may constitute a serious hazard.

The time needed for a contaminant to reach the biosphere interface is the second vital factor. The time required for nuclides to reach the biosphere allows for decay. In addition, arrival time is the linking factor to many other aspects of repository analyses such as nuclide sorbtion and material heterogeniety.

The last of the three factors, the location of contaminant emergence, is important because a contaminant isolated from the biosphere may represent little or no hazard, even in rather large quantities. Under other conditions, small amounts of contaminants arriving at critical locations over short periods of time will involve severe hazard.

Knowledge of these three interrelated factors provides the data needed by the decision makers, enabling them to choose between site alternatives and to specify margins of safety in the repository design. Realistic determination of the same three factors is provided by detailed hydrologic modeling.

The factors of contaminant quantities, arrival times, and outflow locations can be interrelated in two ways. The most general approach is to use the outflow location as the predominant variable. This approach provides the arrival distribution summary, which is described in detail elsewhere [5,6,7,8,9]. The second approach uses the cumulative quantity of containment as the outflow predominant variable and has become known as the "Geohydrologic Response Functions." Response functions, though somewhat less general than arrival summary distributions because they are restricted to analyzing steady-flow systems, are simpler to apply and are particularly useful in the site-selection evaluation outlined previously.

GEOHYDROLOGIC RESPONSE FUNCTIONS

Geohydrologic Response Functions are a pair of summary relationships that interrelate the three factors of contaminant quantity, arrival times, and outflow locations. The first, or quantity/time, response is the more useful of the response function pair, while the second, quantity/location response function, may be of particular importance when considering the outflow location. They will be discussed in more detail following the repository flow-system example. As the name response functions suggests, they summarize all of the hydrologic effects between the contaminant input and outflow of the subsurface system.

Repository Flow-System Example

A repository flow-system example illustrates the response functions and shows their usefulness in performance evaluations. Such a subsurface flow system is shown in Figure 1 for a worst-case release from a hypothetical repository. The shape and location of the contaminated front depicts the gradual movement of contaminant from the repository toward a river. Beginning at the repository, the contaminated front slowly moves outward in expanding areas. Contaminants seeping along the shortest paths first reach the river about 15,000 yr after the initial contaminant release from the repository.

The longer more or less horizontal curves starting at the repository in Figure 1 and extending to the river represent some of the flow paths of contaminated water. The first contaminated water to reach the river moves along pathline Number 1, directly to the river and would arrive in slightly more than 15,050 years. More time is required for the fluid moving in the longer flow paths. For example, Pathlines 5 and 9 require about 15,495 yr and 17,337 yr, respectively, for the contaminated water to reach the river. For longer pathlines, such as 12 and 13, the elapsed times are 22,290 yr. Additional arrival times for other flow paths and data are summarized in Table 1.

Paths of flow are often referred to as pathlines when discussing travel times, and as streamlines when quantity or amount of flow is involved. The pathlines and streamlines are the same in Figure 1 because the flow system is steady. Accordingly, pathlines and streamlines will be used interchangably in the following discussion.

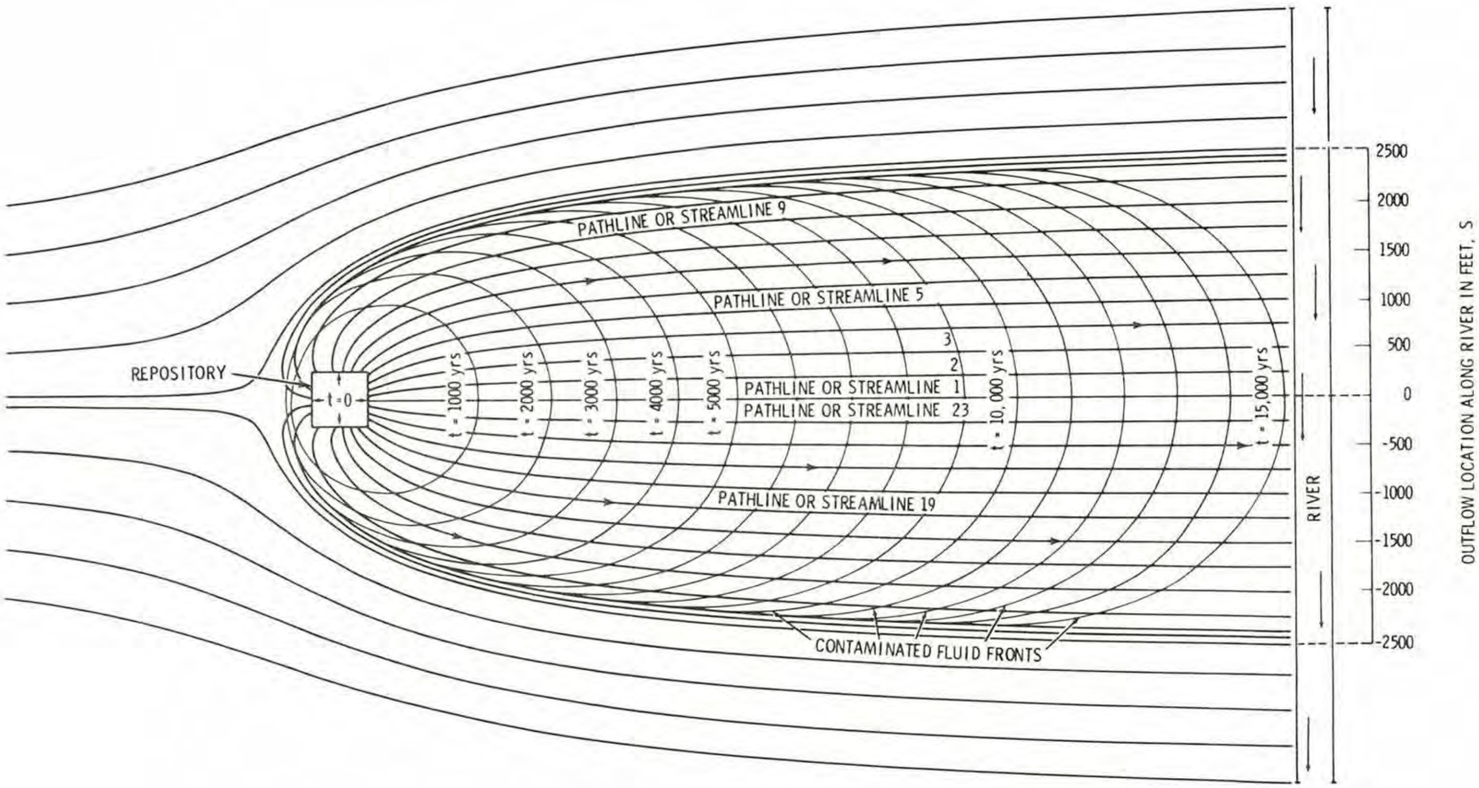

Fig. 1. The gradual movement of contaminated fluid from a repository toward the river (sample case)

Table 1. Summary of Contaminated Water Arrival Data at River Bank for Example Repository Analysis

Pathlines or Streamlines (Number)	Cumulative Relative Flow Rate Entering River (q/Q[a])	Arrival Time at River (T in Years)	Location of Contaminant Outflow into River (S in Feet)
1	0.00	15053	0
2 and 23	0.10	15079	± 248
3 and 22	0.20	15159	± 496
4 and 21	0.30	15299	± 745
5 and 20	0.40	15495	± 993
6 and 19	0.50	15768	± 1242
7 and 18	0.60	16134	± 1490
8 and 17	0.70	16629	± 1739
9 and 16	0.80	17337	± 1988
10 and 15	0.90	18546	± 2237
11 and 14	0.96	20300	± 2393
12 and 13	0.98	22289	± 2456

[a]$Q = 3.785 \times 10^5 ft^3/yr$ or 7.757×10^3 gal/day (2.932×10^4 l/day)

The streamlines shown in Figure 1 are spaced so that the same amount of fluid passes between successive streamlines. For example, 0.05 or 5% of the total outflow rate from the repository passes between Streamlines 1 and 2. Similarly, another 0.05 or 5% of the total flow rate enters the river between Streamlines 23 and 1. The flow between most of the other successive streamlines is the same increment of the total repository outflow.

The incremental outflow rates of the contaminated water entering the river are easily correlated with the arrival times at the river bank; in fact, from this correlation will emerge the response functions. In Figure 1, the contaminated fluid gradually moves along Streamline 1 and reaches the river at 15,053 yr (see Table 1). Before that first arrival time, the cumulative amount of contaminated fluid (denoted by q/Q) that has entered the river is zero. Our correlation in interrelating quantities, arrival times, and outflow location has established the first row of entries in Table 1. Specifically, for the contaminated fluid traversing Streamline 1, (first column in Table 1) the cumulative amount of fluid that has entered the river (second column) is $q/Q = 0$, which had outflowed into the river by the first arrival time, T = 15053 yr after the hypothetical repository release. Streamline 1 reaches the river at outflow location S = 0, the location where the first contaminated fluid enters the river.

For the second example of correlating quantities, arrival time, and locations, consider Streamlines 2 and 23 in Figure 1. From the symmetrical shape of the advancing contamination fronts, the contaminated water moving along Pathlines 2 and 23 could be expected to outflow into the river at the same time. As expected, the arrival for both is the same time, i.e., T = 15079 yr. The cumulative quantity of contaminated fluid that has entered the river by time (T = 15079) is the sum of the fractional flows between Streamlines 23 and 2, i.e., $q/Q = 0.05 + 0.05 = 0.1$. By the time the contaminated fluid traverses Streamlines 23 and 2 to reach the river, the cumulative inflow rate of contaminated water to the river includes all of the flux between those streamlines that have arrival times less than those in Streamlines 23 and 2. The second row of entries in Table 1 interrelate all of the factors associated with Streamlines 23 and 2. They are: 1) the cumulative quantity of contaminated fluid in terms of q/Q, 2) the arrival time, t, and 3) and location, S, along the first bank where the contaminated fluid emerges into the river.

The First (Quantity/Time) Geohydrologic Response Function

Interrelationships between quantity, arrival time, and outflow locations, are perhaps easier to visualize and discuss if displayed graphically. In Figure 2, the contaminant quantity expressed as the cumulative relative outflow rate, q/Q, is shown as a function of the arrival time, T, from Table 1. The result in Figure 2 is the first type or quantity/time Geohydrologic Response Function for the sample repository.

The response function, shown in Figure 2, has been discussed in terms of the continuous system flow of the advancing contaminated fluid front, gradually progressing from the repositiory through the subsurface flow system and emerging at the river. Actually, the advancing front in Figure 1 depicts the locations at various times of an instantaneous contaminated fluid pulse leaving the repository at time zero.

The instantaneous pulse is illustrated in Figure 2. A single vertical line is seen at T = 0, which is the contaminant input entering the subsurface flow system from the repository. The single vertical line at T = 0 really represents the instantaneous input pulse of contaminated water of height $q/Q = 1.0$ but lasting for only an infinitesimal time, Δt. If we consider the product of the repository outflow rate, Q, and Δt, the result is a small volume of contaminated water departing from the repository at zero time. We can visualize Δt as smaller and smaller until a sufficiently small volume is reached to allow distributing one fluid particle around the entire periphery of the repository. Each such fluid particle would be poised at zero time to depart from the repository along its particular streamline, as

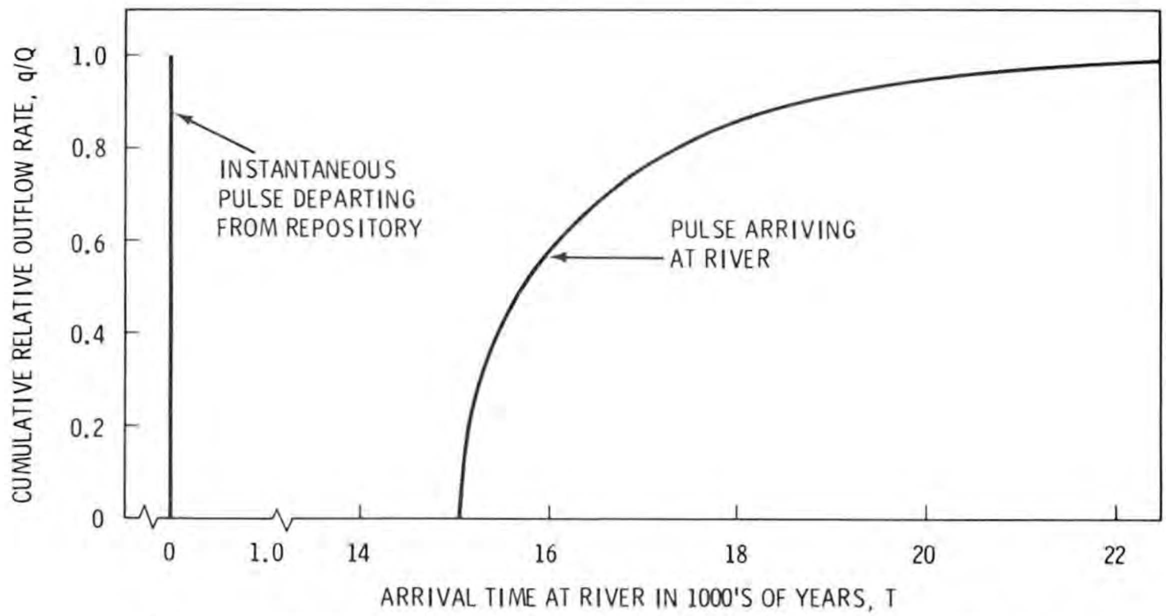

Fig. 2. Geohydrologic Response Function (Quantity/Time) for sample repository

represented by the vertical line input pulse in Figure 2. In other words, the pulse of height equal to unity and of infinitesimal width represents the line up of all the contaminated water particles, as each one starts along its particular streamline. Along any of the paths, each contaminated fluid particle encounters permeability variations that will affect the time when the fluid particle emerges from its streamline at the river [10].

The response function or right hand curve in Figure 2 is the time history of all the individual fluid particles emerging from their respective streamlines at the river. All of those factors causing changes in the system alter the time when the contaminated fluid particles arrive; hence, each effect changes the response function. From this nature of the response function springs the real utility of geohydrologic response functions.

The quantity/time geohydrologic response function provides the amount of contaminated fluid leaving the subsurface flow system, with time as a result of an instantaneous pulse input at time, t_o. Three items are important in this definition and its use:

- The response function gives the cumulative quantity of outflow as the overall system response to all the factors affecting the flow paths and interim delays in the subsurface system.

- Each response curve is the result of a specific infinitesimal volume leaving the contaminant source.
- For each successive infinitesimal volume of fluid leaving the repository there is a specific departure time, t_0; hence, for each t_0, there is an associated response or arrival curve.

The Second (Quantity/Location) Response Function

Although the quantity/time response function is of primary usefulness, an auxiliary response function is introduced here for those cases where the outflow location may be particularly important. Its origin is in interrelating the location where contaminated fluid leaves the subsurface system as related to the cumulative outflow rates.

From Figure 1 and Table 1, we note that associated with each pair of streamlines there is in general a cumulative flux ratio q/Q as a function of the outflow locations, S. The outflow locations are a double valued function of the cumulative flux because there are always two bounding streamlines associated with each cumulative relative flux. For example, in Figures 2 and 1, q/Q = 0.30, or rather, 30% of the cumulative outflow has emerged between the locations of S = -745 ft and S = +745 ft where Streamlines 4 and 20 enter the river, respectively.

The auxiliary response function, as shown in Figure 3, interrelates the quantity of contaminant outflow with the location where that contaminant leaves the subsurface flow system and enters the river. When used in connection with the principal response function, it is easy to determine where the contamination is entering the river for various contaminant inputs to the overall subsurface flow system.

USE OF GEOHYDROLOGIC RESPONSE FUNCTIONS

Response functions may be used for a variety of repository source terms. We begin by considering a steady constant release of contaminated wastes from the repository of 750 yr duration. In Figure 4, the reponse functions for the first and last parts of the constant release are shown. The first pulse input at time $t_0 = 0$ and the associated response curve to the left represents the first contaminated water departing from the repository and its later arrival at the river, respectively. The right side of the square input pulse departing the repository at $t_0 = 750$ yr and the associated right hand response curve represents the last of the $t_0 = 750$ yr pulse of contaminated water to reach the river. Accordingly, only the stippled area between the two response curves represents contaminated water entering the river. The complete result is the lower cross-hatched curve in Figure 4, which is the contaminated water outflow rate with time resulting from the 750-yr release at the repository.

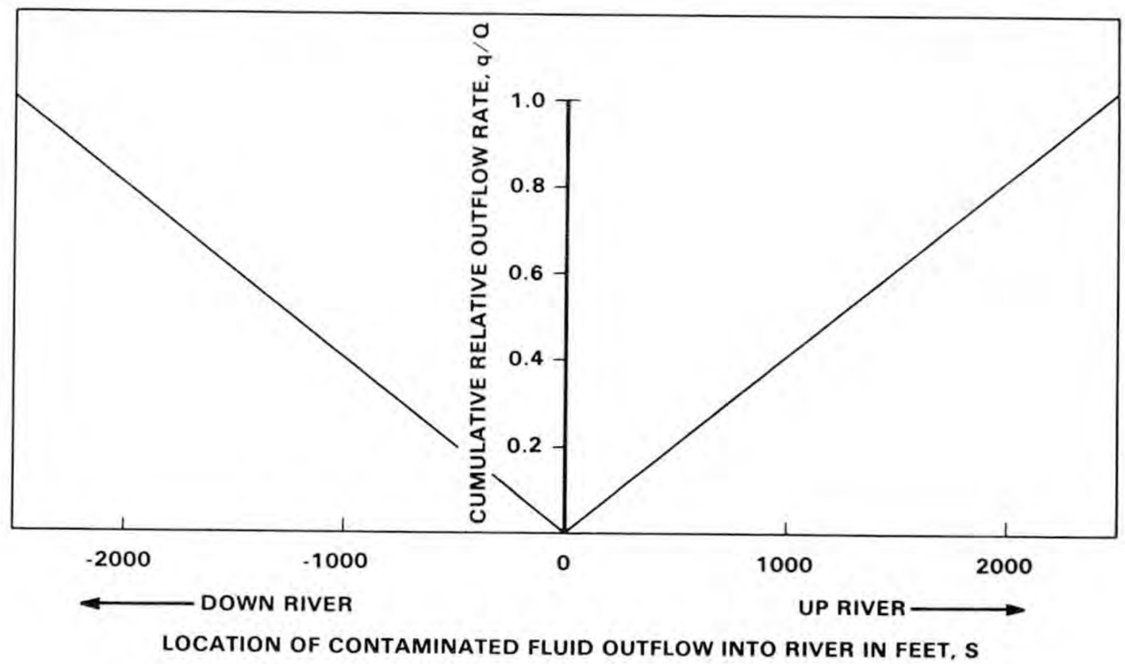

Fig. 3. Geohydrologic Response Function (Quantity/Location) for sample repository

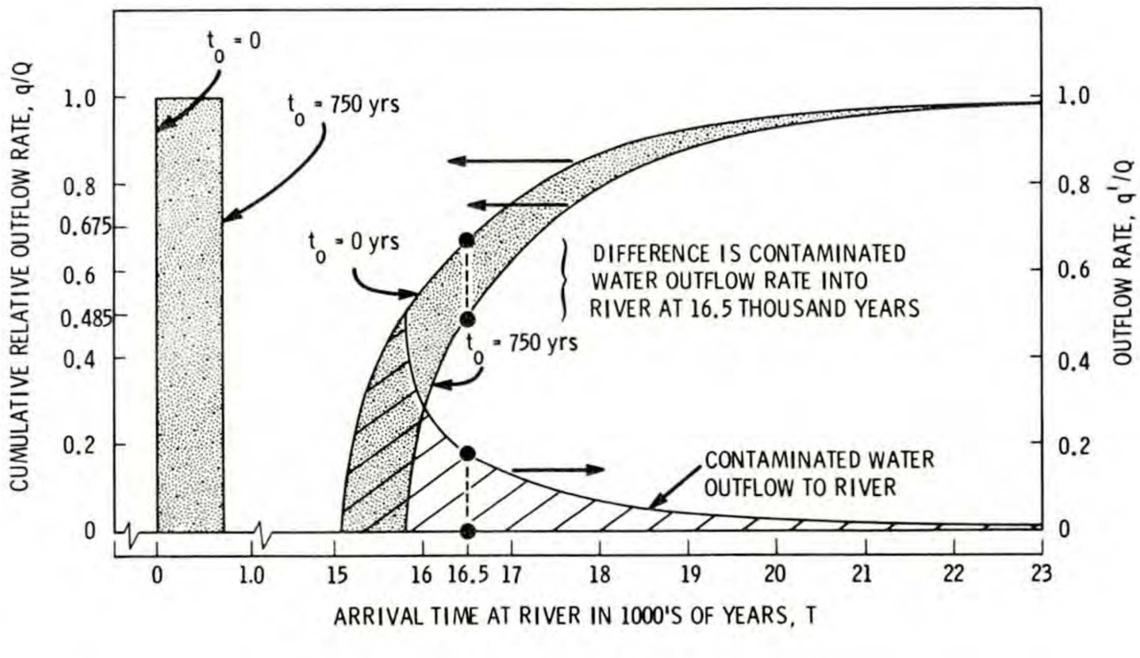

Fig. 4. Use of Quantity/Arrival Time Response Function to determine the release at the river from a constant contaminant input at the repository

The outflow/location response curve can be considered in Figure 4 in greater detail, which shows that at a time of 16.5 thousand yr, q/Q = 0.675 as observed from the upper response curve (t_o = 0), and the lower response curve (t_o = 750 years) is q/Q = 0.485. If these two values are transferred to Figure 5, which is a quantity outflow/location response curve like Figure 3, a number of interesting results are seen. From the upper q/Q = 0.675 value, Figure 5 shows two outflow locations along the river at S = ±1680 ft. From the lower response curve value of q/Q = 0.485, Figure 5 gives the lower contaminated water outflow locations of S = ±1200 ft. These results show that at 16.5 thousand yr, contaminated water is entering the river between locations -1680 and -1200 ft along the river bank and also between +1200 ft and +1680 ft along the river. At all locations between -1200 ft and +1200 ft uncontaminated water that departs from the repository after the 750-yr contaminated pulse is entering the river. Also between ±240 ft and ±1680 ft, respectively, uncontaminated water that departs from the repository before the 750-yr pulse is also entering the river. This example provides insight concerning the location and amount of contamination that finally enters the river.

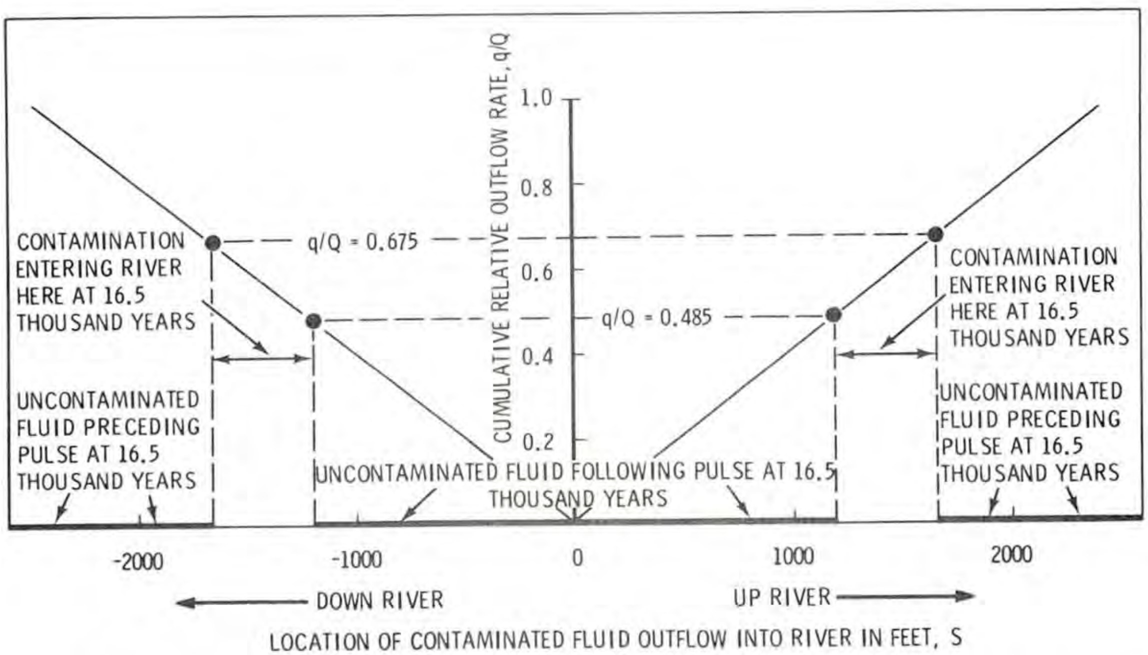

Fig. 5. Use of the Quantity/Location Response Function to determine the contaminant outflow location

REFERENCES

1. C. R. Cole, Radionuclide Transport Sensitivity Studies, PNL-SA-8927, Pacific Northwest Laboratory, Richland, Washington, (1980).

2. C. R. Cole, and F. W. Bond, Summary of Four Release Consequence Analyses for Hypothetical Nuclear Waste Repositories in Salt and Granite, PNL-3548, Pacific Northwest Laboratory, Richland, Washington, (1980).

3. J. R. Raymond, et al., Test Case Release Consequence Analysis for a Spent Fuel Repository in Bedded Salt, PNL-2782, Pacific Northwest Laboratory, Richland, Washington, (1980).

4. M. O. Cloninger, C. R. Cole, and J. F. Washburn, An Analysis of the Use of Engineered Barriers for Geologic Isolation of Spent Fuel in a Reference Salt Site Repository, PNL-3356, Pacific Northwest Laboratory, Richland, Washington, (1980).

5. R. W. Nelson, Evaluating The Environmental Consequences of Groundwater Contamination, BCSR-6/UC-11, BCS Richland, Washington, (1976).

6. R. W. Nelson, Evaluating the Envirnomental Consequences of Groundwater Contamination, 1, An Overview of Contaminant Arrival Distributions as General Evaluation Requirements, Water Resources Research, Vol. 14, No. 3, p. 409-415, (1978).

7. R. W. Nelson, Evaluating the Environmental Consequences of Groundwater Contamination, 2, Obtaining Locaiton/Arrival Time and Location/Outflow Quantity Distribution for Steady Flow Systems, Water Resources Rearch, Vol. 14, No. 3, p. 416-428, (1978).

8. R. W. Nelson, Evaluating the Environmental Consequences of Groundwater Contamination, 3, Obtaining Contaminant Arrival Distributions for Steady Flow in Heterogeneous Systems. Water Resources Research, Vol. 14, No. 3, p. 429-440, (1978).

9. R. W. Nelson, Evaluating the Environmental Consequences of Groundwater Contamination, 4, Obtaining and Utilizing Contaminant Arrival Distributions in Transient Flow Systems, Water Resources Research, Vol. 14, No. 3., p. 441-450, (1978).

10. R. W. Nelson, Use of Geohydrologic Response Functions in the Assessment of Deep Nuclear Waste Repositories, PNL-3817, Pacific Northwest Laboratory, Richland, Washington (May, 1981).

ANALYSIS OF FRACTURE FLOW AND TRANSPORT IN THE NEAR-FIELD OF A NUCLEAR WASTE REPOSITORY

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ABSTRACT

Numerical models were recently developed for use in modeling the near-field processes in fractured-porous rock systems. The theoretical framework of the models is based on a continuum description of rock mass and Darcian flow concepts. Groundwater flow is related to the rock stress state and thermal regime through the effects on hydraulic conductivity and buoyancy. Heat transport through the water-rock system is modeled in a coupled manner accounting for processes of convection, dispersion, conduction, and thermal energy sources. The variations in the rock stress state are related to the temperature patterns in the near-field zone. The complete set of model equations are solved using a finite element numerical method.

Parametric and sensitivity modeling studies have been performed in evaluating the waste isolation capability of a hypothetical repository in basalt media. Analysis of various hydrologic release scenarios has been performed. In this paper, simulation results are presented which estimate the time-dependent response of a basalt waste isolation system over a 30,000-year period. Calculations of particle pathlines and travel times are presented in conjunction with nuclide transport patterns.

INTRODUCTION

The Basalt Waste Isolation Project (BWIP) [1] under way at Rockwell Hanford Operations, is chartered with the investigation of basaltic rock as a candidate geology for construction of a nuclear waste repository. The Columbia River basalts which underlie a large portion of the Pacific Northwest are currently being characterized and studied to provide a comprehensive information base which will be used to assess the technical feasibility of a repository in basalt. The principal focus of the BWIP studies is on the basalt formations beneath the Hanford Site, near Richland, Washington, which is contained within the Pasco Basin. Accumulations of basalt in this area are particularly promising because of their extraordinary thickness, lateral extent, and relatively low permeability.

The BWIP hydrologic studies are divided into two major efforts, site characterization and repository performance assessment activities. Under the performance assessment category, various ongoing hydrologic modeling studies are being performed to: (1) develop a quantitative description of the large-scale flow system in the Pasco Basin and (2) evaluate the basic waste isolation characteristics of candidate sites in the deep basalts. The latter aspect entails the application of hydrologic and transport models to the "very near-field" (canister to room scale) and "near-field" (room to repository scale) regions. Computer simulation studies of both natural and disruptive event conditions are being performed to estimate and bound the potential waste isolation capability of a reference repository site.

Recognizing that future technical decisions regarding repository design and site selection may place much reliance on model predictions, consideration of uncertain elements in the modeling process is of fundamental but key importance. For the most part, the uncertainty in model predictions can be attributed to four major sources:

1. Limitations in the mathematical theories which describe hydrologic and transport processes

2. Random and systematic errors in field measurements of hydrologic properties

3. Errors arising from subjective interpretations of the spatial variations of hydrologic parameters from discrete data points

4. Incompleteness of geohydrologic characterization.

The first source, which may be termed "model uncertainty," can be addressed on a limited scale by performing detailed comparisons between numerical simulations and experimental data; these results, in turn, can be analyzed to determine the degree of correlation between measurement and calculation, i.e., model validation [2]. The other three sources, which represent "data uncertainty", can be evaluated using a number of approaches. McLaughlin [3] has reviewed various statistical techniques which estimate the impact of uncertainty elements, given a probabilistic description of the uncertain model input, i.e., a probability density function for each hydrologic parameter. The last two elements can also be grouped into a "descriptive uncertainty" category which is, perhaps, the most difficult to analyze in a rigorous fashion; Kriging techniques [4,5] in combination with a systematic scenario analysis may provide a pragmatic approach to: (1) developing continuous representations of hydrologic data with uncertainty bounds and (2) evaluating hydrologic significance of possible undetected geologic features.

The large quantity of measured data required for a rigorous uncertainty analysis, however, appears to be a major obstacle to their application to diverse geohydrologic systems. This indication is further reinforced by the simple fact that a candidate site may be characterized to a limited degree to assure that natural barriers are not disturbed or compromised. An alternative approach to the problem of addressing predictive uncertainty is to adopt a systematic and conservative methodology which "compensates" for uncertain elements in the modeling process. Such a methodology should provide a framework for guiding the system simulations so that bounding estimates of nuclide migration are obtained.

A methodology based on these concepts is currently being used in the BWIP hydrologic modeling studies. The principal components of this analysis approach include:

1. Simulation models for coupled heat, groundwater flow, and nuclide transport in fractured-porous media

2. Parametric and sensitivity analysis of postulated release scenarios

3. Decision- or logic-tree strategy to guide parametric studies.

The purpose of this paper is to present and discuss selected results from recent modeling studies for a reference repository site in the Columbia River basalts.

STATEMENT OF MODELING PROBLEM

Analysis Framework

A detailed analysis of waste isolation in a hardrock geology requires the consideration of three major types of phenomena: Heat transfer, groundwater flow, and nuclide transport. The extent of coupling and interdependence between these processes is important and depends on the physical scale and location of the analysis region. For example, within a relatively small region around a deep geologic repository, the rock medium will exhibit a behavior distinct from the overall geohydrologic system by virtue of the physical and thermal perturbations created by repository conditions. Such a region is referred to here as the "near-field" zone, i.e., repository backfill, room pillars, and disturbed rock zone.

To provide a logical and systematic framework for parametric and sensitivity analysis of release scenarios, the so-called decision-tree [6] approach is used. A decision tree consists of a set of subtrees, each of which is made up of "branches" representing parameter variations and "decision nodes". By properly connecting the subtrees, a pictorial representation is constructed which identifies sets of parameter combinations which lead to lower bound, nominal, and upper bound predictions. By virtue of the couplings between the physical processes, the subtrees can be ordered for parameter variations in the following categories: (1) thermal, (2) hydrologic, (3) radionuclide, and (4) barrier longevity properties. This simple approach eliminates the need to simulate a large number of cases.

Geohydrologic Features of Basalt System

The Pasco Basin, which extends over an area of 5,180 square kilometers within the Columbia Plateau, is underlain by a bedrock section of at least 1,460 meters in thickness. These vast accumulations of basalt, referred to as the Columbia River Basalt Group, are overlain by up to 220 meters of sediment material. The Columbia River Basalt Group is subclassified into five major formations: the Saddle Mountains, Wanapum, Grande Ronde, Imnaha, and Picture Gorge basalts. Of these five, the first three formations underlie the Pasco Basin. In the Saddle Mountains and Wanapum, several sedimentary interbeds and flow-top breccias exist which are major water-bearing zones or aquifers. In the Grande Ronde formation groundwater is much less abundant, with most water-bearing zones occurring along distinct interflow zones, e.g., flow tops and flow contacts.

In formulating a "geohydrologic conceptual model" for a reference repository site, available stratigraphic information [7] is used from four deep boreholes which are located on the Hanford Site. For the purpose of this analysis, the model study area was restricted to the Grande Ronde Formation, located some 700 meters below the ground surface. The four boreholes define a vertical cross section through the basalts which parallels the actual siting area. Interpretations of data for the Grande Ronde suggest a rather complex stratigraphy which may be grouped into nineteen major rock layers.

Available hydrologic data for these and other boreholes have been compiled by various organizations. The comprehensive report by [8] summarizes much of the published field data and hydrologic interpretations compiled to date. In specifying various properties for the 20 rock types, nominal values for hydraulic conductivities, porosities, and storage coefficients were used. With regard to hydraulic conductivities, anisotropic ratios (K_{zz}/K_{xx}) of 10.0 and 1.0 were assumed for the dense basalt and interflow zones, respectively. Specific values used in the analysis may be found in Arnett et al. (1980) [9].

REPOSITORY FEATURES

For this analysis, a reference repository site is assumed to be located in the middle of the Umtanum flow of the Grande Ronde Formation at a nominal depth of 1,000 meters below ground surface. A proposed physical layout of the repository consists of 22 panels distributed over an area bounded by the dimensions of: length 3,000 meters, width 2,400 meters, and room height 6.5 meters. The room backfill in the repository is assumed to consist of a bentonite/clay mixture.

The spent-fuel inventory in the repository is assumed to be one-half of the year 2000 projection for the U.S. commercial nuclear industry, which is approximately 47,000 metric tons of heavy metals. For a particular radionuclide such as ^{129}I, the initial inventory at closure would be 9,200 kilograms. For the heat generation rate used, 10-year-old spent fuel was assumed. The peak heat rate was assumed to be 16 W/m^2 (65 kW/acre).

NUMERICAL MODELING APPROACH

Governing Equations

A general set of governing equations applicable to transport processes in a fractured-porous medium can be derived from the basic conservation laws and Darcian flow concepts. For a nonisothermal case, the Darcy flow equation can be written in terms of hydraulic and buoyancy driving forces [10], namely:

$$\bar{q} = - \bar{\bar{K}} \quad (\nabla h + \delta_b \nabla z) \tag{1}$$

where

$\bar{q}$ = Darcian velocity vector (m/sec)

$\bar{\bar{K}}$ = hydraulic conductivity tensor (m/sec)

h = hydraulic head (m)

δ_b = density disparity (unitless)

z = vertical coordinate (m)

The density disparity term is a function of fluid density and is given by the equation:

$$\delta_b = \frac{\rho}{\rho_o} - 1 \tag{2}$$

where ρ is the fluid density (kg/m^3), which is temperature dependent, and ρ_0 is the fluid density (kg/m^3) at the initial or reference temperature.

Fluid continuity for a double porosity system is expressed by two equations which describe the processes of flow through the fractures, storage in the rock matrix, and exchanges between fractures and rock matrix. The fluid flow equations are coupled to the heat transport equation which is derived by applying the principle for conservation of thermal energy in a water-rock system. The general mass transport equation is derived from considerations of mass continuity for a multicomponent system, accounting for chain decay. The specific governing equations for coupled groundwater flow, heat transport, and nuclide transport are presented in Baca et al. (1980) [11].

Numerical Models

The general solutions of the governing equations are obtained using two numerical models. The first model, referred to as MAGNUM, solves the fully coupled equations of groundwater flow and heat transport, given a set of initial and boundary conditions. Groundwater velocities computed by MAGNUM are stored for later input to the nuclide transport model CHAINT. Given a specified waste inventory and a release period, the CHAINT model solves the mass transport equations to provide a simulation of the migration and transformation of radionuclides.

Both numerical models are based on a Galerkin finite element technique in conjunction with a Newton-Raphson algorithm [12]. Continuum portions of the rock mass are represented using two-dimensional triangular and quadrilateral elements. Discrete features such as large-scale fractures are modeled with one-dimensional line elements which are embedded along the sides of the two-dimensional elements. Basic features of these numerical models include:

- Accommodate complex stratigraphic features and variable media properties.
- Spatial approximations are based on quadratic shape functions and isoparametric finite elements.
- Time integration is based on variable order implicit technique. Numerical codes provide options for simultaneous or sequential solution of the governing equations.
- Accommodate any combination of single or multicomponent sets of nuclides, i.e., single components such as activation or fission products and diverse multicomponent actinide decay chains.

The MAGNUM and CHAINT computer codes have been extensively verified with various boundary value problems and by benchmark tests with other existing computer codes [13,14]. Validation studies are currently planned using available data from laboratory experiments involving natural convection in porous media.

MODEL APPLICATIONS

As part of repository performance assessments, various numerical modeling studies are under way at Rockwell to quantify the waste isolation effectiveness of a reference site in the Columbia River basalts. Natural (nonperturbed) as well as disruptive event conditions are being considered to evaluate potential waste migration. Representative results for a base case scenario are presented here.

Base Case Scenario

One intuitively expects that, after tens of thousands of years or longer, groundwater ingress to the repository will eventually produce degradation of the engineered barriers, gradual leaching of the waste form, and migration of dissolved radiocontaminants. The significance of this waste release, however, depends on the rate and extent of nuclide migration over the required waste isolation period, e.g., 10,000 years after closure. Indeed, if the waste migration over this period is confined to the host formation, the basic objective of the repository will have been achieved and the radiologic risk posed to man will be insignificant.

In defining the base case scenario, the following set of conservative assumptions are made:

- The repository contains spent-fuel inventory equivalent to one-half the year 2000 projection for nuclear power industry.
- Groundwater completely fills the repository immediately after closure.
- Loss of waste package and engineered barrier integrity occurs immediately after closure.
- The waste inventory is released at a constant rate over a 10,000-year period.

With these assumptions, the numerical models were applied to calculate basic performance assessment parameters, namely, groundwater pathlines from the repository, groundwater and solute travel times, and contaminant distributions for key radionuclides.

Pathlines and Travel Times

Pathlines, computed from the groundwater pore velocities, indicate the trajectories of particles moving with the groundwater. The cumulative time of travel along individual pathlines is the associated travel times; time lines, which are lines connecting points of equal travel time, depict the relative location of particles at different times. These quantities were computed for the base case scenario and are presented in Figure 1. These results show that the groundwater flow above the repository is predominantly upwards because of the buoyancy effects; the hydraulic gradients in the vertical direction, which are significantly greater than those assumed for the horizontal, also contribute to the upward flow direction. The separations between flow fronts or time lines indicate that these bouyancy effects are particularly dominant during the peak thermal period, i.e., the first 500 years. After 30,000 years, all the particles leaving the repository are still contained within the Grande Ronde Formation.

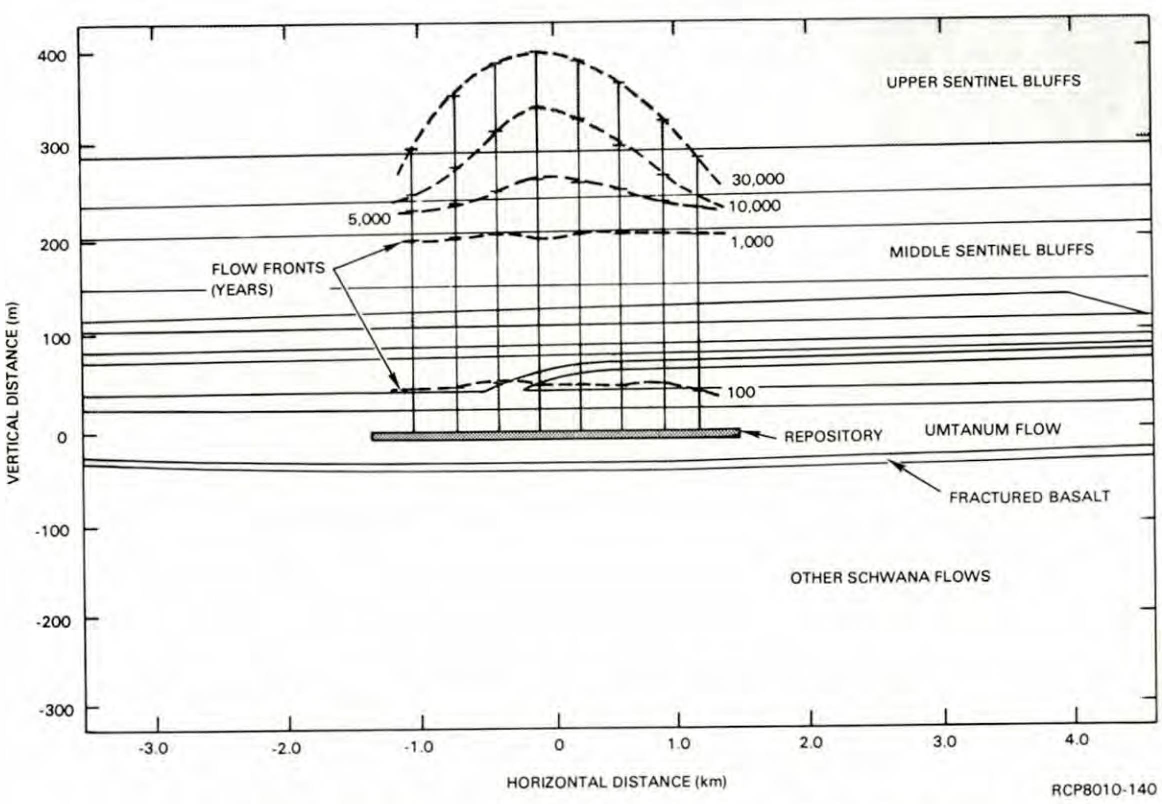

FIGURE 1. Pathline and Travel Time Calculations for Baseline Conditions Within the Grande Ronde Formation.

Waste Migration Simulations

The postulated release, over the 10,000-year release period, creates a contaminant plume which migrates through the rock mass. For mobile nuclides, advective and dispersive/diffusive transport processes determine the rate and extent of plume movement. The base case simulation results for a reference repository in basalt are shown in Figure 2, which consists of the contaminant plumes for ^{129}I at various time planes. This nuclide is a key indicator of potential waste migration by virtue of its large initial inventory in spent fuel, long half-life, and high mobility; the observed range of sorption (K_d) values is 0 to 3 milliliters per gram [15]; a K_d value of 2 milliliters per gram was assumed for the basalt and a value of 0 for the repository.

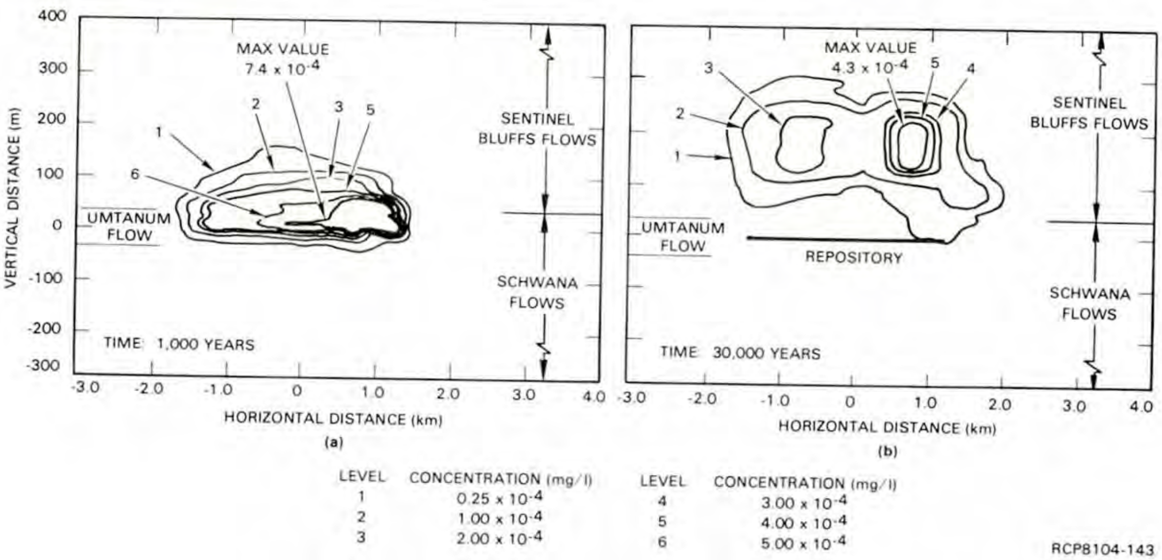

FIGURE 2. Iodine-129 Contours for Baseline Conditions Within the Grande Ronde Formation.

In this base case simulation, conservative values for hydraulic gradients and boundary conditions were assumed so as to emphasize migration of the nuclides along the shortest path to the accessible environment, i.e., upwards to pervious aquifers in the upper formations. These assumptions, in combination with buoyancy effects, produce a displacement of the contaminant plume from the repository. At the end of 30,000 years the ^{129}I plume is still confined to the Grande Ronde Formation and the peak concentrations are near the maximum permissible concentrations.

CONCLUSIONS

Comprehensive parametric and sensitivity analyses have been performed to evaluate the waste isolation effectiveness of a reference site in basalt. A decision-tree approach, used in conjunction with simulation models for fractured-porous media, has provided a systematic and conservative analysis of repository performance in a hardrock geology. Principal findings of the parametric and sensitivity studies conducted to date are:

- The most sensitive model parameters are hydraulic conductivity and sorption coefficients.
- Buoyancy effects significantly influence groundwater flow and nuclide transport patterns.
- Heat transfer through the water-rock system is dominated by conduction through the rockmass.

Overall, the basic conclusion of this analysis is that a basaltic rock geology, such as the Columbia River basalts, can provide a high degree of waste isolation. This conclusion is in line with intuitive expectations, by virtue of the basic features of the geohydrologic system: (1) vast rock strata with relatively low permeability, (2) absence of major water-bearing zones in the Grande Ronde Formation, (3) large distance from major aquifers (and the accessible environment), and (4) low solubility of the waste form in the anoxic groundwater environment in the Columbia River basalts [15].

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REFERENCES

1. Basalt Waste Isolation Project staff, The Basalt Storage Program, RHO-SA-11, Rockwell Hanford Operations, Richland, Washington, 1977.

2. R. E. Shannon, Systems Simulation - The Art and Science, Prentice Hall, Inc., Englewood Cliffs, New Jersey, 1975.

3. D. B. McLaughlin, Hanford Groundwater Modeling Statistical Methods for Evaluating Uncertainty and Assessing Sampling Effectiveness, RHO-C-18, Rockwell Hanford Operations, Richland, Washington, 1979.

4. J. P. Delhomme, Kriging in Hydroscience, Ph.D. Thesis, Centre D'Information Geologique, École National Superieure Des Mines de Paris (University of Paris), 1976.

5. P. G. Doctor, An Evaluation of Kriging Techniques for High-Level Radioactive Waste Repository Site Characterization, PNL-2903, Pacific Northwest Laboratories, Richland, Washington, 1979.

6. H. Raiffa, Decision Analysis: Introductory Lectures on Choices Under Uncertainty, Addison-Wesley, New York, New York, 1968.

7. C. W. Myers et al., Geologic Studies of the Columbia Plateau: A Status Report, RHO-BWI-ST-4, Rockwell Hanford Operations, Richland, Washington, 1979.

8. R. E. Gephart et al., Hydrologic Studies Within the Columbia Plateau, Washington: An Integration of Current Knowledge, RHO-BWI-ST-5, Rockwell Hanford Operations, Richland, Washington, 1979.

9. R. C. Arnett et al., Preliminary Hydrologic Release Scenarios for a Candidate Repository Site in the Columbia River Basalts, RHO-BWI-ST-12, Rockwell Hanford Operations, Richland, Washington, 1980.

10. J. Bear, Dynamics of Fluids in Porous Media, American Elsevier Publishing Co., New York, New York, 1972.

11. R. G. Baca, J. B. Case, and J. B. Patricio, Coupled Geomechanical/Hydrological Modeling: An Overview of Basalt Waste Isolation Project Studies, RHO-BWI-SA-82, Rockwell Hanford Operations, Richland, Washington, 1980.

12. R. G. Baca, R. C. Arnett, and I. P. King, Numerical Modeling of Flow and Transport Processes in a Fractured - Porous Rock System, RHO-BWI-SA-113, Rockwell Hanford Operatons, Richland, Washington, 1981.

13. A. K. Runchal, J. Treger, and G. Segal, Two-Dimensional Fluid Flow, Heat, and Mass Transport in Porous Media, Technical Note TN-LA-34, Dames and Moore, Los Angeles, California, 1979.

14. R. T. Dillon, R. B. Lantz, and S. B. Pahwa, A Model for Calculating Radioactive Waste Migration, the Sandia Waste Isolation Flow and Transport (SWIFT) Model, SAND 78-L1267, Sandia Laboratories, Albuquerque, New Mexico, 1978.

15. B. J. Wood, Estimation of Waste Package Performance Requirements for a Nuclear Waste Repository in Basalt, RHO-BWI-ST-10, Rockwell Hanford Operations, Richland, Washington, 1980.

Session V:

RISK ASSESSMENT — THE EPA PERSPECTIVE

Chairman

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Pacific Northwest Laboratory

MODELS OF RELEASE MECHANISMS FOR CONCEPTUAL NUCLEAR WASTE REPOSITORIES

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(Paper Not Submitted)

MATHEMATICAL MODELS FOR ESTIMATING POPULATION HEALTH EFFECTS FROM THE DISPOSAL OF HIGH-LEVEL RADIOACTIVE WASTE

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ABSTRACT

Mathematical models have been developed to estimate population dose equivalents and health effects due to releases of radionuclides from a high-level waste repository. The mathematical models consider releases to a river system, an ocean system, and directly to land surfaces. For the river and ocean systems, the time-dependent equation used to predict releases from the repository to the river or ocean was controlled by the rate of leaching of radionuclides from the waste into groundwater. Releases during violent geologic interactions with the repository, such as would occur during a volcanic eruption or a meteorite impact, were also considered. For releases directly to the land surface and releases during violent geologic interactions, an instantaneous release of radionuclides was assumed. The equations consider environmental transport of radionuclides in air and in water and cover both internal and external exposure pathways.

INTRODUCTION

The Environmental Protection Agency is developing high-level radioactive waste disposal standards (40 CFR 191) to limit the public health impact from waste disposal activities [1]. Mathematical models have been used by EPA [2] to estimate population dose equivalents and health effects (fatal cancers and genetic effects to the first generation descendents of the irradiated parent or parents). These models have been used to determine the permissible amounts of radionuclides which can be released from a high-level waste repository for inclusion in the EPA high-level waste standard.

The models were developed for generic calculations for high-level waste repositories. The environmental dose equivalent commitments (EDC's) [3] and calculated health effects are long-range estimates and, in some cases, simplifying assumptions were made to the procedures which would normally be used in typical analyses of specific nuclear industry sites over relatively short periods of time. For example, to estimate air concentrations around a small ground-level source of radionuclides we assume that the wind blows equally in all directions with an average wind speed rather than using a more specific joint frequency distribution of wind direction and speed.

Four different release modes are considered: river, ocean, land surface, and air (violent releases). The first three modes are considered to encompass events which we would expect to happen during the useful life of a repository. The fourth mode, violent release to air, encompasses unlikely events such as a volcano or a meteorite interaction with a repository. The release models will be discussed briefly along with their respective environmental transport pathways.

REPOSITORY RELEASE MODELS

River Source Terms

For the river pathway, we assume that the repository is breached after an initial containment period so that groundwater can circulate through the repository, dissolve some high-level waste, and carry radionuclides into the surrounding area and eventually to an aquifer. In some scenarios, the release from the repository is limited by the leaching coefficient for each nuclide; in others, the release of some nuclides from the repository is limited by their solubility in groundwater. The models in this paper are restricted to cases where leaching rate controls the releases.

Radionuclides are released to an aquifer that flows underground until it intersects a river and is diluted into the river flow. The equation for the rate of entry of radionuclides to the river is

$$Q'_{np}(t) = \lambda_{Ln} f_L Q_{on} \exp\left[-\lambda_{Dn} t - \lambda_{Ln} (t - t_{Rn})\right] \qquad \text{for} \quad t \geq t_{Rn} \tag{1}$$

where

$Q_{np}(t)$ = rate of entry of radionuclide n into the river or ocean for pathway p (Bq/y),

λ_{Ln} = leaching rate constant for repository (y^{-1}),

f_L = fraction of repository which is being leached (dimensionless),

Q_{on} = initial inventory of radionuclide n in the repository (Bq),

λ_{Dn} = radioactive decay constant for nuclide n (y^{-1}),

t = time after placement in repository at which EDC is calculated (y), and

t_{Rn} = time after placement in the repository that radionuclide n enters the river (y).

The total amount of a radionuclide that has entered the river to time "t" is then obtained by integrating the rate equation over time to obtain

$$Q_{np}(t) = \frac{f_L \lambda_{Ln} Q_{on}}{\lambda_{Dn} + \lambda_{Ln}} \left[\exp[-\lambda_{Dn} t_{Rn}] - \exp[\lambda_{Ln} t_{Rn} - (\lambda_{Dn} + \lambda_{Ln}) t] \right] \quad \text{for } t \geq t_{Rn} \qquad (2)$$

where

$Q_{np}(t)$ = total release of radionuclide n to the environment for pathway p (Bq).

Ocean Source Terms

For the ocean pathway, we assume that all radionuclides released from a waste disposal facility reach the ocean after transport through a river system. Travel time in the river to the ocean is assumed to be negligible, and depletion of radionuclides in the river due to removal by irrigation and sedimentation is not considered. Based on these assumptions, the source terms for the ocean pathway are the same as those for the river pathway.

Land Surface Source Terms

For the river and ocean pathways, we assume continuing releases of radionuclides to the biosphere after leakage begins. But, for the land surface pathway, we assume instantaneous release of radionuclides, and, to calculate resuspension from ground to air, we assume the release to originate from a point source. For each event considered under the land surface pathway, a fraction of the radionuclides remaining in the repository at the time of release is assumed to be brought to the ground surface and available for resuspension and subsequent redistribution in the environs.

Air Release Source Terms

Only a violent disruption of the waste repository, such as a volcano or a meteorite, can cause materials to be widely dispersed from the repository to air. Due to the violent nature of the release, we assume that the material would be quickly dispersed upward into the air and, eventually, distributed uniformly in the troposphere. The airborne material is divided into the fraction over land and the fraction over water using the ratio of earth land surface area to total earth area and earth water surface area to total earth area. This division of the released material simplifies the calculation, as will be discussed later.

TRANSPORT WITHIN ENVIRONMENT AND POPULATION DOSE EQUIVALENT AND HEALTH EFFECT ESTIMATES

The transport of radionuclides within the biosphere and the mechanism for delivering radiation dose equivalents to humans are evaluated for each of the four release modes. The pathways considered are shown in Table 1.

Releases to a River

The radionuclides are assumed to be diluted immediately upon entering the river. Thus we can express the concentration of nuclides in the river at any time t as

$$WC_{np} = \frac{Q'_{np}}{R} \tag{3}$$

where

WC_{np} = river water concentration of radionuclide n for use with pathway p (Bq/l), and

R = river flow rate (l/y).

This river water concentration is used to compute the environmental dose equivalent commitment and the resulting health effects for a population exposed through the pathways shown in Table 1. The equations for computing environmental dose equivalent commitments for the river release mode will be discussed for each environmental pathway.

Table 1. Environmental Pathways Considered for the Four Release Modes

Environmental Pathway	Releases to a River	Releases to an Ocean	Releases to Land Surfaces	Releases to Air
Drinking water	X			
Ingestion of fish and/or shellfish	X	X		X
Ingestion of above surface foodcrops	X		X	X
Milk ingestion	X		X	X
Beef ingestion	X		X	X
Inhalation	X		X	X
External exposure from air submersion	X		X	X
External exposure from ground contamination	X		X	X

Drinking water. The population receives drinking water from the river with no reduction in radionuclide concentration due to water treatment. The integrated river water concentrations are needed for the drinking water calculations and are calculated by integrating Eq. 3 over time to obtain

$$IC_{np} = \frac{Q_{np}}{R} = \frac{1}{R} \int_0^t Q'_{np}(t')dt' \qquad (4)$$

where

IC_{np} = integrated radionuclide concentration in river water (Bq-y/l).

The environmental dose equivalent commitment to the population from drinking water is obtained by multiplying this integrated water concentration by an intake rate for man, by a dose equivalent conversion factor, and by the population size. The equation is

$$S_{nop} = \left(\frac{Q_{np}}{R}\right) \cdot I_w \cdot D_{nop} \cdot P_R \tag{5}$$

where

S_{nop} = environmental dose equivalent commitment integrated to time t for nuclide n, organ o, and pathway p (person-Sv),

I_w = annual individual water ingestion rate (l/y),

D_{nop} = dose equivalent conversion factor for nuclide n, organ o, and pathway p (Sv/Bq intake), and

P_R = population drinking water from river (persons).

Freshwater fish ingestion. The equation for determining environmental dose equivalent commitment is

$$S_{nop} = \left(\frac{Q_{np}}{R}\right) \cdot CF_{np} \cdot I_f \cdot D_{nop} \cdot P_{FF} \tag{6}$$

where

CF_{np} = bioaccumulation factor for fish or shellfish for nuclide n and pathway p (Bq/kg fish per Bq/l water),

I_f = freshwater fish individual annual consumption rate (kg/y), and

P_{FF} = population eating freshwater fish from the river (persons).

Ingestion of food raised on irrigated land. By spray irrigation, river water containing radionuclides from the repository is deposited directly onto the crops and the land surface below the crops. Radionuclides reach plants both through their leaves and through the root systems. It is assumed that some of the irrigated plants are consumed by humans as food and the rest are consumed by either dairy or beef cattle, with transfer of the radionuclides to milk and meat which are consumed by humans. The equation used to compute the environmental dose equivalent commitments is

$$S_{nop} = \left(\frac{Q_{np}}{R}\right) \cdot W \cdot f_p \cdot RI_{np} \cdot D_{nop} \cdot CP_p \cdot A \qquad (7)$$

where

W = irrigation rate for crop land ($1/m^2$-y),

f_p = fraction of irrigated land used for various food crops for pathway p (dimensionless),

RI_{np} = intake of nuclide n by standard man for crop p and for a unit total deposition to the surface (Bq intake per Bq/m^2 deposited on soil surface),

CP_p = number of standard men who can be fed per unit area of land (persons fed/m^2), and

A = irrigation area (m^2).

In an area where ditch irrigation is employed, the parameter values used for RI_{np} would need to be derived for deposition of radionuclides only onto soil with no deposition directly to the plant foliage.

Inhalation of Resuspended Material. Some of the radioactive material placed onto soil by irrigation of farmland is resuspended into air and can be inhaled by people. To account for this we determined the air concentration of radionuclides by multiplying the ground surface concentration by a resuspension factor. This method is reasonable because the contaminated surface (the irrigated land) is large. The time-dependent soil surface concentration is obtained from the differential equation

$$\phi_n'(t) = \left(\frac{Q_{np}'(t)}{R}\right) \cdot W - (\lambda_{Dn} + \lambda_{sn}) \cdot \phi_n(t) \qquad (8)$$

where

$\phi_n'(t)$ = rate of change of ground surface concentration for radionuclide n (Bq/m^2-y),

λ_{sn} = rate constant for transfer of nuclide n from available to unavailable soil (y^{-1}), and

$\phi_n(t)$ = ground surface concentration of radionuclide n as a function of time (Bq/m^2).

In this equation, the assumed equilibrium between the resuspended material and its redeposition cancel in writing Eq. 8. It can be shown that this assumption is conservative. The source term equation, Eq. 1, is used to solve this differential equation:

$$\phi_n(t) = \frac{\lambda_{Ln} \cdot f_L \cdot Q_{on} \cdot W}{R(\lambda_{Ln} - \lambda_{sn})} \left\{ \begin{array}{l} \exp[\lambda_{sn} t_{Rn} - (\lambda_{Dn} + \lambda_{sn})t] \\ -\exp[\lambda_{Ln} t_{Rn} - (\lambda_{Dn} + \lambda_{Ln})t] \end{array} \right\} \tag{9}$$

The time-dependent air concentration due to resuspension is

$$\chi_{Rn}(t) = RF \cdot \phi_n(t) \tag{10}$$

where

$\chi_{Rn}(t)$= air concentration of radionuclide n due to resuspension from the ground surface (Bq/m^3), and

RF = resuspension factor measured at the center of a large, uniformly contaminated area (m^{-1}).

and the environmental dose equivalent commitment can be computed using the equation

$$S_{nop} = \int_0^t \chi_{Rn}(t')dt' \cdot I_B \cdot D_{nop} \cdot PD_p \cdot A \tag{11}$$

where

I_B = breathing rate for standard man (m^3/y), and

PD_p = population density applicable for pathway p ($persons/m^2$).

When the expression for $\chi_{Rn}(t)$ is substituted and the integration performed, the result is

$$S_{nop} = \frac{\begin{bmatrix} RF \cdot \lambda_{Ln} \cdot f_L \cdot Q_{on} \cdot W \\ \cdot I_B \cdot D_{nop} \cdot PD_p \cdot A \end{bmatrix}}{R(\lambda_{Ln} - \lambda_{sn})} \left\{ \begin{matrix} \frac{\exp(\lambda_{Ln} t_{Rn})}{(\lambda_{Dn} + \lambda_{Ln})} \left[\exp[-(\lambda_{Dn} + \lambda_{Ln})t] - 1 \right] \\ - \frac{\exp(\lambda_{sn} t_{Rn})}{(\lambda_{Dn} + \lambda_{sn})} \left[\exp[-(\lambda_{Dn} + \lambda_{sn})t] - 1 \right] \end{matrix} \right\} \tag{12}$$

<u>External Dose Equivalent from Air Submersion</u>. The radioactive material resuspended into air exposes the population by submersion. To estimate the submersion population dose equivalent, for each organ, we used the integrated air concentration equation discussed for the inhalation pathway. The environmental dose equivalent commitment is computed using the equation

$$S_{nop} = \int_0^t \chi_{Rn}(t')dt' \cdot [SOF \cdot D_{nop}] [PD_p \cdot A] \tag{13}$$

where

SOF = household shielding and occupancy factor (dimensionless), and

D_{nop} = dose equivalent conversion factor for nuclide n, organ o, and pathway p (Sv/y per Bq/m^3).

When the equation for $\chi_{Rn}(t)$ is substituted and the integration performed, the solution is

$$S_{nop} = \frac{\begin{bmatrix} RF \cdot \lambda_{Ln} \cdot f_L \cdot Q_{on} \cdot W \\ \cdot SOF \cdot D_{nop} \cdot PD_p \cdot A \end{bmatrix}}{R(\lambda_{Ln} - \lambda_{sn})} \left\{ \begin{matrix} \frac{\exp(\lambda_{Ln} t_{Rn})}{(\lambda_{Dn} + \lambda_{Ln})} \left[\exp[-(\lambda_{Dn} + \lambda_{Ln})t] - 1 \right] \\ - \frac{\exp(\lambda_{sn} t_{Rn})}{(\lambda_{Dn} + \lambda_{sn})} \left[\exp[-(\lambda_{Dn} + \lambda_{sn})t] - 1 \right] \end{matrix} \right\} \tag{14}$$

External Dose Equivalent from Ground Contamination. The radioactive material deposited on the ground during irrigation results in external dose equivalents to persons in the area. The dose equivalent to the population receiving this external exposure is computed, for each organ, using the integrated form of the soil surface concentration discussed as part of the inhalation pathway. The environmental dose equivalent commitment is determined from the equation

$$S_{nop} = \int_o^t \phi_n(t')dt' \cdot [D_{nop} \cdot SOF] [PD_p \cdot A] \tag{15}$$

where

D_{nop} = dose equivalent conversion factor for nuclide n, organ o, and pathway p (Sv/y per Bq/m^2).

Substituting for the ground surface concentration of radionuclides (Eq. 9), and performing the integration yields

$$S_{nop} = \frac{\begin{bmatrix} \lambda_{Ln} \cdot f_L \cdot Q_{on} \cdot W \\ \cdot D_{nop} \cdot SOF \cdot PD_p \cdot A \end{bmatrix}}{R(\lambda_{Ln} - \lambda_{sn})} \left\{ \begin{array}{l} \dfrac{\exp(\lambda_{Ln} t_{Rn})}{(\lambda_{Dn} + \lambda_{Ln})} \left[\exp[-(\lambda_{Dn} + \lambda_{Ln})t] -1 \right] \\ - \dfrac{\exp(\lambda_{sn} t_{Rn})}{(\lambda_{Dn} + \lambda_{sn})} \left[\exp[-(\lambda_{Dn} + \lambda_{sn})t] -1 \right] \end{array} \right\} \tag{16}$$

Health Effects for Releases to a River. Health effect conversion factors ($HECON_o$) are applied to the environmental dose equivalent commitments (S_{nop}) to estimate fatal cancers (FHE) and genetic effects (GE). Summations are performed over organs and pathways to yield the total fatal cancers or total genetic effects for a particular radionuclide. The equation used to estimate fatal cancers is

$$FHE = \sum_{o=1}^{8} \sum_{p=1}^{8} S_{nop} \cdot HECON_o \tag{17}$$

and for first-generation genetic effects is

$$GE = \sum_{o=9}^{10} \sum_{p=1}^{8} S_{nop} \cdot HECON_o \tag{18}$$

where the summation over pathways (p) extends over the eight pathways considered for the river release mode. The organs used to compute fatal cancers are bone, red marrow, lung, liver, GI-LLI, thyroid, kidney, and other soft tissue. The organs used to compute first-generation genetic effects are ovaries and testes.

Other Release Modes

The details of the environmental dose equivalent commitment equations for each environmental pathway considered for the river release mode have been discussed. For the other three release modes only the conceptual basis for the modeling efforts is discussed because of space limitations. The equations and a detailed discussion of all the models are contained in an EPA report [2] which is being prepared for publication.

Releases to an Ocean

The ocean was divided into two compartments: A shallow upper layer in which it is assumed that all edible seafood is grown, and a deeper lower layer. Radionuclides are added to the upper compartment from flow of the river into the ocean and by back transfer from the lower to the upper layer. They are removed from the upper compartment by radioactive decay, sedimentation, and water transfer from the upper to the lower layer. Radionuclides are added to the lower layer by water transfer and sedimentation from the upper layer and they are removed by back transfer of water from the lower to the upper layer, by radioactive decay, and by sediment transfer to the ocean floor. A differential equation can be written for each compartment to express the change in radionuclide inventory with time. These equations are coupled and they can be solved analytically to yield the inventory of each radionuclide in each compartment as a function of time. The equation for upper compartment inventory is divided by the volume of the compartment to yield the concentration of radionuclides in the upper compartment as a function of time. This upper compartment water concentration equation is integrated to compute the environmental dose equivalent commitment and the resulting fatal cancer and genetic effects for a population exposed through ingestion of ocean fish and shellfish.

Releases Directly to a Land Surface

For the land surface pathway models, we assume that some of the radioactive material initially placed in the repository is brought to the earth's surface due to an event which penetrates the repository such as drilling for resources. The release to the earth's surface is assumed to be over a small area and over a short period of time so that it can by modeled as an instantaneous point source to the earth. The mechanism for distributing this material to human receptors is by resuspension to the atmosphere. When the initial quantity of radionuclides released to the land surface is determined, a time-dependent release rate to the air due to resuspension can be estimated using a simple exponential model that depletes the land surface source by resuspension and radioactive decay. This time-dependent air release equation is applied with an atmospheric dispersion equation to predict air concentrations of radionuclides as a function of time and distance from the source. Using this air concentration equation, time-dependent ground surface concentrations of radionuclides are estimated as a function of distance. Integrations can be performed over time to yield the integrated air and ground surface concentrations of radionuclides.

As for releases to the river, the basic approach is to compute the population intake of radionuclides due to ingestion and inhalation, convert these intakes to population dose equivalents, compute the external population dose equivalents for air submersion and ground contamination and then calculate fatal cancers and genetic health effects. These are summed to obtain an estimate of total fatal cancers and total first generation genetic effects due to the initial release of radionuclides from the repository to the land surface.

Violent Releases Directly to Air

For this release mode, radionuclides are released from the repository directly into air from violent events with a low probability of occurrence. An example of this type of release would be a meteorite or volcano interaction with a repository, violently dispersing material into the air. Since a violent reaction is involved in distributing the radioactive material into the air, it is assumed that the material would be dispersed throughout the troposphere. This airborne material is divided into the fraction above the land surface and that above the oceans in proportion to the surface areas of the oceans and the land. The simplifying assumption is made that the airborne material above land remains over land and that above water remains over water. The population dose equivalent and health effects calculations for these two cases will be discussed separately.

Releases to Air Over Land. The radionuclides released to the air over land surfaces are assumed to be distributed uniformly in a volume determined by multiplying the land surface area of the earth by the average height of the troposphere. With the material distributed in this manner a two compartment model is established for predicting radionuclide movement between the air and the soil. The upper compartment is the tropospheric volume above the earth's land surface and the lower compartment is the soil root zone. Radionuclides enter the upper compartment at the instant of a violent release to air and no further quantities of radionuclides are introduced into the two-compartment system after the initial input.

Radionuclides enter the air compartment during the initial violent release and by resuspension from soil; they leave it by deposition onto soil and by radioactive decay. Nuclides enter the soil root zone only by deposition and leave it by resuspension, movement into the unavailable deeper soil layer, and radioactive decay.

A system of two coupled differential equations that describes the nuclide movements was solved giving the time-dependent quantities of nuclides in air and on the land surface from which integrated air and land surface concentrations are calculated.

Release to Air Over Oceans. The radionuclides released to the air over the oceans are assumed to be distributed uniformly in a volume defined by multiplying the earth's ocean area by the average height of the troposphere. With the material distributed in this manner, a three compartment model is established to describe radionuclide movement between the air and the two ocean compartments.

Compartment 1 is the tropospheric volume above the earth's oceans. Compartment 2 is the upper compartment of the ocean and compartment 3 is the lower compartment of the ocean. It is assumed that radionuclides enter the air compartment at the instant of a violent release to air and that no additional radioactivity is injected into the system after this initial input; radionuclides leave the air compartment by radioactive decay and by deposition into the ocean. Radionuclides enter the upper ocean compartment by deposition from the air and by back transfer with water from the lower ocean compartment. Radionuclides leave the upper compartment by radioactive decay, and by water diffusion transfer and sedimentation transfer to the lower ocean compartment. Radionuclides enter the lower ocean compartment by water diffusion transfer and sedimentation transfer from the upper ocean compartment and leave it by radioactive decay, sedimentation to the ocean floor, and by back transfer to the upper ocean compartment.

The dose equivalent pathways considered for the ocean are consumption of ocean fish and shellfish. A system of three differential equations (two of which are coupled) that describe the nuclide balance in the compartments was solved to calculate the quantity of radionuclides in the three ocean compartments. The time-dependent water concentration of radionuclides in the upper compartment in the ocean is needed in order to compute population dose equivalents and health effects. To obtain the water concentration of each radionuclide in this compartment, the activity of each nuclide in the compartment is divided by the volume of the compartment. The population dose equivalents, fatal cancers, and first generation genetic effects were calculated by the same procedure described for the ocean release mode.

PARAMETRIC DATA APPLIED IN CALCULATIONS

In many cases, simplifying assumptions have been made in choosing the parameter values for the calculations. This was acceptable because of the generic nature of the analysis. For example, in the drinking water calculations for the river pathway, the ratio of the population consuming drinking water to the river flow rate was needed. The numerical value for the ratio was obtained by dividing the projected world population by the total rate of flow of the world's rivers. Data from Annex D of the 1977 UNSCEAR Report [4] was used for this calculation. Upon comparison, we found that the "world" value for this ratio was midrange of values obtained for various areas of the United States. Another example is the use of the world average per capita fish consumption rates which were found to be mid-range of U. S. values. Similar results were obtained in comparing world values for other parameters with a range of values for the United States. In most of the equations, site and region specific numbers could be substituted if it were desired to perform an analysis for a specific site location.

Intake rates of radionuclides for the food pathways from a unit deposition to the ground surface were calculated using the AIRDOS-EPA computer code [5]. For the majority of the radionuclides, internal dosimetry factors are based on calculations performed using the INREM II computer code [6]. Dose equivalent to risk conversion factors were based on information in the BEIR report [8]. Additional dose equivalent and health risk methodology [9,10,11] have been developed since the models presented in this paper were derived. The newer methodology may be applied to future health impact assessments of the disposal of high-level radioactive waste.

SUMMARY

This paper discusses, generally, the models used to perform environmental dose equivalent commitment and health effect calculations for EPA's high-level radioactive waste disposal standard.* The four release modes addressed by EPA were releases to a river, releases to an ocean, releases directly to a land surface, and releases directly to air. The pathways for environmental transport were described for each release mode. The mathematical equations used to predict environmental dose equivalent commitments for the environmental transport pathways for releases to a river were derived and discussed. Results of health effects per becquerel release to the environment are presented in Table 2 for several radionuclides which will be included in the standard. The numerical values for parameters used in the models and the results of the calculations performed to derive the release limits included in the draft EPA standard are discussed in an EPA report by Smith et al. [12].

* Readers desiring more detail on these models should request the EPA report [2] which contains a complete detailed description of all the mathematical models including the technical development of the equations and parameter values used in the calculations as technical support for the standard.

Table 2. Health Effects per Unit Activity Released to the Environment by the River and Land Surface Release Modes

Radionuclide	Health Effects per Bq (Health Effects per Ci)	
	Release to River	Release to Land Surface
Am-243	7.6E-11 (2.8)a	2.7E-11 (1.0)
I-129	3.0E-13 (1.1E-2)	6.2E-16 (2.3E-5)
Pu-239	1.9E-12 (6.9E-2)	1.5E-12 (5.5E-2)
Sn-126	3.2E-12 (1.2E-1)	1.1E-12 (4.1E-2)

a For a unit activity release of Am-243 to the river, about 80% of the health effects were due to the ingestion of above surface food crops, about 10% were from drinking water, and the remaining 10% were from the other pathways.

REFERENCES

1. "Environmental Radiation Protection Standards for High-Level Radioactive Waste", Federal Register, Vol. 41, No. 235, page 53363 (December 6, 1976).
2. J. M. Smith, T. W. Fowler, and A. S. Goldin, Environmental Pathway Models for Estimating Population Health Effects from Disposal of High-Level Radioactive Waste in Geologic Repositories, EPA 520/5-80-002 (to be published).
3. U. S. Environmental Protection Agency, Environmental Radiation Dose Commitment: An Application to the Nuclear Power Industry, EPA 520/4-73-002 (February,1974).
4. United Nations Scientific Committee on the Effects of Atomic Radiation, Sources and Effects of Ionizing Radiation: UNSCEAR 1977 Report, United Nations Publication Sales No. E.77.IX.1 (1977).
5. R. E. Moore, C. F. Baes III, L. M. McDowell-Boyer, A. P. Watson, F. O. Hoffman, J. C. Pleasant, and C. W. Miller, AIRDOS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Airborne Releases of Radionuclides, EPA 520/1-79-009 (Reprint of ORNL-5532) (December, 1979).

6. G. G. Killough, D. E. Dunning, Jr., S. R. Bernard, and J. C. Pleasant, Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities Vol. 1, NUREG/CR-0150 (ORNL/NUREG/TM-190) (June 1978).
7. D. E. Dunning, Jr., S. R. Bernard, P. J. Walsh, G. G. Killough, and J. C. Pleasant, Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities Vol. II, NUREG/CR-0150-V.2 (ORNL/NUREG/TM-190/V2) (October, 1979).
8. National Academy of Sciences, National Research Council, The Effects on Populations of Exposure to Low Levels of Ionizing Radiation, Report of the Advisory Committee on the Biological Effects of Ionizing Radiation (BEIR I), Washington, DC (November, 1972).
9. D. E. Dunning, Jr., R. W. Leggett, and M. G. Yalcintas, A Combined Methodology for Estimating Dose Rates and Health Effects from Exposure to Radioactive Pollutants, ORNL/TM-7105 (December 1980).
10. C. L. Begovich, K. F. Eckerman, E. C. Schlatter, and S. Y. Ohr, DARTAB: A Program to Combine Airborne Radionuclide Environmental Exposure Data to Generate Tabulations of Predicted Impacts, ORNL-5692 (to be published).
11. National Academy of Sciences, National Research Council, The Effects on Populations of Exposure to Low Levels of Ionizing Radiation: 1980, Report of the Committee on the Biological Effects of Ionizing Radiations (BEIR III), Washington, DC (July, 1980).
12. C. B. Smith, D. J. Egan, W. A. Williams, J. M. Gruhlke, C-Y. Hung, and B. Serini, Population Risks from Disposal of High-Level Radioactive Wastes in Geologic Repositories, EPA 520/3-80-006 (to be published).

RISK ASSESSMENTS FOR SETTING HIGH-LEVEL WASTE DISPOSAL STANDARDS

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(Paper Not Submitted)

SYMPOSIUM SUMMARY AND CONCLUSIONS

Chairman

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Report on Workshop I: Input to Risk Assessment Methodologies

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Report on Workshop I: Input to Risk Assessment Methodologies

The goal of this workshop was to discuss the collection and analysis of data needed for site characterization and risk assessment. Topics which were considered included the relative importance of different parameters, the use of sensitivity analyses to guide data collection, the applicability of laboratory data to field problems, and the question of public confidence in the conclusions drawn from complex analyses. The following sections outline in more detail some of the points discussed in the workshop.

Discussion of the Usage and Determination of K_d Values

Elly K. Triegel

A number of participants in Workshop I expressed concerns about the importance of K_d in modeling repository performance and the uncertainties in determining representative values of the parameter. K_d is a distribution coefficient, equal to the mass of solute retained on a unit mass of the solid phase divided by the concentration of the solute in solution. It is a lumped parameter, in that it incorporates a large number of geochemical processes in a simple ratio of solid to liquid concentrations. The concept of K_d arose from a real need of the groundwater modelers to incorporate a simple parameter which characterizes the geochemical processes acting to retard movement of radionuclides in groundwater. Sensitivity studies indicate that the final concentration and arrival time of the radionuclide at the point of consumption or discharge is very sensitive to the K_d value used in the transport model.

The experience of several of the participants suggests that K_d values for some radionuclides are not representative of repository conditions. The degree of confidence in laboratory measurements is difficult to determine since the geochemical conditions in the field are complex and not easily characterized or reproduced. The lack of correspondence of K_d values and field results is related to artificial laboratory conditions, insufficient time for testing very slow reactions and the natural variations in aquifer geochemistry which cannot be characterized by a simple lumped parameter. In addition, the users of groundwater transport models are faced with the problem of choosing appropriate values of K_d. The values for some radionuclides vary greatly depending upon the materials tested and the conditions of the experiment. One study found more than an order of magnitude difference in the K_d values in two cores taken a few meters apart in the same rock layer. K_d also varies with liquid:solid ratio, time of reaction and temperature. Choice of an appropriate value is hampered by (1) the lack of a full understanding of the important factors controlling retardation, (2) incomplete reporting or characterization of the experimental conditions, (3) uncertainties as to which species of the radionuclides exist in the field, (4) incomplete knowledge of the geochemistry of the rock surrounding the repository, (5) the effect of other factors (e.g. the waste form, thermal effects, fracture geometry) on transport.

Several goals in modeling the geochemistry were discussed, including the need to (1) simulate actual conditions in the host rock and nearby aquifers, (2) better understand the geochemical controls, the influence of the waste form, and other factors in nuclide transport, and (3) better coordinate the modeler's needs, the field aquisition of data and the geochemist's contributions. Cooperation among computer, field and laboratory personnel would avoid costly development of new methodologies which cannot be used by the other groups or which require data and techniques which are not available. In addition, contributions from one group may help direct or simplify the tasks of the others. Examples were given of the need for modelers to understand the type of data which may be reliably collected in the field and the use of models to predict those measurements which must be made with the greatest accuracy.

A number of comments were directed at methods for dealing with the geochemical component of radionuclide transport. In general, it was felt by the group that the limitations of using a laboratory derived, lumped parameter should be recognized and extrapolations to complex or untested cases avoided. It should also be recognized that point-to-point predictions of transport cannot be made with any certainty. Field determinations of K_d were suggested as a means of incorporating factors which cannot be easily transported or duplicated in the laboratory. Examples of such factors include fracture and pore geometry, redox and pH fluctuations, and large scale inhomogeneities of the host rock.

The use of a minimum K_d value was suggested as an alternative to the use of unrealistic K_d values or the elimination of K_d entirely. A minimum K_d value would represent the lowest value to be expected in natural environments and, hence, the minimum retardation case. Use of this minimum value would reduce the problem of choosing the most appropriate K_d while avoiding either the underestimation of transport or the unrealistic no-retardation case.

Additional information on the chemical factors affecting transport may be obtained from theoretical geochemical models. However, the complexity of the geologic system and the interrelationship of parameters makes their application difficult, and the results uncertain. Experimental results from K_d determinations may be used in conjunction with the models to calibrate the models and reduce these uncertainties. In addition, consideration of the fundamental chemical relationships may indicate which parameters exert the most control over transport, which situations would produce the fastest migration and what the long-term behavior of the host rock and aquifers would be.

The use of K_d as a site selection criteria was discussed. The principal advantage in such an approach is that the geochemistry of sites can be compared more easily and on a more quantitative basis. Lack of a geochemical criteria might result in the selection of a site which has unsuitable adsorptive properties. Ideally, K_d standards should be used as flexible guidelines, to be custom tailored to each site or changed as more data is collected. It was generally thought by the participants, however, that such flexibility may not be standard regulatory practice or acceptable to the public.

In summary, the importance of the geochemical behavior of the radionuclide in predicting groundwater transport was well recognized. Although the lumped parameter, K_d, is extremely useful to the modeler, it can incorporate large errors and uncertainties. The limitations should be recognized and additional tools, such as field testing, minimum K_d values and geochemical models should be used to reduce uncertainties to acceptable levels.

WORKSHOP II. RISK ASSESSMENT METHODOLOGIES AND THEIR OUTPUT

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Workshop II began with discussion of risk assessment methodologies and their output. However, it soon became apparent that there were two general areas of interest; namely, (1) technical aspects of risk methodologies and their outputs and (2) how to demonstrate compliance with the Environmental Protection Agency (EPA) draft environmental standard and the Nuclear Regulatory Commission (NRC) proposed regulations for geologic disposal of radioactive waste. Therefore, the workshop was split based on interest in the above two subject areas. This report will summarize workshop discussions on the draft EPA environmental standard and the proposed NRC regulations.

The EPA environmental standard, in its current form, is expressed in terms of radionuclide discharge to the accessible environment. The numerical limits on radionuclide discharge to the environment are based on estimates of risk to human health which could result from exposure to discharged material. It was acknowledged in the workshop that having the environmental standard based on risk to human health, but not stated as risk expressed in health effects, is reasonable because of the considerable uncertainties inherent in attempting to predict important biosphere characteristics for the 10,000 year period of the standard. However, the opinion was expressed that the technical basis for the EPA standard should have considered uncertainties in surface environment characteristics and health effects factors that influence risk rather than simply using environmental pathways analysis to interpret results of radionuclide discharge calculations. The opinion was also expressed that the EPA analysis of health effects should consider both maximum and average individual radiation dose. The maximum dose case would be appropriate for such scenarios as drilling into the site where one or a few individuals could experience a relatively high radiation dose whereas the average dose calculation would be appropriate for evaluating population exposure.

During the second half of the workshop session, discussion focused on the proposed NRC regulations for geologic disposal of radioactive wastes. Specifically, the technical basis and the form of the regulations were discussed. In their proposed form, the NRC regulations would place numerical design criteria on specific repository system components. The opinion was expressed that the draft NRC regulations do not have adequate technical basis. If design criteria are to be applied to system components (e.g., requiring that the waste package provide contaminent for a period of 1000 years), these component criteria should be related to the overall system criterion; namely, the EPA environmental standard. The opinion was also expressed that such a technical basis is needed to provide a basis for future changes in the criteria which may be required should present assumptions prove faulty.

Several individuals in the workshop felt that the NRC criteria would be difficult to meet in their present form because of the absolute nature of the criteria. One example cited was the proposed requirement that the waste package contain the waste for 1000 years. To show compliance with such a criterion could require considerable overdesign of the waste package. Under any circumstance, it would be difficult to state with absolute assurance that no container would leak for 1000 years. There seemed to be general agreement among several workshop attendees that the NRC criteria should be softened in some fashion. For the example of the waste package criterion, it was suggested that wording such as, "...provide reasonable assurance that no radioactive materials will leak from the waste package for a period of 1000 years after repository closure" should be used.

NRC PERSPECTIVE ON THE SYMPOSIUM

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As a prelude to my remarks I would like to thank Dave Kocher and his associates at ORNL for their efforts to make this symposium a successful one. I can assure you that from my vantage point we have accomplished the objectives which we set down in the early stages of planning this symposium last summer:

- To bring together individuals vitally concerned with the job of HLW disposal in geologic repositories to consider the sources of uncertainty connected with assessment of post-closure performance of the repository and to consider how various methods, including the tools of modeling, might be used in the regulatory process.

Thank you, Dave.

My purpose in presenting these remarks to you this morning is twofold:

- to try, at least in a preliminary way, to tie back the accomplishments of this symposium to our original objectives;
- to leave you with a charge.

On the first morning of the first day of the symposium, our NRC speakers conveyed to you the job of regulating geologic disposal of HLW.

We told you briefly about our procedures for licensing geologic disposal of HLW, established in the form of a final regulation, a copy of which you had received. We told you that we had evolved - through a bootstrap process of active dialogue with the public - licensing procedures which:

- provided for key decision points - at site characterization, construction authorization, waste emplacement, and permanent closure;
- keyed the strength of each decision about a repository to the information available at the time of that decision, with the strength of each successive decision reflecting increased confidence in reasonable assurance of protection of the public health and safety and the environment should waste be permanently disposed at the repository;

- set forth information requirements so that the information needed to make each decision would be available when the decision was to be made;
- provided mechanisms for public participation in and public scrutiny of the process by which each decision was made.

I am hopeful that the procedures we have established will do much to remove at least some of the concerns which Ms. Yuan, Sheldon, and Olson expressed during our session on public and private interest group perspectives.

I believe these procedures do provide for early and meaningful public participation in the process of licensing disposal of HLW in a geologic repository. However, I think it is important to understand that even though we've attempted to grapple with the institutional issues as well as the technical in our regulatory approach in a manner acceptable to the public, as John Stucker stated, "consultation and concurrence is not NRC's to grant." That is, the Commission must be the decisionmaker so that if we are to carry out our responsibilities satisfactorily, the stakeholders must be satisfied with our decision processes, with our regulatory approach.

Let me explain further. The Commission has been given authority to license and that authority does not carry with it authority to allow partnerships between itself and such publics as the states, local and tribal governments in the decisionmaking or to allow bottom lines, including regulations, to be formulated as recommendations by other than its own staff.

Thus, the purpose, and the limit, of the Commission's authority vis a vis public participation, is <u>not to reach a decision that everyone will like</u>--that is not possible; nor to reach a decision made collegially with any public--the Atomic Energy Act won't permit that; but rather to make decisions that have considered all relevant aspects and perspectives, explored all pertinent uncertainties, and have been subject to public scrutiny from beginning to end.

Then we turned to the technical criteria to tell you how and why we were going about the job of their development. Just as we engaged in active dialogue with those outside NRC in the development of the licensing procedures, so we did in developing the technical criteria. Just as we partitioned the licensing procedures into a series of decisions of increasing commitment based on more comprehensive information of better quality, so we partitioned our technical approach so that we could deal with the questions which must be answered for a <u>licensed</u> repository. And I emphasize <u>licensed</u>. Recall Craig Roberts' <u>keynote</u>: a <u>licensed</u> repository is one for which both the technological <u>AND</u> institutional issues will have been satisfactorily resolved through <u>the</u> licensing process itself. But to do that, the process has to be capable of bringing about resolution. Hence, recall Jack Martin's practical consideration: one of our motivating factors has been to develop an approach--for both the procedures and the technical criteria--which would avoid contentious licensing hearings; to key on another way it was expressed, "to avoid useless rhetoric that leads nowhere."

Therefore, the approach we have taken considered first, what we were trying to do--have confidence that the waste would be disposed safely; and second, how to do that--expose all the uncertainties up front, see what they mean, and find a way around the lack of confidence spawned by these uncertainties. Obviously easier said than done. But we think we've succeeded by redefining geologic disposal of HLW into containment for a time and isolation thereafter and by placing reliance on both engineering and the site as appropriate.

Now all of this development of the technical criteria is going on against the backdrop of a still emerging EPA standard. The absence of a standard creates uncertainties. These uncertainties have policy and programmatic implications of nontrivial import that boil down to dollars. Drawing upon the concerns expressed by Drs. Eichholz and Lieberman: where does one jump off in development of regulatory standards? Should one start from consideration of some limiting conditions that must be met--in this way maintaining maximum flexibility in making choices and allowing cost optimization to be a dominant factor in determining a particular repository system? Or, akin to the approach we've taken with our technical criteria, is it more appropriate to consider what can be done reasonably during containment and isolation, for engineering and siting, and to place requirements on certain subsystems so as to increase confidence in our decisions? As Jack Martin indicated, we think we've struck the right balance; and asking you to consider our approach further will be part of the charge I put to you at the end.

As I listened to the various speakers, heard the questions and answers following the talks, and chatted informally in the breaks, some thoughts occurred to me which I'd like to share with you:

- I was glad to see so many individuals who are working in the field of modeling coming together in order to forward the job--from the NRC perspective--of safe disposal of HLW in geologic repositories; and identifying areas in modeling that require activity in order to get on with the job, and seeing which of these areas are receiving the greatest activity today and which need more attention tomorrow.

- Modeling cannot be the only decision tool that is used; we just aren't there in terms of our ability to quantify. Nor should that disturb us; numbers have never provided all of the answers. As Nestor Ortiz pointed out, review by disinterested experts must be an important part of any process. And why not? Quantitative work must always be preceded by qualitative and semiquantitative analysis: cf. Dr. Burkholder's "design specs" as the forerunner of coding; and Dr. Bradstetter's approach to the entire problem of modeling.

- I am concerned that in our rush to calculate we've overlooked the accuracy of quantitative predictions with respect to whether they represent physical processes and can be meaningfully interpreted physically. I am afraid we've become enamoured of the precision of the computation. In sum, I think we must give emphasis early to the model validation and accuracy questions.

- The question of how realistic one can reasonably expect the quantitative models to be is as yet unresolved; hence, so too is the question of the balance that needs to be struck between the direct use of calculations in making decisions and the use of such qualitative or semiqualitative techniques as judgments, arguments by analog, etc. To my mind this is an area requiring continuing thought and effort. We don't want to get into the licensing process and find we haven't given this question adequate forethought, so that, in spite of our best intentions, the hearing can't be closed or closed only with great difficulty.

- I would note, too, a continuing debate emerged, at least implicitly, during the symposium over the utility and appropriateness of health effects vs. some other measure of repository performance. It seems to me that at heart of this, is the question of whether, given the nature of the hazard involved, it is better to regulate according to the effects on the population or on the individual.

And now to my charge:

Go back, think about what you've heard here. You may disagree with what you've heard. But the points of view of your colleagues may indeed have merit. As you continue in your individual efforts, take time to reflect on what you have heard here. Keep in mind that we all are members of the public, we all are stakeholders in this enterprise, and that we all, as experts in this field, have an obligation to see that the technical solutions which we develop are valid in the societal context of regulation and licensing of geologic disposal of HLW. Soon the technical criteria will be available for public comment in the form of a proposed regulation, and we urge you as individual stakeholders to take the opportunity to provide your comments to the NRC.

SYMPOSIUM ATTENDEES

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