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# Mitigative Techniques for Ground-Water Contamination Associated With Severe Nuclear Accidents

## Analysis of Generic Site Conditions

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## ABSTRACT

The Pacific Northwest Laboratory evaluated the feasibility of using ground-water contaminant mitigation techniques to control radionuclide migration following a severe commercial nuclear power reactor accident. The two types of severe commercial reactor accidents investigated are: 1) containment basemat penetration of core melt debris which slowly cools and leaches radionuclides to the subsurface environment, and 2) containment basemat penetration of sump water without full penetration of the core mass. Six generic hydrogeologic site classifications were developed from an evaluation of reported data pertaining to the hydrogeologic properties of all existing and proposed commercial reactor sites. One-dimensional radionuclide transport analyses were conducted on each of the individual reactor sites to determine the generic characteristics of a radionuclide discharge to an accessible environment. Ground-water contaminant mitigation techniques that may be suitable, depending on specific site and accident conditions, for severe power plant accidents were identified and evaluated. Feasible mitigative techniques and associated constraints on feasibility were determined for each of the six hydrogeologic site classifications. Three case studies were conducted at power plant sites located along the Texas Gulf Coast and the Ohio River. Mitigative strategies were evaluated for their impact on contaminant transport. Results show that the techniques evaluated significantly increased ground-water travel times and reduced contaminant migration rates.

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# PREFACE

This report is divided into two volumes. Volume 1, "Mitigative Techniques for Ground-Water Contamination Associated with Severe Nuclear Accidents: 1) Analysis of Generic Site Conditions," examines generalized aspects of a nuclear core melt accident. The characteristics of core debris, ground-water transport of contaminants, and mitigative techniques in idealized circumstances are discussed in the first volume. Volume 2, "Mitigative Techniques for Ground-Water Contamination Associated with Severe Nuclear Accidents: 2) Case Study Analysis of Hydrologic Characteristics and Interdictive Schemes," considers the site-specific aspects of selected hydrogeologic environments and individual mitigative techniques. A case study format is used to demonstrate the type of conditions and considerations that would need to be addressed following a severe accident. The two volumes achieve different objectives but are designed to be complementary in nature. In composite, they provide a comprehensive guide to mitigative actions in ground-water bodies following a core melt accident.

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# EXECUTIVE SUMMARY

## INTRODUCTION

### Purpose

Pacific Northwest Laboratory conducted a study of mitigative techniques for ground-water contamination associated with severe commercial nuclear power plant accidents for the U.S. Nuclear Regulatory Commission. The purpose of this study was to evaluate the feasibility and desirability of using specific ground-water contaminant mitigation techniques (e.g., constructed barriers to subsurface flow and transport, hydraulic barriers created by ground-water withdrawal and/or injection) to control radionuclide migration in ground-water flow systems following a nuclear core melt accident. The terms "severe accident" and "core melt accident" are synonymous in this study and are defined as an accident where molten nuclear fuel, reactor components, and/or sump water exits the containment structure.

### Objectives

The objectives of this study are:

- identification of hydrogeologic factors that affect the release and ground-water migration of radionuclides following a severe or core melt nuclear accident,
- evaluation of the feasibility and desirability of interdicting radionuclide contaminants to mitigate environmental consequences based on a generic hydrogeologic classification of power plant sites in the United States,
- development and demonstration of the methodology for the characterization and evaluation of contaminant transport and the necessity of contaminant interdiction in contrasting geologic environments, and
- development and demonstration of the methodology for evaluation of the feasibility, design, implementation, and performance assessment of mitigative schemes on a site-specific basis.

The first two objectives are met by conducting a generic analysis of ground-water conditions following a severe accident. This information is contained in Volume 1 and represents an inductive process wherein a large volume of diverse information is reduced to generalized or generic descriptions of a core melt accident. The common properties concerning core melt formation, contaminant migration and arrival at an accessible environment are combined to form a generic hydrogeologic classification system. Ninety-seven existing and proposed nuclear power plant sites in the United States are classified by this scheme. A large body of geologic and hydrologic information is included for the convenience of the nontechnical reader and to serve as a reference guide to further information.

The third and fourth objectives are achieved by conducting three site-specific case studies which are contained in Volume 2. Each of the case studies examines a different aspect of selection, design, and implementation of a contaminant mitigation. The case studies constitute deductive analyses that provide insight into the site-specific conditions that would need to be considered following severe accident. The three case studies examine mitigative techniques in greater detail than is possible in the generic analysis contained in Volume 1. The case studies also serve as a validation of the conclusions of the generic analysis. The determination of an appropriate method to interdict ground-water contaminant and the design of engineering structures can only be made at the case study level of analysis. The core elements of the three case studies are given in Table S.1.

TABLE S.1. Case Study Topics of Emphasis

| Case Study No. | Name | Topics of Concentration |
|---|---|---|
| 1 | South Texas Plant | Unconsolidated hydrologic unit, hydrogeologic characterization, evaluation of mitigative methods. |
| 2 | South Texas Plant | Performance assessment, cost effectiveness, mitigative scheme selection |
| 3 | Marble Hill, Indiana | Consolidated and fractured hydrologic unit, anisotropic flow field, plant structures |

## Scope

The scope of the study was limited to consideration of the necessity and feasibility of mitigative techniques to reduce the environmental consequences along the liquid pathway. Contaminated ground water could reach accessible environments through: 1) controlled discharge points (e.g., abstraction wells), 2) uncontrolled discharge points such as springs, and 3) ground water to surface water flow interface at discharge locations. This study assumed that obvious and available measures to protect the public, such as prohibiting the use of contaminated wells, will be taken. It also assumes that contaminants discharging through natural collection points such as springs will be collected, as necessary, and isolated from the environment. This study concentrates on actions taken to reduce transport rate of radionuclides migrating in ground water and moving toward surface water bodies. Issues concerning atmospheric releases, site restoration and long-term, low-level radioactive discharges to surface water bodies are not a part of this study. Considerations for monitoring systems such as optimal placement, statistical confidence, and detailed mitigative designs are explicitly outside the statement of work as formulated by the U.S. Nuclear Regulatory Commission.

## ANALYSIS OF GENERIC SITE CONDITIONS

### Introduction

The severity of ground-water contamination and subsequent discharge of radionuclides to the surface environment is, in part, a function of the hydrogeology of the site. The geology of the site affects the radionuclide release rate, transport time to the accessible environment, and the ability to mitigate the environmental impact of the accident. Thus, the analysis of liquid pathway contamination resulting from a severe accident is foremost a study of contaminated ground water and its eventual discharge into surface water.

The two types of accidents investigated are: 1) containment basemat penetration by molten core debris which slowly cools and leaches radionuclides to the subsurface environment, and 2) containment basemat penetration by contaminated sump water. The release and transport of radionuclides following a core melt accident is a complex process which is dependent on many accident and site-specific parameters and events. These processes are defined in as much detail and as accurately as possible. Where there is uncertainty in the examination, conservative yet realistic parameter estimates are made.

The contaminant liberation by the release from containment (i.e., leach rate) and the discharge to the accessible environment is described as an activity flux rate reported in pico curies/year. The environmental contact point is assumed to be the nearest surface water body. The major topics and findings of this study are presented by topic in the following sections.

### Leach Release from Core Debris

The chemical composition of the aggregate in the concrete basemat and the underlying geologic materials has a large influence on the rate of solid material leach release. Calcine debris derived from concrete and carbonate rock would be:

- relatively porous with a high surface area,
- contain a high density of radionuclides per unit volume,
- melt to a depth of about 3 meters below the basemat,
- release radionuclides to the ground-water flow system through a diffusion process, and
- attain a peak release rate for strontium-90 of about 1 x 10 x $10^{17}$ pCi/yr.

Silicic debris produced by the melting of sand or igneous rock would:

- be more glass-like with a porosity and permeability determined by the density of the fracture network,

- have a relatively lower surface area and porosity,
- involve a larger melt zone extending to 10 meters below the basemat,
- release radionuclides through a dissolution process, and
- obtain a peak release rate for strontium-90 of about 2 x $10^{15}$ pCi/yr.

As a result of these fundamentally different characteristics and leach mechanisms, calcine debris would release radionuclides at a rate two orders of magnitude greater than silicic debris. The difference in leach rates would increase if less conservative assumptions of material properties are assumed. The composition of a severe melt debris would be an admixture of both silicic and calcine components resulting in a leach rate between the two limiting cases presented.

Both calcine and silicic debris would leach release radionuclides to a ground-water flow system for long periods of time. The quantity of radionuclides released by leaching would eventually reach insignificant levels because of radioactive decay and a decreasing leach release rate. Based on the calculated leach rates and a ground-water velocity of 1 m/yr, the concentration of strontium-90 adjacent to the melt debris would reach 10 CFR 20 limits of 300 pCi/l about 800 years and 1100 years after the accident for a silicic and a calcine melt, respectively.

Sump Water Release Rate

Sump water drainage rates through the containment basemat and core melt debris would be highly site and accident specific. Feasible rates based on the hydraulic properties of each site indicate that sump water drainage rates can potentially release radionuclides at a greater rate than core melt leach rates. However, the actual drainage rate could be considerably more to much less than predicted. The time over which an actual sump water release would occur is a period of days to months. Very slow drainage rates could allow removal of liquid contaminant from the containment structure before it entered the ground-water flow system. Conversely, a release of sump water driven by a pressurized containment dome (pressurized water reactors only) could produce a rapid hydraulic spreading of contaminant and decrease the travel time to the surface environment. This would result in the largest possible radiological flux to the ground-water environment and the greatest need for contaminant interdiction.

Generic Hydrogeologic Classification of Nuclear Power Plant Sites

A hydrogeologic classification system for contaminant interdiction at nuclear power plants must consider the geological factors of a core melt accident from the creation of the melt debris to the eventual contaminant arrival at land surface. The hydrological factors used to determine a classification scheme are:

- the rock chemistry of the contaminant source for the determination of leach rate,

- feasibility of contaminant mitigation in different geologic environments (i.e., consolidated and unconsolidated materials), and

- hydraulic transport parameters to land surface.

Six types of generic sites result from application of the classification scheme to proposed and existing nuclear power plant sites in the United States:

1. Fractured Consolidated Crystalline Silicates,

2. Fractured and Solutioned Consolidated Carbonates,

3. Porous Consolidated Silicates,

4. Porous Consolidated Carbonates,

5. Porous Unconsolidated Silicates, and

6. Fractured Consolidated Silicates - Shale.

Assigning of representive or "average" hydraulic parameters to a generic classification and generating "average" radionuclide discharges to a surface water body is undesirable and was not attemped because:

- there is a wide range in hydraulic values among geologically similar sites,

- such "average" conditions may not occur at any real site,

- averaged parameters that are inversely related (e.g., hydraulic gradients and permeabilities) may not produce an average result, and

- the variability of transport within a given hydrogeologic classification would be lost.

Simulation of indivdual sites and analysis of the results by generic groups is applicable and demonstrates the large differences in contaminant release and transport among and within the generic classifications. The major findings of the generic hydrogeologic analysis is presented below.

The discharge of radionuclides to the surface water environment is more a function of site hydrogeology than the type of accident sequence (e.g., PWR 1-7 and BWR 1-4). The range of contaminant quantities available for transport because of a less probable accident is small (several tenths of a percent) in comparison to the large range of values (up to 6 orders of magnitude) for hydrologic transport parameters. The different accident sequences would alter the quantity of contaminant by a linear function (a percentage of the total fuel inventory) while changes in hydrologic parameters allow for longer transport times which exponentially would decreases the total quantity discharged to the environment. The major hydrogeologic factors determining the severity of an accident (listed in order of relative importance) are:

1. chemical composition of the containment structure basemat and underlying bedrock,

2. effective porosity,

3. sorption of contaminant, and

4. hydraulic gradient and conductivity.

Indicator Radionuclides

Three radionuclide indicators of contamination are used in this study because of their initial quantity, longevity, and mobility. The analysis is conducted for strontium-90, cesium-137 and ruthenium-106. Ruthenium-106 is found to be sorbed and retarded under core melt conditions. Previous studies (RSS-1975 and LPGS-1978) assumed that 50 percent of the ruthenium was complexed by nitrate and formed a water coincident comtaminant. This assumption was based on the migration of ruthenium-106 in high nitrate level processing wastes at the Hanford site near Richland, Washington. Nitrate concentrations found in natural ground water are not sufficient to complex and thus mobilize ruthenium-106. Retardation of ruthenium allows decay processes to reduce the amount of contaminant to low levels prior to reaching surface water. Only 7 percent of the nuclear power plant sites in the United States would have a discharge of rutheniun-106 before it experienced 40 half-lives of decay.

Cesium-137 would be released in the sump water from pressurized water reactors. Empirical testing indicates that cesium-137 is more strongly sorbed than strontium-90, but the retardation mechanism is phenomonologically complex and not fully described by present retardation models. Retardation of cesium-137 has been noted as being time and concentration dependent. Cesium-137 would arrive at the discharge location at 37 percent of the sites before 40 half-lives of decay.

Strontium-90 is the preferred contaminant to determine the relative sensitivity to a core melt accident. Strontium would be released in sump water and as leachate from the core debris. It is more mobile than cesium-137 and would arrive at the discharge location at relatively early times and at activity rates comparable to the cesium-137 in a sump water release. Strontium-90 could be expected to arrive at a surface water body at 55 percent of the sites prior to 40 half-lives of decay.

Contaminant Discharge to Accessible Environments

The generic discharges of contaminant are examined at two basic levels. The first level determines whether the contaminant will arrive within a short time at a high flux, or at a much longer time at an insignificant level. A conservative definition of significance is based on a 40 half-life travel time to surface water. Over this time period, radionuclides are decayed to very low levels or fall into the category of a non-imminent situation requiring mitigation. The analysis of significant discharges to the environment demonstrates that:

- 43 percent of all sites do not produce a significant discharge to a surface water body that would require immediate contaminant interdiction to prevent severe environmental consequences,

- interdiction would be desirable at 85 percent of the fractured geologic sites, and

- interdiction of contaminant would be desirable at 42 percent of the nonfractured sites.

Within the first level of analysis the generic sites are ranked as to their relative environmental sensitivity to a core melt accident by comparison of the percentages of sites that would result in a significant radionuclide discharge and those that would produce a minor discharge. The ranked generic sites are presented with the percentage of significant discharges in Table S.2.

TABLE S.2. Generic Sensitivity to a Severe Nuclear Accident

| Rank | Generic Classification | Percent of Sites with Significant Surface Water* |
|---|---|---|
| 1 | Fractured Consolidated Crystalline Silicates | 94 |
| 2 | Fractured and Solutioned Consolidated Carbonates | 83 |
| 3 | Fractured Shale | 60 |
| 4 | Porous Unconsolidated Silicates | 49 |
| 5 | Porous Consolidated Silicates | 38 |
| 6 | Porous Consolidated Carbonates | 20 |

* All three indicator radionuclides considered.

The second level of analysis of generic sites is more detailed and examines the generic trends in arrival times and discharge fluxes of the significant discharges. Observations of the premitigative radionuclide discharges indicates that:

- the earliest time of contaminant arrival of individual sites are in the fractured media classifications at about 6 months for carbonates and 8 months for silicates,

- contaminant arrival at a surface water body at 90 percent of the sites would be greater than 5 years, allowing detailed monitoring, simulation, and planning to proceed mitigative actions,

- the generic average of arrival times at a surface water body ranged from 5 years in fractured and solutioned carbonates to over 200 years for porous consolidated silicates,

- the greatest radionuclide flux entering a surface water body is produced by a sump water release of cesium-137 in a fractured and solutioned carbonate at $2.5 \times 10^{17}$ pCi/yr,

- peak flux rates of cesium-137 and strontium-90 in sump water discharges are similar to an order of magnitude at contaminant arrival times of less than 30 years,

- silicic media has a peak contaminant discharge 100 times less than carbonate media because of the difference in leach rates, and

- although the core debris contains 10 times more strontium-90 than the sump water, the more rapid release of sump water produces a higher radionuclide flux to the environment.

When there are no trends within a hydrogeologic classification of first arrival times or quantity of radiological outflow, the site-specific hydraulic parameters and/or reactor siting (i.e., distance to surface water) are more important than generic classification. This situation occurs for porous consolidated carbonates and fractured shale. These sites are best evaluated for environmental sensitivity by observing the percentage of sites that produce a significant discharge (i.e., prior to 40 half lives of decay).

## Mitigative Techniques for Contaminant Interdiction

There are two general classes of ground-water contaminant interdiction techniques that may be used to mitigate the environmental effects of a severe nuclear accident: 1) static or passive techniques, and 2) dynamic or active strategies. The individual techniques or schemes that comprise each class are designed to interact directly with ground-water flow, and consequently the contaminant being transported, to achieve an acceptable level of contaminant mitigation.

## Static Barriers

Static or passive mitigation techniques are typically engineered/constructed barriers to contaminated ground-water flow. The primary objectives of a constructed barrier is to redirect the ground-water flow away from potentially accessible surface environments or to retard the flow, and allow radioactive decay to reduce the environmental hazard. Achievement of these objectives usually results in ground water being forced to follow more circuitous routes with longer travel times or at slower migration rates. Constructed barriers are considered static ground-water contaminant mitigation techniques because once in place they are not readily adaptable to changing conditions of ground-water contamination. Engineered/constructed barriers do not normally require a significant amount of maintenance or energy. Three basic types of constructed barriers were analyzed for their feasibility and suitability as mitigation measures for ground-water contamination resulting from a severe power plant accident: grout curtain cut-off walls, slurry trench cut-of walls, and steel sheet piling.

Dynamic Barriers

Dynamic or active ground-water contaminant mitigation techniques are primarily conceptual strategies for actively influencing the state of ground-water contamination. Active influence is accomplished by either changing the ground-water flow regime by pumping and/or injection, directly treating the contaminated ground water or combinations of both approaches. Active ground-water contaminant mitigation schemes are generally better able to respond to changes in the state of ground-water contamination than static barriers. However, dynamic schemes typically have relatively high maintenance costs. Also extensive monitoring feedback is usually recommended to ensure adequate performance. The dynamic ground-water contaminant mitigation schemes analyzed for their feasibility and applicability are:

1. Ground-water withdrawal for potentiometric surface adjustment,

    1a. prevent discharge at receiving surface water body
    1b. prevent saturated contact with core melt debris
    1c. prevent contamination through leaky aquifers

2. Ground-water withdrawal and/or injection to control the contaminant plume,

    2a. withdrawal and injection
    2b. withdrawal without injection
    2c. withdrawal with surface treatment and recharge
    2d. injection only

3. Subsurface drains,

4. Selective filtration via permeable treatment beds,

5. Ground water freezing,

6. Air injection to form a permeability barrier.

## Feasibility Criteria

There are several important considerations for determining the suitability of mitigative techniques for ground-water contamination: 1) design, 2) construction, 3) performance, and 4) implementation. Design considerations include the variations in specific types of techniques (e.g., particulate versus non-particulate grout), appropriate host geologic media, size, location, and orientation of the various mitigation measures and design limitations. Passive ground-water barriers (i.e., slurry trenches, grout curtains, and steel sheet piling cutoffs) have better defined engineering design considerations than typically do dynamic ground-water contaminant mitigation strategies, which are less rigorously defined from an engineering standpoint.

Construction considerations are a major concern in determining the feasibility of specific mitigation strategies. Construction considerations include appropriate methods of installation, limitations of construction methods, equipment required for construction, etc. Several of the mitigation strategies (i.e., slurry trenches, subsurface grouts, and permeable treatment beds) require extensive excavation. Trenching is realistically feasible only in unconsolidated media and soft, easily ripped semi-consolidated media.

Performance considerations include permeability reductions, durability, continuity, and contaminant compatibility. All of the performance considerations vary with time. For example, steel sheet piling can be expected to corrode in approximately 40 years, thus significantly reducing its effective performance. Durability is closely related to permeability reduction and maintenance requirements. How long a barrier will perform as designed is a function of quality control during construction and ground-water chemistry. Cement-based constructed barriers will lose their integrity more rapidly if exposed to freeze and thaw cycles or high levels of sulfate. Most, if not all, dynamic mititgation strategies are temporary and energy extensive, and their design with respect to the overall mitigation plan should reflect this condition.

Implementation considerations to construction are centered around the practical engineering feasibility of the technique. These considerations include:

- installation and construction time,
- cost,
- equipment mobilization and availability,
- toxicity, (some chemical grouts are toxic), and
- exposure hazards to workers.

Worker safety during the installation and maintenance of a mitigation scheme is of primary concern. In most cases the closer to the contaminant source the mitigation scheme is to the core debris, the more effective it will be. A site-specific investigation of the radionuclide hazards from core debris, contaminated ground water, and surface contamination must be conducted prior to construction activities. Another safety issue involves the safe handling, treatment, and disposal of contaminated ground water. Several of the mitigative schemes require aboveground handling of contaminated ground water thus requiring special care to ensure the safety of workers and the integrity of the surface environment.

Summary of Generic Analysis

The implementation considerations for ground-water contamination mitigation schemes are extremely important in the overall assessment of the applicability of each measure. However, these issues are also highly sensitive to specific and individual site characteristics ranging from the physical plant configuration, to local meteorological condition at the time of the accident. Therefore it is difficult, if not impossible, to detail the absolute effect these issues in a generic manner. For this reason the general limitations are presented in the generic analysis, and the performance of a mitigative scheme is examined through case studies.

In a generic framework, the constraints on feasibility are related to site geology, hydrology, and accident characteristics. The criteria for determining feasibility for each hydrogeologic classification are:

1. consolidation of the geologic media,
2. host geologic material grain size or fissure width,
3. low hydraulic conductivity,
4. high ground-water velocity,
5. surface handling of contaminated ground water,
6. depth to a basal confining layer
7. depth to contaminant plume.

The generic examination of mitigative techniques is summarized to its most basic level in Table S.3. The table presents the feasibility of each major mitigative technique for the generic hydrogeologic classifications.

TABLE S.3. Feasibility of Mitigative Techniques for Each Generic Hydrogeologic Classification

| Mitigative Technique | A | B | C | D |
|---|---|---|---|---|
| | Generic Classification* | | | |
| | Feasibility** | | | |
| 1. Grouting with Particulate and Chemicals | Y | Y | Y | Y |
| 2. Slurry Trenches | N | N | Y | N |
| 3. Steel Sheet Pilings | N | N | Y | N |
| 4. Ground-Water Withdrawal for Potentiometric Surface Adjustment | M | Y | Y | N |
| 5. Ground-Water Withdrawal and/or Injection for Contaminant Plume Control | Y | Y | Y | M |
| 6. Interceptor Trenches | N | N | Y | Y |
| 7. Permeable Treatment Beds | N | N | Y | N |
| 8. Ground-Water Freezing | M | Y | Y | Y |
| 9. Air Injection | M | M | M | M |

* Generic Hydrogeologic Classification: A = Fractured Consolidated Silicates and Fractured and Solutioned Carbonates, B = Porous Consolidated Carbonates and Porous Consolidated Silicates, C = Porous Unconsolidated Silicates, D = Fractured Shale.

**Feasibility of Mitigative Techniques: Y = yes, N = no, M = marginal.

## CASE STUDY ANALYSIS

### Introduction

The components of the case studies are designed to start with information gained from the generic analysis and follow an iterative process of collecting additional information and developing more sophisticated conceptual and numerical models. In the event of a severe accident this process would be continued until either the analysis indicated that no contaminant interdiction was necessary or that the mitigative scheme in place would be an effective safeguard of environmental concerns. All analyses discussed are strictly hypothetical.

Case Study No. 1 South Texas Plant

Selection of appropriate mitigative actions following a severe accident requires a detailed evaluation of pre- and post-mitigative radionuclide transport through the ground-water system to potentially accessible environments. The South Texas Plant (STP) Case Study No. 1 was conducted as a demonstration of a methodology for evaluating mitigative techniques on the basis of site-specific characteristics. Emphasis is focused on the characterization and evaluation of ground-water flow and contaminant transport phenomena important at the South Texas Plant. The STP was selected on the basis of having adequate available data and being located on unconsolidated silicate (e.g., sand, silt, and clay). Relative to the other generic hydrogeologic classifications, the porous unconsolidated silicate sites have high hydraulic conductivity, high effective porosity, and low hydraulic gradient.

The STP is situated in south-central Matagorda County, Texas, on the Texas Gulf Plain approximately 4.9 km due west of the Colorado River. The STP is influenced by the coastal hydrometorologic regime and tidal effects of the Gulf of Mexico. The geomophology of the site is typical of a slightly eroded plain, characterized by low relief, abandoned river valleys, marshes, and offshore barrier bars. The near-surface geology of the STP site consists of the Pleistocene Beaumont Formation which extends at least 700 ft below the ground surface. The formation is characterized as layers of clay, sandy clay, and thick sand units. The layers of sand are up to 100 ft thick and produce significant amounts of water. Clay layers of up to 150 ft thick hydraulically isolate the various sand layers. Based on the geologic evidence and piezometric data, three major sand layers underlie the site; these layers are seperated into two hydrostratigraphic units, a deep aquifer and a shallow aquifer. The shallow aquifer is artesian and extends to about 150 ft below the surface and consists of upper and lower units. The two units are separated by a 20-ft clay layer and have slightly different potentiometric levels.

The STP is composed of two pressurized water reactors, each capable of producing 3800 MW thermal and 1250 MW electric power. The two units are roughly 590 ft apart and use certain shared facilities including the cooling reservoir, spillway and blowdown facilities, and essential cooling pond. The reactor core-rated thermal power is 3800 MWt. High-pressure light water serves as the coolant, neutron moderator, reflector and solvent for the neutron absorber. The reactor containment building is 148 ft in diameter and has a 18-ft-thick basemat.

The accident scenario assumed for the a single reactor of the STP is a loss of coolant accident leading to penetration of the core melt into the earth below the containment structure to about 35 ft below the basemat. At this depth the core debris would reside in the lower unit of the shallow aquifer. Using strontium-90 as the indicator radionuclide, an estimated $4.53 \times 10^{18}$ pCi would be released based on a prescribed release rate determined under the same assumptions as used in the generic examination of silicic core melts.

The recommended methodology for the evaluation of selected techniques for mitigation of possible ground-water contamination caused by severe accidents consists of four main steps:

1. survey of regional ground-water hydrogeologic characteristics and regional glow analysis to determine local boundary conditions,

2. pre-mitigative local ground-water flow and transport analysis,

3. performance evaluation of feasible mitigative techniques based on ground-water and contamiant transport simulation, and

4. sensitivity analyses of contamiant transport to hydrogeologic parameters,

The approach taken for the Case Study No. 1 is consistent with this methodology. Specifically, a regional hydrogeologic analysis is conducted using the TRANS ground-water flow and transport code. The analysis was accomplished using only previously published data. When data are sparse or unavailable, hypothesized data are generated based on the best information available.

The results of Case Study No. 1 include:

- a detailed hydrogeologic characterization of the aquifer system underlying the STP site,

- a complete discussion of data requirements and sources for the characterization,

- development of a two-dimensional ground-water flow and contaminant transport model for the site,

- a baseline pre-mitigative analysis of radionuclide transport, and

- a limited evaluation of the effect of selcted engineered barriers and hydraulic barriers on radionuclide transport.

On the basis of the case study results it's concluded that ground-water contaminant mitigation would not be necessary at the STP site to prevent discharge of radionuclides into the Colorado River. This is primarily due to the naturally low hydraulic gradient and associated long travel times. Nevertheless, for demonstration purposes, mitigative stategies are evaluated for their impact on contaminant transport. Results show that the techniques evaluated (i.e., a low permeability cutoff wall placed upgradient from the plant, a low permeability cutoff wall placed downgradient from the plant, a near-field hydraulic barrier, and a far-field hydraulic barrier) significantly increase ground-water travel times. Increased travel times resulting from more circuitous travel paths allow for both greater natural decay of radionuclides and increased sorption of radionuclides by the geologic host material. Hydraulic barriers appear to be more effective, in this case, than cutoffs in increasing ground-water travel times to the accessible environments. Hydraulic barriers are, however, more energy entensive and require regular maintenance and refurbishing.

Case Study No. 2

The South Texas Plant (STP) Case Study No. 2 is a continuation of analyses and results of Case Study No. 1. Building on these results, Case Study No. 2 illustrates a more comprehensive (though not exhaustive) performance evaluation and trade-off analysis of mitigative strategy conceptual designs. Also provided is a discussion of the STP site configuration and the accompanying constraints the layout of plant facilities could have on the design, construction, and implementation of mitigative measures. The STP case studies in composite are an illustrative example of a site-specific, reconnaissance level analysis and evaluation of strategies to mitigate the environmental consequences resulting from migration of radionuclides in a porous, unconsolidated geologic formation following a severe nuclear accident.

CASE STUDY OBJECTIVES

The objectives of STP Case Study No. 2 are to utilize the conceptual and numerical models developed for STP Case Study No. 1 to accomplish the following:

- evaluate performance of an extensive array of mitigation alternatives including upgradient and downgradient engineered barriers (linear, L-shaped and U-shaped) and hydraulic barriers,
- assess the sensitivity of mitigation measure performance to design characteristics such as length, distance from the source, and effective barrier permeability,
- investigate the effects of hydrogeologic characteristics (e.g., hydraulic conductivity, retardation, dispersivity, etc.) on mitigation,
- consider the importance of the STP facilities spatial configuration on mitigation measure design, and
- discuss cost as a factor in the evaluation and selection of mitigative strategies.

Taking advantage of the site characterization conducted for the first case study, Case Study No. 2 begins with a pre-mitigative flow and transport analysis and continues with a comprehensive performance evaluation of numerous mitigation alternatives and the sensitivity of their performance to specific hydrogeologic parameters. The TRANS two-dimensional ground-water flow and transport code was employed throughout the two studies and only previously published data are used. Required data that are unavailable are estimated based on the best information available and/or engineering judgment.

## STP FACILITIES DESCRIPTION AND CONFIGURATION

For several reasons, the general configuration of the STP facilities would be an important factor in the design and implementation of possible mitigative actions subsequent to a severe accident. The power station is composed of two identical pressurized water reactors (PWR). Several of the plant structures important to safe operation and shutdown of the plant are shared by both units including the cooling reservoir, makeup pumping station, spillway and blowdown facilities, essential cooling pond, emergency transformer, and switchyard. If continued operation of one unit is important, a key element of any mitigation design would be maintenance of essential functions such as reactor cooling and power transmission. For example, engineered barriers would have to be located outside the cooling reservoir and essential cooling pond to maintain their integrity as reliable sources of cooling water.

Plant structures also serve as physical obstacles that would influence location of engineered barriers (e.g., it might be difficult to construct a grout curtain or slurry wall within the switchyard). Another consideration would be the likelihood that water impounded by the cooling reservoir and essential cooling pond will be contaminated by atmospheric fallout. To prevent release of contaminated reservoir water, it would be necessary to avoid damaging the impoundment embankments during mitigation construction.

## MODEL DEVELOPMENT

The ground-water flow and transport analyses for the STP site are accomplished using a two-stage modeling approach. The first stage utilizes a course grid regional hydrologic flow model. The purpose of the regional model is to establish boundary conditions for the local model under both pre- and post-mitigation conditions. The local model is then used to simulate the ground-water system in the immediate area of the plant in greater detail. While the regional model used in this case study is the same as that developed for STP Case Study No. 1 without modification, a modified, higher resolution local model is employed.

The boundary and initial conditions for the new local model are determined directly from the regional model. The procedure followed is to first run the regional model, and then determine the potentials for the local model boundary from the regional model simulation results. Other local model parameters were also interpolated directly from the regional model including the lower shallow-zone aquifer top and bottom, hydraulic conductivities and the recharge/discharge rates from/to the upper shallow-zone aquifer.

Because of the total lack of observed radionuclide transport data at the STP site, estimates of transport modeling parameters (i.e., soil bulk density, effective porosity, retardation factor and dispersivity coefficients) are based entirely on previously published information. The values used for the local model utilized in Case Study No. 2 are essentially the same as those used in the first case study.

## PRE-MITIGATIVE LOCAL TRANSPORT RESULTS

The purpose of the severe accident pre-mitigative transport analysis is twofold:

1. quantitatively assess the need for mitigation, and

2. when mitigation is found to be necessary, provide a baseline for evaluating relative mitigation performance.

On the basis of Case Study No. 1 results, it was determined that after 1000 years subsequent to the assumed severe accident, transport of significant quantities of radionuclides was limited to a distance of approximately 2400 ft. The results also indicate that by 1000 years dilution and natural decay would reduce the maximum concentration within the contaminant plume to about 20 x $10^{-4}$ pCi/ml, a level well below the maximum permissible concentration of 0.3 pCi/ml set for strontium-90 by 10 CFR Part 20.

However, the STP still provides a vehicle for analyzing mitigation performance. Conclusions regarding the need for mitigation in Case Study No. 1 were based on the prevention of significant radionuclide releases to the Colorado River. The new local model, which encompasses a smaller area with greater spatial resolution, facilitates incorporation of an alternative objective for possible mitigation measures, i.e., containment of radionuclide contamination within, or close to the immediate plant area. The approach used to assess mitigation performance is to determine the resulting contaminant flux as a function of time at a section 800 ft downgradient from the reactor site (referred to as the breakthrough section). The effectiveness of a given mitigation measure in reducing contaminant flux serves as an index to its performance in contamination containment. For the pre-mitigated case using the modified local model the simulated travel time for strontium-90 to the breakthrough section is greater than 200 years. The maximum flux rate is about 6.2 x $10^9$ pCi/yr and decays to less than 3 x $10^5$ pCi/yr by the year 1000. The main implication from the pre-mitigated results is that on the order of 200 years are available for implementation of mitigation. Also, if site restoration were desirable prior to that time, the contamination would be limited to a distance of less than 800 ft from the plant.

## EVALUATION OF MITIGATIVE TECHNIQUES

Grout cutoffs and hydraulic barriers are identified as the most feasible techniques for the STP. A total of 28 different designs are considered in this case study including upgradient and downgradient cutoffs, one combination design and a limited number of downgradient hydraulic barriers. Keeping the mitigation objective in mind (i.e, minimizing contaminant migration from the site vicinity), the purpose of the evaluations presented are to:

1. analyze the general effectiveness of selected mitigation alternatives in limiting contaminant migration from the immediate reactor site,

2. investigate the relative importance of specific design parameters including barrier length, distance from the site, orientation to the site (i.e., centered or offset), shape (linear, L-shaped, U-shaped), permeability of grout cutoffs and upstream vs. downstream location, and

3. consider the sensitivity of mitigation performance to hydrogeologic and transport parameters such as hydraulic conductivity, dispersivity and retardation factor.

The selection of conceptual designs to be considered, while somewhat arbitrary, was based on several factors. Because of the potential for high levels of surface and subsurface contamination in the immediate vicinity of the reactor due to atmospheric fallout, it was assumed that barriers would have to be placed approximately 1000 ft away from the reactor site. A sufficient number of alternatives were considered to provide insight into the performance characteristics of a wide range of design variations. Both downgradient and upgradient cutoffs were considered while only downgradient hydraulic barriers were assessed. Also, one combination design was included which utilizes both an upgradient and a downgradient barrier. Costs, discussed in a later section, played no role in the initial design selections.

## DOWNGRADIENT PLANT CONFIGURATION DESIGN CONSIDERATIONS

The primary plant feature in the downgradient direction from the plant which might affect design and placement of mitigation measures is the cooling reservoir. There may be certain circumstances wherein, following a severe accident at one unit, the cooling capacity for the second unit would have to be maintained. Or, if the reservoir water is heavily contaminated by atmospheric fallout, for a period of time following a severe accident it might not be feasible to drain it. If either of these possibilities were the case, construction of a grout curtain in the shallow aquifer through and beneath the reservoir might be infeasible. Worker safety considerations might preclude drilling activities in and around contaminated reservoir water, and the presence of the reservoir water might greatly increase the difficulty of the drilling and injection operations associated with grout cutoff construction.

The results from the simulations for these alternatives show that, downgradient cutoffs, if constructed outside the cooling reservoir, provide no benefit and actually increase transport of radionuclides from the STP site. Therefore, if grout cutoffs are to be constructed in the downgradient direction, it will be necessary to locate them with a more centered orientation relative to the reactor site. If mitigation were delayed sufficiently to allow the potentially contaminated water in the reservoir to be disposed of or to evaporate such that dangerously high surface contamination is removed, grout cutoffs could be effectively implemented downgradient of the reactor site. Assuming this to be the case, the remainder of the downgradient grout cutoffs evaluated are conceptualized without the constraint of remaining outside the reservoir area.

The next set of alternatives considered consists of four linear cutoffs, centered relative to the reactor site, located 1000 ft downgradient. As noted above, it was assumed that the closest possible location for cutoff construction following a severe accident was about 1000 ft from the reactor. This assumption is largely arbitrary; however, it was made recognizing that in reality a limit will exist. Contrary to the previous designs, this set is intended to maximize, per unit length of linear barrier, the impact on the transport of radionuclides from the reactor site. Therefore, they are located at the assumed minimum distance from and centered relative to the reactor. Four designs are evaluated at this location, including lengths of 500, 1000, 2000, and 3000 ft.

The simulations show that the 500-ft design actually increases the flux rate with arrival of strontium-90 occurring at the breakthrough point after less than 200 years while the 1000-ft design only marginally reduces the flux rate. The results for the two longer cutoffs demonstrate that for increasing lengths beyond 1000 ft, there is substantial increase in first arrival time. Consequently, because of natural decay, the flux rates are reduced. The total flux of strontium-90 for the 1000-year simulation period is reduced from $1.3 \times 10^{12}$ pCi for the pre-mitigated case to $8.4 \times 10^{11}$, $1.7 \times 10^{10}$, and $1.2 \times 10^{8}$ pCi for the 1000-, 2000-, and 3000-ft designs, respectively.

A number of other downgradient cutoff designs were evaluated, varying design length, distance from the plant, orientation, and shape. The most significant of these parameters in improving performance is cutoff shape. Designs utilizing barriers to flow in both directions (i.e., L-Shape and U-Shape) were the most effective per unit length of cutoff. In fact, these designs practically eliminated strontium-90 flux for the 1000-year simulation period and delayed any flux at the breakthrough section (800-ft downgradient) until after 800 years.

The final set of downgradient designs evaluated are three injection schemes. The schemes consist simply of an injection well (or wells) located directly 1000 ft downgradient of the reactor site. The total injection rates for the three schemes are 20, 30 and 40 gpm, respectively. The impact of injection on the potential field is creation of a mound which increases in magnitude and areal extent with increasing injection rate. The mound for the 20-gpm injection scheme is barely discernible while that for the 40-gpm scheme is quite prominent. For the 40-gpm case, though some spreading has occurred, the majority of the plume is contained upgradient of the hydraulic barrier. In general, the contaminant flux rates produced by the injection schemes show increasing effectiveness in containing the strontium-90 with increasing injection rate. All three schemes reduce flux relative to pre-mitigation. Doubling the injection rate from 20 to 40 gpm delays the first arrival from about 350 years to about 450 years, decreases the maximum flux rates by over two orders of magnitude, and reduces the total flux from $3.8 \times 10^{10}$ pCi to $2.8 \times 10^{8}$ pCi.

UPGRADIENT PLANT CONFIGURATION DESIGN CONSIDERATIONS

The upgradient plant features that may directly influence mitigation design at the STP, with the exception of the buildings in the immediate plant area, are the essential cooling pond and the switchyard. The concerns associated with the cooling reservoir are equally applicable to the pond (i.e., the water may become highly contaminated due to atmospheric deposition and/or it may be necessary to keep the pond functional in the near term following a severe accident). Similarly, the the operability of the switchyard facilities may be required under post-accident conditions, or the facilities and their foundations may simply be obstacles to convenient or expedient cutoff construction. Therefore, because of these concerns, the primary location for upgradient cutoffs was selected just upgradient of the essential cooling pond (approximately 800 ft upgradient of the reactor), maintaining integrity of the pond and minimizing interference with the switchyard. A secondary location approximately 1800 ft upgradient of the reactor was also evaluated. At these locations there are no constraints on cutoff placement.

While downgradient cutoffs serve to increase the travel path length and reduce the potential gradient, upgradient cutoffs accomplish only the latter. However, they do offer an advantage in that because they are upgradient, the possible hazards associated with contaminated ground water are greatly reduced. For the same reason it might also be possible to initiate upgradient cutoff construction sooner than downgradient alternatives following an accident.

The set of upgradient cutoffs evaluated in detail are centered and located 800 ft from the reactor. Lengths for the five designs are 500, 1000, 2000, 3000, and 4000 ft, respectively. This location has hydraulic conductivities of about 1100 gpd/sq ft, values considerably higher than where the down-gradient cutoffs were located (e.g., 600 gpd/$ft^2$ at y-coordinate 50,450 ft). Consequently, the impact of the cutoffs on hydraulic gradients, for a given length, is attenuated relative to the downgradient barriers. In terms of performance, as with the downgradient cutoffs, flux from the reactor site was actually increased for cutoff lengths less than 1000 ft. Likewise, for longer cutoffs, increased length produces decreased flux. From the limited analysis conducted, it appears that for a given length, downgradient cutoffs are more effective than upgradient cutoffs in reducing contaminant flux from the site. However, with regard to future efforts to remove all contaminated soil, it's noteworthy that because the upgradient barriers do not obstruct the flow (i.e., they simply reduce gradient and velocity), less lateral spreading of contaminant occurs.

The only combination design evaluated consists of a 1500-ft cutoff located 1000 ft downgradient of the reactor and a 1500-ft cutoff placed 800 ft upgradient. The two barriers, though the same length, produce different head drops, demonstrating the influence of hydraulic conductivity on design performance.

The combined effect of the two cutoffs is to produce a gradient at the reactor site of $1.5 \times 10^{-4}$ ft/ft. The first arrival of contaminant is delayed to between 400 and 500 years and the peak flux rate is reduced over two orders of magnitude relative to the pre-mitigated case.

## SENSITIVITY ANALYSIS: CUTOFF DESIGN PARAMETERS

One of the keys to identifying a "best" design, given specific design objectives, is to develop insight into how performance changes with variations in major design characteristics. Mathematical models are ideally suited for accomplishing this in that once a model is developed for a site, any number of designs can be simulated without appreciable additional cost. Though there is no attempt to select an optimum design in this study, for demonstration purposes a limited sensitivity analysis of cutoff design parameters is conducted. The approach used is to compare mitigation performance of different alternatives as a function of selected design parameter values. The parameters evaluated include length, distance from the reactor, cutoff permeability, shape and orientation. Performance is measured on the basis of the strontium-90 flux at the breakthrough section 800 ft downgradient of the reactor. In an actual mitigation design situation, many more simulations than are presented here would be conducted, perhaps in conjunction with optimization techniques, to select the most appropriate design.

## CUTOFF LENGTH

The general effect of increasing cutoff length is to decrease velocities in both the upgradient and downgradient directions. It follows that velocities approaching zero could be achieved with a sufficiently long cutoff. However, infinitely long cutoffs are not necessary or practical. The simulation results show that significant reductions in flux can be achieved with cutoffs of several thousand feet. The question then is how long of a cutoff should be constructed? This question must be answered based on specific performance criteria. As an hypothetical example, it might be determined that site restoration is planned 300 years following a severe accident at the STP; consequently, specific mitigation performance objectives might be to extend the first first arrival time at the breakthrough section to greater than 300 years. For the pre-mitigation case (cutoff length equal to zero) the first arrival of strontium-980 occurs at approximately 250 years. For both downgradient and upgradient designs, for lengths less than 1000 ft, the arrival time is actually shortened. For downgradient cutoffs with lengths greater than 1000 ft, there is an approximate increase in first arrival time of 150 years for a 1000-ft increase in length. The upgradient designs, which are less effective for the STP, produce an average increase of less than 50 years for each 1000-ft increment over 1000 ft. For a first arrival of 300 years, the minimum acceptable linear cutoff is approximately 1500 ft for a downgradient design and 2500 ft for an upgradient alternative.

## CUTOFF DISTANCE FROM THE REACTOR

Selection of the exact location for construction of mitigative measures will be determined by a number of factors such as the nature and location of plant facilities, the level and extent of both surface and subsurface contamination, prevailing wind conditions and the type of design. Therefore, in selecting a final design it's important to understand how distance from the contaminant source will affect the performance of designs in question. For both the downgradient and upgradient designs, an increase in distance from the source significantly impacts performance, both in terms of flux rate and first arrival time. For the downgradient designs, an increase in distance from 1000 to 2000 ft reduces the first arrival time by about 200 years while increasing the initial flux rate by approximately three orders of magnitude. The same increase in distance for the upgradient designs produces a 100-year increase in arrival time and an order of magnitude increase in flux rate. Though limited in scope and extent, this brief analysis points out the importance of distance from the site to mitigation performance and the potential trade-offs that could be made between distance and cutoff length.

## GROUT PERMEABILITY

An important aspect of grout cutoff performance as a barrier to groundwater contaminant migration is the grout permeability, in terms of both the "as constructed" condition and the change in permeability with time. For simplicity, all of the grout cutoff flow and transport simulations discussed assume cutoff permeabilities equal to 0.0 gpd/sq ft. Under actual conditions it's not realistic to expect total permeability reduction. Laboratory tests with silicate-based grouts achieved permeability values averaging approximately 4.8 x $10^{-7}$ cm/sec or about 0.01 gpd/sq ft. Chemically grouted sands exhibit permeability reductions relative to the host media of three to six orders of magnitude. For the STP site this would indicate possible values on the order of about 0.001 to 0.1 gpd/sq ft.

To gain insight into how variable or deteriorating cutoff permeability may affect performance, four simulations are conducted varying the cutoff permeability. The analysis is based on the 3000-ft design located 1000 ft downgradient of the reactor with permeability values of 0.001, 0.01, 0.1, and 0.1 gpd/sq ft. The 0.01 and 0.1 permeability cases show only minor increases in flux. The increase to 1.0, however, results in a dramatic change, increasing the maximum flux rate from about 1 x $10^6$ to almost 6 x $10^7$ pCi/yr. The time of first arrival is also reduced about 200 years. Nonetheless, the simulation results show that even with appreciable deterioration of the grout's permeability reduction properties (i.e., over a couple orders of magnitude) significant reduction in contaminant flux is achieved. Also, over the ranges of achievable permeability reduction (i.e., 0.001 to 0.1 gpd/$ft^2$) performance is essentially unchanged.

## CUTOFF SHAPE

It's quite evident that perhaps the single most important design parameter, with regard to reduction of flux, is design shape. For the purpose of evaluating relative performance, several designs are compared including a linear or straight line, U-shape, L-shape and an upgradient/downgradient combination. All four designs consist of a total cutoff length of 3000 ft and are 1000 ft downgradient from the reactor. The L-shape design has the leg section attached to the west end (high conductivity region). The combination design consists of two 1500-ft sections, one located 1000-ft downgradient and the other 800 ft upgradient. Comparison of the 1000-year flux rates for the four designs shows the dramatic differences in performance. Clearly, the L-shape and U-shape designs are the best evaluated. This is attributable to the leg sections obstructing lateral flow through the high conductivity region of the contaminant flow path. The other two designs obstruct the flow and decrease the potential gradient but do not have the same beneficial effect. Also, the combination design, per unit length, is not as effective as the single linear design. This comparison points out the advantages to accurately determining the hydraulic and geohydrologic conditions of a site and tailoring the mitigation design to those conditions. Though the L-shape and U-shape designs would most likely be more difficult to construct, it appears the added difficulty would be compensated for by reduced total cutoff length for a desired level of performance.

## SENSITIVITY ANALYSIS: HYDROGEOLOGIC/TRANSPORT PARAMETERS

Perhaps the most important activity in the design of appropriate mitigation measures is the site characterization which describes the pertinent soils, geology, and hydrology. Because mitigation selection, evaluation, design, and implementation are based on understanding gained from this characterization, it follows that improving the reliability of the hydrogeologic data for the site will improve the selection of the appropriate strategy. Likewise, uncertainty in determining key site parameters can result in overestimation of mitigation performance.

In this study, as a limited demonstration of what should and can be done using numerical models, three parameters are considered: hydraulic conductivity, retardation, and dispersivity. The base case for the sensitivity studies is the 3000-ft linear cutoff located 1000 ft downgradient from the reactor.

## HYDRAULIC CONDUCTIVITY

Hydraulic conductivity is a measure of the capacity for flow through a unit area of aquifer. Estimates of hydraulic conductivity at a site are determined by testing core samples in the laboratory or by field pump tests. Typically hydraulic conductivity values are highly variable due to heterogeneities in the geologic materials of an aquifer, ranging over several orders of magnitude. In most cases the available number of pump tests is inadequate to fully characterize the distribution of hydraulic conductivities in an aquifer. The lack of fully adequate data is compensated for through model calibration, (i.e., the process of adjusting model parameters, based on understanding of the ground-water flow system, until simulated results compare favorably to observed results). Conducting the model calibration provides an appreciation for how the ground-water flow model responds to changes in hydraulic parameters. However, recognizing that calibration is an inexact process, it's just as important to gain an appreciation for the sensitivity of the transport processes to hydraulic properties of the model.

Parameter sensitivity studies are warranted based on the importance of the results and uncertainty associated with the available data. Here, a simple analysis was done to gain some understanding of how mitigation performance might be affected by hydraulic conductivities different from those assumed in the initial performance evaluations. In addition to the base case, two simulations were made, one assuming all hydraulic conductivities are 50% greater and the second assuming all conductivities are 100% greater. As expected, the adjusted hydraulic conductivities result in reduced mitigation effectiveness, increasing flow velocities, thus reducing travel times and increasing flux rates. The increase of 50% produced a decrease in first arrival time of approximately 200 years and increased the maximum flux rate by approximately two orders of magnitude. The incremental effect of increasing conductivities by an additional 50% is markedly less. The first arrival time is reduced only an additional 100 years (300 years overall) and the maximum flux rate by just over one order of magnitude (three orders of magnitude overall). The results of the analysis show that indeed uncertainties in the aquifer hydraulic characteristics could result in overestimation of mitigation performance and should be quantified and factored in to the design process.

## RETARDATION

Under ideal conditions determination of transport parameters parallels that of hydraulic parameters whereby initial parameter values are estimated from available data and are subsequently calibrated based on comparisons of field-measured and model-predicted contaminant transport. In reality, the necessary field data related to radionuclide migration are not likely to exist. Therefore, parameter estimates are based entirely on available information. In the case of retardation coefficients, their value is related directly to the equilibrium distribution coefficient ($K_d$) which is determined empirically in the laboratory and is a function of both the contaminant properties and the aquifer geologic material.

For this study the base case mitigation results (retardation equal to 46.0) are compared to mitigation results assuming three different retardation factors: 35.0, 23.0, and, as a worst-case, 1.0. Reduction of the retardation coefficient by 50%, in effect, doubles the convective portion of the transport velocity. The impact of this is evident in the increased flux rates for each of the runs relative to the base case. For each 25% decrease in retardation, there is about a 100-year decrease in first arrival time and a two order of magnitude increase in the maximum flux rate. The results for retardation equal to 1.0 (i.e., no retardation) closely resemble the source release rate, indicating that practically all of the contaminant would reach the breakthrough section within the 100-year time steps used in the transport simulations. Clearly, retardation effects are a very important consideration in mitigation design and values should be estimated conservatively.

## DISPERSIVITY

Like retardation coefficients, changes in dispersivities directly effect the rate and extent of contaminant transport. There are significant problems in considering spatial variability of aquifer hydraulic properties and their effects on field-scale dispersion processes. Given the total lack of transport data available at the STP site, the longitudinal ($D_L$) and transverse dispersivity ($D_T$) coefficients (164.0 and 8.0, respectively) were estimated based on information in the literature. Available field observations illustrate the variability of dispersivity as a function geologic material and travel distance. To evaluate the importance of dispersivity to mitigation performance at the STP, strontium-90 transport was simulated using dispersivity coefficient values 1.5 and 2.0 times that of the base case. For $D_L$ equal to 328.0 ft and $D_T$ equal to 16.0 the maximum flux rate is increased just over one order of magnitude and the first arrival time is reduced about 100 years. As noted with the previous parameters evaluated, the initial increment in change produces the greatest change in transport while subsequent value changes have incrementally less effect on simulation results. Overall, the simulations results are less sensitive to incremental variations in dispersivity than to comparable changes in retardation and hydraulic conductivity.

## MITIGATION COSTS

In the event of a severe nuclear reactor accident, the immediate concerns related to the ground-water pathway will be prevention of releases to accessible environments such as wells or surface waters. Once these concerns are alleviated, either by interim mitigation or determination that an immediate problem does not exist, the focus will likely be toward site restoration. Whichever is the case, if mitigation is deemed warranted, the selection of an appropriate strategy will be based on, in order of priority, engineering feasibility, effectiveness and cost. First and foremost, one or more feasible alternatives will be identified that meet pre-determined performance objectives. From these alternatives then, the least cost strategy will be implemented.

The incorporation of costs into the selection of appropriate mitigation measures must be based on a site-specific, detailed investigation of ground-water flow and contaminant transport in conjunction with an accurate assessment of the levels and extent of both surface and subsurface contamination at the time of construction. While cost considerations may be secondary to meeting the ultimate objective of minimizing risk to man and the environment, they may be a deciding factor in the selection of a "best" alternative.

## MITIGATION SCHEME SELECTION

Selection of appropriate mitigation techniques for ground-water contamination associated with a severe reactor accident is highly site-specific and requires thorough evaluation of the nature and extent of the contaminant release, site characteristics and feasible mitigative alternatives. Additionally, a myriad of other factors are integral to the selection process including the nature of accessible environments; worker safety during mitigation design, construction, and operation activities; costs; etc. At present there is no known way to directly integrate all of these factors and quantitatively determine an "optimal" mitigation strategy. The alternative is to address the problem systematically and methodically, using a psuedo-decision tree approach based on detailed site characterization and modeling studies. The desired result is sufficient information to initiate detailed engineering design studies of one or more recommended strategies. The key elements of the selection process can be addressed in a hierarchical fashion at four levels:

Level 1: Is mitigation required? If yes,
Level 2: Is mitigation feasible? If yes,
Level 3: Select and evaluate performance of feasible strategies.
Level 4: Rank feasible alternatives on the basis of engineering feasibility, performance, reliability, costs and other factors deemed appropriate.

## CONCLUSIONS

The South Texas Plant Case Study No. 2, using the conceptual and numerical models developed in Case Study No. 1, presents a detailed, though not exhaustive review of mitigation design alternatives. The purpose is to gain an increased understanding of how mitigation performance is related to design parameters (e.g., size, shape, permeability, location) and hydrogeologic characteristics. The numerical model proved to be extremely useful in performing the necessary flow and transport computations and facilitated evaluation of numerous alternatives within the confines of limited time and cost constraints. The model also is quite flexible in representing a range of mitigation types, sizes, and shapes (28 different designs were evaluated). General conclusions developed in the process of conducting the case study are listed below.

1. Selection of appropriate mitigation techniques is highly site specific and requires thorough evaluation of the nature and extent of the contaminant release, site characteristics, and feasible alternatives.

2. Barrier performance (cutoffs or slurry walls) is closely tied to the hydraulic characteristics of the aquifer in question. Thus, a very important aspect of mitigation design is accurate, detailed characterization of aquifer properties. Barriers improperly placed may in fact modify local ground-water velocities such that contaminant migration is increased.

3. An important consideration in mitigation design is to exploit the occurrence of natural decay as an in situ treatment process by containing contaminant releases close to the plant.

4. Downgradient designs decrease hydraulic gradients, reduce flow velocities and increase the contaminant path length. Upgradient designs serve to just reduce the gradient and velocity.

5. In general, downgradient designs produce greater lateral spreading than do upgradient designs.

6. Cutoffs constructed in low hydraulic conductivity areas create greater backwater effects than cutoffs constructed in areas having relatively higher conductivity.

7. Cutoff effectiveness decreases with increasing distance form the contaminant source.

8. Barriers which obstruct flow in both the x- and y-directions (L- and U-shaped) appear to significantly out perform linear barriers.

9. In the normal range of achievable permeability reduction (i.e., 0.001 to 0.1 gpd/sq ft) performance is relatively unchanged.

10. Understanding the sensitivity of a given system to the assumed retardation coefficient is very important given the uncertainty associated with determining its value and its direct impact on transport results.

11. Incorporation of costs into the selection of appropriate mitigation measures must be based on a site-specific, detailed investigation of ground-water flow and contaminant transport in conjunction with an accurate assessment of the surface and subsurface contamination at the time of construction.

Case Study No. 3--Marble Hill, Indiana, Nuclear Generating Station

The Marble Hill case study is designed to address the special features of a fractured flow system and how plant configuration effects a mitigative strategy. The Marble Hill site also exhibits a greater degree of hydrologic diversity than the South Texas Plant. The ground-water flow system is anisotropic with highly variable hydraulic properties and contains multiple flow units. The ground-water flow direction, geologic units transporting the contaminant and predominant discharge location(s) are not apparent from an inspection of site data. The transport of radionuclides in a fractured system would be mainly along the preferential flow channels created by fracturing and solutioning. The characterization of these fracture flow paths is of prime importance to the determination of contaminant transport rates, flow direction and design of a mitigative scheme. Detailed examination of factors unique to fractured systems is a prime objective of this study.

Conceptual Models of Radionuclide Release and Transport

Several conceptualizations of contaminant release and flow are formed and are examined by mathematical models. Such multiple conceptualizations might occur in any first round of characterization, especially at a site where existing hydrogeologic information is sparse or the hydrogeology is complex.

A severe accident at the Marble Hill site could consist of two contaminated components: sump water and core melt debris leachate. The geology of the Marble Hill site allows two feasible sump water pathways dependent on the hydraulic head in the containment structure. The site is evaluated under two sump water assumptions: 1) a low hydraulic head release that allows contaminants to seep into the hydrologic units adjacent to the core melt debris and 2) a high hydraulic head release that forces contaminants upward into a more permeable zone near land surface. A low head release would contaminate the Osgood, Brassfield, and Saluda Formations which are collectively referred to as the lower hydrologic unit. A high head sump water release would enter the Laurel Member of the Salomonie Dolomite and is referred to as the upper hydrologic unit.

A high head sump water release is less likely than a low head release. To produce a high head release, sump water would have to travel upward along a path not created by the core melt mass. A violent vaporization of ground water under the plant would be required to fracture a pathway up into the overlying unit. A high hydraulic head release is considered because it provides a pathway to the most permeable hydrologic unit and therefore has the potential to create the most severe environmental consequences. The core debris would be subject to a downward hydraulic gradient and contaminate the lower hydrologic unit at or below the melt zone. Therefore, two hydrologic units are simulated in this case study, an upper unit for high head sump water releases, and a lower unit for low head sump water releases and core melt leachate releases.

The plant site is situated on a ground-water divide. The apparent crest of the divide passes between the two reactors for both the upper and lower hydraulic units. Under these circumstances the direction of contaminant migration is not apparent from inspection. The precise location of the discharge areas is difficult to determine because anisotropic fracture networks do not allow contaminant to travel directly down the hydraulic gradient except when the gradient and the fractures are aligned along a common orientation. As may be possible at a severe accident site, uncertainty exists as to the preferential contaminant pathway. Assuming that the direction of flow will be down the hydraulic gradient and along fracture avenue, the contaminant pathways are to the northwest into a tributary of Little Saluda Creek and toward the southeast into an unnamed creek along the Ohio River. It is conceivable that radionuclides from the two reactors would travel in separate directions or that the plume from either reactor would bifurcate and enter both surface water drainages. The direction and flow rate of contaminated ground water would be determined by accident-specific conditions that at the Marble Hill site cannot be determined prior to a core melt accident. Post-accident monitoring would be necessary to determine which unit(s) were contaminated by the accident. Four accident scenarios are chosen to demonstrate the differences in the feasible contaminant discharges. The selected cases are:

1. sump water migrating to the east in the upper hydrologic unit,
2. sump water migrating to the west in the upper hydrologic unit,
3. sump water migrating to the east in the lower hydrologic unit,
4. core debris leachate migrating to the east in the lower hydrologic unit.

## Plant Configuration

Topography and plant structures limit the available construction space for a mitigative technique at this site. The site is located on a river bluff which has steep hill slopes leading to surface water drainage into the Ohio River. These slopes are tree covered and contain cliffs at outcroppings of dolomitic units. If cost is not a concern of the construction project, the available space for interdiction can be extended through extensive site preparation. However, the short distance to the contaminant-receiving water body and the limited space for construction suggests that the mitigation scheme be designed to accomplish the performance objectives by the first system installed.

The cooling towers along the northern section of the plant also would effect the placement of mitigative and monitoring bores. Adjacent to the northern portion of the containment building, the profusion of under- and overground utilities would restrict construction access to the contaminated units. The western side of the plant is restricted by the sewage treatment plant, the associated waste water lagoon, and the steep hill sides along Little Saluda Creek. These spatial restrictions would tend to compress any mitigative scheme that relied on multiple barriers (i.e., rows of injection wells or grout barriers behind withdrawal wells). The space restrictions would also require that montioring systems would be close to engineered barrier(s) and the discharge area(s), giving a short response time to a failure of the mitigative scheme.

Hydrologic Characterization

The permeability of porous media often are lognormally distributed about an average value. The reported transmissivities for the lower unit range over four orders of magnitude have an arithmetic mean of 13.5 ft/yr. These values are not grouped about an average or median value. The transmissivities in the upper hydrologic unit have an arithmetic mean of 136 ft/yr and a data spread of three orders of magnitude. These transmissivities are also strongly skewed about the mean and do not follow a lognormal distribution. The large spread of values requires that the characterization be more detailed than a simple mean or range of extreme values. The transmissivity data for both hydrologic units are described with cumulative probability distributions.

Assuming an average transmissivity value would severely underestimate the contaminant migration along some preferential flow paths. In this case, mitigative techniques could be designed based on an improper response time and safe distance from the reactor for construction of a mitigative scheme. Assuming an upper extreme value of transmissivity would overestimate the quantity of contaminant migrating at that rate, this would severely overestimate the necessity and design basis of a mitigative technique.

The spatial limitations in the data base also influence the approach taken to characterize the site. The Marble Hill Final Safety Analysis Report (FSAR 1982) contains a large amount of hydrologic data for areas near the reactors. However, other locations along the probable pathway are not hydrologically characterized. The lack of spatial data for a significant portion of the contaminant pathway requires that additional hydrologic data be estimated or interpolated.

## Approach to Modeling the Marble Hill Flow System

The modeling approach for the Marble Hill site is based on three major considerations:

1. the desirability of conducting a discrete analysis,
2. the necessity of retaining the parameter variability of the fracture flow system and,
3. the spatial limitations of the existing data base.

The approach taken to this ground-water flow problem is to statistically retain the variability of transmissivity (expressed as an aperture width), fracture orientation and fracture length in a stochastic realization of the flow field. A stochastic representation of the flow fields based on cumulative data distributions of site parameters (i.e., aperture width, fracture length and fracture orientation) can preserve the variability of permeability and anisotrophy in the system and demonstrate the key factors of transport in fractured hydraulic units. Low permeability fractures comprised of small apertures are of great importance to overall system function for two reasons.

First, the small aperture fractures are an intergral part of the fracture system interconnection. These fractures can provide critical interconnections among the larger fractures. When small apertures are the only interconnections among the larger aperture fractures they form impediments to flow and transport. Secondly, restricted flow pathways and low velocity fractures delay contaminant migration and release radionuclides to higher velocity pathways over long periods of time.

## Comparison of Strontium-90 Discharges

The results of the model simulations are summarized in Table S.4.

TABLE S.4. Summary of Contaminant Discharges.

| Case No. | Peak Flux (pCi/yr) | First Arrival (yr) | Last Arrival (yr) |
|---|---|---|---|
| 1 | $2.2 \times 10^{17}$ | 9 | 235 |
| 2 | $1.9 \times 10^{17}$ | 13 | 275 |
| 3 | $1.5 \times 10^{8}$ | 705 | 1150 |
| 4 | $1.4 \times 10^{-6}$ | 2245 | 5000 |

Contaminant breakthrough curves for this fractured system are characteristically different from the results of porous media modeling. The major items that distinguish the fractured flow system at this site are: 1) the breakthrough curves are irregular and contain time periods when all fractures discharging to the surface are swept clean of contaminants, 2) the peak flux is less than predicted by an isotropic-homogenous model caused by a portion of contaminant being delayed in low velocity pathways, and 3) the total period of a contaminant release to the environment is extended by the late arrival of radionuclides from low velocity pathways that require long time periods to reach the discharge location.

Necessity of Mitigation

The various contaminant discharge scenarios considered for examination produce a wide range of results. Clearly, the severity of a radionuclide release and the decision to mitigate the environmental consequences must be based on what constitutes an acceptable level of contaminant discharge. This study will define that discharge rate as any amount that would result in the Ohio River having concentrations of strontium-90 above the 10 CFR Part 20 limit of 300 pCi/l. This methodology is given not as an absolute technique for determining the biological hazard associated with a severe accident. Rather, it is intended to provide a somewhat realistic and relative guide to the biological hazard posed by the contaminant pathways at this site. Given the average dilution factor of the Ohio River, and the maximum allowable concentration for strontium-90, yields a maximum strontium-90 discharge rate of $3.11 \times 10^{16}$ pCi/yr.

The evaluation of strontium-90 flux discharging to the surface environment indicates that contaminant interdiction in the lower unit to protect the adjacent Ohio River would not be necessary. The upper hydrologic unit is capable of transporting sump water contaminant to the discharge area(s) at activity levels of concern.

Mitigation

Mitigation at this site could be accomplished by several means. The method selected is a grout barrier to retard ground-water flow and radionuclide transport. The location and configuration of the barrier is based on what is considered as minimal post-accident characterization and design. Upgradient and downgradient barriers are found to be effective in reducing the environmental consequences. The simulated grout barrier reduced strontium-90 concentration in the Ohio River to less than the 10 CFR Part 20 limit of 300 pCi/l. This level of mitigation is within the stated performance objective and could be improved if desired by extention of the grout barrier or coupling the grout barrier with other mitigative techniques (i.e., contaminant collection wells).

# 1.0 INTRODUCTION

## 1.1 PURPOSE OF STUDY

The purpose of this study was to evaluate the feasibility and desirability of using specific ground-water contaminant mitigation techniques to control radionuclide migration in ground water following a severe commercial nuclear power reactor accident. In the unlikely event of a core melt accident, the hydrogeology of the site serves as the final buffer between the radionuclide contaminant in the ground and the biosphere. Understanding the key factors of hydraulic transport following a core melt accident becomes paramount to an evaluation of the environmental consequence of liquid pathways. When the natural site conditions do not provide sufficiently long travel times to the surface environment, engineered barriers may have to be constructed to protect the public. The evaluation of the necessity and feasibility of mitigative strategies for ground-water contamination resulting from a severe accident is accomplished by two separate levels of analysis. The first level of analysis involves examination of core melt characteristics through an inductive process where a large volume of diverse information is reduced to a small set of generalized (i.e., generic) data concerning broad characteristics of core melt accidents, hydrogeologic properties, and ground-water contaminant mitigative strategies. The second level of analysis is more of a deductive process where insight into site-specific ground-water contaminant mitigation is developed through the performance of case studies. These two levels of analyses are complementary in nature and follow a logical progression in the development of methodology for evaluation of mitigative techniques for core melt accidents.

The generic analysis (presented in Volume 1) determines the basic geology and hydrologic factors that affect radionuclide release and transport following a core melt accident. The key hydrogeologic factors are used to classify existing and proposed nuclear power plant sites into generic groups. Evaluation of nuclear power plant sites in a generic manner provides a screening tool that determines:

1. the importance of various hydrogeologic factors related to a core melt accident,

2. the suite of mitigative techniques that are applicable to each generic classification, and

3. the relative environmental sensitivity of a generic classification to a nuclear release.

The second level of analysis (presented in Volume 2) determines the site specific aspects of a severe reactor accident and the methodology for evaluating the impact and the response. Case studies are developed to detail the individual characteristics of a site and demonstrate how these characteristics affect the evaluation of mitigative strategies. In addition to demonstration of methodological approaches to the analysis of severe accidents, the case studies address:

1. site-specific hydrogeologic conditions,
2. development of a conceptual model,
3. selection of mitigative techniques or schemes for evaluation,
4. selection of analytical procedures,
5. design and performance evaluation of mitigative schemes,
6. plant configuration aspects which affect the selection and performance of the mitigative strategies, and
7. validation of generalized results determined by generic analysis.

The generic analysis is limited by definition, in scope by the requirements for simplifying assumptions and generalizations concerning release and transport of ground-water contaminants. The case studies focus on site conditions and unique factors that affect the design of mitigative schemes. Therefore, the generic examination presents a rough approximation of the relative environmental consequences of a core melt accident and the feasibility of various mitigative techniques that could be used to interdict the resulting contaminants. This information can be used for screening purposes to help plan the necessary elements of a detailed site-specific analysis. Detailed design and implementation plans of a mitigative scheme are properly considered only within the framework of a specific site and accident scenario.

## 1.2 ORGANIZATION OF REPORT

This report is divided into two volumes. The generic or generalized analysis is contained in Volume 1, "Mitigative Techniques for Ground-Water Contamination Associated with Severe Nuclear Accidents: Analysis of Generic Site Conditions." The chapter sequence is designed to follow the processes of a severe accident. A brief description of the contents of each chapter follows:

Section 1 - presents an overview of the study including: purpose of study, report organization, study objectives, and the scope of the study with associated limitations.

Section 2 - discusses the major types of severe commercial nuclear power reactor accidents considered for this study. Chapter 2 includes discussion of radionuclide release mechanisms and rates expected following a reactor core melt accident.

Section 3 - describes the generic hydrogeologic classification scheme and presents the definition of each generic classification. Ground-water flow parameters (e.g., hydraulic conductivity, effective porosity, etc.) and contaminant transport parameters (e.g., longitudinal dispersion, retardation, etc.) are discussed.

Section 4 - identifies the various ground-water contaminant mitigation techniques and strategies that may be applicable to ground-water contamination resulting from a severe accident.

Section 5 - presents the results of the evaluation of the radionuclide flux for each generic hydrogeologic classification with an assessment of appropriate mitigation measures.

Volume 2, "Mitigative Techniques for Ground-Water Contamination Associated with Severe Nuclear Accidents: Case Study Analysis of Hydrologic Characteristics and Interdictive Schemes." A brief description of the contents of each chapter follows:

Section 6 - discusses the geologic and hydrologic conditions at the South Texas Plant. Included are simulations of premitigative contaminant migration and the mitigative benefits of a cut-off wall and injection wells.

Section 7 - continues the discussion of the South Texas Plant in greater detail. Emphasis is placed on near-field simulations, design considerations and performance assessment.

Section 8 - presents an analysis of the special features of plant configuration and hydrologic characterization of a fractured anistropic unit.

Section 9 - discusses the "Lessons Learned" and suggestions for further research.

Section 10 - presents the summary of conclusions.

Appendix A - presents a glossary of geotechnical terms used in this report.

Appendix B - provides a generalized guide to site characterization and code selection.

Appendix C - provides a description of the TRANS ground-water flow code.

Appendix D - gives a list supplemental references on contaminant mitigation.

## 1.3 BACKGROUND

Several studies related to this study in purpose and scope have been made previously completed. This study draws on these previous studies for basic definitions involving core melt accident types, reactor designs related to radionuclide releases, history of events, and characterization of a core melt accident. The previously completed studies that have influenced the direction and focus of this study are:

1. Niemczyk, S. J. et al. 1981. The Consequences From Liquid Pathways After Reactor Meltdown Accident. NUREG/CR-1596, 80-1669, Sandia National Laboratories, Albuquerque, New Mexico.

2. Office of Nuclear Reactor Regulation. 1981. Technical Bases for Estimating Fission Product Behavior During LWR Accidents. NUREG-0772, U.S. Nuclear Regulatory Commission, Washington, D.C.

3. Office of Nuclear Reactor Regulation. 1981. Preliminary Assessment of Core Melt Accidents at the Zion and Indian Point Nuclear Power Plants and Strategies for Mitigating Their Effects. NUREG-0850, Vol. 1, U.S. Nuclear Regulatory Commission, Washington, D.C. This report is alternately referenced as: PACMA 1981.

4. Office of Nuclear Reactor Regulation. 1978. Floating Nuclear Power Plants. NUREG-0502, U.S. Nuclear Regulatory Commission, Washington, D.C.

5. Office of Nuclear Reactor Regulation. 1978. Liquid Pathway Generic Study. NUREG-0440, U.S. Nuclear Regulatory Commission, Washington, D.C. This report is alternately referenced as: LPGS 1978.

6. U.S. Nuclear Regulatory Commission. 1975. Reactor Safety Study - Appendices VII, VIII, IX, and X. WASH-1400 (NUREG 75/014), U.S. Nuclear Regulatory Commission, Washington, D.C. This report is alternately referenced as: RSS 1975.

7. Houston Power and Light. 1978. Final Safety Analysis Report, South Texas Units 1 and 2. Houston, Texas. Also referred to as STP-FSAR.

8. Public Service of Indiana. 1976. Final Safety Analysis Report, Marble Hill Nuclear Generating Station Units 1 and 2. Madison, Indiana.

The conclusions of this study are predicated on the results of these previously completed studies of reactor safety and consequences of a severe nuclear power plant accident.

Throughout this report several terms or phrases are used interchangeably to denote a severe power plant accident (e.g., core melt accident, severe accident, reactor core accident, core meltdown, etc.). The phrase "severe accident" is most encompassing of the conceptualizations of the problem addressed by this study. Within the context of this study a severe accident is considered any extraordinary sequence of events involving the breach from the reactor containment of significant amounts of the reactor core radionuclide inventory which subsequently contact the subsurface environment. This accident definition includes both a molten core melt-through of the containment basemat and/or a significant sump water release through a cracked or otherwise damaged basemat.

## 1.4 STUDY OBJECTIVES

The objectives of this study are:

- identification of hydrogeologic factors that affect the release and ground-water migration of radionuclides following a severe or core melt nuclear accident,

- evaluation of the feasibility and desirability of interdicting radionuclide contaminants to mitigate environmental consequences based on a generic hydrogeologic classification of power plant sites in the United States,

- development and demonstration of methodology for the characterization and evaluation of contaminant transport and the necessity of contaminant interdiction in contrasting geologic environments, and

- development and demonstration of the methodology for evaluation of the feasibility, design, implementation and performance assessment of mitigative schemes on a site-specific basis.

These objectives are accomplished with data from previously published literature and responses from geotechnical engineering firms and government agencies to a letter survey of expertise and experience. Field level studies and primary data collection efforts were entirely beyond the scope of this study.

The intent or purpose of this study is to neither verify nor repudiate previous studies on which this current effort is based. However, judgment is exercised in acceptance of the information provided in previously completed reports. For example, the hydrogeologic classification scheme is based primarily on data provided by S. J. Niemezyk in an unpublished report by Oak Ridge National Laboratory.

Some of the early Final Safety Reports (FSAR's) and Preliminary Safety Analysis Reports (PSAR's) do not contain an extensive review of geologic site conditions. Consequently certain hydrologic parameters had to be estimated. The content of the hydrogeologic data base was reviewed in three respects prior to acceptance for use in this study:

1. a spot check of values was made based on FSAR's available at Pacific Northwest Laboratory,

2. hydrologic values were examined for "reasonableness" in the context of the geologic classification, and

3. the ground-water velocities determined from the hydrologic parameters were examined for unrealistic results. In a few cases combined conservative values and/or gradients based on possibly perched water tables resulted in unrealistic ground-water velocities.

In these instances the hydrologic parameters were adjusted with the maximum ground-water velocity restricted to less than 75 m/day. One site was removed from the data base because all hydrologic parameters were extreme values. In addition, four sites where the core melt would reside in the partially saturated zone above the water table were excluded from analysis. These data, pertaining to the hydrogeologic properties of commercial reactor sites, were considered acceptable for use in a generalized manner in this study for the following reasons:

1. the data were compiled from PSAR's and FSAR's that have been reviewed by NRC,

2. limited comparisons of the data with information contained in individual FSAR's indicate the data are reasonable estimates,

3. the data pertain to actual reactor sites in the United States. Therefore, these data may reflect peculiarities in hydrogeologic properties that may be unique to nuclear power plant sites because of siting restrictions. Such peculiarities would not be evident by simply assuming general properties for various geologic unit types,

4. the data used to develop, from a statistical perspective, generic attributes that can be grouped in a few general categories, and

5. the data represent the most thorough and complete description of hydrogeologic site conditions at nuclear power plant sites in the United States.

This study is intended to provide U.S. Nuclear Regulatory Commission staff, and other interested parties, guidance in making reconnaissance level estimates for the urgency and necessity of mitigating the effects of a severe nuclear power plant accident on ground-water quality. The study also provides reconnaissance level information on the feasibility and constraints on feasibility of implementing a wide range of potentially applicable ground-water contaminant mitigation schemes.

## 1.5 PROJECT SCOPE

Because of the multidisciplinary nature of the informational requirements of the project objectives, several independent, but functionally related, tasks were performed in order to provide a thorough and sufficiently detailed analysis. These tasks include:

1. evaluation of contaminant release following a severe nuclear power plant accident,

2. classification of nuclear power plant sites based on the hydrogeologic regime.

3. analysis of radionuclide transport in ground water,

4. identification and evaluation of ground-water contaminant mitigation techniques,

5. determination of feasible mitigative techniques for specific hydrogeologic classifications, and

6. performance of case study analysis of the effectiveness of feasible mitigative techniques.

Only contaminant mitigation schemes that directly affect the long-term environmental consequences by active and/or passive interaction with the contaminant are considered in this study. Contaminant mitigation schemes that involve redesign of reactor containment structures or direct manipulation of reactor core materials (e.g., in situ vitrification or core debris removal) are not within the scope of this analysis. Also not within the scope of this project are issues related to site restoration and health effects of long-term, low-level radionuclides to surface water. In addition, it is assumed that contaminated ground-water supplies would no longer be used.

Figure 1.5-1 is a schematic diagram of the principal technical elements of the first phase of this study (i.e., the generic analysis). The interactions and interdependencies of the technical elements of the first phase are also presented in the figure. The generic analysis is basically one of decreasing specificity. An intensive review of literature pertaining to postulated core melt features, hydrogeologic site conditions of nuclear power plant sites, and ground-water contaminant mitigative techniques is conducted. Based on the review, a vast amount of information is reduced to generalized guidelines concerning hydrogeologic properties of nuclear power plant sites, radionuclide release and transport following a severe accident, and feasible mitigative strategies for resulting ground-water contamination. The generic analysis does not provide sufficient detail required to describe individual sites and such was not the intent of the first phase of this study.

The case study level of analysis, as schematically presented in Figure 1.5-2, complements the generic analysis. The case studies are designed to highlight differing aspects of site-specific considerations and methodologies that are required to evaluate the necessity and feasibility of implementing ground-water contaminant mitigation following a severe accident. The site-specific framework of analysis is one of increasing specificity with increasing detail in the hydrogeologic characterization, radionuclide transport, and evaluation of the performance of mitigative strategies. In summary, the first phase (i.e., generic analysis) of the study is designed to provide broad general information concerning severe power plant accidents and the interdiction of contaminants entering the ground-water pathway. The second phase (i.e., case study analysis) of the study is designed to demonstrate, to the extent possible, methodologies and approaches to the analysis of a severe power plant accident and ground-water contaminant mitigation at a specific site. The scope of each chapter is presented below.

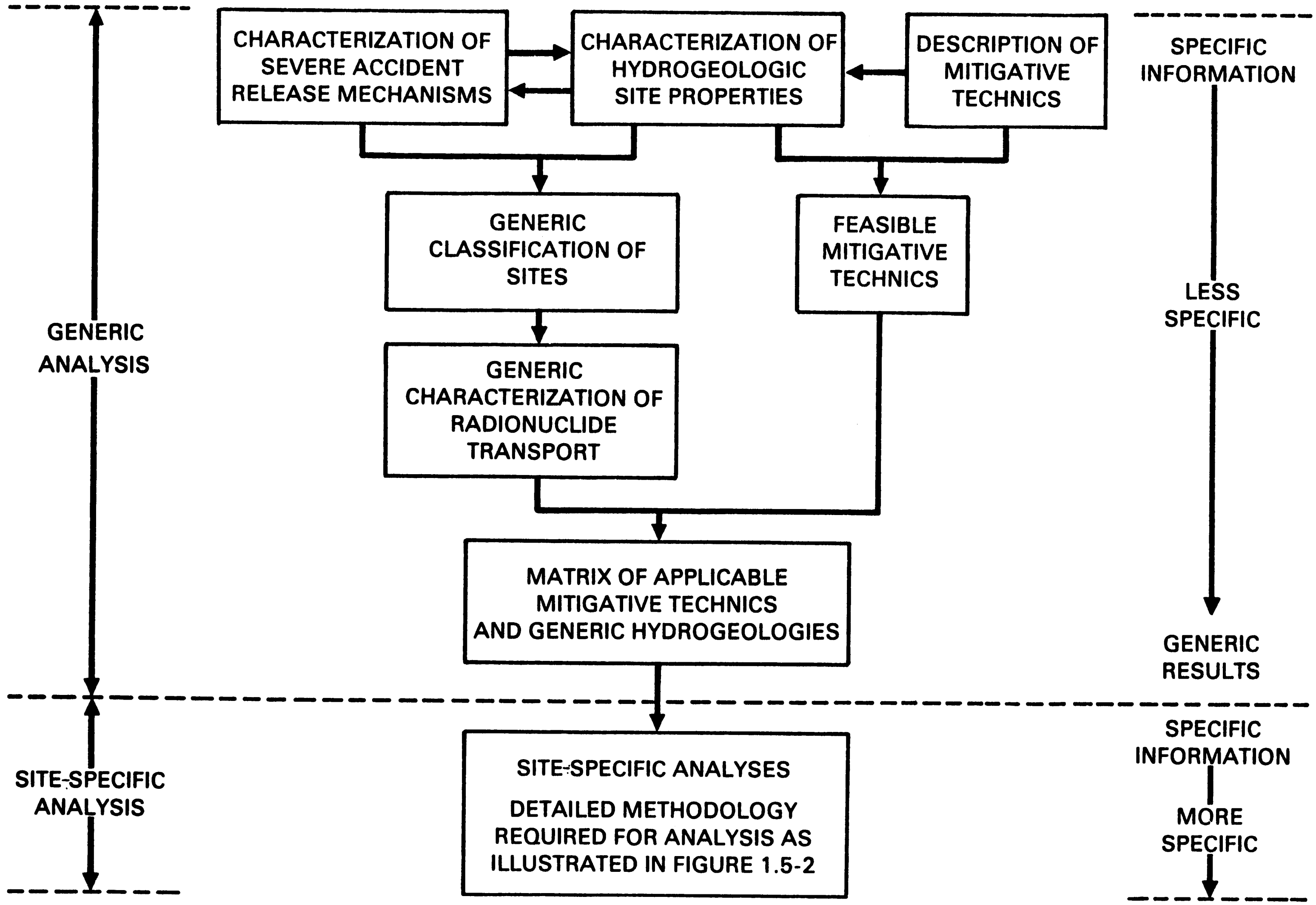


FIGURE 1.5-1. Schematic Diagram of Phase 1 Study Components and Interrelationships for the Generic Analysis

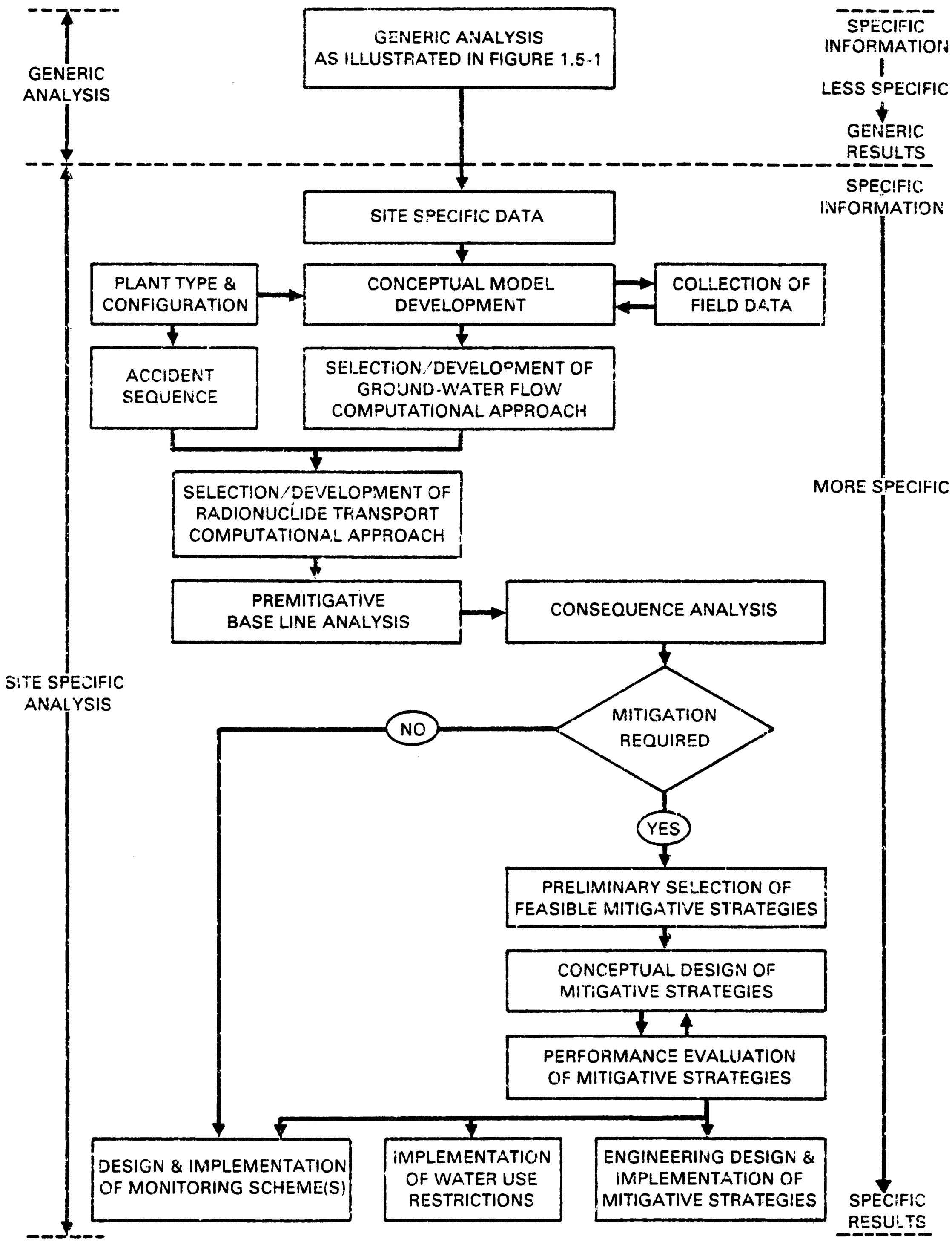


FIGURE 1.5-2. Schematic Diagram of Phase 2 Study Components and Interrelationships for the Site-Specific Analysis

### 1.5.1 Release of Contaminant into the Ground Water Flow System

The release of contaminant following a severe nuclear accident is a complicated site- and accident-specific event. The term "contaminant release" is used in reference to the release of contaminant by leaching, or by flow of reactor sump water into the ground-water system. Contaminant residing below the containment structure in the solid phase of the core debris and not being transported by ground water is not considered to be released.

The release rate, as well as the quantity, is important in characterizing the release of contaminant. This is especially true for accident situations where the environmental consequences are evaluated in terms of the time and spatially dependent concentrations or population doses. The dominant mechanism controlling the time-dependent discharge of core melt leachate to surface water is not found in the characterization of transport (i.e., dispersion) but rather in the much greater effect of solid material leaching and hydraulic restrictions of a liquid release. In describing the release of contaminant following a core melt accident, the goal is to quantify the major characteristics that control the time dependence of contaminant migration to an accessible environment. Conservative estimates for parameters that govern the release mechanisms allow conservative yet realistic examination of the consequences of a core melt accident.

The release mechanisms that would liberate contaminant to the ground water are not precisely described by basic assumptions and simple processes. There are a host of inner-dependent chemical, hydraulic, thermal and morphological reactions that would control the release rate of contaminant into the ground-water flow system. This study has concentrated on the long-term, far-field effects of a core melt accident and, as such, does not address the following:

- less than full core melt penetration of the containment basemat,
- partial saturation or resaturation adjacent to the core melt debris,
- transient thermal effects on melt debris leach release rates and sump water flow rates,
- less conservative assumptions of core debris morphology (i.e., specific surface area),
- multi-component contaminant release mechanisms (i.e., diffusion and corrosion),
- mixed chemical melt types (e.g., melts composed of partially silicic and partially calcine material), and
- multi-phase flow of super-heated gases and water away from the core debris.

### 1.5.2 Hydrogeologic Classification of Nuclear Power Plant Sites

The classification of nuclear power plant sites is based on the concept that there are common physical characteristics that control the release, transport, and interdiction of contaminants in ground water. Combining the criteria for classification of contaminant release and contaminant transport with feasible mitigative measures results in a matrix of generic hydrogeologies versus mitigative actions. The classification system serves to define the generic characteristics of a severe accident (i.e., magnitude, transport rates) and the techniques that may be used to achieve a potentially necessary reduction in environmental hazard.

The scope of the generic classification system has been constrained to a workable number of categories such that a representative number of sites are in each category. The development of a generic hydrogeologic classification is limited by the hydrogeologic data base on which the classification scheme is made. The hydrogeologic descriptions are based on data extracted from PSAR's and FSAR's. The extent of the geologic/hydrologic description of contaminant pathways varies among the reports. This does not, however, imply that the original data or its extraction by other researchers from the safety reports are in error. It should be recognized that the data were compiled for the purpose of conducting generic studies and do not possess the resolution that would be expected from a detailed examination of a specific site. For example, the geologic description portion of the classification scheme is often based on one word descriptions of the materials found beneath the containment structure at a site (e.g., sandstone, clay, limestone). Layered media of different compositions or geologic changes along the contaminant flow path are indiscernible in these circumstances. Consequently, the recognized limitations of the generic hydrogeologic classification scheme are:

1. geologic descriptions in the hydrogeologic data base are limited in detail and apply only for the materials under the containment structure, and

2. geology of multi-layered sites are classified based on the predominant unit. Variations in geologic materials can cause unique contaminant transport characteristics and require special or composite mitigative strategies.

### 1.5.3 Analysis of Radionuclide Transport in Ground Water

For the analysis of radionuclide transport in ground water, a one-dimensional transport analysis is conducted for each individual existing and proposed commercial nuclear power plant site in the United States. The results of the analyses are "lumped" or categorized according to the hydrogeologic regime into generic classifications. Transport analysis on individual nuclear power plant sites when examined as hydrogeologic groups provides considerable insight to the generic factors of a core melt accident. This approach has distinct advantages over an approach based solely on a series of average or representative conditions:

1. hydraulic transport parameters may have a negative correlation (e.g., gradient and hydraulic conductivity) and "average" values may not occur at actual sites,

2. mass transport equations for radionuclides are nonlinear and average values may not produce an "average" or representative contaminant discharge to an accessible environment,

3. the range of hydraulic and transport factors can cover several orders of magnitude even in a single generic hydrogeologic classification. Bracketing the feasible range of several key parameters can create more transport scenarios, producing a broader range in results, than found at actual sites,

4. the variations in contaminant transport that can be expected within each classification are contained within the hydrogeologic data base and can be carried through the analysis,

5. actual site data may contain associations and correlations unique to nuclear power plants due to siting requirements that could be masked by incorporating averaged or assumed data for similar materials existing elsewhere.

In this manner, the generic characteristics of release, transport, and discharge to the environment are analyzed. The hydrogeologic data base contains sufficient information for a one-dimensional transport analysis at each site.

However, there are certain limitations to this approach. The degree of modeling accuracy is less than if an exhaustive site and modeling study were conducted for each site. The description of spatially dependent hydraulic characteristics (i.e., hydraulic conductivity and effective porosity) by a single representative value permits transport calculations of an approximate nature. The accuracy of the transport analysis is a function of how well each site can be described by single dimensional parameters.

The hydrogeologic data base for nuclear power plant sites used in this study is a combination of measured, extrapolated, and estimated parameters. When hydraulic data were unavailable (e.g., effective porosity) conservative values, that is, values that are somewhat biased toward producing more rapid contaminant transport were selected. Successive conservative estimates of hydrologic parameters used in transport analyses can produce overly pessimistic results. Specifically, the hydrogeologic transport analysis is limited by:

- one-dimensional, saturated, steady-state transport at each site,
- a single hydrologic unit is considered,
- the discharge of contaminant is assumed to be at the nearest surface water body,

- hydrologic conditions are assumed to be the same as when the power plant was constructed,

- hydraulic spreading of contaminant during sump water release is not considered,

- data deficiencies (i.e., dispersivities and effective porosities) are filled by estimations based on judgment.

These limitations are most sensitive to the analysis at an individual site. However, when analyzed as a generic group, the individual variations are less important and the central tendencies (if present) serve as the generic descriptor. If the analysis results show a broad spectrum of contaminant discharge rates and arrival times at surface water bodies, then the analysis is useful in that it becomes known that the site-specific factors (i.e., distance to environmental contact) are more important than generic hydrogeologies.

### 1.5.4 Identification and Evaluation of Ground Water Contaminant Mitigation Techniques

The goal of the identification and evaluation of ground-water contaminant mitigative techniques is to provide a detailed description of feasible methods for controlling and/or reducing ground-water contamination in various geologic environments. Each mitigative measure is described in terms of:

1. design considerations,
2. construction considerations,
3. performance considerations, and
4. implementation considerations.

Information provided in this report serves as a guide to feasible mitigation schemes and discusses their advantages and limitations in comparison with each other and in relation to the geologic medium in consideration.

Considerations not included in the analysis of mitigative techniques are:

- multi-layered systems of very different properties requiring a unit by unit evaluation of feasible mitigative measures,

- complex hydrogeologic environments where spatial changes in material properties require a strategy of multiple mitigative measures,

- site-specific restrictions to access at desired distance from accident site (i.e., topography and existing structures),

- mitigative measures interacting directly with the core debris (i.e., in situ vitrification, injection of sorbing agents along core debris, or removal of core debris).

The determination of engineering feasibility of the various mitigative schemes requires an in-depth evaluation of implementation considerations which include:

1. installation time,
2. construction cost,
3. equipment mobilization,
4. toxicity of chemical treatments, and
5. worker safety.

Unfortunately, these issues are highly site specific and site sensitive; especially worker safety. Thus, great difficulty arises in analyzing these issues in a generic manner. This study identifies these issues and as far as possible, within the generic context of the study, describes their implications in regard to the feasibility of each mitigative technique. To go beyond the level of information provided in this report would be unfounded and potentially misleading and inaccurate within the current scope of the project.

### 1.5.5 Determination of Feasible Mitigative Techniques for Specific Hydrogeologic Classifications

The approach taken for the determination of hydrogeologic sites versus feasibile mitigative techniques is to couple the geohydrologic information pertaining to the generic sites with information compiled on appropriate geologic properties for mitigative technique feasibility. The coupling is based on the range of conditions for which the mitigative technique is designed and the hydrogeologic characteristics describing each generic site. As a result, a practical guide to feasible mitigative techniques with limitations on their feasibility in each generic geologic environment is provided.

### 1.5.6 Case Study Analysis

A series of three case studies are presented in this report. The case studies describe the methodological approach necessary to perform a reconnaissance level assessment of the need and feasibility of implementing mitigative actions at selected commercial reactor sites in the United States. The case studies are not intended so much to answer concerns regarding specific courses of action at the selected sites as to develop and demonstrate a methodology for evaluating mitigative alternatives in a site-specific manner. The methodology must be broad-based because of the complicated nature of the problem involved in evaluating the suitability of various mitigative techniques. In general, the methodology can be subdivided into two components: 1) the ground-water system which dictates the need for and acceptability of mitigative actions; and 2) the plant configuration and accident scenario which dictate, in a large part, the feasibility of implementing mitigative actions. The case studies are designed to focus attention on different aspects of these two components with the hope that in composite they will provide insight into the overall approach necessary for evaluation of the broad range of issues involved in determining the necessity, feasibility, suitability, and implementability of mitigative techniques for ground-water contamination following a severe power plant accident.

The determination of an appropriate method to interdict ground-water contaminant and the design of engineering structures can only be made at the case study level of analysis. The core elements of the three case studies are given in Table 1.5.6-1.

## 1.6 REALISTIC VERSUS CONSERVATIVE ANALYSES

The analysis of the consequences of nuclear accidents is often of a conservative nature. Simplifications and estimations are made such that underestimation of the consequences is unlikely. For many aspects of a consequence analysis, this approach is valid. However, when a complex series of interrelated events are examined, as in the case of a simulation of a core melt accident, successive estimates that are conservative can affect the realism of the analysis. In extreme cases very conservative analyses produce physically impossible results.

TABLE 1.5.6-1. Case Study Topics of Emphasis

| Case Study Number | Name | Topics of Concentration |
|---|---|---|
| 1 | South Texas Plant | Unconsolidated hydrologic unit, hydrogeologic characterization, evaluation of mitigative methods. |
| 2 | South Texas Plant | Performance assessment, cost effectiveness, optimization of mitigative scheme. |
| 3 | Marble Hill, Indiana | Consolidated and fractured hydrologic unit, anisotropic flow field, plant structures. |

For the analysis of core melt accidents, a balance must be made between conservatism and realism. The analysis of the environmental consequences and the need for ground-water contaminant mitigation must be based on a realistic examination of the situations. Overestimation of the amounts of contaminant by many orders of magnitude may not provide a proper basis for an evaluation of mitigative measures. At the same time however, it must be recognized that underestimation of the consequences of a core melt accident is far less desirable than an overestimation of the consequences. This study follows a conservative yet realistic approach, to the degree possible. When phenomena can be simulated at realistic levels of expectation, conservatism is avoided. However, for analysis of events subject to large uncertainties (i.e., leach rates) conservatism is preserved.

## 1.7 UNITS OF MEASUREMENT

Metric units are the authors' preferred reporting format and are used in the generic analysis of Volume 1. However, site studies (i.e., Final Safety

Analysis Reports) are reported in English units. So that this document is compatible with existing site studies at South Texas Plant and Marble Hill Nuclear Generating Station, English units are used to report results in Volume 2 case studies.

# 2.0 DESCRIPTION OF CORE MELT RELEASE OF RADIONUCLIDES

## 2.1 INTRODUCTION TO CORE MELT ACCIDENTS

The accident sequence and the type of nuclear power plant affect the amount of radionuclides that may be released during a severe accident. Commercial nuclear power plants utilize either a pressurized water reactor (PWR) or a boiling water reactor (BWR). Fundamentally, BWR's use one coolant loop with water flowing through the core allowed to boil and flow directly to the turbine-generator. In contrast, PWR's have pressurized water in a double loop incorporating a steam generator. The two reactor types exhibit different characteristics related to core melt accidents. Reactors of either design are capable of undergoing a severe accident in which the reactor core containing nuclear fuel and support materials overheats to the point of melting (RSS 1975). The resulting molten mass could contain sufficient heat to subsequently penetrate (i.e., melt through) the reactor containment structures and enter the geologic strata beneath the power plant (RSS 1975). After sufficient cooling, ground-water flow would contact the core melt debris and initiate the hydraulic transport of radionuclide contaminants away from the site. Pressurized water reactors could also release a significant amount of contaminated water through the melted opening in the containment structure (RSS 1975). The water would originate from reactor cooling water and from the operation of emergency sprays during the accident sequence. The water would be exposed to core materials and would contain a portion of the radionuclide core inventory. This liquid would collect in the reactor sump and could be released as "sump water" during a core melt accident. Severe nuclear power plant accidents of both types (i.e., core melt-through and sump water release) are illustrated in Figure 2.1.1-1. A core melt accident involving the penetration of the containment structure has never occurred. Therefore, the hypothesized sequence and impact of core melt events contain varying degrees of uncertainty related to the size of the radionuclide release, leach rate, and ground-water transport.

### 2.1.1 Definition of Core Melt Accident

In the context of this study an "accident" refers to an unplanned sequence of events which leads to the release of fission products into the ground-water flow system. Although throughout the nuclear fuel cycle there are circumstances which could give rise to the accidental release of radionuclides attention is focused on the class of accidents which result in heating of the reactor core sufficiently to cause some form of breach in the reactor containment. This type of accident is extremely unlikely. However, a core melt accident could be the most catastrophic, in terms of radionuclide release to the environment, of all potential nuclear power plant accidents types. For this reason this investigation is limited to the class of severe nuclear power plant accidents characterized by some form of breach of the reactor vessel and subsequently the containment building basemat.

The characterization of a severe nuclear power plant accident is limited to facilities which employ light-water reactor technology since this technology is the most common for commercial reactors. Specifically, consideration is given to a core melt scenario that would pertain to a reference pressurized

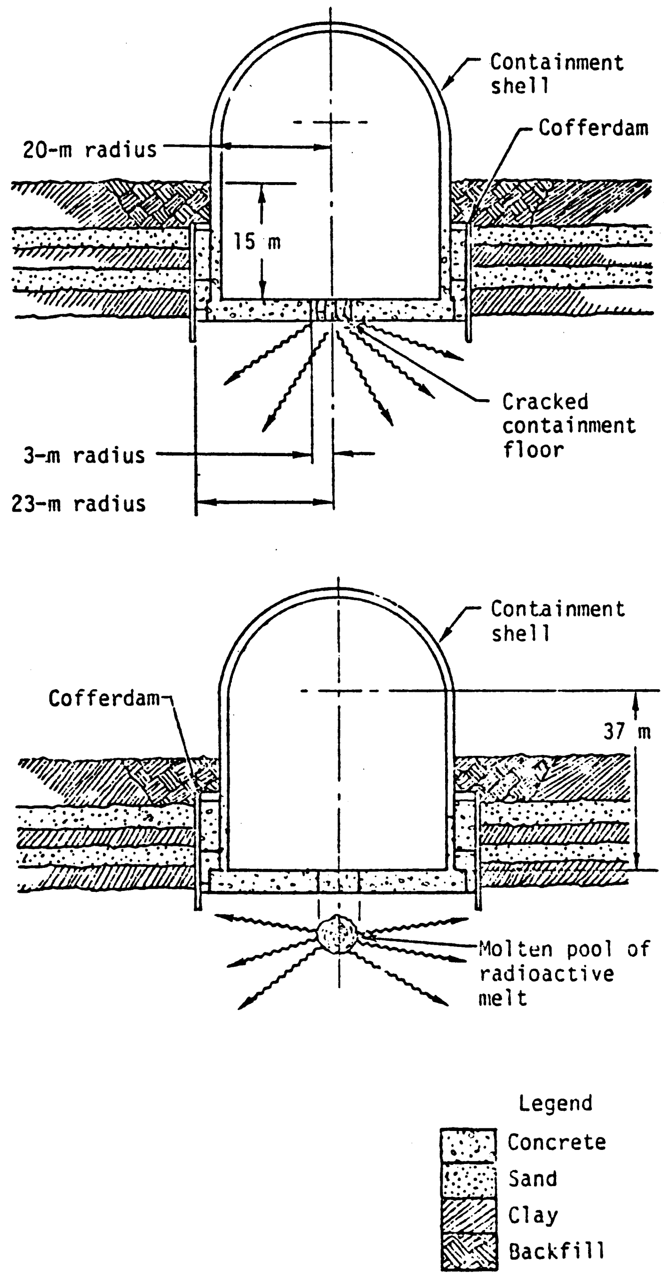


FIGURE 2.1.1-1. Two Possible Configurations of a Severe Nuclear Accident Involving Failure of the Reactor Basemat (Source: Pitts 1975)

water reactor or a reference boiling water reactor since these design types encompass most of the commercial power reactors either in service or planned in the U.S.

The intent of this discussion is not to provide a detailed assessment of the engineering aspects of a core melt sequence. It is intended, however, to outline, in conceptual terms, the possible causes for the initiation of a potential core melt sequence and the subsequent steps leading to the breach of the containment basemat. This outline of a severe accident should provide a common foundation for the analysis of radionuclide transport and contamination of ground water, determination of hydrogeologic site conditions, and ultimately the assessment of ground-water contamination mitigative techniques.

### 2.1.2 Causes of a Severe Accident

In order for an accident to occur in which the containment is breached there must be sufficient heat generated to cause a loss of integrity of the reactor core by either a melting or partial melting of fuel elements. Overheating of the fuel can occur only if more heat is generated in the fuel than can be removed. This circumstance can be brought about by one of two events. First, a loss of coolant will allow fuel to overheat because of the continued decay of radioactive materials in the fuel after the reactor has been shut down. Second, a heat imbalance can occur if the reactor power is increased beyond the heat removal capability of the cooling system or the cooling system capability is reduced to a level below the generation rate. This study is primarily concerned with the loss of coolant accident which results in the uncovering of the reactor core and subsequent overheating of the fuel.

This class of accident can lead to the initiation of meltdown of the reactor core if there is associated with the loss of coolant failure of the emergency core cooling system (ECCS). The most probable cause of a severe loss of coolant accident is a break in one of the main coolant loop pipes followed by operational failure of the ECCS. In the event of a severe core accident there are specific power plant barriers that must be breached in order for a significant release of fission products and hazardous chemicals to the subsurface environment. The plant barriers that must be breached, in succession, are (RSS 1975):

1. Fuel matrix ($UO_2$ pellets in most cases),
2. Fuel cladding (Zircaloy casing or tube for most plants),
3. Reactor vessel and primary system piping, and
4. Containment basemat.

During normal reactor operation the majority of radioactivity remains in the fuel matrix with a small percentage migrating to the gap between the fuel pellets and the cladding. In order for a significant amount of radioactivity to be released to the reactor vessel and primary system piping the fuel cladding must fail thus allowing direct exposure of the fuel pellets. It is assumed that if conditions exist which are severe enough (i.e., high temperature) to cause melting of the fuel cladding then the fuel pellets would also melt causing complete core melting. This assumption is conservative however, due to the higher melting point (~5000°F) of $UO_2$ than the surrounding metal. Once the above sequence of events has taken place the molten core could melt through the bottom of the reactor vessel (RSS 1975).

It is difficult to predict the physical processes that may occur as a result of a core melting loss of coolant accident. It is considered "likely" that sufficient thermal mass would be available to eventually melt through the lower concrete structure (i.e., the basemat) of the containment (RSS 1975). In this study it is assumed that the core melt mass penetrates the basemat and enters the geologic materials under the power plant.

There are several accident sequences for both PWR's and BWR's that could result in a core melt. Each of these accident sequences has a characteristic release of radionuclides. This study assumes that the most probable core melt accident for a PWR or a BWR would occur. The sensitivity of contaminant discharge to the environment to various accident sequences is discussed in Section 2.2.2. The assumed accident sequences are conservative in that: 1) PWR's would partition the radionuclides between core debris and sump water in a ratio that favors early hydraulic release; and 2) BWR's and PWR's would release the maximum amount of radioactivity into the ground-water system rather than the surface or above-ground environment.

The most probable accident sequence for a pressurized water reactor (designated PWR-7 in the Reactor Safety Study 1975) is summarized as follows:

- PWR 7 - This accident sequence involves a core meltdown due to failure in the core cooling systems. The containment sprays would operate, and the containment barrier would retain its integrity until the molten core proceeded to melt through the concrete containment basemat. The radioactive materials would be released into the ground, with some leakage to the atmosphere occurring upward through the ground. Direct leakage to the atmosphere would also occur at a low rate prior to containment vessel melt through. Most of the release would occur continuously over a period of about 10 hours. The release would involve 0.002% of the iodines and 0.001% of the alkali metals present in the core at the time of release. Because leakage from containment to the atmosphere would be low and gases escaping through the ground would be cooled by contact with the soil, the energy release rate would be very low.

The most probable accident of a BWR core melt accident (designated BWR-3 in the Reactor Safety Study 1975) is summarized as follows:

- BWR 3 - This release category represents a core meltdown caused by a transient event accompanied by a failure to scram or failure to remove decay heat. Containment failure would occur either before core melt or as a result of gases generated during the interaction of the molten fuel with concrete after reactor-vessel melt through. Some fission-product retention would occur either in the suppression pool or the reactor building prior to release to the atmosphere. Most of the release would occur over a period of about 3 hours and would involve 10% of the iodines and 10% of the alkali metals. For those sequences in which the containment would fail due to overpressure after core melt, the rate of energy release to the atmosphere would be relatively high. For those sequences in which overpressure failure would occur before core melt, the energy release rate would be somewhat smaller, although still moderately high.

### 2.1.3 Core Melt Penetration of Reactor Basemat

In the final stage of a core melt sequence the core of the reactor containing nuclear fuel, steel support structure and piping would liquify (Niemczyk et al. 1981). The molten mass sequently would flow under gravitational force to the bottom of the reactor vessel. The molten mass could then melt through this structure and contact the final barrier--the containment basemat. The basemat is a limestone or silica sand-based concrete approximately 3 m in thickness which serves as part of the structural foundation. The decay heat content of the core melt mass would decompose and melt the basemat. Experimental data indicate that the rate of basemat penetration is 3 to 7 cm/h (PACMA 1981). During this process the core melt mass would be accreting material from the structures it contacts. The core melt mass may penetrate the basemat and continue to melt into the geologic materials underlying the power plant (PACMA 1981). In reference to an analysis of two PWR's the staff of NRC concluded that basemat penetration can be precluded only if the core debris is kept separated from the concrete or if the core debris is cooled to temperatures below the penetration threshold for concrete (PACMA 1981). The melt mass would accumulate geologic materials and thus initiate cooling due to convection and conduction. After about one month the melt mass would no longer contain sufficient heat to melt additional material and the core melt debris would begin to solidify (Niemczyk et al. 1981).

### 2.1.4 Chemical Composition of Core Melt Debris

The physical and chemical properties of core melt debris are, in part, a function of the construction material comprising the basemat and the undisturbed geologic units under the power plant. The precise characterization of a generic core melt debris is not possible due to physical uncertainties, accident dependent factors, and site specific conditions. The melt debris can however be generalized into two basic classifications: 1) core melts into silic materials; and 2) core melts into carbonate materials. The chemical composition, solidified geometries, and release of radionuclides would be fundamentally different in these two melt types. In classification of core melts based on chemical composition of liquified geo-materials there is an assumption that the basemat and the underlying geologic units are chemically similar. In general, this is correct as limestone or silicic aggregate from local sources is used to construct the basemat. The chemical composition of cement produced from silicic and carbonate aggregate is given in Table 2.1.4-1. Core melt masses containing a mixture of silica and carbonate are quite possible and would have physical properties between these two chemical extremes. The two types of chemical melts span the range of feasible debris conditions (Niemczyk et al. 1981).

Silicic materials are more easily melted than carbonates and the molten mass would extend about 11 m below the basemat (RSS 1975). The idealized configuration of the solidified silicic melt debris is roughly cylindrical. The geometry of the core melt would be determined by the specific heat of silicic material encountered (this may change with depth if a layered geologic unit is penetrated) and the presence of open fractures. The silica melt would

TABLE 2.1.4-1. Concrete Compositions (Source: Levine 1977)

| Material | Siliceous Weight Percent | Carbonate Weight Percent |
|---|---|---|
| $SiO_2$ [(a)] | 55.7 | 15.3 |
| $CaCO_3$ [(b)] | 0.2 | 64.9 |
| $Ca(OH)_2$ | 21.6 | 12.7 |

(a) Includes $TiO_2$, $Na_2O$, $K_2O$
(b) Includes $MgCO_3$

not be a massive block of glass (Niemczyk et al. 1981). Mixing of unmelted materials and some degassing of volatiles would produce a somewhat porous mass along the outer boundary. Fracturing during cooling, especially if cooling was rapid, would greatly increase the surface area and permeability of the debris.

Carbonate materials require an order of magnitude more heat than silicate materials to melt an equivalent volume of rock (Niemczyk et al. 1981). The depth of penetration below the reactor vessel would be less in carbonate materials at about 3 meters as shown in Figure 2.2.2-2. The shape of the solidified melt mass would be strongly influenced by penetration into solution cavities. If the cavities contained water, the melt debris would be rapidly cooled. The core debris would chemically resemble a calcine material and would have a high density due to degassing of carbon dioxide during melting. The degassing of a carbonate melt could also impart a relatively high porosity to the core debris.

### 2.1.5 Sump Water Release Following Basemat Penetration

Pressurized water reactors have a probability of a sump water release in addition to a core melt release of $5 \times 10^{-6}$ per reactor year (RSS 1975). The sump water would originate from cooling sprays used in the accident sequence and would acquire radionuclides from the containment atmosphere. The rate of liquid release is dependent on the permeability of the core melt and surrounding areal position of the water table or perched water tables, size of the basemat penetration, partially saturated flow characteristics, and pressurization of the containment building. The range of variables involved in determining the rate of sump water release indicates that the liquid could slowly leak into the ground-water system over a period of months or could be jetted into the earth in a few hours. A description of the sump water releases is given in Section 2.5.

## 2.2 INDICATOR RADIONUCLIDES

### 2.2.1 Initial Amount of Indicator Radionuclides in Core

The reactor core contains an inventory of over 75 radionuclides (Niemczyk et al. 1981). These radionuclides are in various quantities and have different half-lives. However, examination of the entire reactor core radionuclide content is not necessary to characterize generic sites. In this study the radionuclides used to indicate the relative severity of an accident have three properties: 1) a long half-life to assure that they do not undergo significant decay prior to surface water discharge; 2) a high initial amount that could cause environmental concern; and 3) a low degree of sorption so that the contaminant would be easily transported by ground water. The three radionuclides meeting these criteria are listed in Table 2.2.1-1.

By examining the transport of these indicator radionuclides, the severity of radionuclide nuclear discharges to the accessible environment can be evaluated. The initial amount of each radionuclide is based on a theoretical reactor of a 3200 thermal megawatt design (RSS 1975). This reactor size is typical of nuclear power plants in the U.S. which have a design efficiency of 31% yielding 1000 electrical megawatts (RSS 1975). The differences in calculation of fuel burnup rates and power densities between PWR's and BWR's is not a sensitive parameter (Niemczyk et al. 1981). The assumption of a single inventory for both reactor types is conservative in respect to the core melt process (Niemczyk et al. 1981).

### 2.2.2 Radionuclide Partitioning

Boiling water reactors would have minor water releases below the containment structure and the radionuclide inventory would reside in the core melt debris. Pressurized water reactors could release a fraction of the core inventory during a core melt accident to the cooling water that collects in the contaminant sump. Release of the sump water through the basemat melt hole or through cracks and fractures in the basemat would also enter the ground-water flow system. The sump water is of note for two reasons: 1) some radionuclides, particularly cesium-137, are concentrated in the sump water; and 2) sump water involves a hydraulic release that could occur over a short period of time thus concentrating and driving contaminant toward the accessible environment. The release fractions of radionuclides are accident and reactor type specific. The core inventory is partitioned into the atmosphere, the core

TABLE 2.2.1-1. Indicator Radionuclides

| Radionuclide Nuclide | Initial Amount (pCi) | Half-Life (days) | Adsorption (Relative to Other Indicator Radionuclides) |
|---|---|---|---|
| Strontium-90 | $3.71 \times 10^{18}$ | 10519 | Low |
| Cesium-137 | $4.67 \times 10^{18}$ | 11042 | High |
| Ruthenium-106 | $2.48 \times 10^{19}$ | 367 | Intermediate |

melt debris, and (if it is a PWR) the sump water. The partitioning ratios for this study are given in Table 2.2.2-1 and are taken from the (Niemczyk et al. 1981).

The assumed accident sequence for the PWR is conservative in that the maximum amount of radioactivity enters the sump water where it can reach the biosphere in the shortest period of time. Radionuclides not in the sump water are assumed to be in the core melt debris where they are leached into ground water over long periods of time.

A core melt accident sequence other than PWR-7 or BWR-3 would release less contaminant into the ground-water flow system than the fractions indicated in Table 2.2.2-1. The magnitude of variations in hydrogeologic parameters are constrasted with variations in documented accident sequences in Table 2.2.2-2. The range of variation in contaminant release fractions due to the occurrence of less probable accident sequences is small in comparison to the large range of core melt source and hydraulic transport parameters. The amount of contaminant discharged into a surface water body is a much stronger function of generic hydrogeologic conditions than accident sequence.

The effect of radioactive decay exponentially magnifies the variations in the hydraulic and transport characteristics when the radionuclide flux at a distant boundary is evaluated. That is, the large site specific variations in hydrogeologic transport result in even larger variations in amounts of contaminant when discharged into surface water depending on half-life. However, the accident sequence determined release fractions are linearly related to the amount of radionuclides discharged from the ground-water system to a surface water body. The accident sequence is therefore an insensitive parameter in the computation of radionuclide discharge fluxes to a surface water body and the maximum amount is assumed to be released in this study.

TABLE 2.2.2-1. Release Fractions for Indicator Radionuclides

| Accident Sequence | Radionuclide | Airborne Release | Sump Water Release | Core Melt Debris Leach Release |
|---|---|---|---|---|
| PWR-7 | $^{90}Sr$ | $1 \times 10^{-4}$%[(a)] | 11% | 89% |
| | $^{106}Ru$ | $1 \times 10^{-4}$%[(b)] | 8% | 92% |
| | $^{137}Cs$ | $1 \times 10^{-3}$%[(c)] | 100% | 0% |
| BWR-3 | $^{90}Sr$ | 1% | - | 89% |
| | $^{106}Ru$ | 2%[(b)] | - | 92% |
| | $^{137}Cs$ | 10%[(c)] | - | 0% |

(a) Includes Ba.
(b) Includes Ru, Rh, Co, Mo, Tc.
(c) Includes Rb.

TABLE 2.2.2-2. Generic Hydrogeologic and Accident Sequence Variations

| Type of Variation | Range of Variations within Generic Classification (given in orders of magnitude) |
|---|---|
| Hydraulic Characteristics | |
| Porosity | 0-1 |
| Hydraulic Conductivity | 3-6 |
| Hydraulic Gradient | 1-2 |
| Transport Characteristics | |
| Retardation | 1-3 |
| Distance to accessible environment | 1-2 |
| Core melt leach rate | 2 |
| Accident Sequence | |
| Leach release fractions | 0-2* |
| Sump water release fractions | 0-2* |

* Not generically controlled. Includes release categories Xe, I, Cs, Te, Sr, Ru, and La range of variations are less for indicator radionuclides.

The molten core melt mass would vaporize the ground water in the vicinity of the debris. Other gasses might also form due to volatilization and chemical reactions in the melting of geologic materials. These gasses would contain some of the radionuclides released and may migrate around the basemat or through a ruptured containment structure and enter the atmosphere. This study conservatively assumes that these releases are negligible and the core inventory is available for ground-water transport.

## 2.3 COOLING OF THE CORE MELT DEBRIS

The molten core materials will initially cool by 1) a decrease in decay-heat generation, 2) incorporation of cooler geologic materials, 3) degassing of volatiles and 4) convection-conduction processes. The mass of the debris can be estimated to a reasonable degree of certainty based on the heat content of the reactor core and the type of materials penetrated. The shape of the solidified mass is dependent upon the melting point, bulk mass density and water content of the geologic materials as well as the vigor of core melt mixing during penetration. The core melt mass would be roughly cylindrical in form. Liquid core material would flow into any openings or voids (i.e., fractures and solution cavities) encountered during melt penetration. The melt would quench quickly if it encountered a highly transmissive saturated fracture.

The emplacement of the core melt will alter an undetermined zone around the debris. Partial melting and dessication of this zone will change its hydraulic properties. Partial melting may lower the effective porosity and seal existing fractures. Dessication adjacent to the core debris would grade into a partially saturated zone and then an undisturbed area. The residual heat of the melt would maintain the dessication zone until temperatures dropped below the boiling point of water. These factors would reduce hydraulic conductivity around the core melt and delay radionuclide transport. The near-field effects of core melt implacement are not considered further in this study. Transient hydraulic events such as the re-establishment of a flow field around the core debris are conservatively assumed to be instantaneous after core debris cool down.

Ground water contact with the core melt debris will cool the melt at a much faster rate. As ground water cools the outer skin of the melt mass contraction with fracturing along radial and axial patterns is likely. With time, ground-water flows would penetrate deeper into the melt debris until the melt debris became saturated throughout its entire mass. The time period for ground water to fully contact the debris would be on the order of one to two years. In PWR's the liquid sump water could initially reside on top of the core melt mass and hasten cooling by evaporation and condensation inside the containment structure. The top portion of the core melt debris could cool sufficiently to allow sump water to flow around the hot debris about six months after the accident (Niemczyk et al. 1981). The central portion of the core debris would remain at an elevated temperature after the sump water release. In this study, radionuclides are assumed to enter the ground-water system one year after the accident for core melt leached contaminants and six months after the accident for sump water releases.

## 2.4 CORE MELT DEBRIS LEACH RELEASE

### 2.4.1 Introduction to Leach Releases

The release of contaminant from the core melt debris would be from leaching of radionuclides into the ground-water flow system. Leach releases by ground water are dependent on many factors including:

1. chemical composition of material leached,
2. temperature,
3. ratio of surface area to volume,
4. density,
5. leachate resaturation rate,
6. dominant leach mechanism (i.e., molecular diffusion or matrix corrosion), and
7. amount of core melt debris saturated by ground water.

There are variations and uncertainties associated with all of these factors. The computation of a long-term leach rate for a core melt mass involves parameter estimates and generalizations with greater ranges than those used to calculate ground-water contaminant transport. There is considerable uncertainty in computing a radionuclide leach release rate for a core melt mass.

Leaching results of glasses under laboratory conditions can vary by over an order of magnitude. In addition there are a variety of test methods and reporting formats. Many tests are conducted on powdered or fine-grained material at elevated temperatures for short periods of time. The extrapolation of these results to a more massive material at ground temperatures over long time periods is somewhat questionable. The initial state of the surfaces being leached has an important effect on the short-term results. For example, samples with a flame-polished surface are not typical of the bulk of the glass, and leaching of such samples will give different initial results. Fractures or cut surfaces are more typical of the bulk glass surface. Leach tests carried out at room temperature often show high initial leach rates which drop by 2 or 3 orders of magnitude over a few days. Many accelerated leach tests (particularly the Soxhlet test) obscure this initial effect (IAEA 1979). Test of glass leaching are infrequently performed for periods more than a few weeks. Chalk River Laboratory in Canada has a leach test of glass blocks in progress since 1960 (Merrit 1977). The difficulty in determining a long-term leach rate has, in part, led to the extremely conservative modeling assumption that the radionuclide release to ground water is instantaneous (LPGS 1978; Niemczyk et al. 1981).

Despite these difficulties the material properties of a silica melt and a calcine melt are recognized as having leach rates that are different by at least one order of magnitude. The leach rate is important when estimates of radionuclide flux over time at a surface water body are used to calculate concentrations and subsequent population doses. Obviously, over estimation of radionuclide release rates by many orders of magnitude (i.e., instantaneous or prompt releases) causes corresponding over estimation of the environmental hazards. In addition, the implementation of contaminant interdiction is predicted by the magnitude and duration of the nuclear release. This study uses conservative yet realistic long-term estimates of leach release rates for silicic and calcine materials.

### 2.4.2 Silicic Melts

#### 2.4.2.1 Leach Mechanisms

For this melt type the geologic materials comprising the basemat and underlying formations are assumed to be predominantly silicon-aluminum-oxides. A glassy (amorphous silica) core melt mixture is calculated by Niemczyk et al. (1981) to contain 86% silica by weight at the time of solidification. Mixing and degassing would incorporate cavities and particles of rock. Consequently, the core debris would not resemble a solid block of glass. However, the melt material can be chemically characterized as similar to a glass or natural occurring volcanic obsidian. Cooling would subject the debris to thermal-induced stress that would cause fracturing. Experiments conducted on nuclear explosion melt glass indicates that the flow rate of water over the samples did not effect the leach rate (Chapman et al. 1980; Failor et al. 1983). The core melt debris is assumed to be sufficiently porous and/or fractured that it would not form a major hydrualic barrier. The position of the water table is conservatively assumed to be above the top of the core debris.

The mechanism of glass leaching has undergone extensive study due to the feasibility of isolating waste products in glass. The leaching mechanism is described by Barkatt et al. (1981) as:

Early work on silicate glasses containing alkali metal oxides and alkaline earth oxides has shown that the attack of water on the glass starts as a diffusive process through which alkali cations are preferentially leached from the surface layers, leaving behind a porous high-silica layer. As the dealkalized layer becomes thicker, the rate of further diffusion of alkali out of the glass through this layer becomes progressively slower, until silica dissolution at the interface between the dealkalized layer and the solution begins to control the rate of the attack.

After having been exposed for a sufficiently long time, silicate glasses with high durability posses silica-rich films which are dense enough to protect the glass from further rapid attack. In these cases, a transition layer, highly resistant to diffusion, is observed to form between the outer porous gel layer and the solid glass. This is probably due to the replacement of ionized oxygen-alkali bonds by undissociated matrix dissolution. In glasses, the formation of a hydration layer generally occurs simultaneously with the depletion of alkali ions.

A protective gel layer develops more slowly with time and the leach rate is of a parabolic type (Lanza et al. 1980). The growth of the hydration layer may be interrupted by cracking or peeling of the gel. There are two mechanisms that cause disruption of the gel layer: 1) as the glass hydrates the gel layer swells; and 2) the exchange of alkali ions by hydrogen (or $H_3O^+$) ions creates stress along the glass-gel layer due to change in ionic size and bond energies (Barkatt et al. 1981). Exposing fresh glass would restart the leaching process without the protective hydration layer and the leach rate would increase. Mechanical agitation in laboratory tests due to boiling, mixing and handling may cause disruption of the hydration rind. These conditions would occur only in the early stages of core melt cooling and saturation.

Early in the leach process diffusion is noted in glass by the preferential release of the radionuclides strontium-90, cesium-137, and alkali ions (Barkatt et al. 1981). At longer times ionic diffusion from the glass is hindered by the protective hydration layer. The migration of radionuclides through the hydration layer is retarded by sorption in the insoluble silicic rind. The leaching of radionuclides from glass over long time periods can be summarized as in Table 2.4.2-1.

Leaching of glass over the short term (days) is diffusion controlled. Long-term (decades to a millennium) leach processes, which are important in determining the severity of a long radionuclide release period are controlled by hydration and corrosion of the glass matrix. Leach rates of radioactive high-level waste glass had not reached a constant value at 639 days and demonstrated the combination of release mechanisms (Bradley 1978). Matrix dissolution is an important part of long-term glass leaching (Clark et al. 1979). Matrix dissolution is probably the dominant mechanism at 25°C and perhaps as high as 75°C (Coles 1981b).

TABLE 2.4.2-1. Silicic Leach Processes

1. At early times (days) ionic diffusion process are dominant which become exponentially less important with time.

2. Hydration of outer layer of glass with subsequent loss of akali ions and corrosion of the glass matrix.

3. Following corrosion of the matrix radionuclides and other ions migrate through the hydration layer and are retarded along the pathway.

4. The radionuclide reaches the outer edge of the hydration rind and enters the ground-water flow system.

### 2.4.2.2 Silicic Leach Rate

The rate of hydration of glass can be estimated by examination of a volcanic glass known as obsidian. These glasses have been exposed to leaching by ground water over thousands of years and provide an example of long term rates. The process of degradation of obsidian forms perlite or hydrated obsidian (Ericson 1981). Obsidian formation is a near surface geologic event and leaching conditions are similar to those of a postulated core melt accident (Ericson 1981). The thickness of the insoluble hydration rind has been correlated with historic and geologic age and is described by Friedman and Long (1976) as:

$$\ell = K\,T^{1/2} \tag{2.1}$$

where:

$\ell$ = thickness of hydration ($\mu$m)
$K$ = hydration rate [$(\mu m)^2$/1000 year]
$T$ = time (yr).

The hydration rate (K) is a function of temperature and chemical composition. In a shallow geologic environment the earth temperature is assumed to be 20°C. The hydration rate of obsidians in Japan was found to be related to temperature (T) by:

$$K = (6.76 \times 10^{-13}) \exp(-8927/T) \tag{2.2}$$

yielding a K of 5 $\mu m^2$/1000 years (Suzuki 1973). This value is in excellent agreement with the hydration rates of obsidian in the western U.S. (Friedman and Long 1976) and (Friedman and Obradovich 1980). The correlation of historical date to thickness of the hydration rind indicates that peeling and loss of the rind due to stress is not a prevalent event at these time periods (i.e., hundreds to thousands of years).

The leach rate of a silica glass is computed by knowing the surface area of the melt and application of Equations (2.2) and (2.3). The geometric surface area of the melt can be computed to a moderate degree of accuracy. The configuration of a silicic core melt is illustrated in Figure 2.4.2-1. The surface area of an actual melt would consist of partially granular to fracture

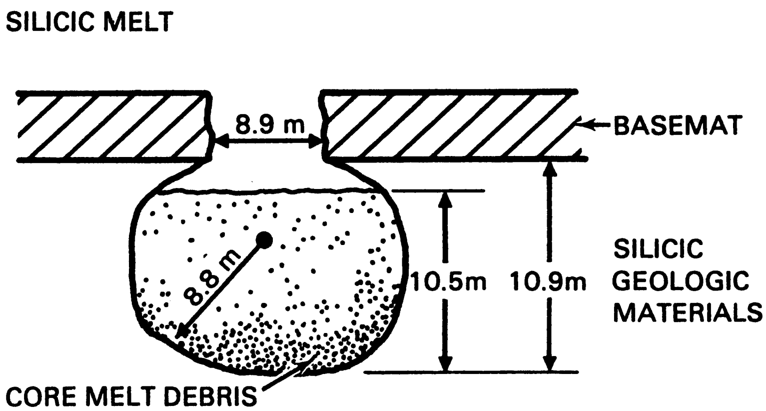


FIGURE 2.4.2-1. Configuration of Solidified Core Debris for Silicic Melt (After: Niemczyk et al. 1981)

surfaces and include irregular fingering into the geologic media. The surface area of a silica melt has been estimated by Niemczyk et al. (1981) to be "at least several times greater than the apparent exterior surfaces". A fractured surface area increase of 1000 over the geometric surface area was considered "more likely" in LPGS (1978). This study will assume a surface area increase factor of 67,000 which corresponds to a very conservative specific surface area of 100 $cm^2/g$ ($\beta$) as the upper limit of a silicic melt.

The bulk density ($\rho$) of the silicic core debris is about 3.01 $g/cm^3$ including steel and fuel. Assuming a basically cubic fracture pattern the length of a representative fracture surface is computed by

$$\frac{V}{A} = \frac{N\,L_o^3}{N\,6\,L_o^2} = \frac{L_o}{6} \qquad (2.3)$$

where:

$V$ = geometric volume = 2.55 x $10^9$ $cm^3$
$A$ = surface area = ($V \cdot \rho \cdot \beta$)
$N$ = number of representative fracture cubes
$L_o$ = length of representative fracture surface

Solving Equation 2.4 for $L_o$ gives 1.9938 x $10^{-2}$ cm in the example silicic melt. The amount of silica and dispersed radionuclides leached at time (t) then becomes:

$$M = \frac{Q}{N}\;L_o^{3} - \;2\;L_o - H_t^{\;3}\;\;N\,W_t \qquad (2.4)$$

where:

$M$ = quantity released to time (t)
$Q$ = amount of initial radioactivity
$W_t$ = radioactive decay fraction at time (t)
$L_o$ = original length of representative fracture cube
$H_t$ = hydration thickness at time (t), and other notation as defined previously.

The number of representative fracture cubes (N) cancels in both Equations (2.4) and (2.5). The release of radionuclides from a core melt by this mechanism is conservative in four major respects:

1. the hydration rate would be more rapid in an admixture of silica, steel, fuel and incorporated partially melted geologic materials,

2. the surface area of 100 $cm^2/g$ is very high and implies extensive fracturing and/or a partially granular material,

3. hydration with subsequent corrosion release of radionuclides would also include the additional time necessary for material to diffuse through the insoluble hydration rind,

4. saturation of core melt is assumed instantaneous after cooling, and

5. the entire core debris is assumed to be below the water table.

These conservative factors of a silicic leach release are judged to be adequate to compensate for the uncertainty in the long-range leach mechanisms not accounted for in the release model. Specifically, there is no method to predict the possible cracking and pealing of the hydration rind and at early times (e.g., days) the hydration rind is not fully formed and diffusion dominates the release of radionuclides.

2.4.2.3 Comparison with Experimental Results

Comparison of this methodology to long-term glass leach data indicates that the results are reasonable. The comparison is based on a leach rate (R) in g $cm^{-2}d^{-1}$ is given as (IAEA 1979):

$$R = \frac{A_t}{A_o} \cdot \frac{W_o}{S \cdot T} \tag{2.5}$$

where:

$A_t$ = amount of "A" (g) removed in time t (days)
$A_o$ = initial amount of "A" (g)
$W_o$ = initial weight (g)
$S$ = surface area ($cm^2$)
$T$ = time (days)

The parameter A can represent the activity of an isotope, however in this case A equals grams of silica. The value of R is not always directly comparable with other published results. In some cases R is based on geometric surface area and in others it is based on true surface area. This can change the leach rate by over three orders of magnitude depending on the material composition. Values for leach rates used in this study are based on true surface area. The leach rate (using Equations 2.4, 2.5, 2.6 and the referenced experimental results) at 45 days and at 14 years is given in Table 2.4.2-2.

The leach rate at 14 years is computed for a standard ground-water temperature of 20°C and a lower temperature of 5°C. The lower leach rate observed at 5°C is comparable to measured rates in the cool ground-water (0-5°C) at Chalk River Laboratory. The hydration rate in near freezing ground water is reduced by over two orders of magnitude. Leach rate calculations for this study were made for ground-water temperatures of 20°C.

Figure 2.4.2-2 presents the long-term silicic leach release for strontium-90 and ruthenium-106. The release of these two radionuclides is assumed to be congruent and at a rate controlled by the hydration-dissolution of the

TABLE 2.4.2-2. Comparison of Long-Term Experimental Data and Silicic Leach Model

| Material | Time | Leach Rate ($g\ cm^{-2}\ d^{-1}$) |
|---|---|---|
| Rock-Glass[(a)] | 57 days | $6.6 \times 10^{-8}$ |
| LWR Glass[(a)] | 45 days | $1 \times 10^{-7} - 1 \times 10^{-8}$ |
| NTS Nuclear Explosion Glass[(b)] | 45 days | $1 \times 10^{-8} - 1 \times 10^{-9}$ |
| NTS Nuclear Explosion Glass[(c)] | <20 days | $5 \times 10^{-8}$ |
| Silicic Leach Model[(f)] | 45 days | $8.3 \times 10^{-8}$ |
| Chalk River Glass Blocks[(d)] | 14 years | $5 \times 10^{-11}$ |
| Silicic Leach Model[(e)] | 14 years | $6.9 \times 10^{-12}$ |
| Silicic Leach Model[(f)] | 14 years | $7.8 \times 10^{-9}$ |

(a) Experimental results from IAEA 1979.
(b) Experimental results from Coles and Ramspott 1982.
(c) Experimental results from Failor, Coles, and Rego 1983.
(d) Experimental results from Merrit 1977.
(e) At 5°C
(f) At 20°C

SILICIC LEACH FUNCTIONS

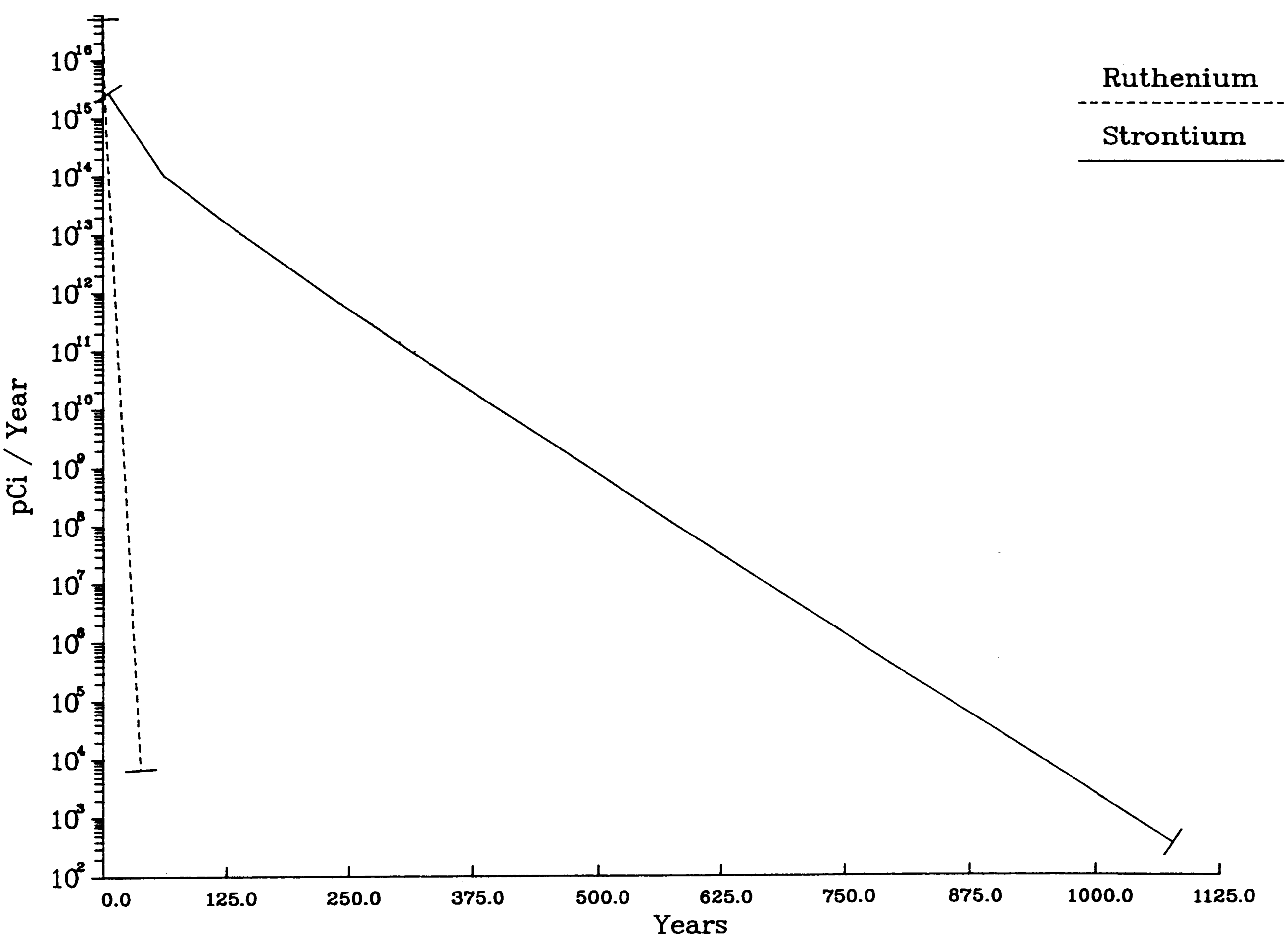


FIGURE 2.4.2-2. Long-Term Leach Release Rate for Silicic Core Melt

glass matrix. Obviously, the initial radioactive quantity, release fraction and radioactive decay is included in the calculation. In long-term leach processes individual chemistries of the various elements become less important as a common leach rate is approached as leaching proceeds (Coles 1981a). The decreasing rate of radionuclide release is due to a combination of both the hydration-corrosion model and radioactive decay. Leach rates of nuclear explosion glass have indicated a continued decrease in rate to 420 days and were on a decreasing trend. Steady leach rates were not obtained in these tests (Failor et al. 1983). Initial amounts and rates of decay are responsible for the differences in slope of the strontium-90 and ruthenium-106 leach release curves.

### 2.4.3 Calcine Melts

#### 2.4.3.1 Leach Mechanism

A core melt into a carbonate material of limestone and dolomite [$CaCO_3$-$CaMg(CO_3)_2$] would produce a debris that would chemically resemble a calcine material. The debris would also have a high bulk density (i.e., 4.3-5.0 g/$cm^3$) and a porous-spongey composition (Niemczyk et al. 1981). The structural properties of calcium oxide are not strong and it is quite soluble in water. The porosity of the melt may be enhanced if the molten debris enters a solution cavity and is rapidly quenched (LPGS 1978). As in the case of a silicic melt, the core debris is assumed to be fully saturated and does not form a hydraulic barrier to ground-water flow. The dominant leach mechanism for this material is diffusion. The diffusion rate is dependent on material properties and the radionuclide of interest. The diffusion model is (LPGS 1978):

$$\frac{\Sigma A_N}{A_o} = 2\ K_e\ \frac{S}{V}\left(\frac{D_e}{\pi}\right)^{1/2}\left(t^{\,1/2}\ e^{-\lambda +}\right) \qquad (2.6)$$

where:

- $\Sigma A_N$ = Sum of radioactivity lost to time (t)
- $A_o$ = Initial radioactivity
- $K_e$ = Relative leach factor (dimensionless)
- S = Surface area ($2.9845 \times 10^{11}$ $cm^2$)
- V = Volume ($5.969 \times 10^{8}$ $cm^3$)
- $D_e$ = Effective diffusivity ($cm^2$/sec)
- t = time
- $\lambda$ = decay constant ($t^{-1}$)

The relative leach factors account for the more rapid release of certain classes of radionuclides. Elements with low valences and/or small ionic radii are generally leached at the faster rates. The leach rates of strontium-90 and ruthenium-106 are intermediate to alkali metals (i.e., cesium) and the actinide elements (i.e., plutonium) (Moore et al. 1976). The relative leach rate can be reduced by up to three orders of magnitude by inclusion of clay or shale (Moore et al. 1976). Limestone commonly contains thin layers of shale as well as interstitial clay which would reduce the leach rate in comparison to cementitious grouts. Surface area and geometric volume of the melt is based on the calculated core melt geometry presented by Niemczyk et al. (1981). The configuration of a calcine core melt mass is illustrated in Figure 2.4.3-1. Depending upon the rate of carbon dioxide ($CO_2$) degassing and the rate of cooling, the bulk density is calculated to be 4.3 to 5.0 g/$cm^3$. The latter value was chosen to represent the core melt. The surface area of the core debris is subject to large uncertainty. In a ionic diffusion release (Equation 2.7) the

CALCINE MELT

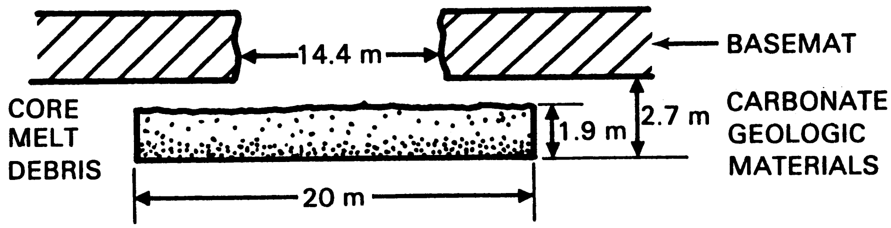


FIGURE 2.4.3-1. Configuration of Solidified Core Debris for Calcine Melt (After: Niemczyk et al. 1981)

accuracy of the leach rate prediction is directly proportional to the accuracy of the estimation of surface area. The core debris surface area can range over two orders of magnitude depending on the chemical composition and water content of the geo-materials under the plant, vigor of the carbonate to calcine reaction and fracturing during post accident cooling. The specific surface area of calcine material is estimated in LPGS (1978) based on geometric surface area and granular particle sizes. The mean diameter of a particle produced by the decomposition of concrete by core debris ranges from 200 to 1000 μm (PACMA 1981). These particle sizes correspond to specific surface areas of 60 to 12 $cm^2/g$, respectively. A summary of these estimates is given in Table 2.4.3-1. A specific surface area of 100 $cm^2/g$ is considered to be realistic of a calcine material not quenched in standing water.

The effective diffusivity is a fundamental measure of the rate at which a contaminant will be removed from the solid matrix. Experimental work with cementitious grouts (Moore et al. 1976) indicates that the release rate is over 100 times greater than for glass. Based on experimental leaching of strontium-90 from hydrofract grout the effective diffusivity based on geometric surface area is $1 \times 10^{-11}$ to $6 \times 10^{-10}$ $cm^2/sec$ (Moore et al. 1976). An effective diffusitivity based on actual surface area is taken as $6 \times 10^{-15}$ $cm^2/s$. Figure 2.4.3-2 presents the calcine leach release rate for strontium-90 and ruthenium-106.

## 2.5 SUMP WATER RELEASE RATES

Pressurized water reactors could also release contaminated water used in cooling during the accident sequence. The water would collect in the reactor sump and may be released due to: 1) the flowing through the hole in the basemat formed by the molten core; or 2) flowing through the fractured basemat if the core did not penetrate the containment structure. Therefore, it is feasible to have a contaminated sump water release even if the core melt does not completely penetrate the basemat. The rate of this liquid release would depend on:

TABLE 2.4.3-1. Estimated Specific Surface Areas for Core Debris

| Specific Surface Area ($cm^2/g$) | Basis for Estimate |
|---|---|
| 0.0025[a] | Geometric surface area |
| 6.3 | Assumes value for land based plant (LPGS-1978) |
| 2.5[a] | Surface area enhancement of 1000 |
| 100[b][c] | Value less than for water quenched material |
| 1000[b][c] | Particle size minimum of 12 μm for core material quenched in standing water[c] |

(a) Determined for calcine core melt geometry of Niemczyk et. al. (1981).
(b) LPGS (1978) melt geometry more indicative of silicic material.
(c) Particles are assumed to be spherical grains. The range of specific surface areas in a calcine debris is probably bounded between 10 and 1000 $cm^2/g$.

1. size of basemat opening,
2. density and viscosity of sump water,
3. hydrogeologic properties of underlying materials, and
4. pressure head.

Assuming that a sump water release occurred at any of the nuclear power plant sites, the liquid release could take place over several days to several months. The sump water release rate and radionuclide release rate used in this study are based on the aquifer properties at each site. There are uncertainty as to the actual conditions that might be present at a core melt accident. Specifically, the permeability of the opening in the basemat (important only when the basemat is fractured and not penetrated) and the pressure head inside the containment building.

If the containment structure ruptured prior to basemat melt-through, the pressure head would consist solely of the hydraulic head difference between the ground water and the fluid inside the containment. The position of the water table at most sites is above the top of the basemat. At these locations if the containment building ruptured prior to melt-through, ground water would flow up through the core melt debris and flood the lower portions of containment. If this ground-water seepage was allowed to equilibrate with respect to the water table the average maximum depth of water inside containment would be 8 to 20 meters (S. J. Niemczyk Personal Communication 1982). As in the case of water flowing out of the containment structure, the rate would depend upon the site specific conditions as noted above. This water could be pumped from the containment structure as a part of the mitigative procedures.

CALCINE LEACH FUNCTIONS

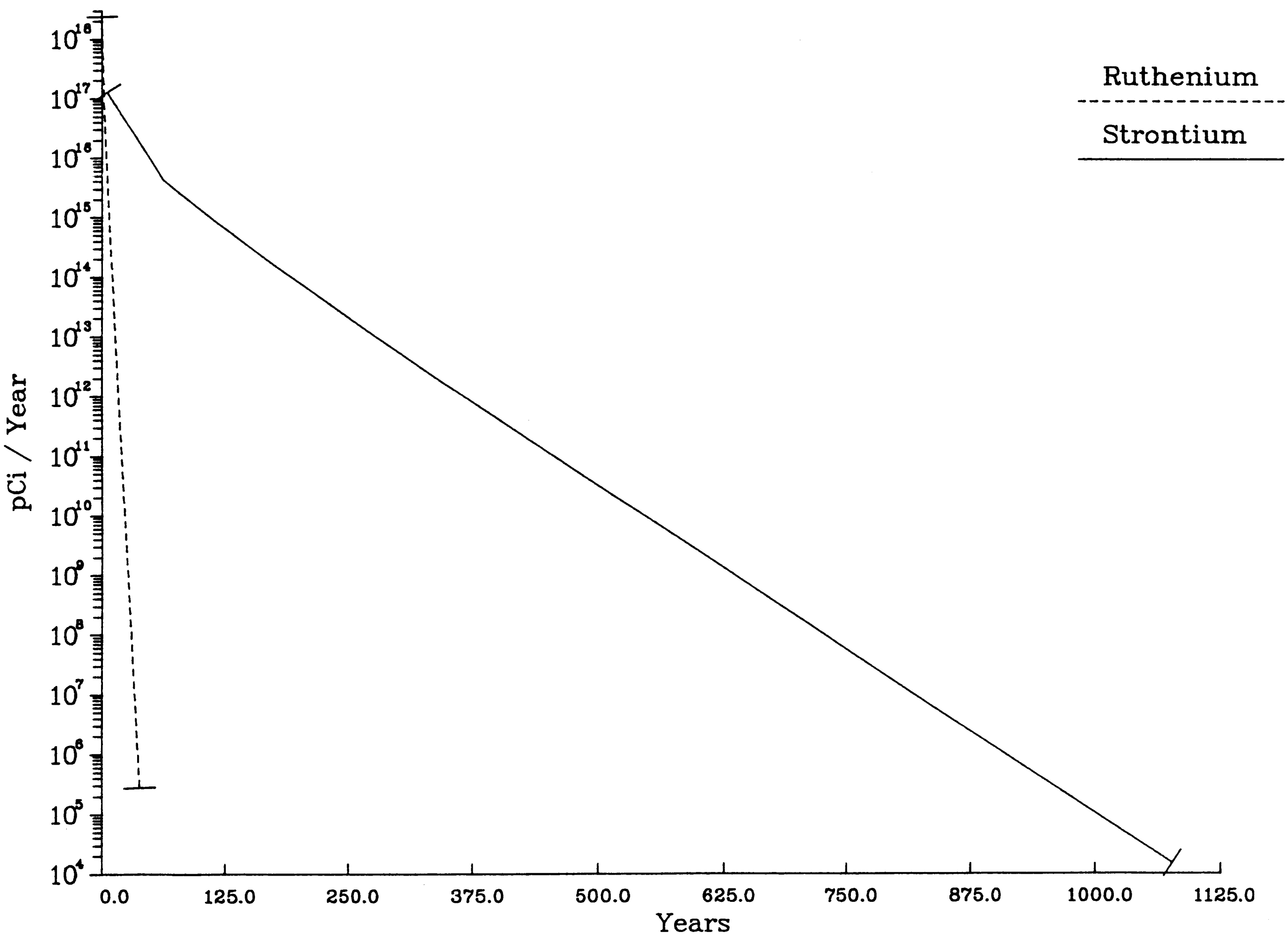


FIGURE 2.4.3-2. Long-Term Leach Release Rate for Calcine Core Melt

If the containment structure had not failed and pressure inside the containment was above the ground-water pressure head, sump water would exit the containment. Initially the heat of the core melt mass would prevent sump water from escaping. However, the constant vaporization and condensation of sump water in contact with the melt mass would result in more rapid cooling of the debris. The top of the core melt mass may become sufficiently cool to allow sump water to enter the ground-water system in as little as six months after the accident (Niemczyk et al. 1981). This time period of six months for delay after the accident to time of liquid release is used in this study.

A sump water release is evaluated by making six practical assumptions:

1. the pressure head inside the containment is above the water table head by distance from the top of the core melt to the top of the basemat plus 4.5 meters standing water above the basemat,
2. the ground-water flow system is unconfined and the effective porosity as reported in Niemczyk (undated) is equivalent to the effective storage coefficient,
3. the radius of flow is equal to the size of the exterior of the solidified core melt debris,
4. radionuclides are dispersed in 1135 $m^3$ of sump water liquid,
5. the ground-water flow conditions are saturated and hydraulic conductivity is as listed for each site in Niemczyk (undated).
6. The sump water has the density and viscosity of average ground-water.

These conditions are known to be conservative with respect to positive containment pressure and flow hydraulics in the disturbed zone beneath the containment structure. This is recognized also to be less conservative, but more realistic than an instantaneous or prompt release. An actual sump water release would be at slower rates unless the containment building was severely overpressurized. Under these assumptions the Theis equation for non-steady radial flow was used to determine the flow rate from the containment (Freeze and Cherry 1979).

$$Q = 4 \pi K b Z \int_{\frac{r^2 f}{4Kbt}}^{\infty} \frac{\ell^{-x}}{x} dx \qquad (2.7)$$

where:

Q = volumetric flow rate ($L^3/t$)
K = hydraulic conductivity ($L^2/t$)
b = geologic unit thickness (L)
Z = hydraulic head (L)
r = radius of flow input to aquifer (L)
f = coefficient of storage (dimensionless)
t = time

The rate was averaged over a short period of two hours which allows the peak release rate of radionuclide to be determined. The exponential form of the Theis equation allows an initial flow rate to be rapid with subsequent diminishing of the flow with increasing time. Under these conditions the peak release rate of radionuclides would occur at the beginning of the sump water escape. Therefore, the release rate is modeled as a single valued peak release at each site. The sump water release was not modeled as a time dependent release due to the short period of release from the containment as compared to the travel time to the accessible environment and the uncertainty associated with the volume of water that would be released. The assumptions of the sump water release are reasonable but do not account for site specific factors other than hydraulic conductivity and effective porosity.

The sump water release is designed to properly scale the release of radionuclides based on aquifer hydraulics. Although there is recognized uncertainty associated with this methodology, it is more realistic than assuming the sump water instantaneously exits. At the least, this method accounts for the gross variations documentated in aquifer hydraulics at the individual commercial nuclear power plant sites. Since each plant site was characterized by a different sump water release rate, there is no standard release rate curve for sump water exiting the containment structure.

## 2.6 DURATION OF RADIONUCLIDE RELEASE

Leach release of radionuclides from the core debris would continue over a long period of time as evidenced in Figures 2.4.2-2 and 2.4.2-3. With sufficient time, the decreasing mass leach rate and radioactive decay would result in an inconsequential release of the more mobil radionuclides. Many of the longer lived and highly retarded radionuclides (i.e., plutonium) would remain near the melt debris. The time period over which ground-water contamination would continue is demonstrated by the release of hazardous amounts of strontium-90. In this context the term hazardous is defined as ground-water concentrations of strontium-90 adjacent to the core debris above the 10 CRF 20 limit of 300 pCi/l. The time required for strontium-90 releases to be nonhazardous is determined from the calculated silicic and calcine leach rates and representative ground-water flow rates. This time period is valid only for the area immediately adjacent to the debris and not at the nearest surface water body. Proper restrictions of ground water use at the site of the accident would prevent exposure through this pathway. The calculated time period of a hazardous strontium-90 release at the source of the accident given in Figure 2.6.1-1 is made by assuming:

- the entire portion of the nuclear inventory partitioned to the core debris is available for leaching,
- the geometry of the melt mass is as described for calcine and silicic materials,
- the effective porosity of the melt debris is 0.01,
- the range of ground-water flow velocities is from .01 to 10 meters per year.

The long time period indicated by these assumptions demonstrates that the area near reactor will remain a contaminant source and require maintenance of monitoring and perhaps mitigative structures beyond the effective life span of present day materials.

## 2.7 CONCLUSIONS CONCERNING RADIONUCLIDE RELEASES FOLLOWING A SEVERE ACCIDENT

1. In the unlikely event of a core melt accident, it is feasible for core debris to degrade and penetrate the containment basemat releasing radionuclides to the ground-water flow system.

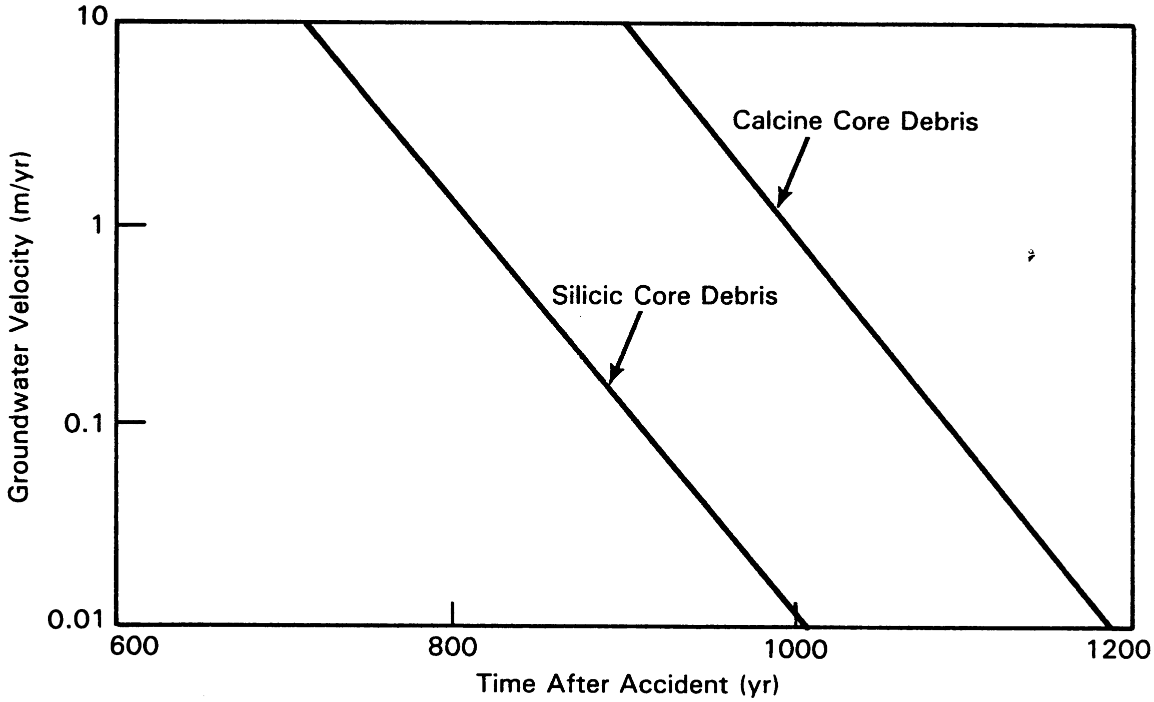


FIGURE 2.6.1-1. Time Period of Hazardous Strontium-90 Releases from Core Debris

2. Chemical composition of the concrete and underlying materials could have a large influence on the physical properties of the solidified core debris. The release of radionuclides from silicic materials is basically a corrosion-dissolution mechanism while calcine materials release contaminants primarily through diffusion processes.

3. The leach release rates could be 100 times greater in calcine materials than in materials that are predominately silicic.

4. Sump water liquid release rates are very site and accident sequence specific. This type of release at a site could occur very slowly or quite rapidly depending on containment pressurization and the hydraulics of the altered zone around the core debris.

5. Radionuclide discharge quantities to surface water are more a function of ground-water transport factor than release fractions determined by accident sequences (e.g., PWR 1-7 and BWR 1-4).

6. Radionuclide releases from core debris would contaminate ground-water adjacent to the reactor for hundreds of years.

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# 3.0 GENERIC HYDROGEOLOGIC SITE CLASSIFICATION

## 3.1 CLASSIFICATION SCHEME

### 3.1.1 Considerations for a Classification Scheme

The classification of existing and proposed nuclear power plant sites in the U.S. is based on their respective hydrogeologies. The classification scheme follows the concept that the hydrogeological site conditions will control the ground-water transport mechanisms and will contribute to the determination of appropriate mitigative strategies following a core melt accident. Combining classification criteria for transport with potentially feasible mitigative measures results in a matrix of generic hydrogeologies and the associated mitigative actions. This classification scheme also facilitates the generic study of important ground-water transport characteristics such as travel times and radionuclide discharge rates associated with accessible environments.

The major factors controlling ground-water transport are site geology, hydrology, geochemistry, and geography. These factors are interrelated and strongly affect the overall characterization of a core melt accident. For example, the rock chemistry determines the penetration depth of the core melt mass, the type and rate of leach release and, in part, the retardation of radionuclides in transport. In this case, the geochemistry effects the depth of borings into the contaminated zone, the time scale for project completion, and the necessity for any mitigative action. The selection of specific mitigative techniques is also a function of the hydrogeologic factors at the site. The hydrogeologic site conditions affect the feasibility of a mitigative technique at that location. For example, the construction of slurry walls requires unconsolidated material or very soft consolidated material.

The classification scheme is based on hydrogeologic parameters that are most sensitive in affecting radionuclide transport but are also readily determined for a site. The classification of nuclear power plant sites was limited in scope so that it would not be unwieldy. However, a representative number of sites are included in each generic classification. This is similar in practice to the determination of generic surface water classifications as found in "The Consequences from Liquid Pathways After a Reactor Meltdown Accident," NUREG/CR-1596. Five criteria were used to determine the hydrogeologic classification of each nuclear power plant site. These criteria are based on:

1. geologic unit,
2. rock chemistry,
3. consolidation of material,
4. porosity, and
5. ground-water chemistry.

All existing and proposed nuclear power plant sites were reviewed according to the above criteria. The generic site classification scheme was then developed by determining commonalities among the hydrogeologic properties for certain groupings of sites based on the above criteria. Individual generic sites embody these common properties.

The information needed to classify the nuclear power plant sites is taken from an unpublished report "A Summary of Subsurface Hydrogeological Information for Light Water Nuclear Reactor Sites" by S. J. Niemczyk at Oak Ridge National Laboratory. The hydrogeologic data base presented in the report lists the geologic unit, distance to nearest suface water body and the aquifer properties for each site. The geotechnical data base used to determine the hydrogeologic properties of each power plant site was developed from Niemczyk's report. Portions of the geotechnical data base are presented in Section 3.5 by generic classification.

#### 3.1.1.1 Geologic Unit Criterion

The basemat of the containment structure at nuclear power plants is constructed of concrete up to 3 meters in thickness. Below the basemat and any intervening engineered backfill at each site lay undisturbed geologic materials of various compositions. The geology may consist of massive units or stratified units of various types. When the geologic media are stratified, the hydraulic properties can range over several orders of magnitude between adjacent units. The geotechnical data base used in this study contains 50 sites where stratified deposits were noted. At 29 sites the hydraulic properties of individual units within the stratified materials are known. The determination of which geologic unit was chosen to characterize each stratified site was based on three conditions:

1. position of the water table, (which may be perched),
2. silicate or carbonate rock chemistry, and
3. ground-water hydraulics.

The geologic unit must lie below the water table. The basemat of most nuclear power plants lies below the water table and therefore a core melt accident would directly impact the saturated zone. Fifteen sites have water tables below the basemat. However, the core melt would penetrate into the saturated zone at all but four of these sites. The sites where the core melt would reside in the partially saturated zone are excluded from further study due to extremely slow contaminant transport rates and the complex site specific data and modeling requirements for characterization. Geologic units above the water table or above the top of the core melt were not considered for the purpose of generic classification. A liquid release of sump water from a pressurized water reactor is assumed to flow through and around the core melt mass and into the selected geologic unit.

The geologic materials were classified as being silicates or carbonates. This distinction is necessary because the rock chemistry determines the ultimate depth of the melt and hence controls which geologic units will be in saturated contact with the core melt. There are 47 sites in the geotechnical data base that list a single geologic unit and it was assumed to be the principal unit in contact with the core melt. That geologic unit was then used in the characterization scheme for the generic sites.

When several diverse geologic units contacted the core melt, the third condition, ground-water hydraulics, was considered. For these cases, the geologic unit with the highest transmissivity was selected. Some sites list extreme and average hydraulic properties. These sites were characterized by the average hydraulic values.

#### 3.1.1.2 Rock Chemistry Criterion

Geologic materials at nuclear power plant sites can be divided into two generalized chemical classifications: 1) silicates and 2) calcium-magnesium carbonates. The chemical composition of a geologic unit is a result of its formation and any subsequent alteration. Silicic rocks are formed from igneous processes (i.e., granitic intrusions) and occasionally biological processes such as deposits of diatoms radiolaria. Silicates are weathered by physical and chemical actions into unconsolidated sedimentary material such as clay, silt, sand and gravel. Sedimentary silicates (e.g., sandstone and siltstone) are consolidated to competent rock by deep geologic burial. Carbonates are formed primarily by marine organisms and deposited as layered media. Both silicates and carbonates are subjected to a variety of processes that alter their physical form and chemical composition. The percentage of silica and carbonate found in common rock types is given in Table 3.1.1-1.

The reaction of these two chemical rock types to a core melt accident would be markedly different. Silicic materials would be melted to a greater depth below containment structures and would be more resistant to leaching. Carbonitic materials would produce a more shallow melt zone and leach radionuclides into the ground water at a faster rate. The characteristics of a core melt are discussed in Section 2.1.1 of this report and in NUREG/CR-1596. Details of the chemical controls of leaching processes are discussed in Section 2.2 of this report. Although there is a general distinction between silicate and carbonate melts and leach rates, there is considerable uncertainty involved in assigning either melt type an absolute leach release rate.

The geochemical rock type also has as strong influence on ground-water chemistry and aquifer-nuclide reactions. Table 3.1.1-2 presents a summary listing of melt formation and transport characteristics based on geologic rock type.

#### 3.1.1.3 Consolidation of Material Criterion

The selection of mitigative strategies is, in part, a function of the workability of the geologic media. Consolidated materials consist of crystalline and sedimentary units which have become competent rock. Unconsolidated units consisting of clay, silt, sand, and cobbles are characterized as packed particulate material. The engineering properties of consolidated and unconsolidated units are fundamentally different. The competency of geologic materials in many instances influences the feasibility of mitigative measures. For example, a radionuclide release into a consolidated limestone will preclude

TABLE 3.1.1-1. Percentage of $SiO_2$ and CaO of Common Geologic Units[a]

| Geologic Material | Percent $SiO_2$ | Percent CaO |
|---|---|---|
| clay | 70 | <2 |
| silt | 60 | <10 |
| till | 60-80 | <1 |
| sand | 70-100 | <5 |
| limestone | <10 | 50 |
| dolomite | <10 | 30[b] |
| basalt | 50-60 | 8-12 |
| tuff | 50-70 | <6 |
| granite | >65 | <15 |
| schist | 60 | <1 |
| sandstone | 50-95 | <15 |
| shale | 58 | <3 |
| arkose | 77 | 3 |
| graywacke | 66 | 3 |

(a) Sedimentary units from (Pettijohn 1975) igneous units from (Bowen 1956).
(b) Also contains 30 to 50% $CO_2$.

TABLE 3.1.1-2. Core Melt and Ground-Water Transport Characteristics Based on Chemical Rock Type

| Core Melt and Leach Characteristics | Silicate | Carbonate |
|---|---|---|
| 1. Depth of melt below basemat | 11 meters | 3 meters |
| 2. Core melt composition | Silica melt glass | Calcine material |
| 3. Dominant leach process of core melt | Hydration-corrosion | Diffusion |
| 4. Relative leach rate | Slow | Fast |
| 5. Porosity of core melt | Fracture controlled | Interstitial-Dependent upon degrassing carbon dioxide |
| 6. Sorption in aquifer type | More slightly | Less slightly |

TABLE 3.1.1-3. Generalized Construction Considerations Versus Type of Geologic Formation

| Construction Method | Type of Geologic Formation: Consolidated | Unconsolidated |
|---|---|---|
| Excavation for trenching or disposal | Requires special equipment, processes are slow and extensive blasting may be required | Common construction equipment, requires support for side walls, limited to practical depths, may require dewatering below water table |
| Bore holes as for: | Drilling can be slow, bores do not usually require casing | Drilling is more difficult, casing and screen required, drilling technique dependent on purpose of bore |
| (injection) | Material enters along fractures and bedding planes | Material enters between particles and bedding planes, may cause deformation or lifting of unit |
| (withdrawal) | Water is from fractures and interstitital pathways drilling technique may seal fractures | Water is interstitial, screen must be of proper size to avoid removal of fine material, drilling technique may clog formation |
| (monitoring) | Drilling technique may disturb chemical analysis because of muds used | Drilling technique may disturb chemical analysis because of muds used |
| Sheet pile driving | Not feasible | Difficulty dependent on particle size and strength of unit |
| Ground water freezing | May not be feasible, karstic limestone with ground water velocities over 1 meter/day | May cause ground heave and damage to existing facilities |

use of mitigative techniques requiring a deformable geologic media (e.g., sheet piling). The basic construction considerations of consolidated versus unconsolidated materials are listed in Table 3.1.1-3.

The consolidation of materials criterion also has a bearing on groundwater hydraulics, geochemistry of the radionuclide source term and sorption. These factors are considered as the remaining criteria.

#### 3.1.1.4 Porosity Criterion

Porosity of geologic materials is due to interstitial voids between adjacent grains and openings along joints and fractures. The percentage of interconnected pathways to bulk rock volume is known as the effective porosity.

Two major distinctions can be made between fractured and interstitial flow systems. First, interstitial flow occurs in porous media which generally affords a higher percentage of open area and secondly, a larger surface area of rock for the contaminant to contact than in fractured media.

Interstitial porosity affects the degree of contaminant retardation due to chemical sorption and the average linear ground-water velocity. Sorption of contaminant is dependent upon bulk mass density of the aquifer, porosity, and the equilibrium distribution coefficient. Interstitial porosity normally results in a large aquifer surface area which provides abundant locations for contaminant to be sorbed onto the grains. The average linear velocity of ground water is a function of hydraulic conductivity, hydraulic gradient, and effective porosity. The larger effective porosity of porous media generally results in slower average linear velocities and greater chemical retardation of the transport of contaminants.

Fractured media channels ground water and contaminants preferentially along open joints. This does not mean that contaminant does not enter the rock matrix, rather that the mass of contaminant will travel along the fracture until hydrodynamic dispersion and/or molecular diffusion forces it into the wall of the aquifer. When the concentration of contaminant in the fracture is less than the concentration in the rock matrix, the contaminant will re-enter the open joint. The aquifer surface area that contaminant contacts is much less in a fractured aquifer. Consequently, less sorption takes place. Secondary mineralization of zeolites or clay filling along the open joints can increase sorption. The effective porosity is relatively lower in fractured rock. When high hydraulic gradients are present in fractured media, the average linear ground-water velocity can be several orders of magnitude greater than in porous media. Rapid flow velocities are more probable when a liquid release occurs such as a reactor sump water release.

#### 3.1.1.5 Ground-Water Chemistry Criterion

Ground-water chemistry is as site specific as ground-water hydraulics. The major effect of water chemistry (i.e., pH and ionic strength) is on the amount of contaminant sorption. Geochemical data are not available for most nuclear power plant sites. This study bases the ground-water chemistry criterion on typical conditions and not on site specific data.

Most categories created by the first four hydrogeologic criteria are assumed to have similar ground-water chemistries. The presence of clay, specifically illite, can strongly affect geochemical processes in the hydrologic unit, especially in regard to sorption/desorption of radionuclides since illite has a specific affinity for cesium.

### 3.1.2 Definition of Generic Sites

Based on the above criteria 16 individual classifications of hydrogeological parameters resulted. However, these 16 classifications produced only six generic sites when applied to the geotechnical data base of existing and proposed power plant sites in the U.S. Some classifications were not filled because of the geologic improbability of occurrence, as in the case of a unconsolidated carbonate aquifer with primarily fracture premeability. Another reason many classifications were empty is due to the limitations of the geotechnical data base. Specifically, the ground-water chemistry at nuclear power plant locations is unavailable. The ground-water chemistry was assumed consistant except for the silicic-consolidated-fractured category. In this classification were both fractured crystalline rock (i.e., basalt) and shale which is a fractured sedimentary rock. These rock types would form similar melts and ground-water flow would be mainly contained along fractures. However, the ground-water chemistry of these units is different with the crystalline material having an oxidizing environment and shale a reducing environment. The pH of both units is acidic. The differing ground-water chemistry contributes to contaminants in shale being more sorbed during transport than for crystalline rock. In addition shale contains the clay mineral illite that can cause irreversible geochemical reactions. An additional classification for shale media was created by this criterion.

A flow chart showing the classification of the sites is given in Figure 3.1.2-1. The final generic classifications are: 1) porous consolidated carbonate, 2) fractured consolidated carbonate, 3) porous consolidated silicate, 4) porous unconsolidated silicate, 4) fractured consolidated silicate in an oxidizing environment (referred to as fractured crystalline silicate), and 6) fractured consolidated silicate in a reducing environment (referred to as shale media). Table 3.1.2-1 presents the generic classification and the associated common aquifer names. Some of the groupings are expected such as fractured limestones and dolomites. An interesting combination of aquifers occurs in the fractured consolidated silicate classification which includes basalt and granite. These geologic units are formed under very different circumstances and can have a large range of hydraulic properties. However, in the near-surface ground-water enviroment they can be expected to have similar transport characteristics. The largest generic classification is porous unconsolidated silicate with 41 sites. This result is not suprising since many nuclear power plants are located adjacent to surface water bodies which are used as a source of cooling water. Many of the surface water bodies are located on alluvial materials. Breaking this classification into further subgroupings was considered. However, flow and transport properties are similar in this classification and further discrimination of generic differences could not be made.

An examination of the generic classifications and associated common aquifers show that the classification scheme is indeed generic. The number of classifications is not excessively large, there is a representative number of sites in each classification, and each classification contains similar hydrogeologic characteristics.

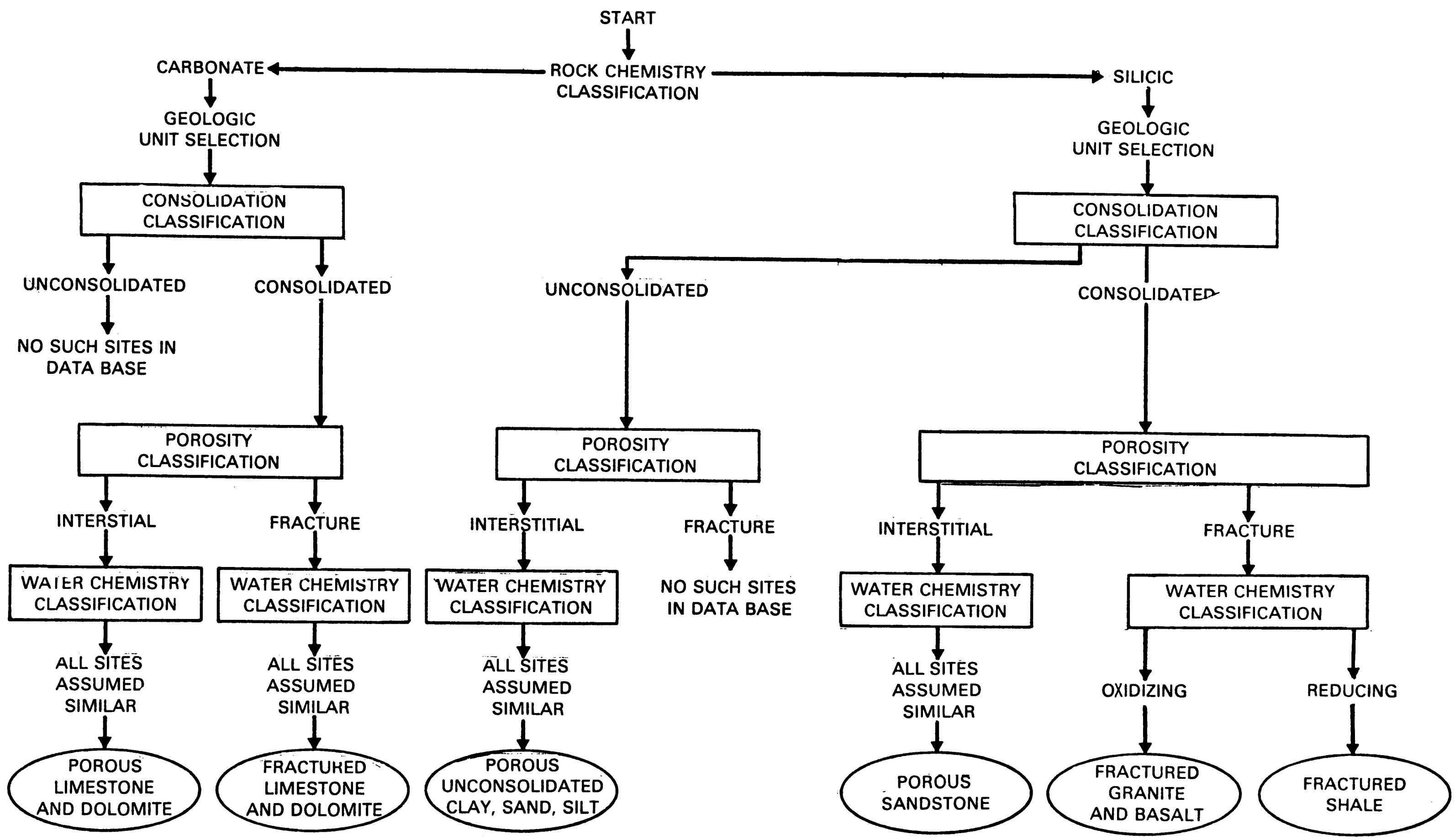


FIGURE 3.1.2-1. Generic Hydrogeologic Classification Scheme for Nuclear Power Plants in the U.S.

TABLE 3.1.2-1. Generic Site Classification

| Generic Classification | Common Aquifers | Number of Sites |
|---|---|---|
| Porous consolidated carbonate | limestone<br>dolomite | 10 |
| Fractured consolidated carbonate | fractured and solutioned limestone<br>fractured and solutioned dolomite | 12 |
| Porous consolidated silicate | sandstone<br>siltstone<br>claystone<br>graywacke<br>arkoses | 13 |
| Porous unconsolidated silicate | clay<br>silt<br>sand<br>conglomerate<br>glacial deposits | 41 |
| Fractured crystalline silicate in oxidizing environment | igneous rocks<br>basalt<br>tuff, granite | 16 |
| Fractured consolidated silicate in reducing environment | shale | 5 |

## 3.2 FLOW PARAMETERS FOR GENERIC SITES

### 3.2.1 Hydraulic Conductivity

Hydraulic conductivity is a property of the saturated geologic medium and the fluid that flows through it. Basically, hydraulic conductivity is a measure of the capacity for flow in a unit area of an aquifer. It is defined by Darcy's Law which states:

$$q = -KI \tag{3.1}$$

where

$q$ = fluid flux rate ($L^3/L^2T = L/T$)
$K$ = hydraulic conductivity ($L/T$), and
$I$ = hydraulic gradient (dimensionless).

Hydraulic conductivity is a spatial parameter which varies in all three dimensions. Inspection of Equation (3.1) shows that hydraulic conductivity has dimensional components of length and time, however this should not be considered ground-water velocity. Hydraulic conductivity at a site is determined by testing of core samples in the laboratory or by field testing. The field tests measure a large volume of rock and test results provide a composite hydraulic conductivity. Even at the field scale, hydraulic conductivities may range over thirteen orders of magnitude for differing geologic materials. Coarse porous media such as gravel and solutioned limestone have the highest hydraulic conductivities. Silt, clay, glacial till, are often tightly compacted and exhibit much lower hydraulic conductivities. Crystalline rocks have the lowest hydraulic conductivities because these materials have few flow channels for water movement. For this study, knowing a site hydraulic conductivity within an order of magnitude was considered adequate for generic characterization.

The hydraulic conductivity, for any particular site, may be imprecise. However, within each generic classification, the grouped hydraulic conductivities are characteristic of the geologic materials existing at nuclear power plant sites in the U.S. The data were fit with a log normal distribution by generic classification and are presented in Figures 3.2.1-1a through 3.2.1-1f. The figures show two standard deviations about the mean of the log hydraulic conductivities. The data transform to log hydraulic conductivities allows a normal probability density function to be fit to the data (Freeze and Cherry 1979). The generic hydraulic conductivities should be examined from two perspectives. First, within each classification, the range of values and the mean value are generically characteristic. Second, comparisons of values among the generic classifications demonstrate which type of site will overall have the highest hydraulic conductivity and the largest expected variations about the mean. The extreme data values from actual nuclear plant sites are indicated by crosses on the left vertical axis. The general ranges of expected hydraulic conductivities for these geologic materials determined for locations not associated with this study (Freeze and Cherry 1979) are given along the right vertical axis for reference purposes.

There is fairly good agreement between the data extremes found at nuclear plant sites and the expected limits. For three classifications (i.e., porous carbonate, porous sandstone, and fractured shale) the site data have values higher than expected. This can be explained as either a possible characteristic of the locations where nuclear power plants have been sited for construction, or is a pessimistic bias in estimation of individual hydraulic conductivity values.

The fractured shale classification contains only five sites and may not be representative of shale media in general. The hydraulic conductivity data are also based on fractured geologic units whereas the expected range of values is given for unfractured shale. This accounts for much of the four orders of magnitude difference between the expected and reported peak values in shale media. The other classifications are within expected limits. None of the classifications have lower than reasonable hydraulic conductivities indicating that the data base is conservative with respect to this parameter.

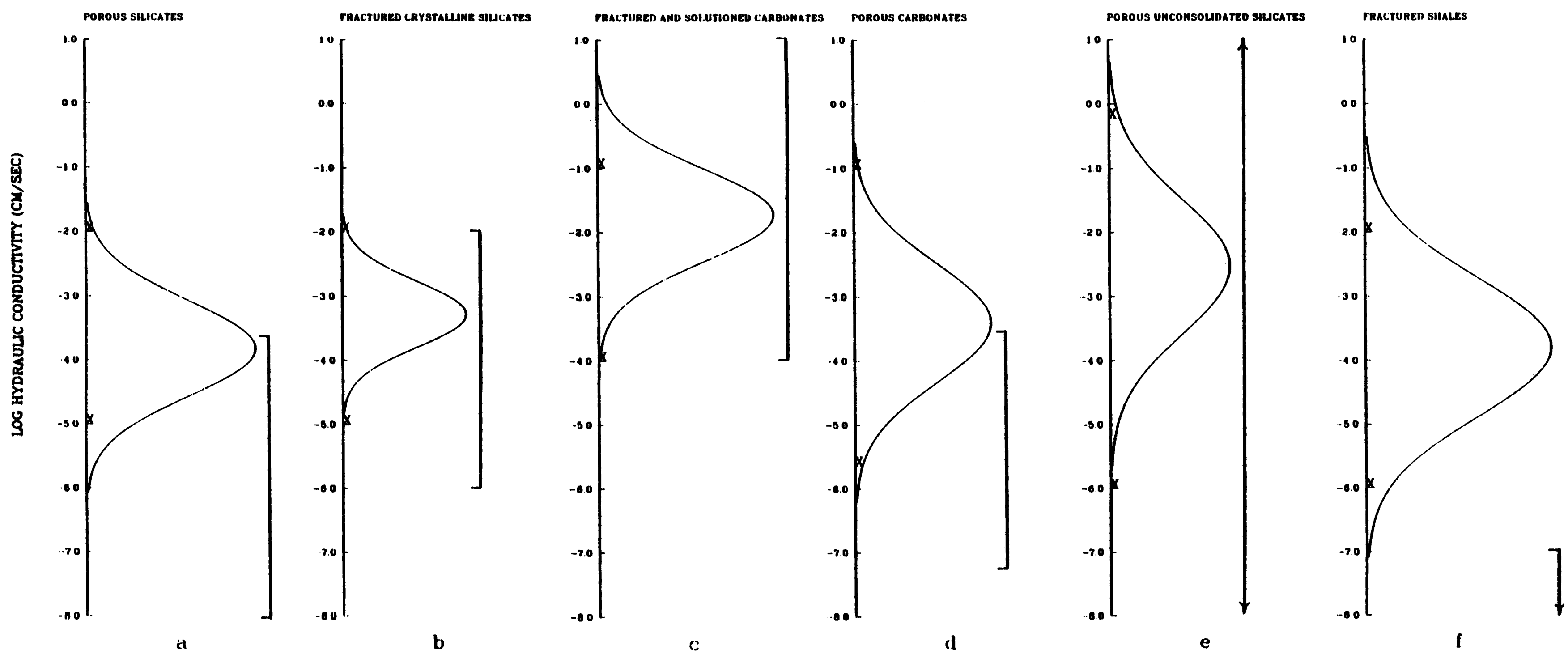


FIGURE 3.2.1-1. $LOG_{10}$ Distribution of Hydraulic Conductivity by Generic Classification. Brackets indicate range of values from Freeze and Cherry 1979.

The highest log mean values are found, as might be expected, in the fractured-solutioned carbonates. These aquifers can achieve open channel flow, and water movement can be relatively unrestricted due to large flow channels. The porous unconsolidated silicate classification has the second highest log mean value of hydraulic conductivity. This result is somewhat unexpected and is probably due to the coarse granular composition of many of the sites. The hydraulic conductivity log mean for fractured consolidated silicates (crystalline), porous carbonates and fractured shale are similar in magnitude. Fractured materials are often most weathered and broken near the surface of the land. The core melt mass would not enter the deep fractured zones (over 50 m) where fracture permeability decreases rapidly with depth (Freeze and Cherry 1979). For this study, the hydraulic conductivities of fractured crystalline silicates may be upwardly biased by their proximity to land surface. The lowest hydraulic conductivities are found in the porous silicates classification. The mean value is not unreasonably low and is a mid-range value as compared to expected limits from Freeze and Cherry (1979). A comparison of the log hydraulic conductivities by generic classifications is given in Table 3.2.1-1.

The absolute hydraulic conductivities are given on a linear scale in Figures 3.2.1-2a through 3.2.1-2f. Again, the data extremes are indicated by crosses along the left vertical axis. These figures highlight the strong data skewness toward low values of hydraulic conductivity. The greatest degree of skewness is found in the fractured and porous silicates classifications. Presented in Figures 3.2.1-2a and 3.2.1-2d these classifications have nearly all of the values less than $5.0 \times 10^{-3}$ cm/sec. There is an intermediate degree of skewness in the porous carbonates (Figure 3.2.1-2c) and in fractured shale (Figure 3.2.1-2f). These data are generally less than $2.5 \times 10^{-2}$ cm/sec. The fractured and solutioned carbonates and porous unconsolidated silicates (Figures 3.2.1-2b and 3.2.1-2e) have relatively less skewness toward zero with the data having values mainly below $5 \times 10^{-1}$ cm/sec. The skewness toward zero is more illustrative of data trends than mean values because of the large range of hydraulic conductivities. Consolidated silicates have the more constrained hydraulic conductivities probably due to the low rate of silica dissolutioning. Another factor affecting consolidated silicate aquifers may be the tendency of secondary mineralization along flow channels. The least skewed data are in the fractured and solutioned carbonates and the porous unconsolidated silicates (i.e., sands, clays and silts). Carbonates may have this characteristic because solutioning varies the hydraulic conductivity with geologic time. Therefore, the chemical changes will tend to add an additional nonrestrictive (whereas secondary mineralization is restrictive) trend to the hydraulic conductivities. The small degree of skewness in the unconsolidated silicates is a function of the wide variations in porosity, permeability, and composition of this classification.

### 3.2.2 Effective Porosity

The volume of interconnected void spaces divided by the total bulk volume. The effective porosity is also reported as a decimal fraction. In this case, the volume of interest is the volume of the continuously interconnected voids which provide pathways for ground-water flow. Ground-water

TABLE 3.2.1-1. Comparison of Generic Hydraulic Conductivity

| Generic Classification | Data Range in Orders of Magnitude | Mean Value, cm/s | Mean Log Value, cm/s | Mean Value[a] Relative to Other Generic Classifications | Standard Deviation About the Log Mean | Standard Deviation[b] Relative to Other Generic Classifications |
|---|---|---|---|---|---|---|
| Fractured crystalline silicates | 3.0 | $1.53 \times 10^{-3}$ | -3.28 | Lower than average | 0.78 | Smaller than average |
| Fractured-solutioned carbonates | 4.0 | $6.42 \times 10^{-2}$ | -1.73 | Higher than average | 1.09 | Average |
| Porous consolidated carbonate | 4.6 | $1.16 \times 10^{-2}$ | -3.41 | Slightly lower than average | 1.40 | Average |
| Porous consolidated carbonate | 3.0 | $1.79 \times 10^{-3}$ | -3.82 | Lower than average | 1.13 | Average |
| Porous unconsolidated silicates | 5.9 | $5.55 \times 10^{-2}$ | -2.53 | Higher than average | 1.59 | Larger than average |
| Fractured consolidated silicates-shale | 4.0 | $2.4 \times 10^{-3}$ | -3.80 | Lower than average | 1.64 | Larger than average |

(a) Average of 6 log mean values is -3.10.
(b) Average of 6 standard deviations about the log mean is 1.27.

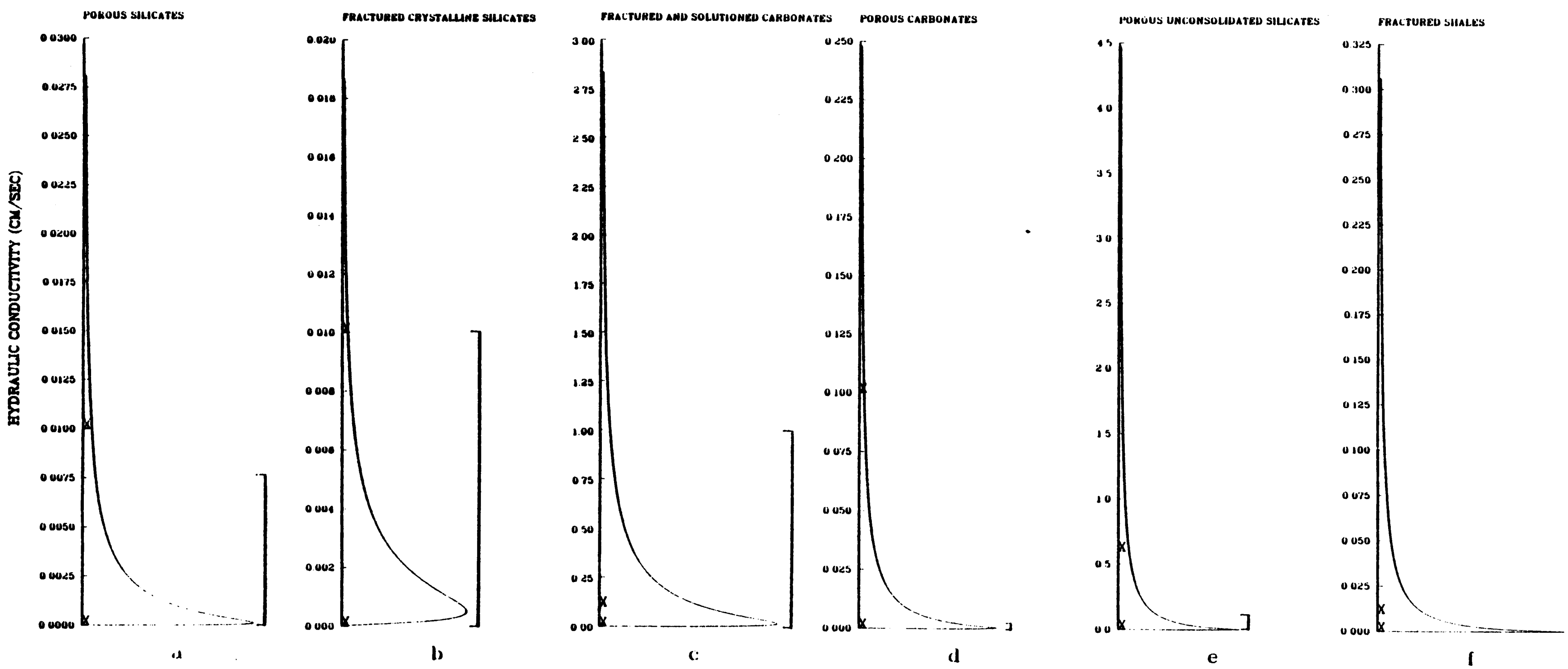


FIGURE 3.2.1-2. Linear Scale Distribution of Hydraulic Conductivity by Generic Classification. Brackets indicate range of values from Freeze and Cherry 1979.

calculations use this parameter to determine flow velocities. There is a positive correlation between effective porosity and hydraulic conductivity.

There are few measured porosities available in the geotechnical data base. At most sites, the effective porosity was estimated by assuming a reasonable value based on geologic rock type. At sites on unconsolidated materials, porosity was estimated for a wide range of geologic materials giving a distribution of values. A statistical analysis on assumed porosities based on a single rock type has little meaning. Representive effective porosities for these generic classifications are presented in Table 3.2.2-1. The effect porosities for unconsolidated silicates are plotted on a log normal distribution in Figure 3.2.2-1 and on a linear scale in Figure 3.2.2-2. Although porosity is not usually considered log-normally distributed this distribution is presented for consistency with other analyses. Also, negative porosity has no meaning which gives credence to a log-normal distribution.

The data extremes in Figure 3.2.2-1 are shown as crosses on the left vertical axis. The effective porosities have a range of about one order of magnitude. The mean log porosity is 0.10 which is typical of unconsolidated sedimentary deposits. The lowest effective porosity for this generic classification is 0.01 for silt and clay materials. The linear plot of effective porosities shows a definite skewness toward values below the mean. This indicates that most sites are located on deposits with some interstitial silt and clay.

### 3.2.3 Hydraulic Gradient

The hydraulic gradient is the slope of the water table or potentiometric surface. The geotechnical data base used for this study determined the

TABLE 3.2.2-1. Effective Porosities for Generic Hydrogeologic Classifications

| Generic Classification | Effective Porosity (Dimensionless) |
|---|---|
| Fractured crystalline silicates | 0.01 |
| Fractured and solutioned carbonates | 0.10 |
| Porous carbonates | 0.10 |
| Porous silicates | 0.01 |
| Porous unconsolidated silicates | Average value 0.16 |
| Fractured Shale | 0.01 |

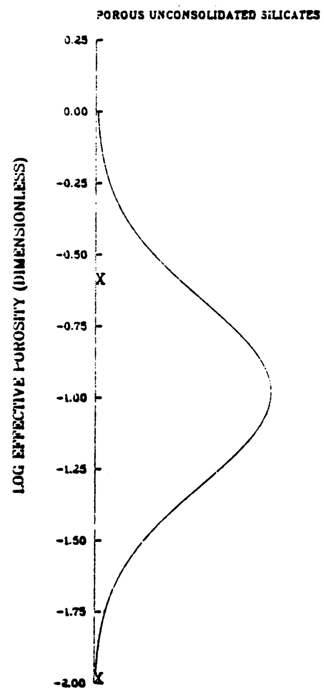


FIGURE 3.2.2-1. $LOG_{10}$ Distribution of Effective Porosity for Porous Unconsolidated Silicates

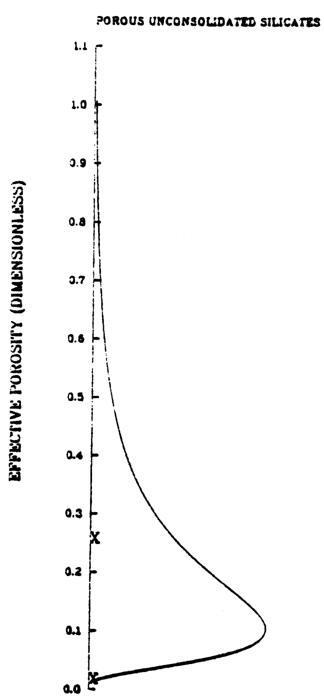


FIGURE 3.2.2-2. Linear Scale Distribution of Effective Porosity for Porous Unconsolidated Silicates

hydraulic gradient by taking the steepest (thus most likely) path along the potentiometric surface from the containment structure to the nearest surface water body. In general, low hydraulic gradients are associated with large hydraulic conductivities and high gradients are found with low conductivities. Hydraulic gradients are a function of aquifer properties and location. Local changes in gradient are related to changes in hydraulic conductivity, the presence of hydraulic boundaries and barriers, as well as hydrologic sources and sinks. A hydrologic source is defined as an addition of water to the aquifer through recharge or injection and a hydrologic sink is where water discharges through wells, springs, interaquifer transfer or into surface water bodies. The hydrogeologic gradient in any given aquifer has a strong relationship to the location in the flow field where it is measured.

The log hydraulic gradient is plotted for two standard deviations about the log mean in Figures 3.2.3-1a through 3.2.3-1f. The data limits are shown as crosses on the left vertical axis as in previous figures. The log distributions show that the highest gradients are found in fractured silicate rock. This generic classification also has a relatively small range of values for hydraulic gradients. Fractured shale and fractured silicates have the smallest range of values which indicate a similarity in fracture hydraulics. The lowest gradients are found in porous carbonates. The relatively low value for this classification may also be related to site physiography (i.e., location of plants sites in areas of low relief). The largest spread between data extremes is in the fractured and solutioned carbonate classification. The upper limit is in fair correlation with other classifications, the lower limit is extremely low due to the possibility of karst conditions and open channel flow. A summarized comparison of the hydraulic gradients is given in Table 3.2.3-1.

TABLE 3.2.3-1. Average Hydraulic Gradient for Generic Hydrogeologic Classifications

| Generic Classification | Mean Log | Arithmetic Mean | Data Range in Orders of Magnitude |
|---|---|---|---|
| Fractured crystalline silicates | -1.2 | 0.070 | 1.2 |
| Fractured and solutioned carbonates | -2.0 | 0.010 | 2.9 |
| Porous carbonates | -2.2 | 0.007 | 1.5 |
| Porous silicates | -1.8 | 0.015 | 2.3 |
| Porous unconslidated silicates | -2.1 | 0.009 | 2.5 |
| Fractured shale | -1.9 | 0.012 | 0.9 |

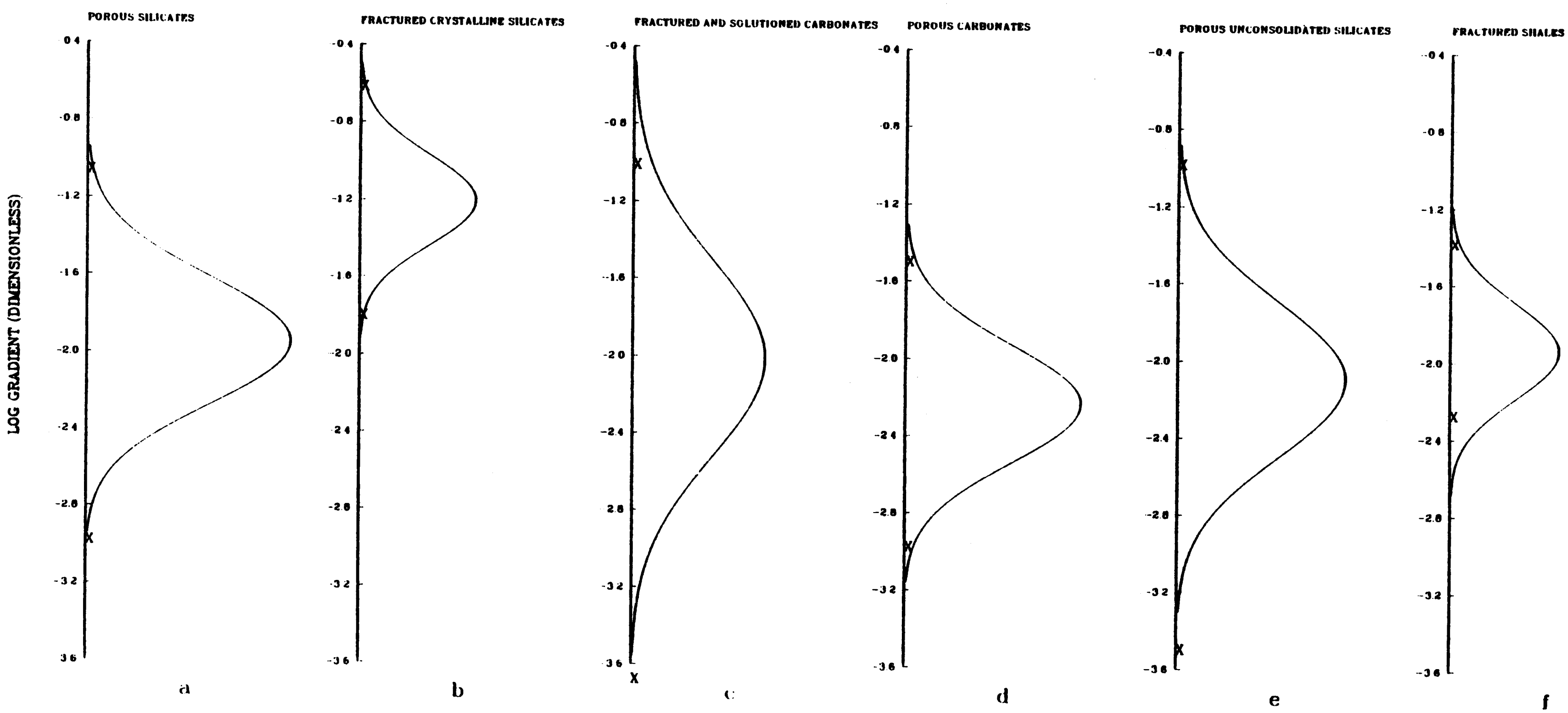


FIGURE 3.2.3-1. $LOG_{10}$ Distribution of Hydraulic Gradient by Generic Classification

The hydraulic gradients are presented in Figures 3.2.3-2a through 3.2.3-2f with the vertical axis as a linear scale. The data extremes are again indicated by crosses. The figures show skewness toward lower values in fractured and solutioned carbonates, porous silicates, and porous unconsolidated silicates classifications. The least skew is seen in Figures 3.2.3-2a and 3.2.3-2f for fractured silicates and fractured shale, respectively. The hydraulic gradients in these classifications are more likely to have high values than in unfractured geologic media. The high gradients in fractured rock are caused by low aquifer transmissivity.

A clear distinction can be seen between the fractured silicate and fractured carbonate classifications (Figures 3.2.3-2a and 3.2.3-2b). The opening of flow channels via chemical dissolution effectively increases hydraulic conductivity and decreases hydraulic gradient.

### 3.2.4 Distance to Nearest Surface Water Body

Nuclear power plants are usually sited adjacent to surface water bodies which serve as a source of cooling water.[a] The geotechnical data base used for this study lists the shortest distance to the surface water body. This distance is assumed to be the approximate ground-water travel distance from the reactor containment to a surface water body. The distance from containment to a surface water body is more a function of site geography and physiography than of the hydrogeological classification. However, there is a relationship between geology and site topography which affects distance to surface water. Such a relationship even if it is slight, will have an important influence on a release of radionuclides into the environment.

The distance to the nearest surface water body is given in meters on a linear scale in Figures 3.2.4-1a through 3.2.4-1f. The data extremes are plotted as crosses on the figures. The distances are skewed toward the lower values but not toward zero. The fractured shale category displays the least skewness which may be due to the limited number (5) of sites. The data are summarized in Table 3.2.4-1.

## 3.3 TRANSPORT PARAMETERS FOR GENERIC SITES

### 3.3.1 Longitudinal Dispersion Coefficient

Contaminant is transported in aquifers by the flow of ground water. Along the flow path contaminant is spread both horizontally and vertically by mechanical dispersion and molecular diffusion. Mechanical dispersion is a result of variations in flow velocity due to aquifer inhomogeneities. These variations in flow velocity can be caused by small scale changes in flow around

(a) The affects of radionuclide release into generic surface water bodies is covered in "The Consequences from Liquid Pathways After a Reactor Meltdown Accident" NUREG/CR-1596.

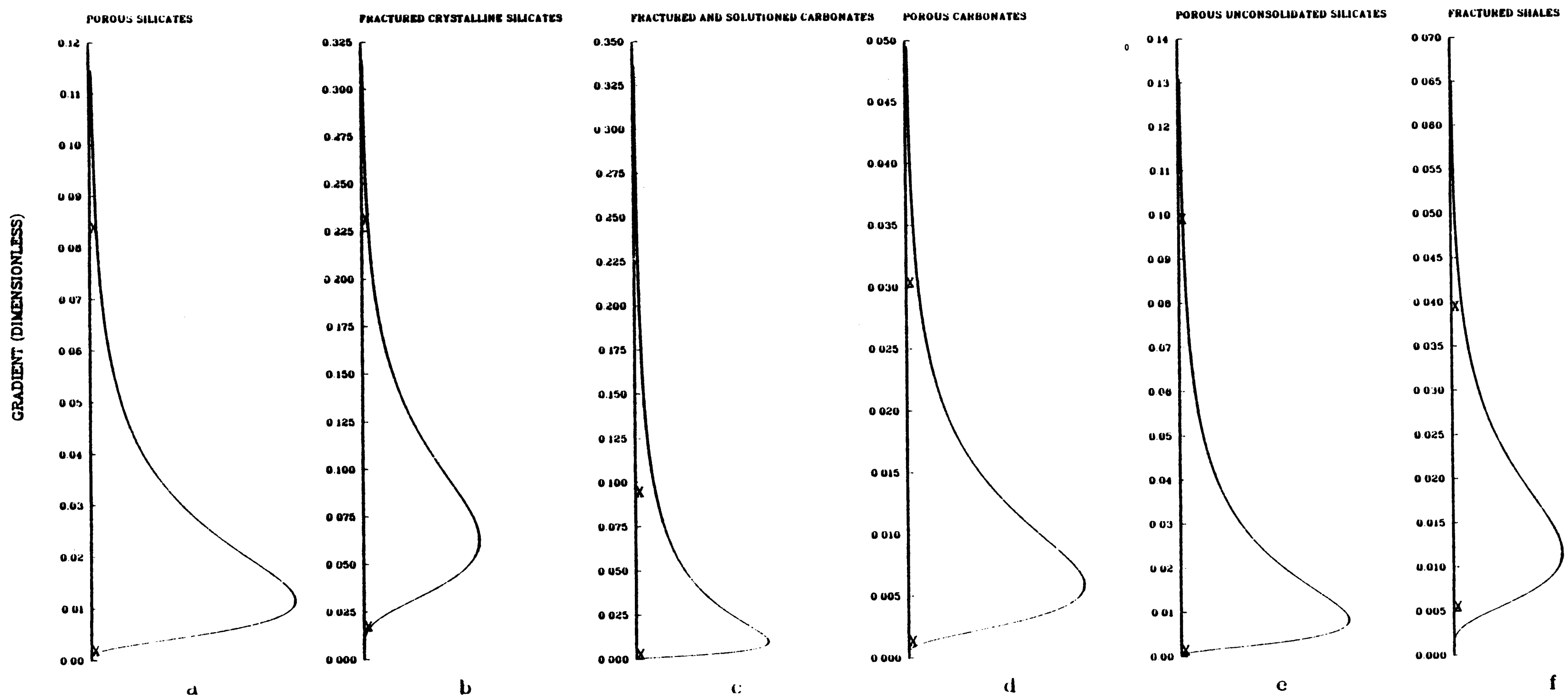


FIGURE 3.2.3-2. Linear Scale Distribution of Hydraulic Gradient by Generic Classification

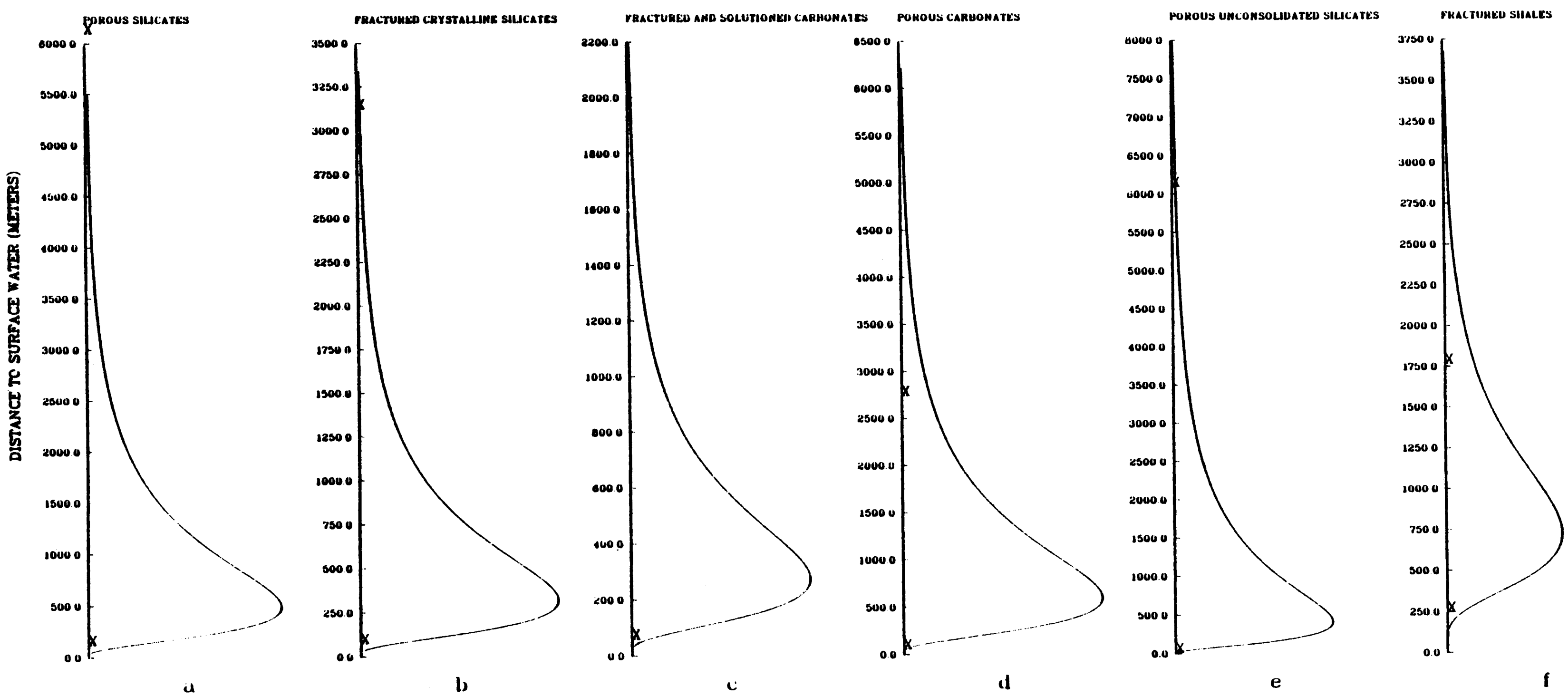


FIGURE 3.2.4-1. Linear Scale Distribution of Reactor Distance from Surface Water

TABLE 3.2.4-1. Average Distance to Surface Water for Generic Hydrogeologic Classifications

| Generic Classification | Arithmetic Mean, m |
|---|---|
| Fractured crystalline silicates | 300 |
| Fractured and solutioned carbonates | 275 |
| Porous carbonates | 650 |
| Porous silicates | 450 |
| Porous unconsolidated silicates | 500 |
| Fractured shale | 700 |

an individual sand particle to regional changes in hydraulic conductivity. The dispersive process is described by the advection-dispersion equation in one dimension as:

$$\frac{\partial C}{\partial T} = D_L \frac{\partial^2 C}{\partial L^2} - \overline{V}_L \frac{\partial C}{\partial L} \qquad (3.2)$$

where

$C$ = concentration ($M/L^3$),

$T$ = time,

$D_L$ = coefficient of hydrodynamic dispersion in the longitudinal direction ($L^2/T$), and

$L$ = ground-water pathline length ($L$), and

$\overline{V}$ = average linear ground-water velocity ($L/T$).

The one dimensional form of the advection-dispersion equation does not account for contaminant spreading transverse, that is at right angles to, the ground-water flow direction. However, in keeping within the scope of the geotechnical data base dispersion values are limited to one dimension.

The coefficient of hydrodynamic dispersion is expressed by its components as:

$$D_L = \alpha_L \overline{V}_L + D^* \qquad (3.3)$$

where

$\alpha_L \overline{V}_L$ = dynamic dispersivity characteristic of the geologic medium (L), and

$D^*$ = coefficient of molecular diffusion ($L^2/T$).

In geologic materials where ground-water velocities are low molecular diffusion is the dominant factor of hydrodynamic dispersion (Freeze and Cherry 1979). Rapid ground-water flow results in dynamic dispersivity being the major factor in dispersion. In this study, molecular diffusion is ignored for two reasons. First, in various geologic media rates of molecular diffusion are uncertain. Values for this parameter can only be estimated within two orders of magnitude. Secondly, molecular diffusion along the flow path is important only when ground-water velocities are slow or when travel distances are long. In these cases, radioactive decay often diminishes the contaminant to very low concentrations prior to surface water discharge. These assumptions provide a realistic yet conservative approach for consideration of this parameter.

There are no measured values of dispersion for nuclear power plant sites. Field tracer tests which inject a nonreactive chemical agent into an aquifer and monitor concentrations at distance are required to determine true dispersion. Few of these long-term tests are conducted in the field and most dispersion data are derived from laboratory experiments involving flow through isolated columns. The applicability of these measurements to field situations and the parametric content of the advection-dispersion equation are currently undergoing critical review by several researchers (Matheron and DeMarsily 1980; Gelhar et al. 1979; Simmons 1982; Molz et al. 1983).

The dynamic dispersivity was estimated for each site based on the geologic materials. Estimates of dynamic dispersitivity are presented in Table 3.3.1-1.

TABLE 3.3.1-1. Estimated Dynamic Dispersivity for Various Geologic Materials (Source: Yeh 1981)

| Geologic Material | Estimated Dynamic Dispersivity, m |
|---|---|
| Clay-Silt | 1 |
| Silty Clay | 5 |
| Silty Marl | 10 |
| Sandy Silt | 25 |
| Sand | 50 |
| Porous Consolidated[a] | 50 |
| Fractured[a] | 100 |

(a) Estimated by authors.

### 3.3.2 Retardation of Radionuclides by Sorption

#### 3.3.2.1 Definition of Sorptive Process

The concentration of contaminant can be altered by chemical reactions along the ground-water flow path. The chemical reactions among the contaminant, native ground-water and aquifer material are complex and interrelated. Possible reactions include: adsorption, fixation, complexing, colloid formation, precipitation, solutioning and ion exchange. These processes tend to alter contaminant concentrations by retarding transport with respect to ground-water velocity, that is causing the contaminant to move at a different rate than the ground water. The reactions may change in time and space as the contaminant moves away from the source. The vigor of these reactions depends upon the chemical species present, ionic strengths, pH (acidity) and Eh (oxidation potential) of the ground water and the type of minerals in the geologic unit. Description of these processes by coupled geochemical ground-water transport models is in the early stages of development by researchers.

Site specific ground-water chemistry is lacking for most nuclear power plant sites. The most basic measure of ground-water chemistry is pH and this parameter is available for less than half of the sites. Most sites have measured or estimated ground-water pH values between 7.0 and 8.5 which is slightly alkaline (Niemczyk et al. 1981).

Sorption is a process by which chemical reactants are adsorbed or adhere as a thin film onto the surfaces of solids and then by desorption re-enter the ground-water flow field. The mechanism of retardation is a result of contaminant not being transported during the time it is sorbed onto the rock matrix. The process is modeled by assuming that reactions are instantaneous, in equilibrium and reversible. The partitioning of contaminant between solid phases and liquid in a porous medium is described by the equilibrium distribution coefficient known as $K_d$. Aquifer properties of porosity and bulk mass density are also parameters in the determination of retardation by sorption.

When more than one chemical species is present, which would most likely be the case for a core melt accident, each species has an individual value of the equilibrium distribution coefficient and the contaminant stream becomes chemically segregated. The retardation by sorption is described by:

$$R_d = \frac{V}{V_c} = 1 + \frac{\rho_b K_d}{n_e} \tag{3.4}$$

where

$R_d$ = retardation factor expressed as a ratio of the ground-water velocity to the radionuclide species velocity.
$V$ = ground-water velocity (L/t),
$V_c$ = contaminant velocity (L/t),
$\rho_b$ = mass bulk density ($M/L^3$),
$K_d$ = equilibrium distribution coefficient ($L^3/M$), and
$n_e$ = effective porosity expressed as a fraction.

The value of the equilibrium distribution is empirically determined in the laboratory by batch (static) experiments or dynamic experiments in which the contaminant flows through a column of aquifer material. The batch laboratory tests give a representative value for the $K_d$ of the chemical species while column experiments can measure the retardation directly. Field tests for this parameter are less numerous. Comparison between laboratory batch and column experiments indicate that values can range over an order of magnitude for similar geologic materials. Ground-water chemistry changes as mentioned above can induce additional large scale variations in $K_d$ values.

Equation (3.4) is valid for granular porous media where there is a large surface area for contaminant to be sorbed. The presence of very fine grained material such as clay enhances the sorption process. By contrast in fractured aquifers the surface area of rock that the contaminant contacts with is much less. The contaminant is largely confined to the fractures until matrix diffusion forces it into the interior of the rock. Therefore contaminant transport and retardation have two fundamental differences in fractured rock. First, the contaminant is hydraulically confined to the fracture for much of its transport. Second, there is much less surface area onto which the contaminant can be sorbed. The texture of the fracture surface and any secondary mineralization are also factors which affect contaminant transport under these conditions.

Examination of Equation (3.4) shows that if typical fractured aquifer parameters are input, the low value of porosity and high value of mass bulk density will cause a large computed retardation. Clearly this approach is incorrect because conceptually less retardation should occur on fracture surfaces. A better description of retardation in fractured systems than given by Equation (3.4) is needed to describe the sorptive process.

When radionuclide transport in fractured rock occurs over long time periods, molecular diffusion of contaminant into the rock matrix is an important factor in retardation (Neretnieks 1980). Diffusion would continue to carry radionuclides into low velocity zones as long as a concentration gradient existed between the fracture and the rock matrix. The net effect would be an equilibrium distribution coefficient that increases with time as observed in tests of retardation of strontium and cesium in granite (Allard et al. 1978). In these tests the contaminant concentration was found to be related to the log of the square root of time. These processes are not considered by $K_d$ mechanisms which assume that reactions are instantaneous, in equilibrium and reversible. Applying $K_d$ values to geologic media and not considering diffusion over long times can lead to an under-estimation of retardation. An equilibrium distribution coefficient based on fracture hydraulics has been proposed (Burkholder 1976) which requires a description of the fracture geometry. Knowledge at this level of detail of the hydrologic characteristics is beyond this study and most other studies on fractured aquifers. In some instances where fracturing is extensive the retardation is computed with Equation (3.4) by assuming that on a regional scale the fracture system performs as an equivalent porous media. The fracture systems in the geotechnical data base for this study do not meet this requirement in that the porosities are much below those found in porous media.

At the present time there is no standard method or accepted means for computing retardation factors in fractured media. However, not accounting for sorption in fractured media is undesirable because: 1) sorption is known to take place in fractures, 2) zero retardation is unduly conservative and would result in unrealistically high rates of contaminant discharge, and 3) it would not allow for the gross variations in sorption among the generic classifications. In keeping with the "order of magnitude" approach of this generic analysis, a contaminant velocity is determined by applying a correction factor to laboratory $K_d$ data. In developing this methodology it was also recognized that the ground-water velocities predicted by the hydrogeologic data base may already be biased toward high values by conservative estimates of hydraulic parameters. Therefore, computation of a retardation factor for fractured media was accomplished with the understanding that it only provided a means of scaling the known differences in retardation among various geologic environments.

A retardation factor in fractured aquifers is determined by computing the $K_d$ value based on a mass value by correcting for the fraction of the aquifer exposed to contaminant. To accomplish this, the equilibrium distribution coefficients determined from laboratory results with crushed rock are divided by the ratio of fracture porosity over crushed rock porosity. The $K_{df}$ value is defined as:

$$K_{df} = \frac{K_d \cdot \text{fracture porosity}}{\text{crushed rock porosity}} \tag{3.5}$$

where

$K_d$ = granular media equilibrium distribution coefficient, e.g.,
(crushed rock porosity = 0.25)
(fractured porosity = 0.01).

Therefore, the resulting $K_{df}$ in this example is 25 times less than the $K_d$ value for porous media of the same rock type.

#### 3.3.2.2 Values of Equilibrium Distribution Coefficients

The values of equilibrium distribution coefficients were determined by an extensive literature search for previous test results for the radionuclides strontium-90, cesium-137 and ruthenium-106.

The choice of these radionuclides to characterize a core melt is detailed in Section 2.3.1. Each generic hydrogeological classification is discussed separately below and summary Table 3.3.2-1 provides a list of representative equilibrium distribution coefficients for each radionuclide.

##### Fractured Crystalline Silicates

Most references for this rock type are based on tests in granite or gneiss. These reports give data for strontium and cesium but not ruthenium. Erdal et al. (1979); Tschurlovits (1979); Skagius et al. (1982); Landstrom (1978); Torstenfelt et al. (1982); and Walton et al. (1982) have conducted the

TABLE 3.3.2-1. Equilibrium Distribution Coefficients

| Reference | Radio-nuclide | $K_d$(mℓ/g) |
|---|---|---|
| Erdal et al. (1979) | 90-Sr | 12 |
| | 137-Cs | 300 |
| Tschurlovits (1979) | 90-Sr | 2-15 |
| | 137-Cs | 60-2500 |
| Torstenfelt et al. (1982) | 90-Sr | 7-30 |
| | 137-Cs | 100-400 |
| Skagius et al. (1982) | 90-Sr | 7 |
| | 137-Cs | 10-15 |

most recent work on silicate materials. For strontium the $K_d$'s range from 2-15 mℓ/g and cesium $K_d$'s range from 60-2500 mℓ/g. The equilibrium distribution coefficients are listed by source in Table 3.3.2-1. A clear understanding of retardation of ruthenium is given in Onishi et al. (1981) which demonstrates that ruthenium is mobile mainly in cases of disposal of fuel reprocessing wastes containing high nitrate. In natural environments with neutral pH and not excessive nitrate concentrations the element ruthenium should not be ground-water coincident. An estimated reasonable value of $K_d$ for ruthenium is 50 mℓ/g.

Fractured and Porous Consolidated Carbonates

These two generic classifications have the same rock chemistry and are discussed as one group. Fractured and solutioned carbonate equilibrium distribution coefficients are corrected for a lower porosity to differentiate them from porous carbonates. There are few references for $K_d$ values in this rock type. In non-saline ground-water conditions $K_d$ values that are most probable are 1.4 to 20 mℓ/g for strontium and 1.3-2000 mℓ/g for cesium from: MaClean et al. (1979); Seitz et al. (1979); Serne et al. (1977); Relyea et al. (1979); and Relyea and Serne (1979). There are no references for ruthenium in this generic rock type and a reasonable estimated value of 50 mℓ/g is used for this study.

Porous Consolidated and Unconsolidated Silicates

Consolidated and unconsolidated silicates are considered together because they have similar rock chemistries. A distinction is made in determining equilibrium distribution coefficients between geologic materials that are described as "dirty" and "clean". Geology literature often refers to aquifers that contain significant quantities of clay and silt as "dirty" (e.g., a dirty sandstone). The inclusion of these fine particles in an aquifer provides a larger surface area and more locations for sorption to take place. A "clean" sandstone does not have interstitial clay or silt and less sorption is expected. For cesium the references are: Baetsle et al. (1964); Barney, and Anderson (1979); Barney and Brown (1980); Berak (1963); Coles et al. (1980); Dosch, and Lynch (1978); Erdal et al. (1979); Erdal et al. (1980); Harwell (1980); Janzer et al. (1962); Meyer et al. (1978); Meyer (1979); Meyer (1980); Nork and Fenske (1970); Nork et al. (1971); Relyea et al. (1978); Relyea et al. (1979); Rhodes (1957); Routson (1973); Schmal (1972); Tamura (1972); and Wilding and Rhodes (1963). The cesium $K_d$ values for porous silicates ranged from 0-100 mℓ/g for clean aquifers and 70-3000 mℓ/g for dirty aquifers.

The equilibrium distribution coefficient for strontium was determined by examining the following references: Baetsle and Dejonghe (1962); Baetsle et al. (1964); Barney and Anderson (1970); Barney and Brown (1980); Berak (1963); Cerrai et al. (1969); Dosch (1980); Dosch and Lynch (1978); Duursma et al. (1974); Erdal et al. (1979); Erdal et al. (1980); Francis and Bondietti (1980); Gardner and Skulberg (1964); Harwell (1980); Nork and Fenske (1970); Nork et al. (1971); Relyea et al. (1978); Relyea et al. (1979); Rhodes (1957); Routson (1973); Schmal (1972); and Wilding and Rhodes (1963). The strontium $K_d$ values range from 1-30 mℓ/g for a clean porous silicate to 50-2000 mℓ/g for a dirty porous silicate.

Ruthenium equilibrium distribution coefficients were examined for cases not involving fuel reprocessing wastes. These wastes contain high levels of nitrate which mobilize ruthenium. At the Nevada Test Site ruthenium migration was observed in an alluvial aquifer flowing through an underground atomic bomb melt glass. About one percent of the total inventory of ruthenium was migrating at the velocity of the ground-water (Coles and Ramspott 1982). The following references were used: Aston and Duursma (1973); Collet et al. (1968); Duursma (1973); Gardner and Skulberg (1964); Kepak (1966); Rhodes (1957b); Schell et al. (1979); Schell et al. (1980). The range of expected $K_d$ values for a clean porous silicate could not be reasonably determined from the ruthenium references. There are too few measurements to state with certainty what the expected range of $K_d$'s might be. Certainly the range of values covers more than an order of magnitude. A $K_d$ of 50 mℓ/g is a reasonable estimate for ruthenium sorption in a clean porous silicate. In a dirty porous silicate there are sufficient tests to set the range of ruthenium $K_d$ values at 200-700 mℓ/g.

Shale Media

Laboratory tests on crushed shale samples have a wide range of results. The references for equilibrium distribution coefficients are: Barney and Grutzeck (1977); Barney and Anderson (1979); Tewhey et al. (1978); and Erdal et al. (1979). Laboratory tests on shale report $K_d$ values ranging from 17-156 mℓ/g for strontium, 183-8000 mℓ/g for cesium, and 300-438 mℓ/g for ruthenium.

Selected Equilibrium Distribution Coefficient

Judgment was used to select a representive $K_d$ value for each radionuclide for each generic site classification. Porosity corrections for fractured media were applied to $K_d$'s determined from crushed rock samples. The resulting $K_d$

values are more realistic than applying a single $K_d$ for each radionuclide to all geologic environments. Equilibrium distribution coefficients used in this study and in previous reports on core melt accidents are presented in Table 3.3.2-2.

#### 3.3.2.3 General Comments on Geochemistry and Sorption

The geochemical effects on sorption are assumed uniform within each generic classification. At a site where mitigative measures would be needed the ground-water chemistry would be an important consideration in the choice of chemical treatment methods and predictions of sorption. We can make several observations as to the effect of ground-water chemistry on sorption that applies to all sites. These are listed below for the three radionuclides of concern (i.e., strontium-90, cesium-137, and ruthenium-106).

##### Variations in Equilibrium Distributions Coefficients

Laboratory values for $K_d$'s can be subject to wide variations as noted in Section 3.3.2.2. The range of $K_d$ values for equivalent rock samples was documented by an interlaboratory comparison of batch tests (Relyea and Serne 1979). In these tests cesium and strontium sorption on linestone showed over an order of magnitude variation in resultant $K_d$ values. There was much less variation in test results for cesium and strontium sorption on silicate (basalt). The mechanisms responsible for the wide range of values have been attributed to a strong dependence of sorption to concentration (Seitz et al. 1978, and Anderson et al. 1981). Diffusion of contaminant into the rock matrix was considered as the mechanism for variations in $K_d$ values by Neretniek's reinterpretation of the data in Seitz et al. (1978) (Neretnieks 1980).

The size of the rock particle has also been observed to effect $K_d$ values. Tests of cesium sorption on carbonate showed that $K_d$ was proportional to particle surface area for large particles (diameter >0.2 mm), but proportional to mass for smaller particles (Rancon 1967). A diffusion mechanism was also believed to be responsible for these $K_d$ variations. The diffusion of contaminant into the rock over time is just as important as the sorption equilibrium values in determining retardation (Neretnieks 1980). Not accounting for diffusion in time plus concentration dependent experiments may be responsible for some of the reported range in values. Incorporation of time dependent $K_d$ values into ground-water transport calculations is in the early stages of development.

In summary, four general statements can be made concerning computational time-concentration dependent retardation:

1. Retardation mechanisms are not presently parametrically defined and $K_d$ values are emperically determined. These values have a wide range of reported results for a single nuclide in similar geologic materials.

2. Time-concentration dependence is observed in long-term laboratory tests. Cesium is noted for this characteristics possibly because

## TABLE 3.3.2-2. Generic Equilibrium Distribution Coefficients

| Generic Classification | Radio nuclide | Range in Kd values mℓ/g | Repre-sentative Kd's mℓ/g | Porosity Correction | Kd Value This Study mℓ/g | Kd Value RSS 1975 mℓ/g | Kd Value LPGS 1978 mℓ/g |
|---|---|---|---|---|---|---|---|
| Fractured silicates | Sr | 2 - 15 | 10.0 | 0.04 | 0.4 | 2.0 | 20.0 |
| | Cs | 60 - 2500 | 200.0 | 0.04 | 8.0 | 20.0 | 200 |
| | Ru | - | 50.0 | 0.04 | 2.0 | 0.0 & 4.0 | 0.0 & 4.0 |
| Fractured carbonates | Sr | 1.4 - 20.0 | 5.0 | 0.4 | 2.0 | 2.0 | 20.0 |
| | Cs | 1.3 - 2000 | 60.0 | 0.4 | 24.0 | 20.0 | 200 |
| | Ru | - | 50.0 | 0.4 | 20.0 | 0.0 & 4.0 | 0.0 & 4.0 |
| Porous carbonates | Sr | 1.4 - 20.0 | 5.0 | - | 5.0 | 2.0 | 20.0 |
| | Cs | 1.3 - 2000 | 60.0 | - | 60.0 | 20.0 | 200.0 |
| | Ru | - | 50.0 | - | 50.0 | 0.0 & 4.0 | 0.0 & 4.0 |
| Porous silicate | Sr | 1 - 30<br>50 - 2000 [a] | 10<br>50 [a] | 0.04 | 0.4<br>2.0 [a] | 2.0 | 20.0 |
| | Cs | 0 - 100<br>70 - 3000 [a] | 50<br>300 [a] | 0.04 | 2.0<br>12.0 [a] | 20.0 | 200.0 |
| | Ru | 200 - 700 [a] | 50<br>200 [a] | 0.04 | 2.0<br>8.0 [a] | 0.0 & 4.0 | 0.0 & 4.0 |
| Porous unconsolidated silicate | Sr | 1 - 30<br>50 - 2000 [a] | 10<br>50 [a] | - | 10.0<br>50.0 [a] | 2.0 | 20.0 |
| | Cs | 0 - 100<br>70 - 3000 [a] | 50<br>300 [a] | - | 50<br>300 [a] | 20.0 | 200.0 |
| | Ru | 200 - 700 [a] | 50<br>200 [a] | - | 50<br>200 [a] | 0.0 & 4.0 | 0.0 & 4.0 |
| Fractured shale | Sr | 10 - 160 | 30 | 0.04 | 1.2 | 2.0 | 20.0 |
| | Cs | 180 - 8000 | 1000 | 0.04 | 40 | 20.0 | 200.0 |
| | Ru | 300 - 440 | 200 | 0.04 | 8 | 0.0 & 4.0 | 0.0 & 4.0 |

(a) Geologic Materials Containing Clay and Silt cesium has a high diffusivity. This would support the diffusion mechanism as the cause of time dependent $K_d$ values. Diffusion rates for contaminant entering the rock matrix would be greater than the rates for contaminant leaving the matrix because of the difference in concentration gradients.

3. Coefficients for these mechanisms in various geologic media have not been experimentally determined. Measurement of the depth and number of micro fractures as well as fracture surface area have been shown to be important (Neretnieks 1980).

4. The dominant mechanisms for retardation are related to the site specific geochemical environment. Geochemical processes that alter contaminant mass transport will vary among geologic sites.

pH Effects

When the pH is constrained between values of 5 to 10 the rock matrix will not be dissolved at an appreciable rate and the sorption of strontium and cesium will not be altered. This lack of sensitivity to pH is caused by the fairly simple aqueous chemistry of strontium and cesium. They remain simple cations $Sr^{2+}$ and $Cs^{+}$ in ground water at these pH values. Ruthenium adsorption versus pH is more complicated. Ruthenium aqueous chemistry suggests that at pH values 5 to 8 there is high adsorption, below pH 5 there would be lower adsorption. Above pH 9 adsorption also decreases because ruthenium solution species favor anionic forms which do not readily adsorb onto geologic media. The following references were used: Baetsle et al. (1964); McHenry (1954, 1955, 1958); Rhodes (1957); Rhodes and Nelson (1957); and Prout (1958, 1959).

Eh Effects

Eh does not affect adsorption of strontium and cesium. As ground water becomes more reducing (i.e., contains less dissolved oxygen) it is predicted that ruthenium adsorption would increase as the oxidized forms of ruthenium $RuO^{4-}$ and $RuO_4^{2-}$ would convert to $Ru^{4+}$ and $Ru^{3+}$ (Relyea and Washburne 1980; Onishi et al. 1981).

Temperature Effects

Between 4° and 60°C cesium-137 adsorption seems to drop slightly (a factor of 2 to 3) with increasing temperature. Between these two temperature ranges strontium-90 adsorption is not affected significantly. The thermal effects on sorption of ruthenium-106 are not mentioned in the literature reviewed. The following references were used: Ames et al. (1981, 1982, 1983a, 1983b); Ames and McGarrah (1980a, 1980b); Barney (1982); Daniels et al. (1981); Erdal (1979, 1980); McKinley and Greenwood (1980); McKinley and West (1981a, 1981b); and Salter et al. (1981a, 1981b).

### Ionic Strength

Both strontium-90 and cesium-137 adsorption are directly affected by increases in ionic strength. As competing cations ($Na^+$, $K^+$, $Ca^{2+}$, $Mg^{2+}$) increase both elements show diminished adsorption. In salt brines, Sr and Cs adsorption is essentially zero. This is due to cation competition for positive exchange sites on the rocks and clays. For limestone and dolomite rock, the high ionic strength (high $Ca^{2+}$) can cause precipitation of more calcite-dolomite and thus some retention of strontium-90 by co-precipitation processes. In general, ruthenium-106 adsorption is not affected by increases in salt content from (distilled water to seawater). Ruthenium-106 sorption is not controlled by a cation exchange process.

### Uniqué Properties of Rocks

Carbonate-based rocks such as limestone and dolomite can enhance, via precipitation, strontium removal when compared to silicic rocks. High cation exchange capacity is observed in sediments and in shale, which is a high adsorber. Also, illite-clay-bearing sediments and rocks such as shale, are very good adsorbents. Cesium fits between the plate crystal structures and gets "locked in". Precipitation without subsequent dissolution would deposit a radionuclide permanently in the rock matrix. The following references were used: Coleman et al. (1963); Lomenick et al. (1967); Sawhney (1964); Jacobs (1962); Tamura (1963a, 1963b); Tamura and Jacobs (1960, 1961).

### Dissolved Organics

In general, strontium-90 and especially cesium-137 are little affected by dissolved organics. They form very weak soluble organic complexes. Most of the strontium-90 and cesium-137 remain $Sr^{2+}$ and $Cs^+$ thus organics do not interfere with adsorption. There are only a few studies on ruthenium-106 adsorption in the presence of organics. The studies split equally between claiming increased adsorption and decreased adsorption. Many of the authors used very high concentrations of organics ($10^{-4}$M. These concentrations are at least 100 times greater than any expected ground-water concentrations. The following references were used: Amy (1972); Bovard et al. (1968); Essington et al. (1965); Essington and Nishita (1966); Kilikov (1968); Kilikov and Molchanava (1972); Schell et al. (1980); and Wilding and Rhodes (1963).

### Effect of Inorganic Ligands

Inorganic ligands ($Cl^-$, $SO_4^{2-}$, $HCO_3^-/CO_3^{2-}$, $NO_3^-$ and $PO_4^{3-}$) do not affect Cs adsorption and only slightly affect Sr adsorption. If the concentrations of sulfate, carbonate or phosphate become quite high, Sr adsorption increases because of precipitation of gypsum $CaSO_4 \cdot 2H_2O$, calcite $CaCO_3$ and apatite $Ca_3(PO_4)_2$. Strontium can substitute for calcium in some of the crystal lattices.

Complexes of ruthenium-106 nitrate are very strong and can keep ruthenium mobile. Reprocessing wastes are very high in nitrate, and at Hanford, Washington, ruthenium migration is observed. In normal ground waters nitrate levels should not be high enough, even in organic rich sediments, to impact ruthenium adsorption. The following references were used: Ames and Rai (1978) and Rai and Serne (1978).

### 3.3.3 Effective Bulk Density

The retardation equation [Equation (3.4)] requires a known mass bulk density ($M/L^3$) for each nuclear power plant site. This information is not contained in the geotechnical data base and was estimated for each site. In consolidated formations the effective bulk density was estimated by Equation 3.6:

$$\rho_b = \rho_m (1 - n_e) \tag{3.6}$$

where

$\rho_b$ = effective bulk density,
$n_e$ = effective porosity, and
$\rho_m$ = average mass density of rock (2.65 $g/cm^3$).

Unconsolidated deposits were assigned a bulk density based on the sand, silt and clay content. Sandy units were given a value of 1.4 $g/cm^3$, silty units were assumed to have a bulk density of 1.6 $g/cm^3$ and clay units were assumed to have a value of 1.8 $g/cm^3$ as suggested by Yeh (1981). The mass bulk density of an aquifer can be estimated to a fair degree of accuracy and this is not considered a sensitive parameter.

## 3.4 TRANSPORT EQUATIONS FOR ONE DIMENSIONAL SIMULATION

### 3.4.1 Modeling Objectives for Generic Classifications

The geotechnical data base, given in Section 3.6, was used to simulate the transport of radionuclides in the ground-water flow system from each of the nuclear power plant sites to their respective nearest surface water body. The purpose of this modeling effort is to determine the generic hydrogeological characteristics of a radionuclide release into surface water. The single valued parameters contained in the geotechnical data base dictate that simple mathematical models be used. There is little to be gained by applying a complex multi-dimensional model to a single dimensional data set under steady state ground-water conditions. At the same time it must also be recognized that simple modeling, in this case one dimensional, does not produce the same degree of accuracy that would result from an extensive site specific field and modeling study.

The modeling approach is centered around the determination of what constitutes a generically characteristic radionuclide release to a surface water body. Simply using average or median hydraulic characteristics for each generic classification to evaluate transport would have produced unacceptable results for four reasons. First, hydraulic parameters at a site may have a negative correlation (e.g., gradient and hydraulic conductivity) and thus average parameters may not occur simultaneously at actual sites. Second, mass transport equations of radionuclides are non-linear and average values would not necessarily produce an "average" radionuclide discharge to surface water. The third reason is that even within a generic hydrogeologic classification the hydraulic parameters cover several orders of magnitude. Average parameter values would not represent the range of feasible radionuclide discharges. The variations in radionuclide discharges are generically contained in the geotechnical data base and should be carried through the analysis. Fourth, the actual site data may contain associations and correlations that are unique to nuclear power plant sites which should also be carried through the analysis.

The generic hydrogeologic characteristics of a radionuclide release were determined by modeling each power plant site and presenting the results by generic classification. The rational of this modeling approach is that consideration of actual sites (with subsequent analysis as a generic group) is more beneficial than simply analyzing aquifer properties with assigned average values. The modeling results for 97 nuclear power plants are presented by generic classification and not as individual nuclear plant sites.

### 3.4.2 Equation for Contaminant Transport in Ground Water

The transport of radionuclides was simulated using the computer code AT123D (Yeh 1981). The code represents an analytical transient model of radionuclide transport and can be applied in one, two, or three dimensions. For this study the one dimensional option was appropriate. A simplified form of the transport equation [Yeh 1981, equation (7)] can be written as:

$$\frac{\partial C}{\partial t} = \nabla \cdot (\bar{\bar{K}} \cdot \nabla C) - \nabla \cdot \vec{U}C - \lambda C + \frac{\dot{M}}{\eta_e R_d} \tag{3.7}$$

where

$C$ = dissolved concentrations of contaminant ($ML^{-3}$),
$t$ = time (t),
$\nabla$ = gradient operator,
$K$ = retarded dispersion tensor $D_L/Rd$ ($L^2\ T_{-1}$),
$U$ = retarded flow velocity (L/T),
$\lambda$ = radionuclide decay constant ($T^{-1}$),
$R_d$ = retardation factor,
$\dot{M}$ = rate of contaminant mass release ($ML^{-3}T^{-1}$), and
$\hat{}_e$ = effective porosity (dimensionless).

This equation is also subject to a series of boundary conditions that are not detailed in this report. The retarded flow velocity (U) is defined as:

$$U = \frac{K \cdot I}{R_d \cdot \eta_e} \tag{3.8}$$

where

$R_d$ = retardation factor (Eq. 3.4)
$\acute{K}$ = hydraulic conductivity (L/T),

$I$ = hydraulic gradient (dimensionless), and
$n_e$ = effective porosity (dimensionless).

A continuous release of contaminant is further simplified to:

$$C(X,Y,Z,t) = \int_0^t \frac{\dot{M}}{h_e Rd} F_{ijk} (X,Y,Z,t;\gamma)\, d\gamma \tag{3.9}$$

where
$(X,Y,Z,t;\gamma)$ = the space and time coordinants, and
$F_{ijk}$ = the integral of Green's Function in three dimensions.

In a one dimensional solution of a release from a point source the integral of Green's Function is expressed as:

$$F_i = \frac{1}{4\pi K_{xx}(t-\tau)} \exp\left[ - \frac{(x-x_s) - U(t-\tau)^2}{4K_{xx}(t-\tau)} - \lambda (t-\tau) \right] \tag{3.10}$$

where
$x$ = distance from source down the hydraulic gradient,
$K_{xx}$ = x component of the retarded dispesion tensor,
$X_s$ = x coordinate of point source,
$\gamma$ = total time of release,
and the other parameters as previously defined.

The release of nuclear mass from the core melt ($\dot{M}$) is described for sump water and the core mass in Section 2.2.

The model AT123D determines concentrations at a point. These values were converted to a radionuclide flux (M/T) in order to effectively judge the potential environmental impacts of a core melt release.

## 3.5 CONCLUSIONS OF GENERIC HYDROGEOLOGIC SITE CLASSIFICATION

1. A hydrogeologic classification scheme for nuclear power plants must consider not only basic hydrogeologic transport factors, but also the geologic properties that affect the radionuclide source term and the feasibility of mitigative techniques.

2. Parameters contained in the hydrogeologic data base have values that tend to produce relatively rapid ground-water velocities. These values are not unrealistic, but rather represent conservatism in parameter selection and/or properties characteristic of nuclear power plant sites.

3. Simulation of ground-water transport by a one dimensional-homogeneous model may not produce extremely accurate results. However, in consideration of the large differences in contaminant release and transport rates among the generic classifications this methodology is

adequate to describe the relative characteristics of severe nuclear power plant accidents in various geologic environments.

## 3.6 GENERIC GEOTECHNICAL SITE DATA[a]

The following sections list the individual reactor site hydrogeologic properties by generic classification. These data were used in the preceding equations to establish the probable range in the hydrogeologic properties for each generic classification. The results of the generic hydrogeologic analyses are presented with the applicable mitigative techniques in Section 5.

### 3.6.1 Generic Hydrogeologic Site Classification: Fractured Consolidated Silicates (Crystalline)

Total Number of Sites: 16

| GENERIC SURFACE WATER CLASSIF. | HYDRAULIC CONDUCTIVITY | EFFECTIVE POROSITY | HYDRAULIC GRADIENT | DISTANCE TO SURFACE WATER |
|---|---|---|---|---|
| - | (CM/S) | - | - | (METERS) |
| EST-06 | 0.10E-02 | 0.01 | 0.120 | 76. |
| OCN-06 | 0.10E-02 | 0.01 | 0.025 | 122. |
| OCN-11 | 0.47E-02 | 0.01 | 0.060 | 3125. |
| RES-03 | 0.25E-03 | 0.01 | 0.075 | 762. |
| RES-06 | 0.17E-02 | 0.05 | 0.025 | 198. |
| RES-07 | 0.87E-03 | 0.01 | 0.144 | 457. |
| RES-10 | 0.10E-02 | 0.01 | 0.220 | 320. |
| RES-12 | 0.19E-03 | 0.01 | 0.042 | 366. |
| RES-13 | 0.10E-02 | 0.01 | 0.060 | 183. |
| RES-14 | 0.35E-04 | 0.01 | 0.150 | 76. |
| RES-18 | 0.10E-01 | 0.01 | 0.033 | 91. |
| RIV-14 | 0.10E-02 | 0.30 | 0.040 | 152. |
| RIV-24 | 0.10E-02 | 0.01 | 0.043 | 976. |
| RIV-27 | 0.58E-03 | 0.01 | 0.015 | 1128. |
| RIV-34 | 0.50E-04 | 0.01 | 0.070 | 1829. |
| RIV-37 | 0.10E-04 | 0.01 | 0.229 | 107. |

(a) These data are taken from: Niemczyk, S. J., unpublished, "A Summary of Subsurface Hydrogeological Information for Light Water Nuclear Reactor Sites," Oak Ridge National Laboratory, Oak Ridge, Tennessee. A discussion of this data base is provided in Section 1.4.

### 3.6.2 Generic Hydrogeologic Classification: Fractured-Solutioned Consolidated Carbonates

Total Number of Sites: 12

| GENERIC SURFACE WATER CLASSIF. | HYDRAULIC CONDUCTIVITY | EFFECTIVE POROSITY | HYDRAULIC GRADIENT | DISTANCE TO SURFACE WATER |
|---|---|---|---|---|
| - | (CM/S) | - | - | (METERS) |
| GRL-02 | 0.28E-02 | 0.10 | 0.006 | 61. |
| GRL-06 | 0.10E-02 | 0.01 | 0.007 | 229. |
| GRL-13 | 0.10E+00 | 0.10 | 0.006 | 122. |
| EST-05 | 0.10E-02 | 0.10 | 0.092 | 133. |
| OCN-01 | 0.63E-01 | 0.10 | 0.001 | 610. |
| OCN-14 | 0.10E+00 | 0.10 | 0.000 | 183. |
| RES-04 | 0.10E-03 | 0.10 | 0.020 | 701. |
| RES-05 | 0.10E+00 | 0.10 | 0.085 | 122. |
| RES-16 | 0.10E+00 | 0.10 | 0.023 | 229. |
| RIV-05 | 0.10E+00 | 0.10 | 0.009 | 2927. |
| RIV-09 | 0.10E+00 | 0.10 | 0.008 | 549. |
| RIV-29 | 0.10E+00 | 0.10 | 0.051 | 198. |

### 3.6.3 Generic Hydrogeologic Classification: Porous Consolidated Carbonates

Total Number of Sites: 10

| GENERIC SURFACE WATER CLASSIF. | HYDRAULIC CONDUCTIVITY | EFFECTIVE POROSITY | HYDRAULIC GRADIENT | DISTANCE TO SURFACE WATER |
|---|---|---|---|---|
| - | (CM/S) | - | - | (METERS) |
| GRL-04 | 0.10E-02 | 0.10 | 0.001 | 732. |
| GRL-05 | 0.38E-02 | 0.10 | 0.003 | 61. |
| OCN-07 | 0.10E-01 | 0.10 | 0.002 | 793. |
| EST-01 | 0.10E-02 | 0.10 | 0.009 | 2744. |
| RES-09 | 0.97E-04 | 0.00 | 0.004 | 183. |
| RES-11 | 0.10E-03 | 0.01 | 0.030 | 1067. |
| RES-20 | 0.23E-05 | 0.01 | 0.016 | 244. |
| RIV-08 | 0.10E+00 | 0.10 | 0.004 | 457. |
| RIV-17 | 0.10E-03 | 0.10 | 0.006 | 2287. |
| RIV-40 | 0.10E-04 | 0.02 | 0.019 | 1998. |

### 3.6.4 Generic Hydrogeologic Site: Porous Consolidated Silicates

Total Number of Sites: 13

| GENERIC SURFACE WATER CLASSIF. | HYDRAULIC CONDUCTIVITY | EFFECTIVE POROSITY | HYDRAULIC GRADIENT | DISTANCE TO SURFACE WATER |
|---|---|---|---|---|
| - | (CM/S) | - | - | (METERS) |
| GRL-07 | 0.10E-04 | 0.01 | 0.010 | 152. |
| GRL-10 | 0.10E-01 | 0.05 | 0.020 | 122. |
| OCN-02 | 0.10E-02 | 0.01 | 0.083 | 183. |
| RES-17 | 0.10E-03 | 0.01 | 0.031 | 488. |
| RIV-04 | 0.10E-04 | 0.01 | 0.003 | 6100. |
| RIV-10 | 0.10E-04 | 0.01 | 0.010 | 1280. |
| RIV-12 | 0.10E-04 | 0.01 | 0.001 | 610. |
| RIV-20 | 0.21E-03 | 0.01 | 0.050 | 305. |
| RIV-22 | 0.10E-01 | 0.05 | 0.007 | 1494. |
| RIV-23 | 0.10E-02 | 0.05 | 0.013 | 122. |
| RIV-32 | 0.15E-03 | 0.01 | 0.010 | 549. |
| RIV-36 | 0.10E-04 | 0.01 | 0.006 | 183. |
| RIV-38 | 0.72E-03 | 0.05 | 0.014 | 1829. |

## 3.6.5 Generic Hydrogeologic Classification: Porous Unconsolidated Silicates

Total Number of Sites: 41

| GENERIC SURFACE WATER CLASSIF. | HYDRAULIC CONDUCTIVITY | EFFECTIVE POROSITY | HYDRAULIC GRADIENT | DISTANCE TO SURFACE WATER |
|---|---|---|---|---|
| - | (CM/S) | - | - | (METERS) |
| GRL-01 | 0.21E-01 | 0.05 | 0.020 | 244. |
| GRL-03 | 0.10E-04 | 0.05 | 0.018 | 152. |
| GRL-08 | 0.10E-02 | 0.25 | 0.040 | 6098. |
| GRL-09 | 0.10E-02 | 0.25 | 0.050 | 183. |
| GRL-11 | 0.17E-02 | 0.25 | 0.003 | 92. |
| GRL-14 | 0.30E-03 | 0.01 | 0.050 | 183. |
| GRL-15 | 0.10E-02 | 0.25 | 0.003 | 183. |
| EST-02 | 0.19E-01 | 0.25 | 0.010 | 168. |
| EST-03 | 0.14E-01 | 0.05 | 0.012 | 640. |
| EST-04 | 0.94E-02 | 0.05 | 0.021 | 274. |
| EST-07 | 0.94E-02 | 0.05 | 0.021 | 152. |
| EST-08 | 0.35E-02 | 0.05 | 0.005 | 2043. |
| EST-09 | 0.10E-04 | 0.01 | 0.003 | 488. |
| OCN-03 | 0.94E-01 | 0.25 | 0.001 | 366. |
| OCN-05 | 0.95E-01 | 0.25 | 0.001 | 457. |
| OCN-08 | 0.94E-01 | 0.25 | 0.001 | 2287. |
| OCN-09 | 0.33E-01 | 0.25 | 0.024 | 61. |
| OCN-10 | 0.13E-01 | 0.05 | 0.010 | 76. |
| OCN-12 | 0.70E-01 | 0.25 | 0.004 | 488. |
| OCN-13 | 0.30E+00 | 0.25 | 0.000 | 823. |
| RES-01 | 0.10E-02 | 0.25 | 0.001 | 122. |
| RES-08 | 0.10E-05 | 0.20 | 0.098 | 122. |
| RES-15 | 0.10E-01 | 0.05 | 0.021 | 91. |
| RIV-01 | 0.23E+00 | 0.25 | 0.002 | 213. |
| RIV-03 | 0.97E-03 | 0.05 | 0.004 | 793. |
| RIV-06 | 0.10E-04 | 0.05 | 0.050 | 3963. |
| RIV-07 | 0.20E+00 | 0.25 | 0.028 | 152. |
| RIV-11 | 0.10E-01 | 0.25 | 0.017 | 76. |
| RIV-13 | 0.10E+00 | 0.25 | 0.003 | 1341. |
| RIV-15 | 0.12E-03 | 0.25 | 0.002 | 5183. |
| RIV-16 | 0.10E-02 | 0.05 | 0.004 | 335. |
| RIV-18 | 0.10E+00 | 0.25 | 0.054 | 107. |
| RIV-19 | 0.10E-04 | 0.01 | 0.011 | 6098. |
| RIV-21 | 0.10E-05 | 0.02 | 0.026 | 274. |
| RIV-25 | 0.50E-04 | 0.02 | 0.029 | 13. |
| RIV-28 | 0.19E+00 | 0.25 | 0.010 | 183. |
| RIV-31 | 0.60E+00 | 0.25 | 0.003 | 2896. |
| RIV-33 | 0.30E-02 | 0.25 | 0.002 | 4268. |
| RIV-41 | 0.10E-04 | 0.01 | 0.012 | 1768. |
| RIV-42 | 0.12E-01 | 0.25 | 0.002 | 3659. |
| RIV-45 | 0.35E-01 | 0.25 | 0.030 | 335. |

### 3.6.6 Generic Hydrogeologic Classification: Fractured Consolidated Silicate (Shale)

Total Number of Sites: 5

| GENERIC SURFACE WATER CLASSIF. | HYDRAULIC CONDUCTIVITY | EFFECTIVE POROSITY | HYDRAULIC GRADIENT | DISTANCE TO SURFACE WATER |
|---|---|---|---|---|
| - | (CM/S) | - | - | (METERS) |
| GRL-12 | 0.10E-05 | 0.01 | 0.005 | 396. |
| RES-02 | 0.10E-02 | 0.01 | 0.005 | 251. |
| RES-19 | 0.10E-02 | 0.01 | 0.014 | 915. |
| RIV-02 | 0.10E-04 | 0.01 | 0.015 | 1768. |
| RIV-35 | 0.10E-01 | 0.01 | 0.039 | 1250. |

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# 4.0 GROUND-WATER CONTAMINANT MITIGATION TECHNIQUES

## 4.1 TYPES OF MITIGATION TECHNIQUES

There are two general classes of ground-water contaminant mitigation alternatives that may be appropriate, in individual cases, for application to ground-water contamination resulting from a severe commercial nuclear power plant accident. These two classes are: 1) static or passive techniques, and 2) dynamic or active strategies. The individual techniques or schemes that comprise each class are designed to interact directly with ground-water flow and consequently contaminant transport to achieve an acceptable level of contaminant mitigation. Indirect ground-water contaminant mitigation schemes that involve redesign of reactor containment structures or manipulation of reactor core material (e.g., in situ vitrification) are not considered.

### 4.1.1 Static Ground-Water Contaminant Mitigation Techniques

Static or passive mitigation techniques are typically engineered/constructed barriers to ground-water flow and consequently contaminant transport. The primary objective of a constructed barrier is to redirect the ground-water flow away from potentially accessible surface environments (e.g., surface water bodies, production well fields, etc.). Achievement of this objective usually results in ground-water being forced to follow more circuitous routes with longer travel times. The longer travel times provide longer time for the natural decay of radionuclides. Also, there may be additional benefits to longer travel times through increased opportunity for contaminant retardation.

Constructed barriers are considered static ground-water contaminant mitigation techniques because once in-place they are not readily adaptable to changing conditions of ground-water contamination. Barriers are also considered passive rather than active because they do not directly influence the ground-water contaminant concentrations as, for instance, pumping of ground water for surface treatment would directly infleunce contaminant concentrations. Engineered/constructed barriers do not normally require a significant amount of maintenance. When properly designed and constructed these types of barriers (except steel sheet piping) are considered, from a practical viewpoint, to be permanent. However, constructed barriers do not last indefinitely. Barriers tend to be more costly than other mitigation alternatives and the time for construction can be significant.

Three basic types of constructed barriers were analyzed for their feasibility and suitability as mitigation measures for ground-water contamination resulting from a severe power plant accident. The barriers considered are grout curtain cut-off walls, slurry trench cut-off walls, and steel sheet piling. Table 4.1.1-1 provides a further breakdown of the static ground-water contaminant mitigation techniques. Only steel sheet pilings are considered because other materials (such as wood or reinforced concrete) are not capable of forming a watertight seal to an effective depth. Reinforced concrete pilings are not normally placed to depths that would allow complete vertical

TABLE 4.1.1-1. Static Ground-Water Contaminant Mitigation Techniques Considered for Application to Severe Power Plant Accidents

1. Grout curtain cut-off walls
    - 1a. Particulate grouts: Cement-based grouts
    - 1b. Non-particulate grouts: Chemical-based grouts
2. Slurry trench cut-off walls
    - 2a. Soil-bentonite slurry trenches
    - 2b. Cement-bentonite slurry trenches
    - 2c. Lean concrete slurry trenches
    - 2d. Vibrating beam slurry trenches
3. Steel sheet piling

cutoff of ground-water flow. Each of the ground-water flow barriers listed in Table 4.1.1-1 are analyzed in detail as to their engineering feasibility for specific applications.

Extensive monitoring is required during and following construction in order to verify performance characteristics of constructed barriers.

### 4.1.2 Dynamic Ground-Water Contaminant Mitigation Techniques

Dynamic or active ground-water contaminant mitigation techniques are primarily conceptual strategies for actively (i.e., directly) influencing the state of ground-water contamination. Active influence is accomplished by either changing the ground-water flow regime by pumping and/or injection, directly treating the contaminated ground-water or combinations of both approaches. Active ground-water contaminant mitigation schemes are generally better able to respond to changes in the state (i.e., contaminant plume velocity, concentration, etc.) of ground-water contamination than constructed barriers. However, typically associated with dynamic schemes are relatively high maintenance costs. Also extensive monitoring feedback is usually recommended to insure adequate performance.

The dynamic ground-water contaminant mitigation schemes may be applicable as temporary mitigation measures while permanent measures are being designed and constructed. Also, several of the dynamic schemes may be most effective in combination with permanent barriers. The design of a dynamic mitigation scheme may necessarily require surface handling of contaminated ground water. This circumstance may cause significant safety problems related to handling, transporting, treating, and disposing of contaminated ground water.

The dynamic ground-water contaminant mitigation schemes analyzed for their feasibility and applicability to mitigate the effects of ground-water contamination following a severe power plant accident are presented in Table 4.1.2-1.

TABLE 4.1.2-1. Dynamic Ground-Water Contaminant Mitigation Techniques Considered for Application to Severe Power Plant Accidents

1. Ground-water withdrawal for potentiometric surface adjustment
   1a. Prevent discharge to receiving stream
   1b. Prevent saturated contact with core melt mass
   1c. Prevent contamination of leaky aquifers
2. Ground-water withdrawal and/or injection for contaminant plume control
   2a. Withdrawal and injection
   2b. Withdrawal without injection
   2c. Withdrawal with surface treatment and recharge
   2d. Injection only
3. Subsurface drains
4. Selective filtration via permeable treatment beds
5. Ground freezing
6. Air injection

## 4.2 FEASIBILITY CRITERIA FOR GROUND-WATER CONTAMINANT MITIGATION TECHNIQUES

There are several important considerations for determining the suitability of mitigative techniques for ground-water contamination resulting from a severe power plant accident. These considerations encompass: 1) design, 2) construction, 3) performance, and 4) implementation issues related to each mitigation measure. These issues are addressed in specific detail for each of the mitigation alternatives. A brief overview of each issue follows.

### 4.2.1 Design Considerations

Design considerations include the variations in specific types of techniques (e.g., particulate versus non-particulate grout), appropriate host geologic media, size, location, and orientation of the various mitigation measures and design limitations. Passive ground-water barriers (i.e., slurry trenches, grout curtains, and steel sheet piling cut-offs) have better defined engineering design considerations than typically do dynamic ground-water contaminant mitigation strategies which are more conceptual in design (i.e., less rigorously defined from an engineering standpoint).

### 4.2.2 Construction Considerations

Construction considerations are a major concern in determining the feasibility of specific mitigation strategies. Construction considerations include appropriate methods of installation, limitations of construction methods, equipment required for construction, etc. Several of the mitigation strategies (i.e., slurry trenches, subsurface drains, and permeable treatment beds) require extensive excavation. Trenching is realistically feasible only in unconsolidated media and soft, easily ripped semi-consolidated media. The strategies requiring extensive trenching are not practically feasible in a consolidated medium such as a crystalline silicate medium, for example.

Several of the dynamic mitigation strategies require well construction. The type of well system developed (i.e., well point versus deep well) can have a significant impact on the overall performance of a particular mitigation alternative.

Grouting and ground freezing operations require special expertise and equipment which may not be readily available. Permeable treatment beds require a permeable material with high ion exchange capacity. Naturally occurring glauconite greensands have been recommended but suitable deposits of glauconite may exist only in the Mid-Atlantic region of the U.S. This may preclude application (because of transportation costs) in other regions of the U.S.

Similar to design considerations, construction considerations are a function of the mitigation technique itself, the physical properties of the site, and the accident scenario.

### 4.2.3 Performance Considerations

Performance considerations include permeability reduction (if appropriate to the technique), durability, continuity, and contaminant compatibility. In a practical sense, the performance is related to how well the strategy can achieve and maintain an acceptable level of ground-water quality at predetermined locations. Embodied in this philosophy is the protection of accessible environments such as surface water bodies or producing ground-water well fields.

All of the performance considerations (e.g., permeability reduction) vary with time. For instance, steel sheet piling may be expected to corrode significantly in approximately 40 years thus reducing its effective performance. Durability is closely related to permeability reduction and maintenance requirements. How long a barrier will perform as designed is a function of quality control during construction and ground-water chemistry. For example, cement-based constructed barriers will lose their integrity more rapidly in a saltwater environment or if exposed to freeze/thaw cycles. Also, sulphate attack on concrete can lead to a loss of integrity. Most, if not all, of the dynamic mitigation strategies are temporary and their design with respect to the overall mitigation plan should reflect this condition.

### 4.2.4 Implementation Considerations

In determining the engineering feasibility of ground-water contaminant mitgation schemes implementation considerations play a key role. The implementation considerations include:

1. Installation/construction time,
2. Cost,
3. Equipment mobilization,
4. Toxicity (some chemical grouts are highly toxic), and
5. Safety of workers.

Difficulty arises in analyzing these issues in a generic sense however, because of their site-specific and site dependent nature. Unit values for the estimated time required to perform a specific task (e.g., well drilling) and associated cost are provided as available. The EPA has estimated the total cost of hypothetically designed mitigation schemes involving several of the techniques analyzed herein. EPA's estimates are included for comparison of alternative methods. In other instances, only very general time for installation, relative to other techniques, is provided.

Equipment mobilization requirements are not only dependent on the mitigation strategy employed but also on the site configuration and geographical location of the site. Standard excavating and/or drilling equipment is used to construct most ground-water contaminant mitigation measures but specialty equipment is required for grouting and ground freezing. Unobstructed right-of-way must be provided for drilling and excavating equipment and subsurface obstacles such as utility services must be avoided by mitigation techniques involving trenching or extensive drilling.

Worker safety during the installation and maintenance of a ground-water contaminant mitigation scheme is of primary concern. However, worker exposure to radiation resulting from atmospheric releases or diffusion of vapor through the unsaturated soil column would be extremely site sensitive and accident specific. Meteorological conditions at the time of the atmospheric release of radiation would greatly influence transport and deposition rates of atmospheric contaminants in the vicinity of the plant. Due to the (in general) several orders of magnitude higher transport properties of airborne contaminants versus ground-water transport of contaminants a time delay from the occurrence of an accident and the implementation of a ground-water contaminant mitigation scheme may enhance worker safety without sacrificing mitigation performance. In most cases however, the closer to the contaminant source the mitigation scheme is implemented, in general, the more cost effective the scheme will be. A site-specific thorough investigation of ground-water flow and contaminant transport should be conducted in relation to the accident scenario to determine the time delay that can be tolerated in implementing a ground-water contaminant mitigation strategy if one is necessary.

Another safety issue involves the safe handling, treatment, and disposal of contaminated ground water. Several of the mitigation schemes require above ground handling of contaminated ground water thus requiring special care to insure the safety of workers and integrity of the surface environment. A related concern is the secondary contamination of drilling and pumping equipment in prolonged contact with contaminated ground water.

In summary, the implementation considerations for ground-water contaminant mitigation schemes are extremely important in the overall assessment of the applicability of each measure. However, these issues are also highly sensitive to specific and individual site characteristics ranging from the physical plant configuration, to local meteorological conditions at the time of the accident, to the time history of events of the accident itself. Therefore it is difficult, if not impossible, to address these issues in a generic manner. In the analysis of individual mitigation techniques each implementation issue (if relevant) is addressed to a level of detail consistent with the generic nature of this study.

## 4.3 ANALYSIS OF STATIC GROUND-WATER CONTAMINANT MITIGATION TECHNIQUES

The static mitigation techniques analyzed for their applicability to mitigate the effects of a severe power plant accident on ground-water quality are constructed barriers to ground-water flow. The primary differenes among these interdiction techniques are their method of construction and composition. They all divert ground-water flow in a similar passive manner.

The three barriers to ground-water flow analyzed are grout curtain cut-offs, slurry trenches, and sheet piling cut-offs. The fundamental purpose of each technique is to redirect ground-water flow and consequently contaminant transport (if placed down-gradient from the contaminant source) away from accessible surface environments of concern. From an engineering standpoint these barriers, except sheet piling cut-offs, are considered permanent even though over time their performance (i.e., imperviousness) will deteriorate. Their ability to redirect ground-water flow becomes increasingly impaired as their permeability increases.

Of the static ground-water contaminant mitigation techniques considered, grouting is the most generally applicable across all generic hydrogeologic site classifications. However, grouting operations can be expensive and time consuming plus special expertise and equipment is necessary for construction of grout curtain cut-offs. Slurry trench construction requires excavation which limits application to unconsolidated media or soft, semi-consolidated media and also limits the depth to which a slurry trench can be installed. Sheet piling application is also limited to unconsolidated host materials.

Constructed barriers to ground-water flow placed transverse to the direction of flow have the propensity to "backing-up" water in unconfined aquifers. A low permeability barrier may cause a "bathtub effect" immediately up-gradient from the barrier under certain flow and ground-water recharge and conditions. Depending on the specific circumstances giving rise to the "bathtub effect" pumping may be required to control the ground-water mounding. If the barrier is placed up-gradient from the contaminant source uncontaminated water may be forced to the surface but cause little concern. However, if a "bathtub effect" causes contaminated water to rise to the surface, then a ground-water dewatering scheme (Section 4.4.1) should be implemented. In confined flow situations, a constructed barrier may cause an increase in the hydraulic head immediately up-gradient from the barrier. The increased head may cause increased vertical leakage downward with the potential from contaminating lower aquifers. In this instance a reduction in head through ground-water withdrawal maybe advisable. In any event, the creation of a "bathtub effect" resulting from an engineered barrier and the potential consequences would be an integral part of the site-specific analysis of the performance of the mitigative strategy. Control or reduction of ground-water mounding would depend on site-specific factors including level of contamination, degree of mounding, ground-water discharge opportunities, and the plant configuration. As a final note, even if the contaminant source was totally contained, the potential for ground-water mounding would exist from local precipitation percolating to the water table.

### 4.3.1 Grouts

Grouting is the process of filling soil voids and/or rock cavities and fissures with some type of stabilizing material which acts as a sealing agent and thereby reduces soil/rock permeability. The stabilizing material, or grout, is injected into the geologic medium in the liquid phase by various mechanisms and, upon curing, results in the increased strength of the host material. Permeability reduction is achieved by consolidation and densification of the grouted material.

There are many grouting mechanisms and a wide variety of grouts exhibiting varied in-place mechanical properties. The choice of grout penetration mechanisms depends largely on the properties of the host material and the purpose of the treatment. Within the context of this study, the grout treatment purpose is to develop ground-water cut-offs to control the lateral movement of radionuclides through the geologic medium under consideration. Structural stability and strength of the resulting soil-grout complex are of lesser importance. The choice of grout material is dependent on its rheological behavior, particularly viscosity, rigidity, and granular state (Harris et al. 1982a).

#### 4.3.1.1. Grout Penetration Mechanisms

There are five basic grouting mechanisms (Attewell and Farmer 1976):

1. Permeation Grouting - Even injection into soil or rock pore spaces resulting in a series of cylinders around the grout sources. Successive grout applications (i.e., primary, secondary, tertiary, etc.) result in the formation of a reduced permeability ground-water cut-off.
   Applicability - pervious sands or gravels; porous rock

2. Fissue Grouting - Well-dispersed, water-cement grout injection into fissures. High pressures are often used to slightly enlarge (i.e., widen) the fissures to facilitate passage of the grout. Subsequent deposition of cement particles is relied upon to "silt up" the fissures.
   Applicability - fissured rock and layered soils having a low intrinsic permeability.

3. Fracture Grouting - Hydrofracture in vicinity of grout injection source to increase penetration rates.
   Applicability - low or variable permeability rocks and soils

4. Compaction Grouting - High pressure injection of grout with high solids content with high penetration characteristics. Compaction grouting is also referred to as consolidation grouting.
   Applicability - Loose, unconsolidated soils or sands.

5. Bulk Grouting - Stabilization of large cavities such as caves or abandoned mine works.
   Applicability - large subterranean cavities and voids.

Grouts are designed to penetrate through the void space of the host material. Penetration grouting requires grouts composed of portland cement or fine-grained clays such as bentonite or non-particulate, synthetic chemicals. Cement-based grouts are used primarily for grouting very coarse sands and gravels. Non-particulate grouts may be used to penetrate fine-grained host material (Dunn et al. 1980).

#### 4.3.1.2 Types of Grouts

Grouts are normally dividied into two groups: 1) particulate grouts, and 2) non-particulate grouts. Particulate grouts are subdivided into cement grouts and clay grouts. Non-particulate grouts (often referred to as chemical grouts) are subdivided into silicate grouts and organic polymers (Attewell and Farmer 1976).

The most important properties of grouts related to their flow and consequently permeation of the host material are (Attewell and Farmer 1976):

1. Size of particles,
2. Viscosity, and
3. Shear strength.

Once the grout has been successfully injected its set properties become important considerations. Grout set properties relate to permeability reduction, strength and durability. There is often a relationship between strength and viscosity of grouts within particular subcategories. A grout with a high strength is usually denser and more viscous as a fluid than is a lower strength grout (Attewell and Farmer 1976). Table 4.3.1-1 presents a summary of the major types of grouts currently in use.

TABLE 4.3.1-1. Types of Grouts (Source: Attewell and Farmer 1976)

| Group | Type | Composition |
|---|---|---|
| Cement Grouts | Cement Suspension | water, cement (ratio > 1) |
| | Cement slurry | water, cement (ratio < 1) |
| | Sand-cement | water, cement, sand |
| | Flyash-cement | water, cement, flyash |
| | Clay-cement | water, cement, bentonite |
| | Alum-cement | water, cement, aluminium sulphate |