# The safety of repositories for highly radioactive wastes

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Many authors have studied the safety of disposal of highly radioactive wastes in excavated cavities beneath the earth. Work has been concentrated in three areas: prediction of future events and processes which could affect waste containment, mathematical modeling of failure scenarios, and estimation of uncertainties in model predictions. The results of past safety assessments are reviewed and compared in this paper. Anything but a very small release of radioactivity from a repository would appear to be quite unlikely; a quantitative evaluation of the probabilities of small releases has not proved possible.

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## I. INTRODUCTION

### A. The problem

The problem of disposal of high-level radioactive wastes has attracted a great deal of attention in recent years. Two general treatments of the topic have previously appeared in *Reviews of Modern Physics* (Cohen, 1977; Hebel *et al.*, 1978).

The radioactivity of high-level wastes is principally due to the fission products and transuranic elements formed in nuclear reactors. The composition of these wastes has been described by Cohen (1977) for the case of light-water reactors and by Pigford and Choi (1976) for other reactors. The wastes emerge as a liquid from the reprocessing plant. If spent reactor fuel is not reprocessed, most of the wastes will remain within the uranium dioxide fuel pellets. Some volatile elements migrate from the pellets but are contained by the fuel cladding.

In the Soviet Union, it is authoritatively reported, liquid high-level wastes are disposed of by injection directly into deep geological formations (Spitsyn and Balukova, 1979). Other countries retain these wastes in storage tanks with the intention of converting them into solids for ultimate disposal. The solidification schemes most commonly advocated incorporate the wastes into blocks of borosilicate glass.

A number of means of disposal of solidified high-level wastes have been proposed (GEIS, 1980). The most developed of these is emplacement in mined cavities, called repositories, deep in the earth; this approach is the subject of this review. Disposal in sediments on the sea floor is also being seriously considered *(Oceanus,* 1977), and some research is being devoted to disposal in outer space (BCL, 1980).

Cavities can be excavated at depths of hundreds of meters and radioactive wastes can be emplaced in them using existing technology or straightforward extensions of it (Koplik *et al.*, 1979a). A more difficult question than whether it can be done is whether it is safe to do so. Although routine radiological exposures and accidents of both radiological and nonradiological nature may occur during construction and operation, these hazards are similar to those encountered elsewhere in the mining and nuclear industries (GEIS, 1980); the principal safety issue is the possibility that radioactivity would leak from a repository and cause damage at some time in the future.

  

The literature addressing this question, the long-term safety of high-level waste repositories, will be reviewed in this paper.

The review includes material published through late 1980. Occasional reference is made to more recent publications.

### B. General approach

The process of evaluating the safety of a repository may be divided conceptually into five steps:

- A comprehensive list of processes and events that could contribute to release of radioactivity from a repository is assembled.
- A list of scenarios is selected for further analysis; in each scenario, the events and processes which control repository performance are specified and described mathematically.
- The likelihood of occurrence of each scenario is described.
- The consequences to human health if the scenario should occur are calculated.
- The results are evaluated to determine whether the repository is safe.

The first of these steps, compilation of a comprehensive list of processes and events, is limited principally by the imagination of the compiler. By now, however, a number of extensive lists have been published which the worker in the field can use. While it is not certain (and can never be certain) that these lists are complete, the published compilations represent an accumulation of many individuals' contributions, and there is no need to rely on the imagination of any single analyst. Section IV describes the various processes and events whose relevance has been suggested.

Typically, it is not certain which events and processes will affect repository behavior. For example, one might not know whether thermal expansion of the rocks around the repository will fracture overlying strata. One would then have two possibilities: fractured rocks and unfractured rocks. In a more complicated sequence of events, movement along a fault might occur before or after fracturing occurs. Other possibilities exist, of course, in vast numbers. Each combination of circumstances which might determine repository behavior is referred to here as a *scenario.* A scenario may be described by equations and by parameters which may or may not be known accurately. Circumstances requiring differing mathematical descriptions are distinct scenarios.

By solving the equations for a scenario, one calculates the amount of radioactivity released, the dose to humans, the number of cancer deaths, or some other measure of repository performance. The results of such a calculation are known as the *consequences* of the given scenario.

To be sure, the definition of scenarios for any particular repository is somewhat arbitrary. For example, one could define either a series of scenarios involving faults of different sizes or, equivalently, a single scenario with a fault whose size is given by an unknown parameter. The second alternative reduces the number of scenarios but introduces a compensating increase in the uncertainty of the consequences.

Despite the presence of this arbitrary element, the approach of defining scenarios and calculating their consequences has proven to be of practical value in repository safety analysis, and we shall follow it in this paper. Section V discusses methods of defining scenarios, of describing their probabilities of occurrence, and of selecting scenarios for further analysis when there are too many to calculate the consequences of each. Methods of predicting consequences are reviewed in Sec. VI. Uncertainties, whether due to the existence of a multiplicity of possible scenarios or to uncertainties in their parameters, are discussed in Sec. VII.

(It should be noted that much of the past literature uses a definition of "scenario" slightly different from that employed here. By analogy to the study of reactor accidents, a "release scenario" is often defined as a set of circumstances in which radioactivity is "released" from the repository. A series of equations is solved to determine whether and when release occurs; the results of these calculations define the release scenarios. Only computations dealing with events after release has occurred are considered to be consequence calculations. As Burkholder (1980a) has pointed out, this conceptualization is inappropriate to repository safety analysis. As compared to reactors, the boundaries of containment in a repository are less well defined, and a greater reliance is placed on delay and dilution rather than absolute containment. It is therefore difficult to specify the moment at which "release" occurs. The definitions adopted in this paper avoid the concept of release by making all calculations of waste movement part of the consequence calculations; the scenario merely gives the equations which are to be solved.)

Cohen (1977, 1980a) has developed an approach to safety assessment which does not define scenarios. His technique is based on observing releases of natural substances which are analogous to buried wastes. Cohen's work is discussed in Sec. VI.F.

The final step in a safety analysis is evaluation of the calculated dangers to determine whether the repository is safe. As absolute safety is impossible, the question to be addressed is whether risks are small enough to be acceptable. While science provides much of the basis for this evaluation, law, ethics, and politics are also involved. For this reason, the acceptability of repositories will not be discussed in this paper. But even when, as here, the literature of disposal safety is being evaluated from a scientific standpoint, it is well to bear in mind that lines of argument may have been chosen on moral, political, or legal considerations rather than purely for scientific merit.

### C. Some conceptual difficulties

In the typical case in which the safety of a technology is analyzed, the device under study and the surrounding environment are reasonably well defined. For example, aircraft are constructed according to prescribed designs, and the weather conditions they can encounter, although quite variable, can be characterized statistically. Even when important facts are unknown (for example, atmospheric chemistry and physics relevant to the safety of releases of fluorocarbons or carbon dioxide), the unknowns are often directly resolvable by additional scientific inquiry.

In evaluating the future behavior of a nuclear waste repository and the effects of any releases of radioactivity from it, however, several factors intervene which are not completely amenable to scientific prediction. Among these factors are the existence, nature, and behavior of future human populations. The importance of nonscientific considerations in determining whether and to what degree a repository could do damage leads to conceptual difficulties in safety analysis.

The most severe difficulties are connected with the possibility that future human actions might cause or exacerbate the release of radioactivity. It is necessary to distinguish a variety of different circumstances in which individuals or groups could act. Actions could be undertaken by persons

- Completely without knowledge of the existence of the repository,
- Aware of the existence of a danger of some sort in the area, but ignorant of its nature,
- Disregarding a known hazard, access to which might or might not be restricted by the authorities,
- With a definite purpose (such as retrieval of the wastes to recover their resource value) after considered assessment of the risks and benefits involved.

Physical science can attempt to estimate the consequences of scenarios occurring in these circumstances. But it can neither assess the probability of their occurrence nor make the ethical judgments necessary to evaluate the acceptability of varying degrees of exposure under such conditions. Such judgments will be affected by both the hypothetical nature of the scenarios and the degree to which those exposed understand what risks they are taking.

Further questions arise in estimating the consequences to humans of any future releases. These consequences will depend on the size of the population affected, its economic infrastructure (especially water-supply systems), its diet, and its living habits. None of these is predictable by the physical sciences. The possibility raised by Cohen (1977) that future generations will be able to cure cancer (or, it might be added, other radiation-caused diseases) muddies the waters even more.

Other conceptual problems arise in obtaining the data needed for calculations. Much of this data must be obtained from geology. As Bredehoft *et al.* (1978) comment, "Geology is basically a retrodictive rather than a predictive science." The methods of the earth sciences require considerable adjustment to be usable to make predictions of the relatively distant future. Similar difficulties arise when evidence from archaeology is used to predict the future behavior of man-made objects (Kaplan, 1979). For many aspects of a repository, the only direct evidence of behavior will come from tests conducted before the repository is closed. Extrapolation of such tests to the future inevitably requires an element of judgment.

One of the recurrent themes of this paper is the role of expert opinion as a source of numbers for use in calculations, in the choice of scenarios, and for other purposes. Experts will, of course, employ accepted theories and observed data in producing their opinions, but the use of expert opinion does imply that there is an element of subjective extrapolation. Studies which clearly describe the role played in them by expert opinion will be subject to criticism by persons with other opinions. But little would be known in science if judgment and discretion could not contribute to the rationale for an accepted truth.

As later chapters of this paper will show, attempts to quantify subjective opinion in repository safety studies are often counterproductive. For example, if instead of subjectively classifying scenarios as "credible" and "incredible," one attempts to quantify their probabilities, the numbers that result are usually no more than opinions and tend to give a misleading impression of precision. This may obscure the role of judgment rather than eliminate it. Studies of repository safety are best evaluated on the overall credibility of their reasoning and evidence, including that used to support the expert opinions they incorporate.

### D. Purposes of safety assessment

Safety assessments may be used for a wide variety of purposes, which can be administrative, technical, or scientific in nature. Two elements of the purpose of a safety assessment which have a particularly strong influence on the scientific content are

- Whether the subject of a study is a particular repository at a particular site, or waste disposal in general,
- Whether the aim is to determine whether disposal is safe, or to predict the harm (or probability of any given level of harm) to be caused.

To date, only a few studies (Claiborne and Gera, 1974; Giuffre and Kaplan, 1979; Bradley and Corey, 1976; Logan and Berbano, 1978; D'Alessandro and Bonne, 1980; KBS, 1978a, 1978b; WIPP, 1979; Raymond *et al.*, 1980) have addressed the safety of disposal at particular sites. More commonly, one studies a hypothetical "generic" site. Studies of generic sites are of considerable value in identifying the role of different elements of a repository system in isolating waste. To the extent that the generic site is typical of real sites, this approach will also provide information on the safety of waste disposal in general. However, the earth tends to be quite variable, and it is difficult to draw conclusions about the safe disposal of nuclear waste at specific sites from studies of generic sites.

The distinction between estimating the actual danger from a repository and determining whether a repository is "safe enough" is of particular importance. In working toward the latter objective, many of the unknowns about repository behavior can be avoided by the use of "conservative" assumptions (that is, assuming the worst of a set of reasonable alternatives to be the correct one). Difficult problems of data collection and prediction can often be avoided in this manner. If a worst-case calculation produces results that fall within the threshold of acceptability (however defined), then safety has been shown. A worst-case analysis which gives *unacceptable* results proves nothing by itself.

Estimation of the likelihood that a repository will in fact cause damage, and of the severity of that damage, is a far more difficult task than worst-cases analysis. It is noteworthy how often studies whose aim is to produce realistic estimates must fall back on conservative assumptions (e.g., Burkholder *et al.*, 1975; Berman *et al.*, 1978). Indeed, the more reliable a disposal system is, the less predictable its failures are, because many of its well-understood failure modes have been suppressed.

### E. Other reviews of disposal safety

Several major reviews of radioactive waste disposal technology and disposal feasibility have been carried out. These include discussions of the safety of repositories containing radioactive wastes. A comprehensive description of the technology is included in the U. S. Department of Energy's generic environmental impact statement on disposal of commerical high-level wastes (GEIS, 1980). Earlier the Jet Propulsion Laboratory prepared a status report on reprocessing and high-level waste disposal (English *et al.*, 1977). In the following year, a study committee of the American Physical Society published a report on nuclear fuel cycles and waste management (Hebel *et al.*, 1978). The Union of Concerned Scientists has also prepared a general report on nuclear waste (Lipschutz, 1980).

A broad review of issues in nuclear waste management was prepared for former President Carter by the Interagency Review Group (IRG, 1979). The report addressed the general issues of planning and decision-making, technical strategies for high-level (and other) waste management, and institutional and management considerations. The report includes a section which briefly summarizes and compares previous studies of disposal safety.

Several recent reports deal specifically with the question of the long-term safety of waste repositories. Two reports in which some of the authors of this paper survey major safety assessment studies (Koplik *et al.*, 1979b; Ensminger *et al.*, 1980) have largely been incorporated into this review. A recent review of the literature on the risks of nuclear power by the National Academy of Sciences also addresses the question of waste disposal safety (NAS, 1981).

The U. S. Nuclear Regulatory Commission is currently drawing up a formal statement of opinion on the storage and disposal of nuclear wastes. An administrative proceeding known as the waste confidence rulemaking is being held for this purpose. The issues under consideration are whether there is reasonable assurance that high-level wastes can be safely disposed of, when such disposal will be available, and whether wastes can be safely stored prior to disposal. The Department of Energy has submitted a document to the proceeding which provides extensive information on present and past research on disposal safety (DOE, 1980). Further documentation has been submitted by interest groups, states, and individuals.

## II. COMPARATIVE ANALYSES

There have been two major approaches to judging the safety of nuclear waste disposal:

- Comparison of the hazards of nuclear wastes with other, more familiar, toxic substances. These comparisons are usually based on some simple measure of the waste's toxicity. Comparisons have been used for determining the time over which isolation of waste from the biosphere is required.
- Analysis of the future behavior of a waste repository. This involves the use of mathematical models of physical processes and events.

The latter method is the most appropriate for quantifying dangers and for designing disposal systems. It explicitly takes into account the various barriers to the release of waste materials (such as canister, waste form, and surrounding rock) and addresses the complex interactions of the wastes with their surroundings. The methods and results of such analyses are discussed in detail in later sections of this paper.

The first approach, which is the subject of this section, has been employed by many researchers. Because a comparison of relative hazards requires subjective judgment, conclusions drawn from this work are particularly controversial.

### A. Hazard indices

A key consideration in any comparison is the measure used to represent the toxicity of the waste. These measures are referred to as hazard indices. Hazard indices typically depend on one or more of the following factors:

- Amount and type of radioactivity,
- Persistence or availability in the biosphere,

- Dose to man from ingestion, inhalation, or external exposure,
- Health effects from a given dose.

A comprehensive review of the many different hazard indices that have been used is given by Voss (1979). [Other hazard index reviews have been carried out by Poston (1978), General Research Corporation (1980), and Smith *et al.* (1980).] Voss finds that, depending on the assumptions used in developing the hazard index, vastly different results can be obtained. This suggests that the use of hazard indices must be tempered with an appreciation of their limitations and inherent assumptions. The limitations of hazard indices are discussed further by Hebel *et al.* (1978).

The simplest measure of waste hazard is the radioactivity of the waste. Although comparisons involving radioactivity appear occasionally in the literature, it is generally recognized to be a poor measure of hazard. Both the type of radiation (alpha, beta, neutron, or photon) and the energy spectrum are important in determining the effect on tissue. The established measure for the biological harm done by exposure to radioactivity is the dose equivalent (in rem), referred to simply as dose in the remainder of this paper. It forms the basis of all measures of the radiological toxicity of nuclear wastes.

If radioactivity is ingested or inhaled, it can remain in the body for some time. In such cases, it is often useful to compute a "dose commitment." This represents the dose accumulated over some period of time (usually taken to be 50 years) because of inhalation or ingestion at a single time. Dose commitments are calculated by using models for uptake and retention of radionuclides in the human body.

For the purposes of radiation protection, values of the maximum permissible concentration (MPC) of radionuclides in air or water have been tabulated. These concentrations are set so as to limit the dose commitment incurred by drinking water or breathing air. The MPC's provide a convenient basis for assessing the hazard of radioactive materials that might be released to the environment, and most hazard indices employ them. The MPC is also referred to interchangeably in the literature as the radioactivity concentration guide (RCG).

The approach taken by most authors is to determine the amount of water or air needed to dilute wastes to MPC levels (McGrath, 1974; Gera, 1975; Bell and Dillon, 1971; Pigford and Choi, 1976; and others). Others have developed more complex hazard indices from this by incorporating additional factors deemed relevant to disposal, such as persistence in the environment, leachability of the waste, or probability of release (Smith *et al.*, 1980; Williams *et al.*, 1980; Smith and Kastenberg, 1976; Gera and Jacobs, 1972; Bruns, 1976; Voss and Post, 1976). As the hazard index becomes increasingly complex and more and more processes are incorporated, the distinction between a safety analysis and a hazard assessment blurs. The usefulness of a hazards index is its simplicity and its value for making crude but reasonable comparisons.

### B. Comparisons with other hazardous materials

High-level reprocessing wastes (HLW) and spent reactor fuel have been compared (on a basis of lethality if ingested) with other nonradioactive wastes that are produced in the United States. The results are shown in Table I (Cohen, 1977). The table shows that initially the nuclear wastes are extraordinarily toxic, but after 500 yr the toxicity has fallen off by a factor of 400 for spent fuel and $5\times10^3$ for HLW. This decline in toxicity is a consequence of decay of the two principal radionuclides ($^{90}$Sr and $^{137}$Cs) contributing to the radioactivity of the waste. At 500 yr such compounds as cyanide, arsenic, and mercury are far more toxic than commercial nuclear wastes if ingested.

In addition to the ingestion hazard, nuclear wastes also present dangers from either external exposure to radiation or inhalation of airborne particulates. Plutonium, for example, is about a thousand times more toxic if inhaled than if ingested. The inhalation and external exposure pathways are less important for deeply buried wastes because the principal means for release of such wastes is by transport in groundwater. For this reason, most hazard comparisons have focused on ingestion toxicity.

The comparisons shown in Table I leave out many important factors relevant to assessing actual risks, including availability in the environment, dispersibility, concentration, and chemical form. Since only acute effects are considered, an extrapolation is involved in extending the results to environmental situations in which subchronic or chronic effects are of more importance.

A comparison that is in many ways directly relevant to the disposal of wastes by burial is that with mineral ores. The idea here is that buried wastes should have no greater potential to affect man than natural ores in the ground. Since people accept buried ores they should accept buried waste. The analogy is useful only in a rough qualitative sense, however, since nuclear wastes are highly concentrated and differ both chemically and physically from the ores.

The toxicity of uranium ores has been compared with that of high-level waste and spent fuel by Hamstra (1975), Haug (1976), Rochlin (1977), and DOE (1980). Cohen and Tonnessen (1977) also present comparisons with ores of various stable toxic elements. The toxicity of the radioactive elements is assessed using drinking water MPC's. The toxicity of nonradioactive elements is assessed by using the allowable concentrations in public drinking water as set by the U. S. Environmental Protection Agency. The results (adapted from Haug, 1976; and Cohen and Tonnessen, 1977) are shown in Fig. 1. The toxicity is normalized to that of 0.2% uranium ore. After 500 yr of decay, the buried waste is significantly less toxic than several of the nonradioactive ores (selenium, chromium, and mercury).

TABLE I. Lethal quantities[a] of hazardous materials if ingested.

| Material | Compounds | Average lethal dose (mg/kg) | Experimental animal | Extrapolated lethal dose to man (g) |
|---|---|---|---|---|
| Selenium | $Na_2SeO_3$ | 5 | rabbit, mouse rat, guinea pig | 0.35 |
| (Cyanide) | KCN | 10 | rat | 0.7 |
| Mercury | $HgCl_2$ | 23 | rat, mouse | 1.6 |
| Arsenic | $As_2O_3$ | 45 | mouse, rat | 3 |
| Barium | $BaCl_2$, $Ba(NO_3)_2$ | 250 | rat | 18 |
| Copper | CuO, $CuCl_2$ | 300 | rat | 21 |
| Nickel | $Ni(NO_3)_2$ | 1620 | rat | 110 |
| Aluminum | $AlCl_3$ $Al_2(SO_4)_3$ | 4000 | rat, mouse | 280 |
| High-level reprocessing wastes | | | | |
| after 10 yr | | | | 0.03 |
| after 500 yr | | | | 170 |
| Spent Fuel[b] | | | | |
| after 10 yr | | | | 0.15 |
| after 500 yr | | | | 57 |

[a]The quantity of material such that half the affected people die; the extrapolation to man from the test animal data is scaled by weight to a 70 kg man. [b]Spent fuel was not included in the original table by Cohen (1977). The values shown here were computed using the same procedures as were used by Cohen for high-level waste.

Cohen and Tonnessen calculate the toxicity index for the buried waste by assuming the waste is uniformly distributed over the areal extent of the underground repository. This reduces the concentration of the buried waste by roughly a factor of $3\times10^3$. If the hazard index is based on the actual concentration of the waste, then the toxicity does not fall below that of pitchblende even by $10^6$ yr (Pigford and Choi, 1976). The fact that the wastes are initially highly concentrated and located in discrete packages might be important if anyone ever intruded into the repository.

Figure 1 compares the hazards of various ores on the basis of volume. An alternative approach is to compare the waste to the original uranium ore which was mined to produce it. It may reasonably be argued that disposal need not attempt to provide any greater safety than had the wastes not been produced.

Using MPC's to characterize the ingestion hazard, Pigford and Choi (1976) found that spent fuel and high-level reprocessing wastes (HLW) are initially several orders of magnitude more hazardous than the ore. However, by 500 yr the hazard from HLW crosses that of the ore, and it continues to diminish past that point. After $10^3$ yr the hazard from spent fuel exceeds that from the ore by less than an order of magnitude.

A problem with the work by Pigford and Choi (and indeed any hazard index using MPC's) is that the cancer and genetic risks from radiation are not explicitly taken into account. Cohen (1977), using MPC's as a starting point, developed an alternative index of hazard based on the number of cancer doses in the waste. A much improved approach is now available as a result of recent recommendations by the International Commission on Radiological Protection (ICRP, 1977). Instead of using MPC's, the hazard of a radionuclide may be evaluated using "effective dose factors" (Adams *et al.*, 1978; ICRP, 1979; ICRP, 1981). The effective dose factor is directly proportional to the risk of cancer or genetic defects that would result from consumption of a given radionuclide.

Figure 2 compares the hazard of nuclear waste to that of the uranium ore needed to produce it using effective dose factors rather than MPC's.[1] Spent fuel is always

[1]This figure can be compared to previously published figures showing volumes of water required to dilute waste to MPC by recalling how the MPC's are derived. Roughly speaking, MPC is calculated by postulating that an individual who drinks 800 l of water in a year should not receive a 50-yr dose commitment exceeding 5 rem to the whole body, 15 rem to soft tissue, or 30 rem to bone or thyroid. The current MPC's are derived from older dose factors (see Sec. VI.E). Volumes of water (in $m^3$) required to dilute waste so that one who drinks 800 l in a year would receive an effective 50-yr dose commitment of 5 rem (using the newer dose factors) are shown on the right-hand scale. The isotopic contents of the wastes are taken from DOE (1979), Appendix 10.D, cases 1 and 3B. The spent fuel is from uranium fuel; the HLW is a composite of the wastes arising in a fuel cycle in which uranium and plutonium are recycled.

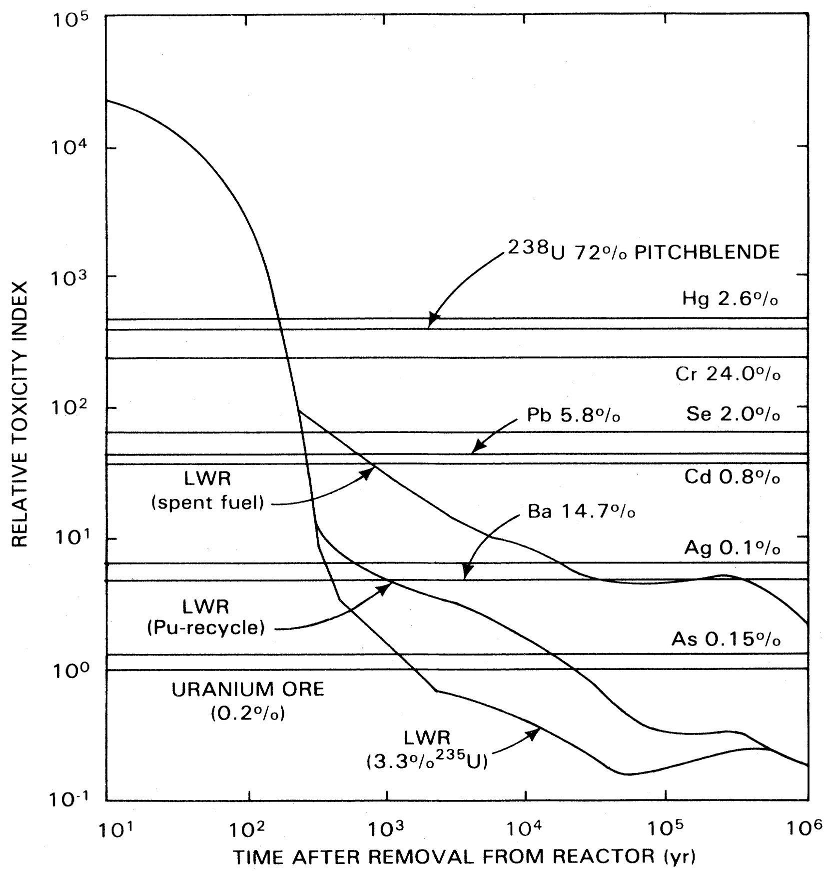


FIG. 1. Toxicity of nuclear wastes relative to ores of toxic elements (adapted from Haug, 1976, and Cohen and Tonnessen, 1977).

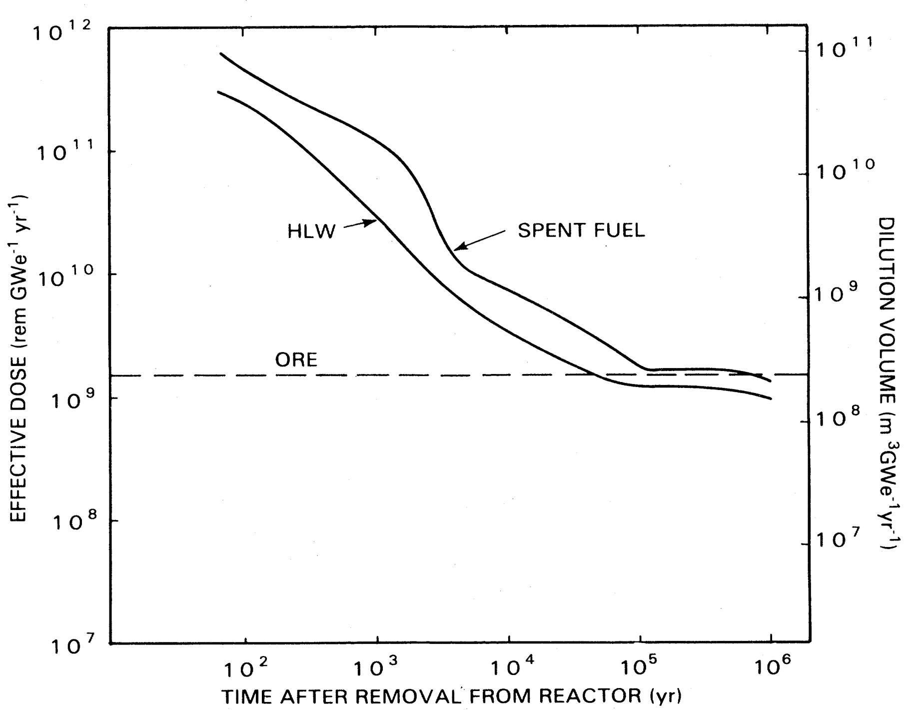


FIG. 2. Effective dose from ingestion of spent fuel, HLW, and uranium ore produced or consumed in generating electricity.

within a factor of ten of the hazard of reprocessing wastes. Both fall to the level of the uranium ore by roughly $10^5$ yr. These results differ from those of Pigford and Choi because the effective dose factor treats uptake of radionuclides in the human body and the resulting risks more realistically than does the MPC. The most hazardous radionuclides are $^{241}Am$ at $10^3$ yr, $^{243}Am$, $^{240}Pu$, and $^{239}Pu$ at $10^4$ yr, and $^{237}Np$ at $10^5$ yr.

The comparisons that have been discussed thus far fail to address possible differences in the availability of various radionuclides once released to the biosphere. Processes that can be considered include dispersion or buildup of radionuclides and pathways to man, such as food, water, and external exposure. One approach to including these processes in a hazards analysis is shown in Fig. 3. The results are presented in the form of retention quotients—the factors of containment required to assure that a selected dose is not exceeded (Williams, *et al.*, 1980). The system retention quotient is calculated by first computing the dose to man from release of a given quantity of waste ($10^4$ GWeyr in Fig. 3) into a surface water body used for irrigation of crops and drinking water.[2] This dose is then divided by a selected dose limit (1 mrem/yr in Fig. 3). The dose calculations employ the effective dose factors previously discussed. The method described here is qualitatively similar to a more general approach first introduced by Kaye *et al.* (1971).

According to Figure 3, HLW and spent fuel have comparable gross toxicities. The wastes fall to the hazard level of the uranium ore originally mined to produce the waste in roughly $10^4$ yr.

Although these various comparisons of buried waste with ores are attractive, they are not entirely satisfactory. Unlike uranium ores, the wastes will be buried at great depths (on the order of $10^3$ m) in specially chosen locations with engineered barriers to prevent or reduce release to the environment. Most uranium ore in the United States occurs in permeable strata with flowing groundwater. Some ores are present at or near the surface, and radium, a uranium daughter, naturally pervades fresh waters and topsoil.

The above observations might suggest that ore body comparisons overstate the hazard of buried radioactive wastes. However, such a conclusion overlooks the fact that wastes are buried in man-made excavations with pathways (shafts, tunnels, and boreholes) connecting the repository to the surface. Furthermore, the repository host rock can be adversely affected by both subsidence after burial and heating by the buried wastes. One of the principal tasks of repository design is to assure that these effects are mitigated and that all pathways are effectively sealed. The perspective provided by hazard indices is most useful if this can be demonstrated.

### C. Implications

#### 1. Time period of concern

It is tempting to try to use the foregoing comparisons to identify the time over which isolation of wastes from the biosphere is required. This is extremely important, as the time period of hazard greatly affects the way in which these wastes are to be disposed of. An early attempt to identify such a time period appears in a preliminary assessment of HLW disposal by the National Academy of Sciences (NAS, 1957).

"Unlike the disposal of any other type of waste, the hazard related to radioactive waste is so great that no element of doubt should be allowed to exist regarding safety .... Safe disposal means that the waste shall not come in contact with any living thing. Considering half-lives of the isotopes in waste this means for 600 years...."

Hazard assessments carried out since 1957 indicate containment times anywhere from 500 yr to more than $10^6$ yr, depending on the method of comparison. There is therefore no clearcut, widely accepted, unambiguous way to define the time required for isolation of nuclear wastes. Clearly a judgment is required that goes beyond what can be determined by scientific analysis alone.

Such a judgment is in the process of being made now by the U. S. Environmental Protection Agency. In drafts of their proposed regulations for disposal of HLW, a time period of $10^4$ yr has been suggested. The time period was arrived at through a comparison of HLW and a uranium ore body.

#### 2. Comparison with coal

Cohen (1979) combines the results from several different hazard analyses in order to compare radioactive wastes from nuclear power to the wastes from burning coal. Three measures of the potential danger are compared: the consequences of ingestion or inhalation of all the wastes, the effects of disposal by inexpensive, straightforward means, and the results of using the best disposal technology. According to Cohen, the hazards from nuclear wastes are hundreds or thousands of times less than from coal-burning wastes.

A comparison between coal and nuclear power is also presented in a recent study by the National Academy of Sciences (NAS, 1979). According to this study, radioactive waste (properly disposed of) contributes only a small fraction to the total risk from the nuclear fuel cycle, and the health effects of coal production appear to be much greater than those from nuclear power. However, the study also emphasizes the extremely large uncertainty

[2]The amount of electric energy whose production gives rise to a given quantity of waste is often used as a measure of this quantity of waste. The energy is conventionally measured in terms of electric (rather than thermal) power production. The symbol GWe rather than GW is often used to emphasize this point.

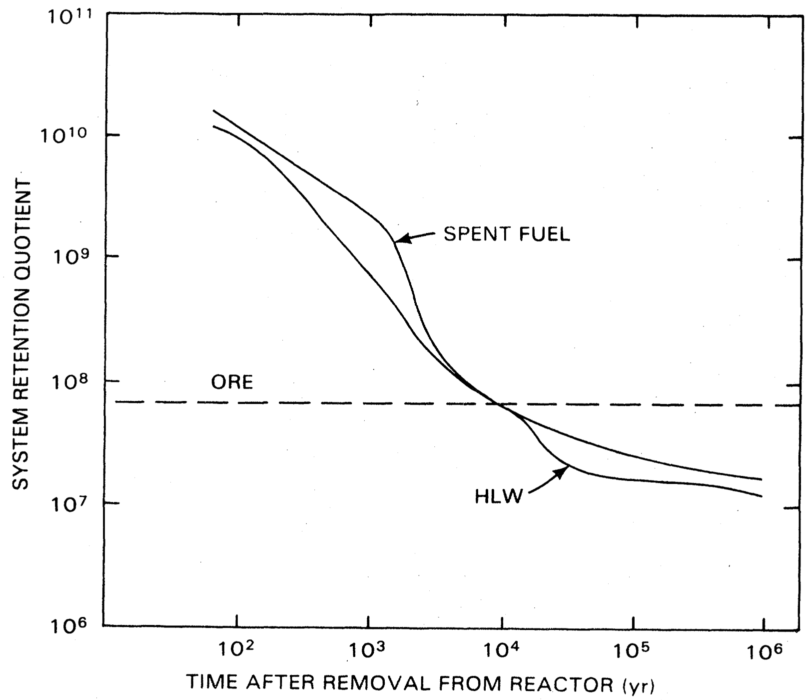


FIG. 3. Retention quotient for spent fuel, HLW, and uranium ore from $10^4$ GW yr of nuclear electricity.

concerning the potential health effects from coal plant emissions. Further emphasized is "the inherent difficulty of comparing one energy system with another, since some of their important risks may be different and thus not strictly comparable."

The principal limitation of the various comparisons that have been made is that many important features of disposal are omitted from the analysis. The most recent hazard analyses attempt to include some of the more tractable features such as movement of radionuclides in the biosphere. Nevertheless, only a detailed safety analysis of geologic waste disposal can comprehensively address the various features of a disposal system. The remainder of this article addresses such work.

## III. MINED GEOLOGIC DISPOSAL

The principal method proposed for disposal of highly radioactive nuclear wastes is burial deep beneath the surface of the earth. A volume of rock in a suitable geologic formation would be mined out and the wastes would be placed within the excavated cavity. Then the mine, the vertical shafts leading to the mine, and any exploratory boreholes drilled into the rock would be sealed. Geologic media that are considered suitable for waste burial include bedded salt, domed salt, tuff, basalt flows, and granites. Figure 4 shows what a nuclear waste repository might look like.

The long-term safety of this disposal concept depends on ensuring that wastes are contained within the earth for as long a time as possible and that any releases have sufficiently small consequences. Three ways in which release can occur have been identified:

- Exhumation of the deeply buried wastes by some natural event,
- Exhumation of the deeply buried wastes as a result of future actions by man,
- Slow transport of waste materials via groundwater from the repository to surface waters or to potable groundwater supplies.

Both the choice of a repository site and the design of the facility are directed toward avoiding these possibilities and mitigating their effects. The first possibility may be largely avoided by locating the repository at sufficient depth within a tectonically stable region. The second dictates locating the repository in an area which is relatively unlikely to be exploited for underground resources. The last possibility is in many ways the most important and has been the focus of most safety research.

Safety analysis requires long-term projections of the behavior of natural and engineered elements of the repository system. Because of uncertainties in estimates of far future behavior, a *multiple-barrier* approach has evolved for providing greater confidence that wastes will be adequately contained. In repositories designed using this concept, there exists a multiplicity of substantial barriers that individually inhibit the entry of waste into man's environment.

Three types of barriers that are expected to be important for mined geologic disposal will be described here briefly; a more extensive discussion can be found in DOE (1980). These barriers are the waste package, the reposi-

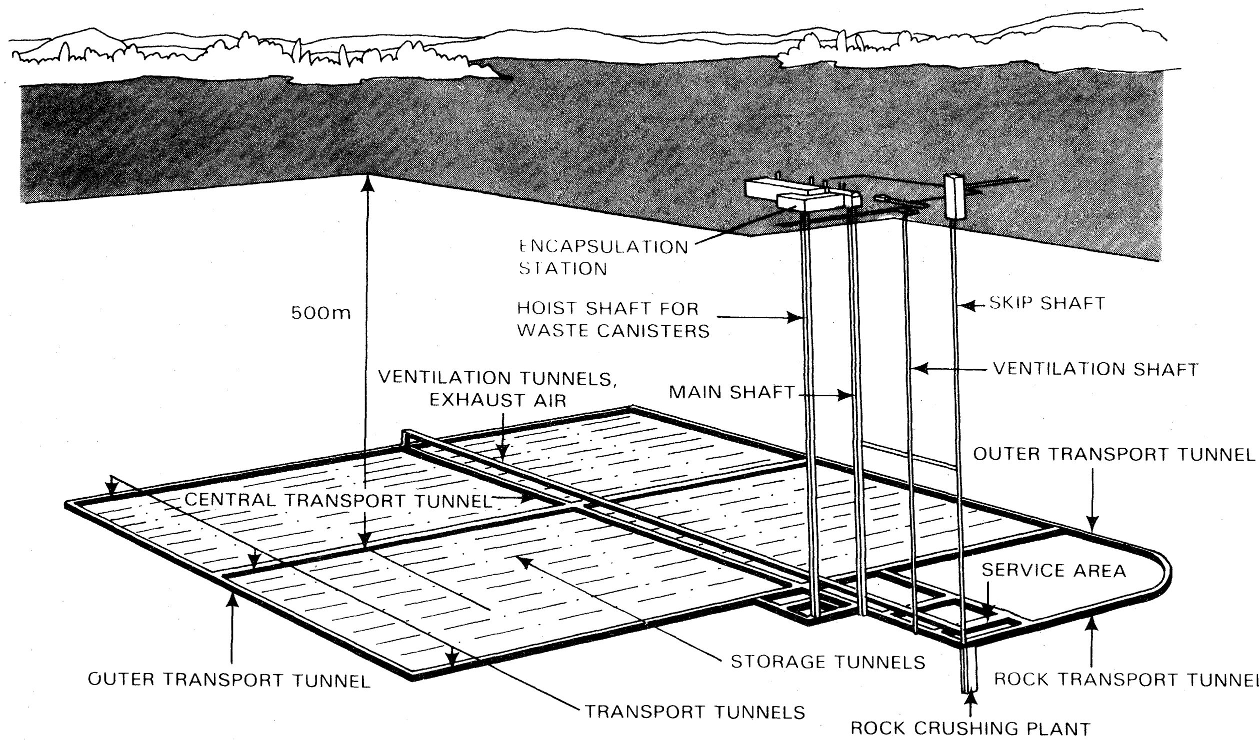


FIG. 4. Artist's conception of a nuclear waste repository (adapted from KBS, 1978b).

tory, and the surrounding geologic formations. Transport, accumulation, and dispersal of radioactivity could occur also in the surface environment; many safety studies model these phenomena as well.

The *waste package* consists of the waste form (liquid wastes are to be converted to a very durable solid material), a canister, and any surrounding protective material. The purpose of these materials is to prevent exposure of the waste form to groundwater for as long a time as possible and to limit the rate of egress of the radionuclides from the waste package if exposure occurs.

A waste canister can provide a significant barrier between the radioactive waste form on the inside and the geologic environment on the outside. The requirements for an effective canister are stringent. It must, while serving as a safe and reliable vehicle for containing the wastes prior to burial, be designed to withstand a severe environment in the repository. Depending on the burial medium, the canister may be subject to high temperatures, high stresses, and a chemical environment conducive to corrosion. Canister designs have ranged from simple enclosures of stainless steel (whose purpose is merely to facilitate waste handling) to complex, highly durable containment systems. Figure 5 shows the waste package proposed by the Swedish Nuclear Fuel Safety Project (KBS) for the disposal of spent fuel (KBS, 1978b).

The waste form consists of the radioactive waste and any associated encapsulating or stabilizing materials. A great variety of waste forms has been proposed for the high-level wastes resulting from the reprocessing of spent reactor fuel. Principal among these is the waste form created by incorporation of the waste into a solid block of borosilicate glass (Grover, 1980). The glass provides a stable and highly insoluble medium for containment of the wastes. An important alternative to the use of amorphous glasses is to use crystalline ceramic materials. Certain crystalline minerals (such as zircon, pyroxenes,

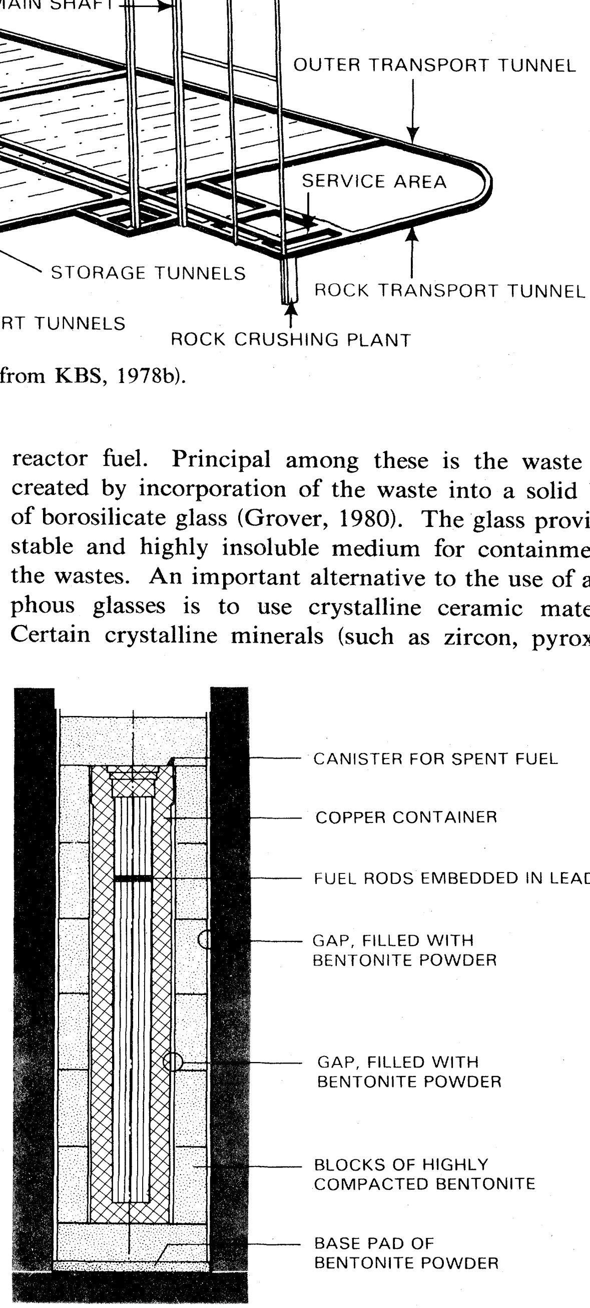


FIG. 5. Waste package and storage hole (KBS, 1978b).

and feldspars) are known to survive virtually forever in nature.

A quite different approach is to encapsulate the waste within another material rather than to form mixtures. Encapsulating materials that have been proposed include both concretes and natural minerals. Historically, however, the major focus of waste form research has been on glass, and almost all safety analyses of wastes from reprocessing have assumed its use.

It is possible to incorporate an additional barrier into the waste package system by surrounding the canisters with a highly sorptive, low-permeability material. This material is intended to slow the migration of radionuclides and to limit contact of the canister and waste with water. It may also be used to control groundwater chemistry so as to inhibit dissolution of canister and waste materials. For example, chemicals which control oxidation potential or pH can be added. KBS has proposed surrounding the canisters with compressed dried bentonite, a highly sorptive clay which swells on wetting (see Fig. 5).

The *repository* may be considered a barrier to the extent its design features are intended to reduce transport of wastes through man-made structures. Low-permeability backfill in these structures can restrict groundwater flow, and backfills can be chosen which expand when they absorb water so that open pathways for water flow will be sealed. Ion exchange material can be used in the backfill to retard the motion of nuclides, and chemical buffers can be added to control groundwater chemistry. Plugs can be used to prevent groundwater flows through backfilled tunnels and storage rooms. Multicomponent shaft and borehole seals can be used to reduce groundwater flow into or out of the repository. Finally, linings similar to those used in tunnels and other underground civil structures can be used to divert groundwater around excavated areas. An illustration of repository sealing applications is given in Fig. 6 (D'Appolonia, 1980). The most critical seals (in terms of long-term repository safety) are those preventing flow through the shafts between the storage chambers and the surface.

The *geologic barrier* consists of the rock surrounding the repository that retards or restricts groundwater flow. In some geologic formations, such as bedded salt, no groundwater flow is expected. Even when small volumes of circulating water do exist, there may be substantial delays before such waters can flow from the buried waste to the surface. In addition, geochemical properties of the geologic media can substantially inhibit waste migration.

The geologic barrier can remain effective over geologic time—tens of thousands of years or more. To be effective, the geologic barier must have rock properties that are compatible with repository integrity—i.e., can withstand the effects of repository construction and the heat output of the waste. The geologic barrier must also remain intact against both natural and man-caused events. This is more likely if the repository is located in a tectonically stable region with a low potential for resource exploration.

Bedded salt has been the focus of attention in the U. S. as a disposal medium. Bedded salt has desirable properties such as uniformly low permeability, high thermal conductivity, abundant availability in thick masses, and plasticity that enables fractures to heal at proposed repository depths. These features are also shared by salt domes. Salt has three disadvantages: it is soluble in groundwater, it is a resource itself and may contain valuable resources, and it provides a hostile environment for

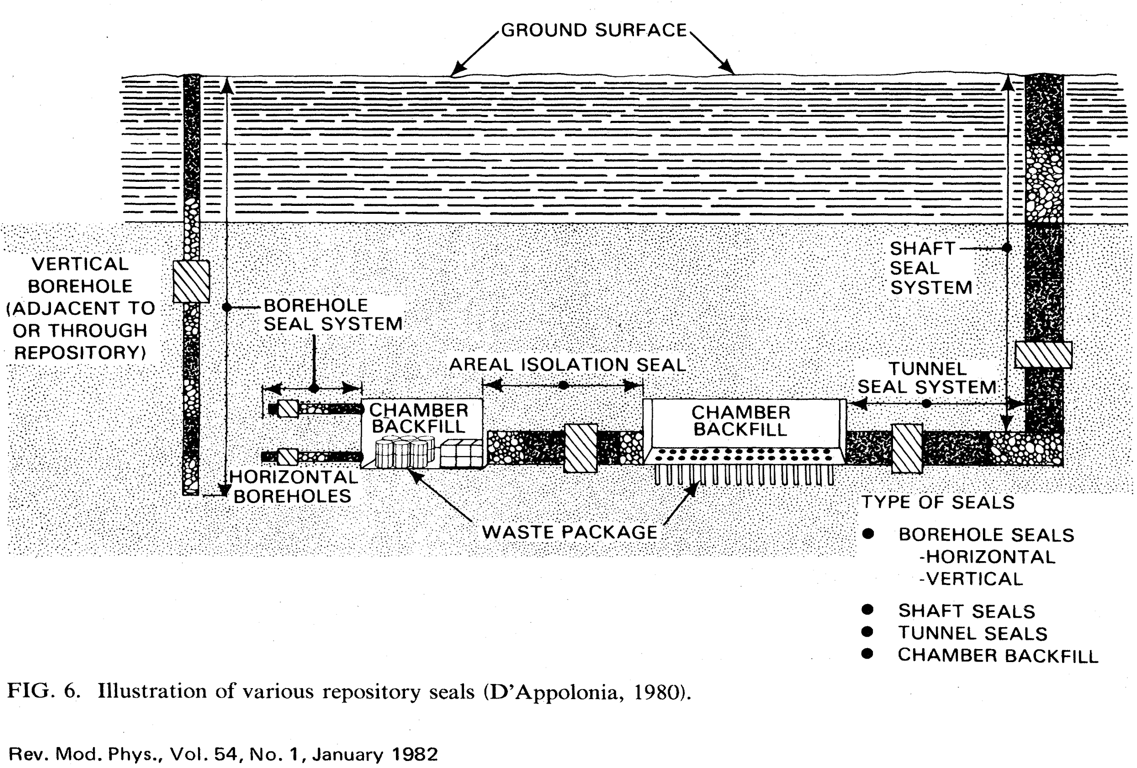


FIG. 6. Illustration of various repository seals (D'Appolonia, 1980).

the waste package.

Other media under investigation include basalt flows (Deju, 1979), granites (KBS, 1978a), clay formations (D'Alessandro and Bonne, 1980) and tuff (Tyler, 1979). An additional possibility is deep burial in desert regions above the water table (Winograd, 1974). By burial in this zone, the repository is isolated from groundwater circulation.

A substantial portion of the research on geologic disposal safety has focused on the performance of individual barriers. A barrier-by-barrier assessment has been suggested by the staff of the U. S. Nuclear Regulatory Commission for evaluating the safety of mined geologic disposal (NRC, 1980).

An alternative but somewhat complementary approach to safety assessment is the systems approach. The systems approach does not emphasize individual components of the disposal system but rather how these components, acting in concert, determine the overall behavior. This requires identification of the salient components of the waste disposal system and delineation of their interactions, relationships, and dynamic behavior mechanisms. In this view, the first step in an analysis of disposal safety is the identification of processes and events that may lead to release of waste. This is the subject of the next section.

## IV. PHYSICAL PROCESSES AND EVENTS

Many different physical processes can affect the future behavior of a nuclear waste repository. Consideration of these processes is necessary in order to determine what could lead to releases of waste materials to the biosphere. The processes that may alter the initial condition of a repository fall into three categories: natural phenomena that occur independently of the presence of a waste repository, phenomena caused by human actions, and phenomena induced by the presence of the repository itself. Tables II—IV list the major processes in each category which may affect repository behavior (adapted from Burkholder, 1980b).

Assessment of *natural phenomena* is largely dependent on work in the earth sciences. The work involves both a description of the process and an evaluation of its likelihood of occurring in the future. Such work is at the very forefront of current research in the geologic sciences. Although prediction of many events that occur on geologic time scales is beyond current capabilities, trends and ranges in behavior can often be determined.

Natural phenomena that have received the most attention are glaciation, dissolutioning in salt deposits, and tectonic activity. The record of past glaciation suggests that renewed glaciation is likely to occur in the next 10 000 years. Research on this topic has been carried out in Sweden, where glaciation is almost certain to occur at any site (Pusch, 1978), and in the U. S. for assessment of disposal in basalt (Stottlemyre *et al*, 1980; Bull, 1980). The U. S. Department of Energy is considering a reposi-

TABLE II. Natural phenomena.

| |
|---|
| Climatic fluctuations |
| Glaciation |
| Denudation and stream erosion |
| Magmatic activity |
| Extrusive |
| Intrusive |
| Epeirogenic displacement |
| Igneous emplacement |
| Isostacy |
| Orogenic diastrophism |
| Near-field faulting |
| Far-field faulting |
| Diapirism |
| Diagenesis |
| Static fracturing |
| Surficial fissuring |
| Impact fracturing |
| Hydraulic fracturing |
| Dissolutioning |
| Sedimentation |
| Flooding |
| Undetected features |
| Faults, shear zones |
| Breccia pipes |
| Lava tubes |
| Gas or brine pockets |
| Meteorites |

TABLE III. Waste- and repository-induced phenomena.

| |
|---|
| Thermal effects |
| Differential elastic response |
| Nonelastic response |
| Fluid pressure changes |
| Local fluid migration |
| Canister migration |
| Convection |
| Chemical effects |
| Geochemical alterations |
| Corrosion |
| Waste package—geology interactions |
| Gas generation |
| Seal—rock interactions |
| Mechanical effects |
| Change in local state of stress |
| Readjustment of rock along joints |
| Local fracturing |
| Canister movement |
| Subsidence |
| Radiation effects |
| Material property changes |
| Radiolysis |
| Criticality |
| Decay product gas generation |
| Stored energy |
| Modification of hydrologic regime |

TABLE IV. Human-induced phenomena.

| |
|---|
| Improper design or operation |
| Shaft seal failure |
| Improper waste emplacement |
| Undetected past intrusion |
| Undiscovered boreholes |
| Mine shafts |
| Inadvertent future intrusion |
| Archaeological exhumation |
| Weapons testing |
| Non-nuclear waste disposal |
| Resource mining (salt, mineral, hydrocarbon, geothermal) |
| Storage of hydrocarbons, compressed air, or hot water |
| Intentional intrusion |
| War |
| Sabotage |
| Waste recovery |
| Perturbation of groundwater system |
| Irrigation |
| Reservoirs |
| Intentional artificial groundwater recharge or withdrawal |
| Chemical liquid waste disposal |
| Biosphere alteration |
| Establishment of population center |
| Climate modification |

tory located near Richland, Washington, which is in the range of past glacial activity.

Salt deposits that have been intensely investigated in the U. S. are too far south to be directly affected by a glacial cycle. Here attention is focused on processes leading to salt dissolution by groundwater (WIPP, 1979; Patchick, 1980). A phenomenon of special interest in bedded salt is breccia pipe formation. A breccia pipe forms as a result of dissolutioning on a very local scale resulting in collapse of overlying rock strata.

Studies of potential tectonic activity include research on earthquakes and fault formation (Scott *et al.*, 1979). The major challenge of this research derives from the fact that repositories will be located in regions of great seismic and tectonic stability. In such regions there is little data on which to base predictions of future activity.

The second major category of processes affecting waste disposal is the *waste- and repository-induced phenomena.* The waste emits considerable amounts of radiation and heat. The presence of the mined excavation affects the surrounding rock mechanically and modifies the local groundwater hydrology. The most localized effects involve interactions between the waste package and the immediately surrounding rock. The behavior of the backfilled mine and connecting shafts might also be modified. The largest-scale effects are principally those due to the effect of heat from the buried waste on surrounding rock masses. The thermal output of the waste is important in several ways: it could lead to expansion of the rock mass leading to fracturing, it could increase rock permeability, and it could result in convection in groundwater.

The third category, *human-induced phenomena,* is the one least amenable to scientific analysis. Predictions of future activities by man are by their very nature entirely speculative. Very little work has been carried out in this area. Further investigation may result in a better understanding of what phenomena could occur and how the consequences could be mitigated. It is unlikely, however, that these efforts could greatly clarify the probabilities of these phenomena.

As has been discussed previously, the most likely way waste materials can be released to the biosphere is transport by flowing groundwaters. The amount of waste released to the biosphere in such scenarios principally depends on the amount of waste released from the waste packages, the permeability of rock and man-made materials to groundwater flow, and the boundary conditions that control hydraulic gradients. Most of the processes listed in Tables II—IV are important because of the way in which they affect groundwater transport. Tectonic activity, for example, is important primarily because of the potential for creating flow paths of higher permeability. Climatic fluctuations and dam building could affect hydraulic gradients.

The processes listed in all three categories can be dependent upon and can affect each other. The next section describes how analysis of these processes is used to assess repository safety.

## V. SCENARIOS

The identification of scenarios and their likelihood and the estimation of the consequences of the scenarios are the heart of any safety evaluation. A scenario may be defined as a possible sequence of processes and events which is describable by equations involving specified physical parameters. This section discusses the various methods of identifying scenarios and calculating scenario probabilities, while Sec. VI describes the techniques for estimating their consequences.

There is an infinite spectrum of repository behavior ranging from complete containment to total, instantaneous release of waste. Both discrete events (geologic or human-induced) and continuous processes (geologic processes, including those induced by the presence of the repository itself) may be of interest. From the various future possibilities, a finite set of scenarios must be chosen for analysis. Section V.A discusses the methods employed for selecting scenarios. The published estimates of scenario probabilities are discussed in Sec. V.B.

### A. Scenario selection methods

#### 1. Informal methods

The majority of studies select scenarios without using any formal procedure. The selection may be directed toward choosing the most likely case, toward defining a worst case in order to bound the consequences, or toward spanning a range of scenarios including both the more likely ones and relatively unlikely scenarios with greater consequences.

Defining the most likely scenario can be either trivial (and therefore uninformative) or extremely difficult. For repositories in salt, it seems likely that the salt would completely contain the wastes and that essentially nothing would escape; this scenario is trivial. For a repository in granite or basalt one would expect some release; a realistic description of such releases would require detailed models of the degradation of the waste package and flow of water through fractured rocks. These processes are not well understood; as a result, the most likely scenario is difficult to specify.

The worst-case approach is often adopted in order to avoid such problems. It has been followed most consistently and thoroughly in the study of a repository in granite by the Swedish Nuclear Fuel Safety Project (KBS, 1978a, 1978b). The attempt here is to bound the possible consequences under "credible" circumstances; that is, one ignores theoretically possible but highly unlikely events such as meteorite strikes. KBS uses repository design and worst-case data to effectively eliminate several categories of scenarios from consideration. For example, pathways to the biosphere via failed repository seals are removed by a bentonite-containing backfill which swells on contact with water and would seal any gap. The exceedingly small transit time to the biosphere utilized in the safety analysis (unwarranted by the field data) would likely bound the effects of such events as geologic faulting or fracturing. The repository would be sited in such a way as to provide little reason for human intrusion (other than entry into the repository itself). In this manner, the range of scenarios is narrowed to consider only a single worst credible situation.

The study which deals with the broadest range of scenarios is by Giuffre *et al.* (1980), who analyze groundwater transport through repositories in salt. (The 34 scenarios were chosen by another group at Lawrence Livermore Laboratory.) The salt may be either impermeable or slightly permeable. Scenarios are chosen by considering all of the various flow paths that might be present (boreholes, shafts, clay partings, fault zones, breccia pipes, etc.) and by modeling all relevant combinations.

As discussed in Sec. I.B, the definition of scenarios is somewhat arbitrary. The possibility that events could occur at different times is one source of the arbitrariness. For example, one can describe a scenario in which a borehole seal fails after a specified time such as $10^3$ yr. Most informal scenario selection uses this approach; however, the choice of a time is arbitrary. Radioactive decay and other gradual processes usually cause the consequences of a disruptive event to depend on its time of occurrence.[3] In studies aimed at placing an upper bound on consequences, this difficulty may be avoided by assuming the event occurs at the worst credible time (usually, but not always, the earliest). An alternative method is to perform the calculations more than once, assuming different times for the event. But neither technique provides a realistic estimate of the danger, which would require taking account of the probability of occurrence.

[3]For example, the consequences of a meteorite impact which fractured the rocks around a repository and created new pathways for groundwater would depend on the integrity of the waste package at the time of impact and the amount, if any, of waste migration which had occurred.

Yet another approach is to define occurrence of an event at time $t$ as a single scenario, with $t$ an unknown parameter describing the time of occurrence. A method has been developed to calculate expected values of consequences when scenarios are defined in this way (Ross and Koplik, 1978). This method is limited to cases where changes in the geology occur only as discrete events and the events are Markov (i.e., the probability of an event depends only on the current situation and not the past history).

#### 2. Fault- and event-tree analysis

Several studies use fault- and event-tree analysis to identify repository scenarios which lead to loss of containment (Logan and Berbano, 1978; Bertozzi *et al.*, 1977; Hill and Grimwood, 1978; Bingham and Barr, 1979; Schneider and Platt, 1974; d'Alessandro and Bonne, 1980). Fault-tree analysis is a deductive technique which begins with careful definition of the failure event and systematically diagrams backward in time to identify events or combinations of events that could cause the failure event. Event-tree analysis is an inductive technique which complements fault-tree analysis. It starts with the basic events and works forward in time to display their logical propagation to system failure events. The probabilities of these different events, if estimated quantitatively, can be used to compute scenario probabilities.

When fault or event trees are used to analyze repository behavior, they must treat both continuing processes and discrete events. Processes are included in trees by describing the effect of the process on repository behavior. Erosion, for example, may be treated by a separate calculation computing the time when the buried waste will be uncovered. The event of uncovering would appear in the tree. Fault and event trees are not useful for analyzing the processes themselves or their interactions. (Methods for developing scenarios through the analysis of dynamic behavior are described in the next section.)

One of the purposes of fault-tree analysis, as it has been applied to repositories, is to compute scenario probabilities. However, it is rarely possible to devise meaningful estimates of the probabilities of the events and processes that occur in the tree. Often expert opinion is used to provide a "best guess" of an event's probability. If a scenario consists of many events whose probability has been so estimated, the reliability of any computed scenario probability is questionable. As a result, fault and event trees have not proved useful in deriving scenario probabilities.

Fault and event trees have been valuable primarily as a means for organizing the thinking of the scenario analyst. Through the construction of a tree, the analyst hopes to ensure completeness by avoiding the omission of important phenomena which might contribute to repository failure. The tree structure also aids in determining which sequences of phenomena are most worthy of detailed analysis. To be sure, the process of logically deriving scenario descriptions can be conducted without the use of fault or event trees. In this case, construction of the tree may occur afterwards as a convenient means of graphically displaying the thinking that went into the derivation of the scenarios.

The most extensive use of these methods is in the draft environmental impact statement on the Waste Isolation Pilot Plant (the WIPP EIS) (WIPP, 1979). Five representative scenarios are chosen from a list compiled from previously generated lists and event-tree analysis. In all, 94 distinct scenarios are identified, of which four result in the direct transfer of wastes to the surface; the remainder introduce the wastes into an aquifer overlying the repository. The complete list of scenarios and how they are derived are discussed by Bingham and Barr (1979). The scenarios are ranked in importance by assigning relative probabilities to events using expert opinion.

On the whole, attempts to apply fault and event trees to repository safety assessment have met with limited success. While these methods are quite useful as a means of cataloging scenarios, the difficulties in obtaining reliable probability data (discussed below) are a formidable obstacle to the use of fault and event trees for quantitative purposes.

### 3. Simulation techniques

The fault- or event-tree approach is based on events which either occur or do not occur. It is therefore inappropriate for analyzing processes which occur continuously at a finite rate. Systematic approaches to the description of scenarios involving such processes are now being developed at Sandia National Laboratory and Pacific Northwest Laboratory (Campbell *et al.*, 1978; Stottlemyre *et al.*, 1980a). These approaches use simulation techniques to describe the effects of continuous processes on a repository.

The simulation methods use ordinary differential equations to describe the evolution over time of a set of variables describing a scenario. In the PNL work, these variables may also take random, discrete jumps due to certain discrete events. The equations are solved numerically.

The values given by the solution of the simulation equations appear as parameters in the equations describing the consequences of the scenarios. No work has yet been published in which consequences are calculated using parameters derived from a simulation.

Sandia's contribution to the identification and study of release scenarios is the numerical simulation analysis of releases induced by perturbations introduced by the repository itself. The method relies on modeling processes by differential equations which are first order in time. Whether this will be a useful tool in understanding the long-term dynamics of geologic and hydrologic systems is unclear. As Campbell states, "simulation calculations only reproduce phenomena already known or reasonably suspected and suggest, but do *not predict* how the real world may behave" (Campbell *et al.*, 1978; emphasis in original).

The Sandia model is designed for repositories in soluble rock. The rate of dissolution of the rock is determined as a function of the disturbances induced by the presence of the repository. This dissolution leads to local collapse followed by fracturing of the rock, which leads to further dissolution. Given the model's reliance on this positive feedback, it seems to be applicable only for salt as a repository medium.

Both the Sandia and PNL simulation analysis efforts are at a preliminary stage. Both are likely to be limited by the amount and quality of data which are obtainable.

## B. Probabilities

The long-term data from which to estimate scenario probabilities must be drawn from geology and archaeology. The difficulties in obtaining these estimates are considerable. Archaeology has never been a predictive science. Geology has only recently developed a predictive nature, and the data and techniques are such that estimates are more qualitative than quantitative.

### 1. Release of waste directly to the surface

The one type of scenario for which a probability can be calculated from observed data without heroic extrapolations is the meteorite strike. Meteorite strikes of sufficient size to cause repository disruption are reasonably randomly distributed and leave identifiable craters. The craters can be counted in order to provide estimates of probability. The earliest study which follows this procedure, by Claiborne and Gera (1974), determines the frequency of impacts producing craters larger than 1 km in diameter on the basis of ancient Canadian meteorite craters. It then calculates the probable number of craters of different diameters on the basis of the frequency distribution observed for the moon. This gives the estimated probability of a meteorite capable of creating a crater 600 m deep as $2\times10^{-14}$ $km^{-2}\,yr^{-1}$. This result is repeated by several other authors (GEIS, 1980; Cohen, 1977; Logan and Berbano, 1978; ADL, 1980).

Logan and Berbano (1978) require the meteorite to be able to exhume material from a depth of 800 m. For this reason, their probability estimate is half that of Claiborne and Gera. KBS (1978a) cites an estimate of $1\times10^{-13}$ $km^{-2}\,yr^{-1}$ for meteorites which can cause craters at least 100 m deep. The highest probability ($1.4\times10^{-12}$ km$^{-2}$yr$^{-1}$) is given by a study by Arthur D. Little, Inc. (ADL, 1980). This last analysis does not study the direct exhumation of the waste, but the increase in water transport through the repository due to the fracturing of the overlying rock by the impact. Recent work by Hartmann (1979) provides relationships between crater size, depth, impact energy, fracture depth, and seismic disturbance.

Direct release by volcanism has been considered by several studies (Claiborne and Gera, 1974; Smith and Kastenberg, 1976; Logan and Berbano, 1978; ADL, 1980; Crowe, 1980). ADL estimates that the national average probability of the formation of a volcanic vent is $1.25\times10^{-11}$ km$^{-2}$yr$^{-1}$. It notes that site selection could reduce this probability. An exception to this is basalt, where the probability is estimated as six times higher than for other media because the presence of basalt is indicative of significant volcanic activity in the past. These estimates are obtained from the average number of volcanic vents per km$^2$ formed within the coterminous United States within the last $10^7$ yr. Logan and Berbano (1978) present an estimate of $8\times10^{-12}$ yr$^{-1}$ for the probability of volcanism affecting a repository in the Delaware Basin with 0.15 of the contents being released. Since the Delaware Basin has had no volcanos since Permian times, a maximum probability of $5\times10^{-9}$ yr$^{-1}$ was chosen for volcano formation. Multiplying that figure by the ratio of the effective area of the repository to that of the basin yields the estimate given. A review of past work in this area and an evaluation of the disruptive effects of volcanic activity is given by Crowe (1980). According to Crowe, the probability of volcanic activity in nonvolcanic areas is undoubtedly quite low and can perhaps be bounded, but a realistic estimate is beyond the current state of the art in volcanology.

The likelihood of disruption by meteorites or volcanos should be placed in perspective. If the universe is approximately 10 to 20 billion years old, there is a one in a hundred chance or less that the events discussed above would occur in the known history of the universe. Given their remote probability, it is surprising that studies continue to consider these events. The Swedish studies reject them because of their improbability (KBS, 1978a; 1978b). The repeated reworking by others suggests that these scenarios are the rare ones for which data can be obtained.

Whereas natural events such as those discussed above leave easily identifiable remains and have occurred over such long periods of time that estimates of probability can be obtained for even extremely rare events, such is not the case with human actions. Although the human genus has existed for several million years, the change from hunting-gathering to an agriculturally-based community has only occurred within the last $10^4$ yr. The possible range in human technology may be seen today, with some cultures capable of space exploration while others are still in the "stone age." This range, together with the short observation time and the inherent difficulties of predicting conscious actions, implies that human intrusion scenarios are the most difficult to predict.

The most commonly mentioned type of human intrusion is drilling and intercepting a canister. WIPP (1979) lists the sequence of events which must occur to bring some of the repository contents directly to the surface:

- Institutional control is lost,
- Knowledge of the repository is lost,
- There is an incentive to explore in the area of the site,
- The repository area is chosen for drilling,
- The contents of the repository go unrecognized as radioactive material before and during drilling,
- Drilling intercepts a high concentration of radionuclides,
- The material brought up is left untreated and exposed.

If any of these events does not occur, then the direct release of radionuclides will not occur. There is no one study which estimates the probability of drilling at the repository site, the probability of hitting a canister, and the consequences of intercepting a canister. The generic environmental impact statement on commercially generated radioactive waste (GEIS, 1980) and WIPP EIS (WIPP, 1979) consider only the last two factors, while the ADL study considers only the first two. A study of the possible scenarios for WIPP (Bingham and Barr, 1979) suggests a relative probability of 0.1 for drilling at the site after $10^3$ yr. However, the authors state that they do not feel it appropriate to use this estimate in a risk analysis. Given that drilling at a site occurs, the probability of hitting a canister has been estimated by taking the ratio of total canister area to the area of the site.

GEIS (1980) also looks at solution mining occurring in a bedded salt repository $10^3$ yr after closure. In this scenario, radioactive wastes become mixed with table salt. No probabilities are assigned to this event, however.

### 2. Groundwater transport scenarios

Most studies do not assign numerical probabilities to groundwater transport scenarios. These scenarios often involve subsurface phenomena which are more subtle than meteorite strikes and volcanic eruptions, and therefore the difficulties in using geology predictively become greater. Stottlemyre *et al.* (1980b) have described a variety of hydrologic and geological phenomena that can affect groundwater release scenarios and have summarized the evidence concerning the likelihood of their occurrence. Additional work on predicting certain natural phenomena (climate, sea level fluctuations, denudation, floods, landslides, glaciation, etc.) is summarized by Scott *et al.* (1979).

Some writers have, however, attempted to assign probabilities to scenarios involving accelerated groundwater transport. Of such scenarios, probabilities are given most often for those in which release is initiated by earth movements along a fault. It is always assumed that a more permeable pathway is created along the fault. In studies of salt repositories, the faulting commonly is assumed to lead to dissolution of the salt.

The probability of fault movement was first estimated by Claiborne and Gera (1974) in their study of the Delaware Basin. Two major faults have been noted in this basin, whose age is $2\times10^{8}$ yr. Claiborne and Gera assume that two additional faults of the same length would become active at random times in the next $2\times10^{8}$ yr. A geometric analysis giving the probability that a random line segment will intersect a circular area the size of the repository is used to estimate the probability that a new fault will intersect the repository.

There are a number of weaknesses in this analysis. First of all, faulting is not a random phenomenon. Where faults exist, they are weaker than the surrounding rock, and stresses tend to be relieved through movement along existing faults rather than through creation of new ones. Furthermore, as noted by Caliborne and Gera, a new fault would not necessarily lead to containment failure. The fault would also have to create a permeable flow path, a significant amount of water would have to flow through it, and salt would have to dissolve faster than the fault zone would be closed by creep. The report makes no attempt to estimate the probability of these occurrences, and it is suggested that no such estimates are presently possible.

ADL (1980) uses the same general approach, although perhaps in a manner even more likely to lead to overestimates, to calculate the probability of formation of new faults in a variety of rock types. KBS (1978a) uses this approach to calculate an upper bound on the rate at which additional fractures will form in already fractured granite.

Fracturing due to remobilization of an existing fault is discussed by ADL (1980) and Logan and Berbano (1978). ADL uses the same approach as for the creation of new faults, with the time since the fault last moved substituted for the age of the formation. Logan and Berbano combine a series of empirical models and extrapolations to obtain the probability from the rate of occurrence of small earthquakes. This reasoning requires a number of poorly supported assumptions.

An alternative approach is employed by Bertozzi *et al.* (1977) for estimating the probability that a fault affects a repository in bedded salt. It is assumed that the faulting frequency for a tectonically stable zone is $2\times10^{-12}$ $\mathrm{km}^{-2}\,\mathrm{yr}^{-1}$. The average fault length is computed from the observed statistical distribution of fault lengths. The probability of faulting within a sensitive area surrounding the repository is then computed. The sensitive area around the repository is derived from estimated dissolution rates for bedded salt following faulting.

D'Alessandro *et al.* (1980) use the same method to estimate the probability that a repository in northeastern Belgium would be intersected by a fault. The probability that new faults will form is taken to be $5\times10^{-9}$ $\mathrm{km}^{-2}\,\mathrm{yr}^{-1}$. This value seems to represent an extreme upper bound. The probability that the movement along the fault would be sufficient to breach a repository is also discussed.

These methods, although sometimes useful in setting upper bounds for probabilities of fault movement, are dependent on so many counterfactual assumptions as to be without value in providing realistic estimates. According to Stottlemyre *et al.* (1980b), who summarize recent work in this area, estimates of faulting should be based on the state of effective stress, material properties, recorded seismicity, observed cumulative deformation, average strain rates, and anticipated changes in strain rates. A number of different models exist for predicting faulting frequency from this data, but the degree of uncertainty in any estimate remains high.

Attempts to quantify the probability of other types of groundwater scenarios have been made by Bingham and Barr (1979) and ADL (1980). Again, the degree of uncertainty in these estimates and the extent to which the estimates rely on expert opinion and recent theoretical work should be emphasized. Not only is it difficult to predict natural geologic events but scenarios also depend on the extrapolated performance of engineered features of a repository. For example, the probability of failure of borehole and shaft seals must, to a large extent, be estimated by use of engineering judgment. Another problem area involves the possibility that certain features, such as faults and breccia pipes in salt formations, could go undetected during site exploration. Further difficulties are presented by the possibility that future human activities might affect groundwater release scenarios. Bingham and Barr acknowledge these issues and emphasize that their results are "intended only to establish *relative* likelihood for the scenarios; they have little absolute significance."

Published estimates of scenario probabilities, including both direct releases and groundwater transport, are summarized in Table V, which is adapted from Burkholder (1980a).

### C. General remarks

An examination of work to date shows that the difficulty of choosing scenarios and evaluating their probabilities profoundly affects the choice of methods for repository safety analysis. Most generic studies do not even attempt to span the range of possible scenarios, let alone systematically evaluate all possibilities. Usually some scenarios are chosen using informal methods and the consequences of each are calculated. In site-specific studies, some attempts have been made to develop worst-case scenarios or to comprehensively sample the range of possible consequences. These efforts as well have relied, to a considerable degree, on informal selection methods.

TABLE Va. Summary of published probabilities.

| Author/ system | Scenario | Cumulative probability | Comments |
|---|---|---|---|
| Claiborne and Gera (1974)/ Los Medanos bedded salt | Meteorite impact | $10^{-10}$ @ $10^3$ yr<br>$10^{-7}$ @ $10^6$ yr | |
| | Faulting—water intrusion—transport to well | $10^{-7}$ @ $10^3$ yr<br>$10^{-4}$ @ $10^6$ yr | |
| Girardi *et al.* (1977)/ Generic bedded salt and domed salt | Water intrusion—transport to surface water body | $10^{-9}$ @ $10^3$ yr<br>$10^{-3}$ @ $10^6$ yr | Probabilities of causative mechanisms not reported separately |
| Logan and Berbano (1978)/ Los Medanos bedded salt | Meteorite impact | $10^{-10}$ @ $10^3$ yr<br>$10^{-7}$ @ $10^6$ yr | |
| | Volcanic explosion | $10^{-9}$ @ $10^3$ yr<br>$10^{-6}$ @ $10^6$ yr | |
| | Volcanic transport to surface | $10^{-8}$ @ $10^3$ yr<br>$10^{-5}$ @ $10^6$ yr | |
| | Faulting—water intrusion—transport to surface water | $10^{-4}$ @ $10^3$ yr<br>$10^{-1}$ @ $10^6$ yr | |
| KBS (1978a)/ generic granite | Meteorite impact | $10^{-10}$ @ $10^3$ yr<br>$10^{-7}$ @ $10^6$ yr | |
| | Fracture Formation | $10^{-6}$ @ $10^3$ yr<br>$10^{-3}$ @ $10^6$ yr | |
| Bingham and Barr (1979)/ Los Medanos bedded salt | Exhumation | | |
| | Drilling | $10^{-3}$ @ $\geq 10^3$ yr | |
| (selected scenarios) | Meteorite impact | $10^{-9}$ @ $10^3$ yr<br>$10^{-6}$ @ $10^6$ yr | Values intended only to establish relative likelihood for the scenarios |
| | Two aquifer connection—transport to surface water body | | |
| | Faulting | $10^{-7}$ @ $10^3$ yr<br>$10^{-4}$ @ $10^6$ yr | |
| | Shaft seal failure | $\sim 0$ @ $10^3$ yr<br>$10^{-7}$ @ $10^4$ yr<br>$10^{-5}$ @ $10^6$ yr | |
| | Igneous intrusion | $10^{-9}$ @ $10^3$ yr<br>$10^{-6}$ @ $10^6$ yr | |
| | Drilling | $10^{-1}$ @ $\geq 10^3$ yr | |
| | One aquifer connection—transport to surface water body | | |
| | Meteorite impact | $10^{-9}$ @ $10^3$ yr | |

TABLE V. *(Continued.)*

| Author/ system | Scenario | Cumulative probability | Comments |
|---|---|---|---|
| Bingham and Barr (Continued) | | $10^{-6}$ @ $10^6$ yr | |
| | Drilling | $10^{-1}$ @ $\geq 10^3$ yr | |
| | Mining | $\sim 0$ @ $\leq 10^5$ yr<br>$10^{-4}$ @ $\geq 10^6$ yr | |
| | Fracturing | $10^{-3}$ @ $\leq 10^5$ yr<br>$10^{-2}$ @ $10^6$ yr | |
| | Shaft seal fracture | $\sim 0$ @ $10^3$ yr<br>$10^{-5}$ @ $10^4$ yr<br>$10^{-3}$ @ $10^6$ yr | |
| | Natural salt dissolutioning | $\sim 0$ @ $10^3$ yr<br>$10^{-5}$ @ $10^4$ yr<br>$10^{-2}$ @ $10^5$ yr<br>$10^{-1}$ @ $10^6$ yr | |
| | Capitan reef potash mine flood | $10^{-6}$ @ $\geq 10^3$ yr | |
| ADL (1980)/ Generic bedded salt, granite, basalt, shale, and domed salt | Undetected borehole | $10^{-5}$ @ $\leq 10^6$ yr<br>$10^{-4}$ @ $\leq 10^6$ yr<br>$10^{-3}$ @ $\leq 10^6$ yr | Bedded salt<br>Granite, basalt, shale<br>domed salt |
| | Drilling | $\sim 0$ @ $10^2$ yr<br>$10^{-2}$ @ $2\times 10^2$ yr | All rock types<br>Bedded salt, shale, domed salt |
| | | $10^{-3}$ @ $2\times 10^2$ yr | Granite and basalt |
| | Faulting | $10^{-5}$ @ $10^3$ yr<br>$10^{-2}$ @ $10^6$ yr | Bedded salt, granite, and shale |
| | | $10^{-4}$ @ $10^3$ yr<br>$10^{-1}$ @ $10^6$ yr | Domed salt |
| | | $10^{-3}$ @ $10^3$ yr<br>$\sim 1$ @ $10^6$ yr | Basalt |
| | Volcanism | $10^{-7}$ @ $10^3$ yr<br>$10^{-4}$ @ $10^6$ yr | Bedded salt, granite shale, and domed salt |
| | | $10^{-6}$ @ $10^3$ yr<br>$10^{-3}$ @ $10^6$ yr | Basalt |
| | Igneous intrusion | $10^{-7}$ @ $10^3$ yr<br>$10^{-4}$ @ $10^6$ yr | Bedded salt, granite, shale |
| ADL (Continued) | | $10^{-6}$ @ $10^3$ yr<br>$10^{-3}$ @ $10^6$ yr | Domed salt |
| | | $10^{-5}$ @ $10^3$ yr<br>$10^{-2}$ @ $10^6$ yr | Basalt |
| | Meteorite impact | $10^{-7}$ @ $10^3$ yr<br>$10^{-4}$ @ $10^6$ yr | |
| | Breccia pipe | $\sim 0$ @ $< 500$ yr<br>$10^{-8}$ @ $500$ yr<br>$10^{-5}$ @ $10^3$ yr<br>$10^{-2}$ @ $10^6$ yr | Bedded salt only<br>Bedded salt only<br>Bedded salt only<br>Bedded salt only |

A variety of methods have been proposed for the systematic, quantitative selection of scenarios. But the development of these techniques has been impeded by several factors, of which the scarcity of data is most prominent. Although ways of systematically generating scenarios from lists of processes and events are available, their heavy reliance on judgmental input data makes the value of the formal calculations hard to see. Furthermore, these methods produce large numbers of scenarios; the calculation of the consequences of all these scenarios would require enormous amounts of time, money, and data. In practice it is possible to select a small number of scenarios whose analysis can give a good qualitative picture which is representative of a repository's performance (e.g., Bingham and Barr, 1979; WIPP, 1979). This winnowing process, however, relies more on informed judgment than on any formal analytical method.

Sets of scenarios which are both complete and tractable, as would be required for a complete description of repository behavior, have proven to be elusive. The quantitative scenario probabilities which a complete description would require have also been difficult to obtain. As a result, the problem of developing scenarios is a formidable obstacle to the conduct of repository safety assessments by quantitative probabilistic techniques whose purpose is the development of a complete description of the probability of various outcomes. Less ambitious approaches to the evaluation of repository safety seem more promising, for it has been possible to produce both worst-case scenarios which bound potential consequences and selections of scenarios which, although not chosen by any formal analytical method, nevertheless seem to give a good general picture of the behavior that may be expected.

## VI. PREDICTION OF CONSEQUENCES

Far more effort has been expended in determining the consequences of repository scenarios than has been spent in identification of scenarios and their probabilities. This is, in part, because the problem is more tractable. Once a scenario has been defined, it is possible in many cases to obtain data from which parameters required in the analysis may be estimated.

The techniques used to predict consequences depend on whether the scenario involves the direct or indirect release of waste to the biosphere. Direct release, whether by natural causes such as meteorite strikes or by human actions such as drilling, circumvents the majority of the barriers placed between the waste and man. The time at which release to the surface occurs (and therefore the amount of radioactive decay) is determined by the time at which the event takes place rather than by the transport delays which dominate groundwater release scenarios. The time of occurrence is therefore paramount in determining consequences. For example, GEIS (1980) investigates the effects of a meteorite strike breaching the repository. The dose to an individual changes by three orders of magnitude when the breach occurs at repository closure rather than a thousand years later.

The methods used to compute the consequences of sudden direct releases of radioactivity are similar to those used for reactor accidents and will not be discussed here. The results of such calculations are summarized in Sec. VIII.A.

Indirect release scenarios, in which groundwater transport is the primary mechanism, must consider the barriers between the waste and man. Once a scenario has been chosen, data are required to describe the system. As described in Sec. III, the system may be divided into four components: waste package, repository, geologic formations, and the surface environment. Although any disposal system will contain these four components, several of the early studies ignore one or another component. For example, Girardi *et al.* (1977) do not consider a geologic barrier between the waste form and the biosphere. Some studies avoid the difficulties in estimating biosphere uptake and dose effects by computing only radioactive releases to the biosphere (Giuffre and Kaplan, 1979; Raymond *et al.*, 1980; Bradley and Corey, 1976).

### A. Waste package

The waste package is usually modeled in a simple manner. Most commonly, the release of waste from the repository is assumed to begin at some time and proceed at a rate given by a simple function.

The delay before release begins may be caused by the time required for canisters around the waste to corrode or by the time required for a repository cavity to fill with water and reach a pressure at which water flows out of it. Clay overpacks such as those proposed by KBS could also delay release, but no studies take any credit for such delays. Grundfelt (1978) examines this effect and concludes that, for the very slow leach rates assumed by the KBS spent fuel study (KBS, 1978b), the additional delays in the clay are of little significance.

The chemical environment around a waste canister can be complicated by temperature increases, radiation fields, and the juxtaposition of differing materials. As a result, many studies decline to predict a canister lifetime and assume that release will begin as soon as the repository repressurizes (Berman *et al.*, 1978; Hill and Grimwood, 1978). Girardi *et al.* (1977) assume leaching begins immediately upon emplacement. Giuffre *et al.* (1980) begin nuclide release at either canister dissolution or repressurization, whichever occurs later.

The KBS group lengthens canister life by heavily shielding the waste to nearly eliminate radiation, diluting the waste and delaying burial by 40 yr to reduce heat generation, and using highly corrosion-resistant materials as an outer shell. The canister illustrated in Fig. 5 is expected to completely protect the waste for times on the order of hundreds of thousands of years, and probably for millions of years. A group of specialists appointed by the Swedish Corrosion Institute concludes that "it is considered realistic to expect a service life of hundreds of thousands of years" (Swedish Corrosion Institute, 1978; see also NAS, 1980). The KBS repository will refill with water long before such canisters corrode; therefore the time required for refilling is neglected in the safety analysis. KBS also investigates the effects of a few defective canisters which offer no resistance to dissolution of the waste within them; doses are much smaller than the later doses caused by the remainder of the waste (KBS, 1978a; Grundfelt, 1978).

Once dissolution has begun, wastes will enter groundwater as they are released from the waste form matrix (typically glass or uranium dioxide). A number of different regimes have been discussed which may control dissolution rates:

- A highly soluble matrix which offers no protection; dissolution is controlled by the solubility of each radioactive element,
- A matrix whose rate of dissolution is controlled by its equilibrium solubility; the radioactive elements are released as the surrounding matrix dissolves,
- A matrix whose rate of dissolution is kinetically controlled (referred to as leach-controlled, as opposed to solubility-controlled); again, trace elements are released as the matrix dissolves,
- A matrix which dissolves so slowly that the release of the radionuclides it contains is controlled by their diffusion through the matrix.

It is not well established which of these regimes will govern the dissolution of nuclear wastes, although the second of them is frequently assumed for spent fuel and the third for vitrified wastes. In the third and fourth alternatives, kinetics dominate, making release rates highly dependent on surface areas. Surfaces can be changed drastically by physical phenomena such as cracking which are difficult to predict, adding to the uncertainty of release rates.

Use of experimentally measured dissolution rates also encounters difficulties. Rates of release of contaminants from a glass matrix appear to change over time. Although these rates appear to level off over a few years after an initial decline (Yen-Bower *et al.*, 1979; Bonniaud *et al.*, 1979), the longest study to date shows a slower decline continuing for twenty years (Walton and Merritt, 1980).

The possibility of future physical or chemical changes in the waste form, such as devitrification of glasses, further complicates the problem. There is a considerable literature on all of these matters, represented most notably in the series of conference proceedings entitled *Scientific Basis for Nuclear Waste Management.*

Most safety analysts have been led to model the release as a simple function, commonly a square wave. A wide range of release rates is used. It is noteworthy, however, that no study uses a rate low enough to correspond to the leach rate of $10^{-13}\ \mathrm{g\,cm^{-2}\,day^{-1}}$ which Walton measures. Also, equilibrium solubility effects are usually ignored even when they might greatly reduce release rates; an example of such neglect is cited in the next section.

### B. The repository

The repository can affect the rock around it in a variety of ways. Paths for groundwater flow can be provided by the repository tunnels and corridors, by shafts and boreholes, and by zones of fractured rock created by construction. Overlying strata can be stressed both by thermal expansion of rocks near the repository and by collapse into void spaces. All of these processes will be strongly affected by details of the repository's design, the methods used to construct it, and the backfills and seals with which it is closed.

Only a few studies examine the consequences of releases through these pathways. The WIPP EIS investigates two such scenarios. The first is a failed borehole which links upper and lower aquifers through the repository. In the second, failed shaft seals and a borehole create a pathway for water in the upper aquifer to reach the repository and return to the upper aquifer. The bounding scenario in the analysis is a variant of the latter one in which all of the water in the aquifer reaches the repository (WIPP, 1979). Berman *et al.* (1978) and Giuffre *et al.* (1980) calculate releases through shafts and boreholes whose seals have failed. In all these studies, the shafts and boreholes provide a means for small amounts of contaminated water to reach overlying aquifers very rapidly. Because the amounts of water involved are very small, the failure of the latter two studies to consider equilibrium solubilities leads to serious overestimation of some releases. One of the cases considered by Giuffre *et al.* provides an extreme example of such overestimation; the concentration of $^{129}I$ in water moving through the borehole may be calculated to be approximately $2\ \mathrm{ton\,m^{-3}}$.

Several studies investigate the effects of backfill material within the repository on system performance. One is the sensitivity analysis performed by Hill (1979). In this analysis, the initial 600 m of the flow path has a higher sorptive capacity than the remainder of the path. It is noted that the effect of the backfill is twofold: the effective rate of nuclide release is reduced, and the time required for the nuclides to reach the biosphere is increased. Berman *et al.* (1978) consider the effects of deteriorated backfill in greater detail. In all their scenarios, flow takes place through fracture zones around the repository corridors, tunnel, and shaft. When the backfill fails, flow may take place through it as well. A sensitivity analysis indicates that the peak concentration of the nuclides and peak dose is affected only slightly by this barrier. Ratigan *et al.* (1977) study the hydrology in and around a repository more carefully than either of the preceding authors. The principal conclusion to be drawn from Ratigan *et al.*'s results is that the repository does not provide a pathway for more rapid release of radionuclides in the rather permeable granite repository they are studying. For that reason, the details of this work do not enter into the safety analysis of the repository (KBS, 1978a).

One further issue is the possibility that thermal convection would accelerate the movement of contaminated water. Maini and Hocking (1977) assume the existence of continuous cracks extending from the repository to the surface and find a very large convection effect. A more thorough analysis by Ratigan *et al.* (1977) finds the effect to be several orders of magnitude less. The difference is due primarily to Ratigan *et al.*'s assumption that the rock has a permeability like that of a porous medium; also, the Swedish repository studied by Ratigan *et al.* has a considerably smaller thermal gradient than that assumed by Maini and Hocking.

A large amount of work is currently in progress to assess the magnitude and effects of heating and other processes caused by the presence of a repository. Much of it is directed toward scenario definition by simulation methods; this is discussed above in Sec. V.A.3. This work has not been carried through to calculation of consequences.

### C. Subsurface transport

#### 1. Theory

The modeling of subsurface contaminant transport in groundwater has been the subject of intense investigation prompted by interest in protecting water supplies. Applied modeling studies invariably assume that transport of contaminants in porous media is described by a diffusivity equation. If the concentration of nuclide $r$ in the interstitial water is $C_r(x,t)$, the interstitial fluid velocity is $\mathbf{v}$, and the medium is of uniform porosity, then one has

$$B_r \frac{\partial C_r}{\partial t} = \frac{\partial}{\partial x_i} D_{ij} \frac{\partial C_r}{\partial x_j} - v_i \frac{\partial C_r}{\partial x_i} - B_r \lambda_r C_r + B_r \lambda_s^r C_s \ . \tag{1}$$

Here $\mathbf{D}$ is the coefficient of convective dispersion, which in practice is invariably assumed to have its principal axes aligned in the direction of the water velocity $\mathbf{v}$. $B_r$ is the "retardation factor," which accounts for material sorbed on the rock. $\lambda_r$ is the radioactive decay constant and $\lambda_s^r$ is the production rate of nuclide $r$ from decay of nuclide $s$. The subscripts $i$ and $j$ refer to the three Cartesian coordinates and are summed when repeated. The subscript $r$ identifies individual radioactive species and is not summed.

The interstitial fluid velocity is determined by the hydraulic head $H$ and the permeability of the porous medium. It is given by Darcy's law as

$$\mathbf{v} = -\frac{1}{n_e} \mathbf{K} \cdot \nabla H \ , \tag{2}$$

where $\mathbf{K}$ and $n_e$ denote, respectively, the hydraulic conductivity and effective porosity of the medium, and $H$ is the piezometric head.

Underground flow cannot always be adequately represented as flow through a porous medium as described by Eq. (2). Flow through a geologic medium may be composed of flow through the pores between the individual grains in the soil or rock (interstitial flow) or flow through fractures or other flaws (fracture flow). If sufficiently fractured, a fractured rock mass is equivalent on a large scale to a porous medium with large grains, and Eq. (2) may be employed. On the other hand, if fractures and joints in the rock are sparse, alternative models may be required. As flow through fractures in sparsely fractured rock can be much faster than in porous media, the value of accurately modeling fracture flow for assessing waste migration is apparent.

Some efforts to model fracture flow for the analysis of waste disposal safety appear in the literature, but no fully accepted approach exists (Gale and Witherspoon, 1979; Bredehoeft *et al.*, 1978). One factor which has limited the amount of attention paid to flow in sparsely fractured rock is that most past work in hydrology has concentrated on the *most* permeable rocks, which are of greatest importance as water supplies. It is the hydrology of the *least* permeable rocks that is crucial in analyzing waste repositories.

The principal difficulty in understanding the hydrology of sparsely fractured rock lies in knowing the density, continuity, spacing, and orientation of fractures and joints in the rock. Fracture flow is an important area for further research. Improved understanding would be valuable in assessing the safety of repositories in some types of rock.

The most problematical element in the transport equation (1) is the retardation factor $B_r$. This factor represents chemical interactions which cause contaminants to move more slowly through a porous medium than does the water in which they are dissolved. In the linear equation (1), $B_r$ is the ratio between the water velocity and the contaminant velocity. A rigorous derivation of the equation requires that contaminant concentration be low, that reactions be rapid enough that a local equilibrium is reached everywhere, and that the mechanisms of importance in retarding the contaminant all be reversible (Grove, 1970). Even then the retardation factor can be highly dependent on the chemical and physical properties of the porous medium.

In fact, there are a variety of processes, collectively referred to as "sorption," which can cause contaminants to move more slowly than the water which carries them. These include irreversible precipitation as well as ion exchange and surface adsorption. Diffusion between the principal pores or fissures and microfissures in which movement is negligible can introduce additional complexities (Neretnieks, 1980). Laboratory measurements of sorption phenomena are extremely difficult to reproduce (Relyea and Serne, 1979). However, sorption is extremely important in evaluating repository safety; retardation factors on the order of $10^3$ and $10^4$ are regularly reported for some elements, especially transuranium elements, and sorption phenomena have been well documented in field observations (Borg *et al.*, 1976). The extensive body of literature concerning these phenomena has been reviewed by Onishi *et al.* (1981). In view of the cloudy theoretical situation, the simple model represented by a constant retardation factor seems appropriate for safety assessment studies; it should, however, be viewed as an approximate phenomenological model and not an exact one. In any case, the values of retardation factors should be supported by experimental evidence developed with careful attention to groundwater and rock chemistry, and the results of field tests are to be preferred when feasible.

The various levels of complexity at which contaminant transport can be modeled are reflected in the number of hydrologic and transport codes presently available. An excellent review of these may be found in Anderson (1979). In general, repository safety assessments have utilized available hydrologic codes, and these will not be reviewed here. Contaminant transport codes which have been utilized in the investigation of nuclear waste disposal may be roughly categorized by complexity according to the number of dimensions in which they solve the transport equation. [In some analyses it can be useful to solve the flow equation (2) in a higher number of dimensions than the transport equation (1).]

### 2. One-dimensional codes

A one-dimensional treatment of contaminant transport is based on approximating the contaminant concentration by an average across some appropriately defined stream tube. This approach is valid when the curvature of the stream tube is not excessive and the variations, if any, in velocity and contaminant concentration across a cross section of the stream tube are not correlated with each other. A derivation and mathematical formulation of these conditions is given by Ross and Koplik (1979).

The GETOUT code (DeMier *et al.*, 1979) uses an analytic solution to the one-dimensional transport equation. Release from the waste package either is instantaneous or proceeds at a constant rate until all waste has been released. Each nuclide is treated independently.

NUTRAN (Ross *et al.*, 1980) allows one to represent flow paths as a (possibly three-dimensional) network of one-dimensional segments, which allows for the representation of small discontinuities. The output of each segment is used as the input to subsequent segments. Within each segment, a Green's function is used to calculate the output during each time interval from the input during previous time intervals. Releases of individual nuclides from the waste package may be constrained by the solubilities of individual elements as well as by the breakdown of the waste form. Campbell *et al.* (1980) have developed a simpler but less flexible program using the same general approach.

Logan and Berbano (1978) also use a one-dimensional model. However their formulation of the transport equation is in error by a factor of $B_r n_e$. Pigford and coworkers have developed one-dimensional solutions of the transport equation [Harada *et al.*, 1979; Foglia *et al.*, 1979; Higashi *et al.*, 1979; in the paper by Foglia *et al.*, Eqs. (4) and (9) have typographical errors]. Hadermann has developed one-dimensional solutions for the transport of radionuclide chains through sorbing layered media (Hadermann, 1980; Hadermann and Patry, 1980).

A principal advantage of the one-dimensional codes is the greater ease of treating radioactive decay chains. GETOUT incorporates an exact solution for two- and three-member decay chains, as do the formulas derived by Pigford's group. Chains with more than three members can, in all cases of practical importance, be adequately approximated by three-member chains (Burkholder *et al.*, 1975). NUTRAN treats decay chains by applying correction factors to the peak release rate of the precursor of the chain and to the total amount of precursor released. Conditions under which this approach gives accurate results are derived by Berman *et al.* (1978) and summarized in part by Giuffre and Ross (1979). The conditions on the peak release rate correction are usually, but not always, satisfied; when not calculated accurately, the peak release will be overestimated. The conditions for accuracy of the correction to total release are considerably less stringent.

The analytic solutions used in GETOUT as originally published (Lester *et al.*, 1975; Burkholder *et al.*, 1975) are incompletely specified and ambiguous. As a consequence, erroneous terms affecting the post-peak "tail" of releases appeared in results from the early versions of the computer program. These terms were eliminated on an *ad hoc* basis (Brandstetter *et al.*, 1979) until the source of the error was detected by Burkholder and Rosinger (1980). The analytic solutions are presented by the latter authors in unambiguous form. Comparisons of early GETOUT results with calculations using the corrected code (Burkholder and Rosinger, 1980) and NUTRAN (Ross *et al.*, 1981) show that peak release rates calculated with early versions of GETOUT are correct as reported, although the "tail" shown in the results of KBS (1978a) may be higher than if correctly calculated.

A number of studies apply finite-difference or finite-element techniques in one dimension. Many of these use two- or three-dimensional codes; such work is discussed below. De Marsily *et al.* (1977) and Center *et al.* (1976) use one-dimensional finite-difference codes; Center *et al.*'s results (which are adopted by Chipman *et al.*, 1979, and Fullwood and Mendoza, 1979) are dominated by numerical errors resulting from use of an excessively large grid spacing and should be ignored.

### 3. Two- and three-dimensional codes

Two- and three-dimensional models of contaminant transport in groundwater usually employ finite-difference or finite-element methods (Pinder and Gray, 1977).

These techniques have been employed with success to analyze a variety of pollution problems. It should be noted, however, that some of the situations encountered in repository safety analysis, such as large contrasts in distance scales and permeabilities, render more likely the presence of the numerical difficulties which these methods sometimes encounter (Strang and Fix, 1973).

An alternative modeling approach is to follow the motion of individual particles or parcels of contaminant. This approach has found a variety of applications (Raymond *et al.*, 1980; Hebel *et al.*, 1978). In it, particles travel independently and dispersion is represented by a random walk. Each particle takes a series of steps. Each step is the sum of a fixed component determined by the water velocity and a random component representing dispersion.

A three-dimensional finite-difference code which treats hydrology, heat, and mass transport together (SWIFT) has been modified to include radioactive decay and sorption and used to solve a two-dimensional problem for the WIPP EIS (Dillon *et al.*, 1978; WIPP, 1979). It should be noted that difficulties with numerical dispersion and oscillatory instability have occurred with this program (Campbell *et al.*, 1978).

A three-dimensional finite-element hydrologic code (DAVISFE) has been modified for coupled solution of water flow and energy and contaminant transport (ONWI, 1979). No applications of this code to safety analysis of a repository have yet been reported.

A three-dimensional, discrete parcel, random walk approach for modeling contaminant transport is used in the MMT code (Burkholder *et al.*, 1979). Several studies use a one-dimensional version of the program (Raymond *et al.*, 1980; INFCE, 1979b; Cole and Bond, 1980).

A review of these applications illuminates an underlying tradeoff. Additional complexity in modeling requires a concomitant increase in the amount and detail of the data, which may be difficult to obtain. As noted by Anderson (1979), the contaminant transport codes currently available cannot realize their full potential until problems of acquiring detailed field data are resolved. Indeed, the three-dimensional transport codes developed for repository safety analysis have invariably been applied to one- or two-dimensional problems; the data required to define the three-dimensional problem has not been available.

Recent theoretical studies have addressed the problem of modeling groundwater flow and contaminant transport in the absence of sufficient data. Work has focused on modeling stochastic representations of the flow system. Such work is closely related to other studies which attempt to model dispersive transport in heterogeneous porous media (Matheron and de Marsily, 1980; Neretnieks, 1981; Ross, 1981). These studies find that dispersive spreading of pulses in proportion to the square root of time, as predicted by Eq. (1), is not a general characteristic of flow in porous media. The conclusion is borne out by some field data. No application of these new approaches to safety analysis of repositories has yet been made.

### D. Transport in the surface environment

Biosphere models are used to investigate the transport of radionuclides through the environment to man. Biosphere models for assessing waste repository safety differ from earlier models for assessing accidental releases of radioactivity in that they attempt to model the cycling of long-lived radioactive elements in the environment. Mauro *et al.* (1977) review models applicable to waste repositories.

There are several codes currently in use (e.g., Soldat *et al.*, 1974; Duffy and Bogar, 1980; Bergman *et al.*, 1977; Campbell *et al.*, 1978; Logan and Berbano, 1978); all are variations on a multicompartment model in which each compartment represents a section of the physical world. The relevant dynamic processes are parametrized into transfer coefficients between the compartments. These transfer coefficients may differ from study to study, as may the modeling of isotope buildup in various parts of the biosphere and the dilution of waste as it enters the biosphere. The conceptual basis for the biosphere transport models has been established; it is the availability and reliability of the model parameters, such as transfer coefficients, which limit their use.

In addition to modeling the transport of radionuclides through soil and water system, biosphere models also evaluate transfers through the food chain to man. Such calculations are highly dependent on the assumptions made as to future human living habits. Among the assumptions usually made when calculating the effect of future release are: population densities, diet, recreational practices, water usage rates, and farming methods. Such predictions of future practices are, of course, purely speculative. The basic philosophy in employing biosphere models is to provide a tool for evaluating the consequences of radionuclide release under conditions not far removed from those experienced today; these models should not be understood as predictions of the future.

### E. Dose to man and health effects

Standard methods are available for computing the dose to humans from ingestion and inhalation of radioactive materials as well as from external radiation exposure. These methods undergo continuous revision as new information on metabolic transfers within the human body and new information on the body's response to ionizing radiation are developed. Tabulated dose factors for converting exposure to dose are readily available (NRC, 1977; Hoenes and Soldat, 1977; Killough *et al.*, 1978a; Dunning *et al.*, 1979; Adams *et al.*, 1978; ICRP, 1979; ICRP, 1981). Because dose conversion factors are updated, however, different safety studies have employed different sets of dose factors. Table VI gives a comparison of dose factors used in five different studies and shows major differences. A careful comparison of the different values in the literature is presented by Dunning and Killough (1981).

TABLE VI. Comparison of 50-yr dose commitment factors for ingestion.

| STUDY | WHOLE BODY (mrem/pCi) | | | CRITICAL ORGAN[a] (mrem/pCi) | | |
|---|---|---|---|---|---|---|
| | $^{129}$I | $^{237}$Np | $^{226}$Ra | $^{129}$I | $^{237}$Np | $^{226}$Ra |
| Logan and Berbano (1978) | $7.2\times10^{-6}$ | $6.3\times10^{-4}$ | $3\times10^{-2}$ | $5.2\times10^{-3}$ | $2.3\times10^{-2}$ | $3.7\times10^{-1}$ |
| Berman *et al.* (1978) and Burkholder *et al.* (1975) | $9\times10^{-6}$ | $5.6\times10^{-5}$ | $2.2\times10^{-1}$ | $7.2\times10^{-3}$ | $1.4\times10^{-3}$ | $3.1\times10^{-1}$ |
| Hill and Grimwood (1978) | $1\times10^{-5}$ | $6.2\times10^{-5}$ | $2\times10^{-2}$ | $1\times10^{-2}$ | $1.3\times10^{-3}$ | $3\times10^{-1}$ |
| KBS (1978a; 1978b) | $9\times10^{-6}$ | $4.6\times10^{-5}$ | $3\times10^{-2}$ | $1\times10^{-2}$ | $1.2\times10^{-3}$ | $3\times10^{-2}$ |

[a]Bone for $^{237}$Np and $^{226}$Ra, and thyroid for $^{129}$I.

Repository studies often reach quite disparate conclusions as a result of choosing different sets of ingestion dose conversion factors. We strongly recommend that future studies use the most recent tabulations, either those by the International Commission on Radiological Protection (ICRP, 1979; ICRP, 1981) or those generated using the INREM II computer code (Killough *et al.*, 1978b).[4] The ICRP tabulation formally supersedes the dosimetry work presented almost twenty years earlier in ICRP Publication 2 (ICRP, 1959). Major differences include much lower dose factors for $^{90}$Sr and $^{226}$Ra and much higher dose factors for the actinides, especially $^{237}$Np. Studies which employ dose factors based on ICRP Publication 2 often conclude that $^{226}$Ra dominates the hazard from HLW at long times, whereas $^{237}$Np dominates in studies based on the more recent data.

The INREM II code, developed for the U. S. Nuclear Regulatory Commission, uses data and models quite similar to those employed by the ICRP. The INREM II results, however, are somewhat more applicable to assessing doses from environmental releases than those of the ICRP, which are oriented towards deriving exposure standards for radiation workers.

Except in the most extreme scenarios, doses received by individuals neighboring a waste repository will be low level (generally taken to mean on the order of 10 rad/yr or less). The health effects of low-level radiation are a subject of considerable debate, and no attempt will be made to review the literature here. Most repository safety analyses that estimate health effects rely on the major studies published by national and international organizations (ICRP, 1977; UNSCEAR, 1977; BEIR, 1972; EPA, 1973). A more recent review is provided by BEIR (1980).

The health effects from low-level radiation are long-delayed somatic and genetic effects. Somatic effects of most concern are the induction of cancer in various body organs. At very low dose levels (below natural background radiation) there is insufficient evidence to directly assess cancer risk. Therefore health effects have been extrapolated from the results of studies at higher dose rates. Two studies of repository safety have gone beyond assessment of health effects to compute the monetary cost of repository risk (Logan and Berbano, 1978; Bradley and Corey, 1976). This approach requires placing a monetary figure on the value of human life and injury to it, which is a matter of great controversy.

Recently a simplified approach to assessing internal doses and health effects has been proposed (ICRP, 1977). An effective dose equivalent is defined which is a weighted average of committed dose equivalents for specific organs. The weights are determined on the basis of the cancer and genetic risks associated with the respective organs. Using this approach, an effective dose from ingestion of a given radionuclide is equivalent in risk to the same dose to the whole body from external radiation. The effective dose factor is useful for several reasons: it simplifies the presentation of safety analysis results, it permits direct comparisons with background radiation, and it facilitates the estimation of stochastic health effects expected in a population.[5]

An important aspect of the fact that doses are expected to be much less than background is that effects will then be strictly cumulative (ICRP, 1977). That is, the number of individuals whose health is impaired will depend only on the total dose to a population and not on how doses are distributed among members of that population. (This result follows from the additional dose be-

[4]At the time this is being written, only Part 1 (ICRP, 1979) and Part 2 (ICRP, 1981) of the ICRP tabulation are available. Part 3 is expected to be issued in late 1981. The INREM II tabulation, which we recommend, also has yet to be published, although a summary of some results is given in Dunning and Killough (1981).

[5]It is not entirely appropriate to use the effective dose equivalent as defined by the ICRP to assess health effects in a population, since the ICRP weighting factors are derived for the protection of individuals. Guidance concerning what are rather minor adjustments to the calculation of an effective dose equivalent to account for this is given in ICRP (1980).

ing smaller than the variations in background exposure among different individuals. It does not depend on assuming a linear, no threshold dose-effect relationship.[6]) Therefore, if total health effects are the measure of repository safety, then as long as doses remain well below background the *amount* of radioactive material released will be more important than the *rate* of release. This has important implications for the use of waste forms to reduce release rates without affecting total releases.

### F. Cohen's natural analog approach

Bernard Cohen has developed an approach to evaluating the safety of nuclear waste disposal which is quite different from those discussed above. This approach is based on a comparison between nuclear wastes and analogous substances which are naturally present in the earth.

Cohen (1977) argues that an atom of waste buried 600 m deep is no more likely to reach man than an average atom of radium in the top 600 m of the earth's crust. He then calculates the probability of ingesting an atom of radium in two ways: comparing the amount of radium in the soil to the amount of radium measured in human bones, and with an analogous procedure based on the amount of radium in surface waters. The probability of ingestion is multiplied by the cancer risk from ingesting all of the waste as a function of time after disposal and integrated over $10^6$ yr. This latter calculation assumes that the probability that an individual will develop cancer depends linearly on the amount of waste ingested. It therefore requires that the amount ingested by each individual be small enough for this assumption to hold.

In a second paper (Cohen, 1980a), he compares the release of radioactive wastes with that of other substances in rocks. Geochemical evidence is presented which indicates that the various components of rocks are removed at similar rates. Cohen then estimates the rate of removal from the chemical composition of groundwater. These removal rates are used to estimate that $8\times10^{-4}$ fatalities will be caused by disposal of the wastes from 1 GW yr of electric power production.

Cohen concludes that the major safety problems are those not addressed by his method, that is, the possibilities that phenomena caused by the presence of the repository will accelerate release. As he points out, many studies employing more usual methods also fail to address this question.

## VII. EVALUATING UNCERTAINTIES

Uncertainties in predictions of the long-term behavior of a repository arise from the possibility that important scenarios have not been considered, from incompleteness or error in the models used to calculate probabilities and consequences, and from uncertainty in the values of the parameters which enter into these models. These three types of uncertainties may be referred to as scenario uncertainty, model uncertainty, and parameter uncertainty. Most work aimed at assessing uncertainties in repository performance has focused on parameter uncertainty, and the bulk of this section will deal with that topic. Some concluding remarks will be devoted to scenario and model uncertainty.

### A. Parameter uncertainty

Three different approaches have been used to assess the doubt which parameter uncertainty creates about the results of safety assessments. These methods are to analyze a worst case, to analyze the sensitivity of results to parameter variations, and to explicitly calculate uncertainty ranges on the results.

Worst-case analysis is the simplest of these techniques. Probably the most consistent and thorough worst-case analyses are those conducted by KBS (1978a; 1978b). These studies calculate consequences using both relatively realistic and worst-case values of parameters. To be sure, worst-case studies only place upper bounds on the dangers from repositories; one does not know whether such a large excursion from the "realistic" estimates is really possible.

Worst-case analysis is certainly the most convincing way to demonstrate that the damage done by a repository will fall within acceptable limits. The difficulty with the technique lies in the existence of possibilities which are conceivable—but usually very unlikely—of relatively high consequences. The meteorite strike discussed in Sec. V.D provides a good example. Usually such very unlikely possibilities are termed "incredible" and ignored. The line between credible and incredible is difficult to draw, however. Even the KBS study, in which the repository was designed so as to eliminate most such problems, was somewhat arbitrary in its choice of worst-case retardation factors.

Several studies (GEIS, 1980; Berman *et al.*, 1978; Giuffre *et al.*, 1980; Hill and Grimwood, 1978; ADL, 1977; Cloninger, 1979; Elert *et al.*, 1979; Hill, 1979) have used sensitivity analysis to ensure that results remain valid even if parameters are widely varied. Generally, sensitivity analysis is performed by varying parameters singly or in groups and noting how the calculated results change. This permits identification of the variables which most influence risk.

Extreme care, however, is required in the use of this technique to evaluate uncertainty. There are a number of ways in which large changes in output values can be missed, including:

- Failure to sample the entire range of parameters upon which the output depends in a nonlinear way,

[6]The conclusion drawn here is formally correct if the small doses are of the same type and intensity of radiation as natural background. The dose equivalent is a measure which is intended to take account of such differences, and, to the extent that it does, the conclusion has general validity.

- Failure to vary all combinations of interacting variables,
- Extension of models to parameter values beyond the range in which the model equations are valid.

Many sensitivity analyses include only a small number of trials and unexpected relationships can be overlooked.

Sensitivity analysis is most useful in cases where only a few parameters are significant sources of uncertainty. This can occur either because these parameters alone determine outputs or because all other parameters have been measured relatively accurately. In such circumstances, sensitivity analysis can be a powerful tool for bounding uncertainty.

A third, more difficult, technique for evaluating the effect of uncertainties in input parameters is to quantitatively estimate the uncertainties in results. In this method a probability density function is estimated describing the range each uncertain input variable may take. The input uncertainties are used to calculate a probability distribution of repository performance. Most commonly, no direct evidence is available to describe the probability functions, and expert opinion must be used. The reliability of the results will then tend to be more limited by the accuracy of the input probability distributions than by any calculational limitations.

Two techniques have been used for the quantitative estimation of uncertainties in safety analyses of nuclear waste repositories. Kaufman *et al.* (1980) have developed a computer code which propagates probability distributions through a series of linked models to provide a probability distribution function of repository performance. The models that have been used so far are relatively simple; it is not known how well the code will function with complex scenarios.

An alternate approach is the "Monte Carlo" method used at TASC and Sandia, in which a number of "sample repositories" are generated from the distributions of the input parameters (Berman *et al.*, 1978; Giuffre and Kaplan, 1979; Giuffre *et al.*, 1980; Iman *et al.*, 1978). The range seen in the outputs is a reflection of the possible range in repository performance due to input uncertainty.

The TASC work utilizes information on the probability distributions, ranges, and correlations of geological parameters supplied by geologists for the study. In other words, the Monte Carlo analysis is built from the field data, incorporating such things as correlations between variables. The Sandia work appears more theoretical, as the comment is made that "If there is a significant distribution effect it may also be important to determine appropriate ranges and distributions for input variables" (Iman *et al.*, 1978, p. 79). This approach runs the risk of being too far removed from the actual data to be practical.

The TASC work generates the repository descriptions using simple random sampling of the inputs. The Sandia work utilizes a "Latin hypercube" sampling strategy which yields a smaller sampling error in the output statistics for a given number of samples (McKay *et al.*, 1979). This sampling strategy has not yet been utilized with linearly correlated inputs, which are included in the TASC analyses. Recent work at Sandia (Iman, 1980) provides a method for dealing with correlated inputs on the basis of their rank correlations. A combination of the TASC approach to the data with the Sandia sampling strategy would provide an analysis with the strengths of both groups' approaches.

One weakness of all the past Monte Carlo studies is their lack of selectivity. Usually, the important uncertainty about a repository concerns how likely it is that consequences would be much worse than expected. In statistical terms, one is interested less in the mean and variance of the outcomes than in the characteristics of the high-consequences "tail" of the distribution. The uniform sampling methods which have been used to date are directed toward determining the shape of the entire distribution; few samples are taken in the tail. Use of stratified sampling techniques and a somewhat different emphasis in the reporting of results would be more appropriate.

Both TASC and Sandia use nonparametric correlation analysis to identify parameters which may contribute to the uncertainty in the results. The question arises of how the information from these analyses is to be used (something not usually addressed in the studies). In one approach, the outcomes may be classified as to whether or not they fall below a stated threshold. A repository may then be judged by whether the percentage of outcomes over that threshold is acceptably low. Correlation analysis would possibly be used to identify those parameters which contribute most to the uncertainty of repository performance. Further experimental work could then focus on these parameters so as to most effectively improve the accuracy of the safety analysis.

### B. Scenario and model uncertainty

Although most of the work explicitly directed to evaluating uncertainties discusses parameter uncertainties, the authors' observation is that most expressions of doubt about repository safety center on uncertainty in scenarios and models. Whether this popular opinion is justified is difficult to say, but it is clear that scenario and model uncertainties are less tractable and have been less studied. Many uncertainties in these areas reduce ultimately to the possibility that something has not been thought of—a possibility which by its nature can never be eliminated. Nonetheless some general remarks on scenario and model uncertainties are possible.

In most scientific fields relevant to nuclear waste disposal safety assessment, the fundamental scientific principles are well understood. The principles of radioactive decay, heat conduction, and fluid flow, for example, are very solidly established. In some other areas, such as the transport of dissolved contaminants in dispersive media and the mechanical properties of rock around mine openings, uncertainty about underlying principles does exist. In most of these areas, however, phenomenological models can provide adequate predictions.

There are three principal areas in which lack of complete scientific understanding contributes significantly to uncertainties in risk assessment: sorption, or chemical interactions between radionuclides and geologic media, groundwater flow in fractured media, and leaching of solid waste forms. Even here, however, it is possible to limit uncertainties through testing and through the engineered design of a repository. For example, uncertainties about repository safety resulting from poorly understood sorption characteristics can be reduced both by designing experiments to provide lower bounds on the strength of sorption interactions and by using chemical additives in backfill to control groundwater chemistry (KBS, 1978b).

## VIII. COMPARISON OF CALCULATED CONSEQUENCES

This section presents the results of repository assessment studies. The studies are listed in Table VII together with a brief description of the repository site, the purpose of the study, and the source of the data used in the analysis. Only five of the 17 studies evaluate disposal at specific sites; three of these consider sites in bedded salt in southeastern New Mexico.

Less than half of the studies had as their principal purpose an evaluation of the safety of disposal. The goal of most studies was to better understand the disposal system, i.e., to learn which site and repository design features are important to safety and which are not. To this end, studies often assume scenarios which result in release of waste, even when the most likely scenario may be no release at all. Such studies quantify the *consequences* of waste release from a repository but say nothing about the probability of release.

Table VII describes the type of data or assumptions used in each study to model the waste form, other engineered barriers, and the geologic barrier. There are significant differences among the studies which result (as will be seen later) in differences in predicted hazards.

The doses given in this chapter can be placed in perspective by some comparisons. Natural background radiation typically gives a dose of about 0.1 rem/yr. An acute dose of more than 500 rem would usually cause death from radiation sickness; the same dose spread over many years would not cause radiation sickness but would lead to an increased risk of cancer in later life.

### A. Direct release by natural events

The most commonly studied scenario involving a cataclysmic natural event is that of a repository struck by a meteorite (Claiborne and Gera, 1972; Logan and Berbano, 1978; Cohen, 1977; GEIS, 1980). The probability estimates for this event are discussed in Sec. V. Some studies, such as those by the Swedish Nuclear Fuel Safety Project (KBS, 1978a; KBS, 1978b), do not calculate consequences of this event in view of its incredibility. Of the studies which pursue the analysis further, the generic environmental impact statement on commercially generated radioactive waste (GEIS, 1980) presents the most detailed results. The consequences depend on the areal density of buried waste, which differs among disposal media. If the breach occurs at the time of closure, the whole-body dose to an individual 4 km from the point of impact ranges from $8.3\times10^3$ to $2.2\times10^4$ rem. If the repository breach occurs after a thousand years, the doses to such an individual range from 6 to 16 rem, while the regional population dose would be comparable to or less than natural background radiation. (These figures are for the disposal of spent fuel; the doses from high-level waste under comparable circumstances would be lower.) Given the remote probability of this event, the GEIS and other studies conclude that the societal risks are negligible, especially when compared with the damage that would be done by the mechanical impact of such a meteorite.

Similar results have been found for direct release by volcanism. Logan and Berbano (1978) estimate a whole-body dose to an individual of 18 rem if the breach occurs at $10^3$ yr. Given that this category of release depends upon events with extremely remote probabilities of occurrence and that the consequences, while considerable, are less than catastrophic unless the event occurs soon after emplacement, it must be questioned whether further effort in this area would result in useful information concerning repository safety.

### B. Human intrusion

There are two studies which perform consequence analyses of scenarios in which someone drills into a waste canister. As in the other direct release scenarios, the time of occurrence strongly affects the magnitude of the consequences. The draft environmental impact statement on the Waste Isolation Pilot Plant (WIPP, 1979) estimates the whole-body dose to a geologist who examines the drilling samples for 1 h at an effective distance of 1 m from the spent fuel to be 90 rem or 1.4 rem if drilling occurs at 100 yr or $10^3$ yr, respectively. The GEIS (1980) postulates an individual who is exposed to the contaminated drilling mud for 12 h/day for an entire year. That individual would receive a whole-body dose of 13 rem (spent fuel) or 19 rem (high-level waste) from a 1-yr exposure if drilling occurs after $10^3$ yr. A point should be noted concerning this scenario: The potential exposures, although sizeable, would not be likely to cause radiation sickness, and they would affect a very limited number of people (principally the drilling crew).

The GEIS also analyzes solution mining of a salt bed containing a repository $10^3$ yr after repository closure. The consequences to a population of $4\times10^7$ are estimated to be, at the very worst, less than what would be re-

TABLE VII. Summary of assumptions of past studies.

| STUDY | SITE | PURPOSE | DATA<br>Waste form | Other engineered barriers | Geologic barrier |
|---|---|---|---|---|---|
| Claiborne and Gera (1974) | Bedded salt in southeastern New Mexico | Safety evaluation | Dissolution rate based on laboratory experiments | Not considered | Uses available data on specific site but data very incomplete. One scenario involves fault connecting waste and aquifer |
| Burkholder *et al.* (1975) | Generic, nonsalt | Safety analysis to compare waste management alternatives | Dissolution rate based on laboratory experiments | Simple canister, short lifetime | Assumes fairly rapid groundwater flow to river |
| Girardi *et al.* (1977) | Generic | Development of methods and application | Dissolution rate based on laboratory experiments | Not considered | Assumed not to be a factor |
| de Marsily *et al.* (1977) | Generic | Analysis of the geologic barrier to waste release | Typical value for intact glass; arbitrary estimate for glass whose structure is destroyed | Simple canister, short lifetime. Effect discounted | Typical values and ranges for rock properties. Assumes vertical groundwater transport to surface |
| ADL (1977) | Generic | Analysis to provide guidance for EPA regulations | Dissolution rate based upon laboratory experiments | Simple canister, short lifetime | Typical values and ranges. Assumes vertical groundwater transport to overlying aquifer and then to river |
| Cohen (1977) | Generic | Safety evaluation | Not directly addressed | Not considered | Uses data on transfer of radium in rock to man |
| Hill and Grimwood (1978) | Generic, hard crystalline rock | Preliminary safety analysis | Dissolution rate based upon laboratory experiments | Simple canister, short lifetime | Typical values and ranges. Assumes groundwater transport to river or aquifer |

TABLE VII. (*Continued*.)

| STUDY | SITE | PURPOSE | DATA Waste form | Other engineered barriers | Geologic barrier |
|---|---|---|---|---|---|
| Berman *et al.* (1978) | Generic, bedded salt and shale | Analysis to provide guidance for NRC regulations | Dissolution rate based upon laboratory experiments | Simple canister, short lifetime. Typical values and ranges for mine backfill, shaft seals, borehole seals, and induced fracture zones | Typical values and ranges. Assumes vertical groundwater transport to overlying aquifer and then to river |
| Claiborne and Gera (1974) | Bedded salt in southeastern New Mexico | Safety evaluation | Dissolution rate based on laboratory experiments | Not considered | Uses available data on specific site but data very incomplete. One scenario involves fault connecting waste and aquifer |
| Burkholder *et al.* (1975) | Generic, nonsalt | Safety analysis to compare waste management alternatives | Dissolution rate based on laboratory experiments | Simple canister, short lifetime | Assumes fairly rapid groundwater flow to river |
| Girardi *et al.* (1977) | Generic | Development of methods and application | Dissolution rate based on laboratory experiments | Not considered | Assumed not to be a factor |
| Logan and Berbano (1978) | Bedded salt in southeastern New Mexico | Development of methods and application | Dissolution rate based upon laboratory experiments | Not a factor for scenarios considered | Uses available data on specific site but data incomplete. One scenario involves groundwater transport to overlying aquifer by fault |
| KBS (1978a) and KBS (1978b) | Granite in Sweden | Safety evaluation | Dissolution rate based upon laboratory experiments | Specially designed canister, buffer material and backfill; values based on laboratory measurements | Values for specific site based on field data from three prospective sites. For these sites groundwater transport from a repository is expected |

TABLE VII. (*Continued*.)

| STUDY | SITE | PURPOSE | DATA Waste form | DATA Other engineered barriers | DATA Geologic barrier |
|---|---|---|---|---|---|
| Lyon and Rosinger (1980) | Generic, plutonic igneous formation | Development of methods and application | Dissolution rate based upon laboratory experiments | Simple canister, short lifetime | Typical values, assumes groundwater transport to river |
| INFCE (1979a) | Generic, hard crystalline rock | Safety evaluation | Dissolution rate based upon laboratory experiments | Simple canister, short lifetime | Representative site parameters chosen. Assumes groundwater transport to lake |
| INFCE (1979b) | Generic, bedded salt | Safety evaluation | Dissolution rate based upon laboratory experiments | Not considered | Representative site parameters chosen. One scenario assumes massive fault connects aquifer to repository |
| Giuffre *et al.* (1980) | Generic, bedded salt | Analysis to provide guidance for NRC regulations | Dissolution rate based upon laboratory experiments | Simple canister, short lifetime. Typical values and ranges for mine backfill, shaft seals, borehole seals, and induced fracture zones | Typical values and ranges. Assumes vertical groundwater transport to overlying aquifer and then to river |
| Raymond *et al.* (1980) | Bedded salt in the Paradox Basin | Development of methods and application | Dissolution rate based upon laboratory experiments | Not considered | Uses available data on specific site but data incomplete. Assumes fault connects aquifer to repository |
| WIPP (1980) | Bedded salt in southeastern New Mexico | Safety evaluation | Uses worst-case bound | Upper-bound values for mine backfill and seals | Values for specific site based on laboratory and field experiments. Although data suggest otherwise, assumes sufficient driving force to move groundwater to upper aquifer in failure scenarios |

TABLE VII. (*Continued.*)

| STUDY | SITE | PURPOSE | DATA: Waste form | DATA: Other engineered barriers | DATA: Geologic barrier |
|---|---|---|---|---|---|
| GEIS (1980) | Generic | Safety evaluation | Dissolution rate based on laboratory experiments | Simple canister, short lifetime | For scenario with faulting and immediate release, worst-case values hypothesized. For scenario with faulting followed by groundwater transport, less extreme values chosen |

ceived from naturally occurring sources. Each individual in the population is assumed to ingest 1800 g of salt per year and to obtain all of the salt from this contaminated source. It is assumed that no radioisotopes are removed during processing of the salt.

If a potable aquifer were to be contaminated by waste, drilling could affect the doses received by man without penetrating the repository stratum. A well drilled into an overlying aquifer could eliminate the delay time involved in transport through that aquifer and avoid the dilution the waste would receive when it entered a larger body of water such as a river.

Only a few studies investigate well scenarios. KBS (1978a; 1978b) assumes that the waste is diluted by $5\times10^5$ $m^3$/yr of water during the transport to a well. The dilution in the lake studied in the same report is $2.5\times10^7$ $m^3$/yr of water. The maximum individual 50-yr dose commitments in the well case (0.26 mrem for HLW and 1.4 mrem for spent fuel) are approximately 15 times higher than those in the lake case.[7]

GEIS (1980) assumes that a large permeable fault intersects the repository and that all the water flowing through the fault enters a well 3 km downstream of the repository. The report indicates that this flow pattern would suggest that the aquifer is of low permeability and that wells are generally not drilled in such aquifers. The resulting doses are quite high (a 70-yr dose commitment of 14 rem to the thyroid if drilling occurs at $10^4$ yr). A high dose (a 50-yr dose commitment of 500 rem to the whole body) is also seen in the ADL (1977) study, where the well is assumed to be located directly above the repository. It should be noted that the ADL scenario assumes direct flow from the repository to the aquifer with the radionuclides reaching the aquifer 300 yr after closure. Both the scenario and the well location are markedly pessimistic choices.

Berman *et al.* (1978) calculate the radionuclide concentration in the aquifer directly above the repository (worst-case location). As in the ADL study, a driving force is assumed to exist which moves groundwater through the repository and up into the overlying aquifer. The concentrations of several nuclides approach or exceed maximum permissible concentration limits. Ross *et al.* (1979) and Giuffre *et al.* (1980) use a two-dimensional extension of the NUTRAN model to evaluate doses from well water. The 50-yr dose commitments to individuals calculated in the worst cases addressed by these studies are as high as nearly 20 rem/yr. These doses are highly dependent on the location of the well and the rate

[7]These dose commitments, and all those discussed subsequently, assume a single year of exposure. Where the sources give only an accumulated dose over some longer period of time, the doses given here have been obtained by dividing by the number of years of exposure. This procedure underestimates the actual dose commitment by no more than a factor of two.

of water withdrawal.

In the results of these studies, there is an inverse relationship between the dose rate to an individual and the number of individuals receiving that dose. If the waste reaches an aquifer with a low flow rate, there is very little dilution and, consequently, a very high dose to an individual using that water. That same low flow rate, however, would limit the number of individuals which the aquifer could supply. Giuffre *et al.* (1980) and Ross *et al.* (1979) calculate upper bounds on population doses and find they exceed the calculated individual doses by factors of 10 to 1200. The effects of drilling a well, therefore, seem to resemble those of drilling into the repository. The potential for sizeable exposures of individuals exists, but the scenarios depend on human actions, and the relatively large doses could affect only a limited number of people. It should be noted that if aquifers near a repository are sufficiently saline, they are unlikely to be used without purification as sources of drinking or irrigation water.

### C. Groundwater transport scenarios

It is generally agreed that the most likely pathway by which wastes could be released from a repository is transport to the surface by groundwater. The plethora of studies concerning this means of release addresses a range of geologic media, methods of estimating waste release and geosphere transport, and initial inventories which preclude immediate and direct comparison of the results. Rather, they must first be placed on a common basis. (A short summary of each study is given by Ensminger *et al.*, 1980).

#### 1. Normalization method

The results of various studies are not directly comparable for a number of reasons. First, studies assume that the repository contains different initial inventories of radioactive waste. A second problem is that studies employ different biosphere models. As discussed in Sec. VI.D., the biosphere model should be thought of as a somewhat arbitrary set of normalization factors rather than a prediction of the future. These problems are addressed here by normalizing each study to the waste produced by generating a fixed amount of electric power and by applying a common environmental consequence model. The method used to normalize the studies was developed by Koplik *et al.* (1979b) and Koplik and Bartlett (1980). The critical portions of any risk study are the models for predicting release rates of waste into the environment, and these are preserved.

The initial quantity of waste is arbitrarily chosen as the waste produced from $10^3$ GW yr of nuclear power generation (roughly the amount of waste expected to be contained in a United States repository). Even after normalization of each study to this quantity of waste, differences in isotopic composition remain. This is a consequence of the different assumptions concerning reactor irradiation and subsequent reprocessing. It would be difficult to normalize to a common isotopic composition, and in any case these variations reflect real uncertainties in future nuclear waste production and management. For several studies (Burkholder *et al.*, 1975; ADL, 1977; Raymond *et al.*, 1980), the amount of electric power generated is estimated from the values they give for the quantity of uranium in the waste. A metric ton of uranium is assumed to produce 32 MW yr of electric power. For CANDU fuel (Lyon and Rosinger, 1980), a metric ton of uranium is assumed to produce 8 MW yr of electric power.

A common biosphere transport, uptake, and dose model is applied to every study. The biosphere model adopted is deliberately quite simple. Release is assumed to occur into a river with a modest flow rate of $10^9$ $m^3$/yr. Water concentrations for human consumption are obtained by dividing the rate of release to the river by the flow rate. Doses are computed for the drinking water pathway alone, assuming an average individual intake of 370 l over one year. The individual dose conversion factors are for a 50-yr commitment from a single exposure and are taken from NRC (1977). This dose commitment is equal to the peak dose rate received from a 50-yr chronic exposure, which occurs in the 50th year. The first quantity has units of mrem, the second of mrem/yr.

The dose to an individual rather than population dose is chosen as the measure of consequences for several reasons. First, individual dose is more sensitive to variations in waste dissolution rate, travel time of the nuclides from the repository to the environment, and the time over which dispersive processes in the rock formation spread the waste (Koplik *et al.*, 1979b). Second, a population dose requires assumptions about the population, a parameter which is not subject to the same intrinsic limitations as the yearly amount of water drunk by an individual.

The first step in the normalization procedure is to convert release rates of nuclides to the biosphere into doses using the common biosphere model described above. The results are then adjusted to reflect a repository containing $10^3$ GW yr of waste.

Two modifications to this procedure should be noted. When only concentrations are given in a study, they are converted to release rates by multiplying by the groundwater flow rate (de Marsily *et al.*, 1977; ADL, 1977; Raymond *et al.*, 1980). In the study of Cohen (1977), only a fractional release rate for the waste as a whole is given, not release rates for individual nuclides. However, sufficient information is available in this study to allow the specific radionuclide composition of the waste to be determined, and hence radionuclide release rates could be computed.

The study by Cohen (1977) which is discussed in Sec. VI.F estimates the risk (that is, the expected value of consequence averaged over all scenarios), rather than the consequences of particular scenarios. It is noteworthy that none of the other studies are able to calculate risk in spite of the expressed intention of several to do so. Some of the reasons that risk is so difficult to estimate are discussed in Sec. V.

### 2. High-level waste results

The normalized results of the studies of high-level reprocessing wastes (HLW) are displayed in Fig. 7. The peak individual dose to the critical organ is shown.[8] An error found in the analysis by Logan and Berbano (1978) made it impossible to include this study's results in the figure (see Sec. VI.B.2). The study by Claiborne and Gera is also not included since no explicit calculations were made in that study for release rates into the nearby river. Some studies have two sets of results shown. For Girardi *et al.* (1977), these represent release at $10^3$ or $10^5$ yr. For de Marsily *et al.* (1977), results are shown for two leaching models. One model assumes the waste glass structure remains intact; the other assumes that the glass structure is destroyed at $10^4$ yr after burial. The Swedish results are a conservative and a more realistic estimate of dose from HLW disposal (KBS, 1978a). The TASC results (Berman *et al.*, 1978) are for a salt and a shale repository.

Results are shown for the two different scenarios that were analyzed in the environmental impact statement on management of commercially generated radioactive waste (GEIS, 1980). Both scenarios were considered to be extremely unlikely to occur and were chosen as "worst cases." In Fig. 7 the term GEIS(1) refers to a scenario in which the creation of a 12-m-wide line fault is followed by direct transport to the surface. Results are shown for release at $10^3$ and $10^5$ yr. The term GEIS(2) refers to a scenario in which faulting is followed by *slow* groundwater transport to the biosphere. The results shown are for the worst of the several cases considered: a $10^{-3}$ $\text{yr}^{-1}$ leach rate with faulting occurring at $10^3$ yr.

INFCE(1) refers to a repository in bedded salt where a violent event (termed "incredible") creates a 2400-m-wide fracture joining the aquifer and the repository. The fault appears 50 yr after repository closure (INFCE, 1979b). INFCE(2) locates the repository in hard crystalline rock and the scenario studied is groundwater transport from an unflawed repository (INFCE, 1979a). The bone doses are many orders of magnitude lower than those for INFCE(1) with the less credible scenario. It should be noted that these two studies employ different nuclide transport models.

In general, the peak doses fall into two classes:

- A class centered at about 1% of average yearly background radiation. This is roughly the same additional dose as is received by an individual taking a five-day vacation in the Colorado mountains. The studies in this class calculate, at best, an upper bound to the hazard. Often the calculated dose is increased by unrealistic assumptions which are introduced for various purposes.
- A class centered at about $10^{-5}$ times background. This yearly dose is roughly equivalent to the dose commitment an individual receives by simply drinking a glass of water. The studies in this class represent either attempts to realistically assess the hazard or analyses with a lesser component of deliberate overestimation.

### 3. Spent fuel results

The calculated consequences from the disposal of spent fuel tend to be higher than those from the disposal of HLW from uranium-fueled reactors. This is so because spent fuel contains at least an order of magnitude more of the actinides per unit of power produced.[9] Any action which could be taken to decrease the risk from spent fuel could be taken for HLW (Koplik *et al.*, 1979b).

The spent fuel studies were subjected to the same normalization procedures as the HLW studies. The normalized results are presented in Fig. 8.

The results shown for the KBS study represent a conservative and a more realistic estimate of the expected peak dose. The peak dose occurs at $10^6$ yr after disposal in the conservative case and $7\times10^7$ yr after disposal in the relatively realistic case (KBS, 1978b). The scenarios for which GEIS and INFCE results are shown are discussed above.

Four scenarios are analyzed in the draft WIPP EIS; Fig. 8 shows results from two of these scenarios. For all scenarios, the analysis is restricted to a period of $10^5$ yr. In scenario 2, water from an upper aquifer flows down through two repository shafts, through the repository and back up to the aquifer through a well bore. This is considered to be a highly unlikely but credible event. The results shown reflect upper and lower bound estimates of the consequences of this event. In scenario 4, all the water in the upper aquifer normally moving above

[8]Most of the studies treat HLW produced by reprocessing uranium fuels. Although in the near future almost all fuels will be of this type, the point of reprocessing is to produce plutonium for use in a reactor of some kind. Whether the plutonium fuels a light-water reactor, a fast breeder, or some other type, the resulting wastes will have a higher transuranic content than wastes from uranium fuels (Pigford and Choi, 1976). If doses are dominated by transuranics or their daughters, wastes from mixed oxide fuels will cause somewhat greater doses.

[9]If reactors are fueled with mixed-oxide fuel containing the plutonium produced by reprocessing, the wastes produced in turn by reprocessing the spent mixed-oxide fuel will contain considerably more of the transuranium elements than does HLW from uranium fuels (Pigford and Choi, 1976). As a result, the consequences of disposal of mixed-oxide HLW would be more like those from spent uranium fuel.

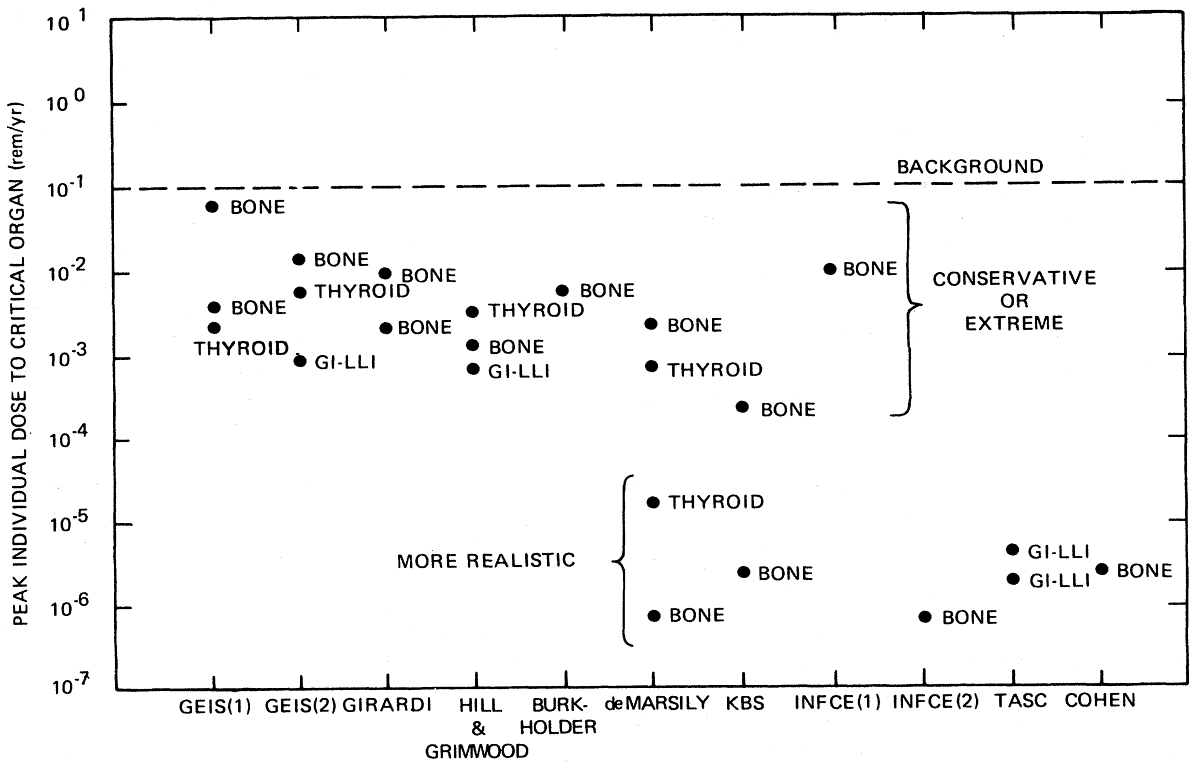


FIG. 7. Normalized peak individual doses for HLW. (The term GI-LLI refers to the gastrointestinal and lower large intestinal tract).

the repository passes through the repository and back to the upper aquifer. This is the worst conceivable groundwater release (WIPP, 1979).

ADL assumes direct advective transport from the repository to the aquifer through a fracture zone or fault. This scenario is chosen as a "worst case." For the comparative analysis, the nuclide concentrations are taken from a point 8 km from the point at which they enter the aquifer. Although this is a relatively short path length, it is the farthest distance for which the study gives nuclide concentrations. The dose to the GI-LLI tract is as high as the other fault scenarios (ADL, 1977).

The scenarios labeled TASC are three of the 34 scenarios analyzed by Giuffre *et al.* (1980). The TASC scenarios all locate the repository in bedded salt. The scenario descriptions are:

- Impermeable salt with failed boreholes [TASC(1)],
- Permeable salt with a downstream shaft [TASC(2)],
- Permeable salt with unsealed boreholes [TASC(3)].

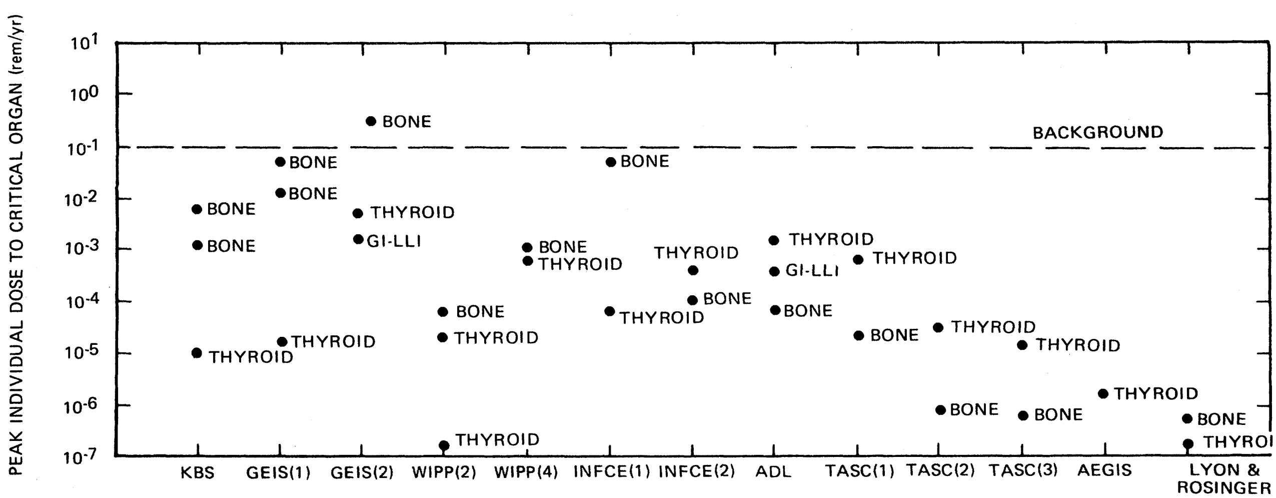


FIG. 8. Normalized peak individual doses for spent fuel.

The effect of the salt's permeability can be seen by comparing TASC(1) and TASC(3). If the salt is permeable, it competes as an exit path for the nuclides, resulting in lower doses.

The work of AEGIS (a U.S. government program called Assessment of Effectiveness of Geologic Isolation Systems) is presented by Raymond *et al.* (1980). The geological and hydrological parameters are taken from the Paradox Basin region (southeastern Utah and southwestern Colorado), making this one of the few studies utilizing site-specific data. The study postulates that a fault 5 ft wide and 10 miles long intersects one room in the repository and links the upper and lower aquifers at the time of repository closure. It should be noted that the fault only affects one of 98 rooms, thereby severely limiting the amount of waste effectively considered in the remainder of the analysis. Lyon and Rosinger (1980) assume release only through unfaulted rock.

The normalized results shown in Fig. 8 do not display the same "conservative, more realistic" patterns seen in the high-level waste studies. In general, the doses are well below background levels. Doses differ from study to study for several reasons. One reason is the different isotopic composition of the waste (e.g., LWR versus CANDU). Another is the time period of the study. ADL calculates nuclide releases out to $10^5$ yr while INFCE(2) identifies a peak dose from $^{226}$Ra at $4\times10^8$ yr (ADL, 1977; INFCE, 1979a). Other studies use time periods between these. The radium release usually peaks somewhere in this $10^5$ to $10^9$ yr time span, and so the cutoff point in time used by the study can affect the peak radium dose.

The normalized results are more easily interpreted by focusing on one organ at a time. Figure 9 shows the thyroid doses. It can be seen that, with a few exceptions, the doses fall within a relatively narrow range (from $10^{-5}$ to $10^{-3}$ rem/yr). This may be explained by the long half-life of $^{129}$I and its meager tendency to be sorbed. For these reasons, iodine is expected to reach the biosphere within the time frame of all the studies with little radioactive decay. The exceptions to this general uniformity are the CANDU reactor waste study by Lyon and Rosinger, the lower bound estimate of consequences for WIPP(2), and the AEGIS study. The low dose seen in the last study is due, in part, to the small fraction of waste affected by the release scenario.

Normalized peak bone doses are shown in Fig. 10. The open circles indicate bone doses for studies in which the time frame of analysis was not limited. The dots indicate bone doses for studies which limit the time period of their calculations. The former results range over three orders of magnitude, from roughly background to 0.1 mrem/yr. The range in results is primarily due to variations among the studies in the retardation of the heavy elements (most importantly $^{226}$Ra and its precursors, although in some studies the $^{237}$Np chain gives a higher dose). Studies with a fixed time period have results which generally fall below 0.1 mrem/yr. The bone dose seen in the AEGIS study is several orders of magnitude below the range shown on the figure because of the small fraction of the inventory affected, the long groundwater transit time, and the time frame of the study.

#### 4. Population dose and health effects

The preceding two sections have focused on the dose which an individual might receive as a result of releases from a waste repository. The dose received by populations is also of interest, particularly since the effects of radiation at very low doses may be treated as additive, as discussed in Sec. VI.E. Several studies have computed population doses by making assumptions on population size and living habits (Logan and Berbano, 1978; Hill and Grimwood, 1978; Berman *et al.*, 1978; GEIS, 1980; ADL, 1977).

The results from the preceding sections can be used to directly calculate normalized yearly population doses for

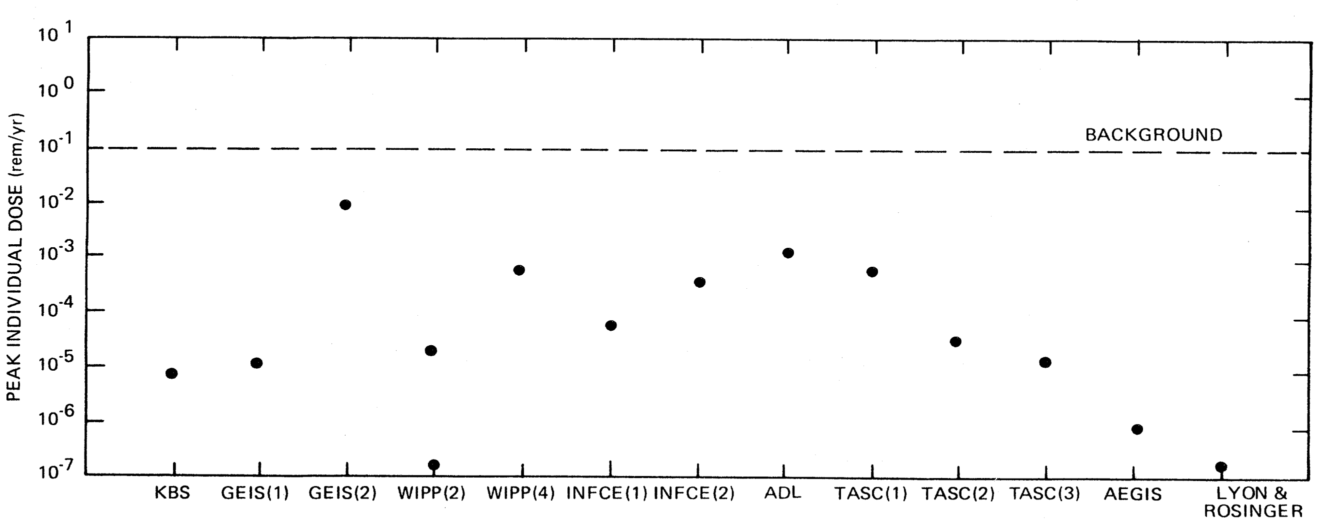


FIG. 9. Normalized peak thyroid doses.

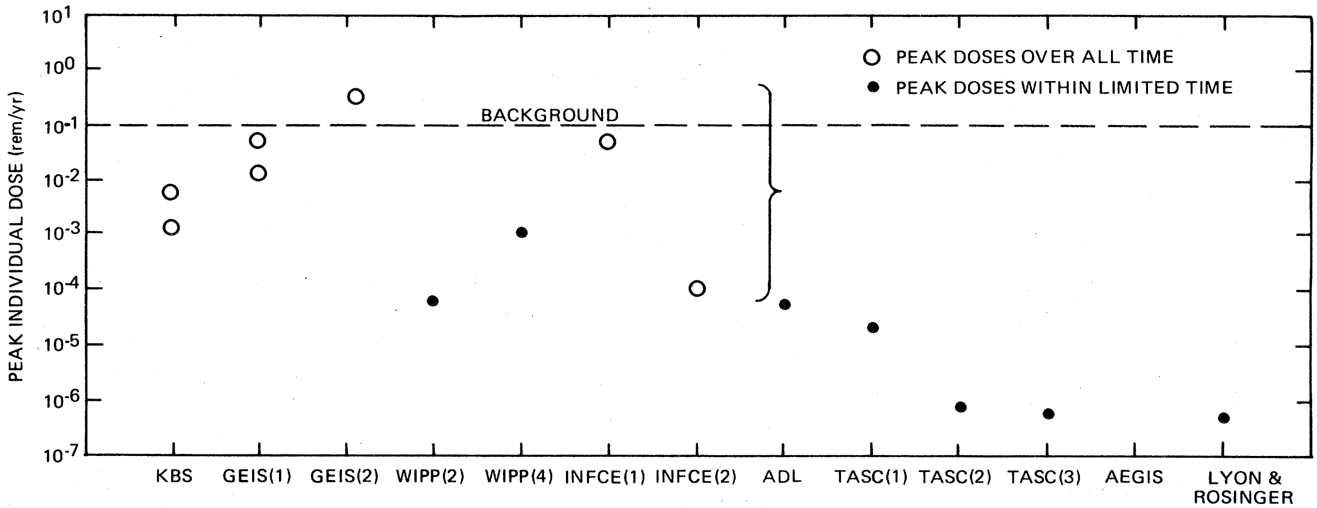


FIG. 10. Normalized peak bone doses.

each study. This is done by assuming a given population at risk. For a river whose flow rate is $10^9$ m$^3$/yr, a population of $10^5$ seems reasonable (alternative assumptions have proportional effects). This leads to predicted yearly population doses ranging from $10^{-1}$ to $10^4$ man-rem/yr for an inventory of $10^3$ GW yr in the various studies.

In addition to consideration of *dose rates,* some studies also emphasize the population dose integrated over time (KBS, 1978b; Cohen, 1980a; Berman *et al.*, 1978; Logan and Berbano, 1978). The meaningfulness of such a measure is debatable, as it involves the summation of very small effects over millions of years. Based on the normalization procedure previously discussed, total doses to a population of $10^5$ from the waste arising from $10^3$ GW yr of power production range from roughly $10^5$ (thyroid) to $5\times10^7$ (bone) man-rem (Koplik *et al.*, 1979b). These values have been integrated over the time period of each study, typically several million years.

Attempts have been made to translate these doses into estimates of health effects. The basis for such calculation is the assumption of a linear no-threshold relationship between dose and effect at low doses. Such an extrapolation is not considered scientifically justifiable (Fabrikant, 1980), but is usually made on the basis that, if not correct, it would overestimate the effects. Those studies which estimate health effects use somewhat different conversion factors, depending on the source relied upon for the estimates. Usually, a factor of about $2\times10^{-4}$ deaths/man-rem of whole body irradiation is used. Smaller factors are used for doses to individual organs. Using a thyroid and bone dose risk factor of $5\times10^{-6}$ deaths/man-rem (ICRP, 1977), health effects may be calculated from the population doses previously given. The results of this computation range from 0.5 to 250 deaths from $10^3$ GW yr over a period of some millions of years. A review of waste disposal studies by Cohen (1980a) produces similar results, i.e., from 1 to 60 deaths from $10^3$ GW yr.

### D. Summary

If extremely unlikely catastrophic scenarios are excepted, the principal hazard from disposal of high-level waste is a release that would slightly elevate the natural levels of radioactivity in groundwater and surface streams neighboring a disposal site. This would, one must assume, cause a small increase in the number of cancers and genetic effects seen in the population using these waters, but the increase would probably not be measurable by the exposed population. The higher levels of radioactivity would persist for probably thousands, perhaps millions, of years.

The range in the numerical values of consequences predicted by different studies if release occurs is quite large. Since the scenarios range from extreme worst cases to attempts at realism, this range in results is not surprising. Most studies predict that the dose to any individual will be extremely small. The higher doses predicted for some scenarios usually depend on very unrealistic assumptions, but they nevertheless serve to emphasize the need for care in disposal of highly radioactive wastes.

## IX. APPRAISAL

### A. Where we are

As the body of work devoted to analyzing the safety of radioactive waste repositories has grown to rather imposing proportions, what can and cannot be learned from present safety assessment techniques has begun to come into focus.

It must be emphasized that safety analysis research has served three different but related purposes. These are:

- To determine if disposal in mined cavities will provide adequate safety,
- To assist in locating suitable repository sites and to provide adequate engineering designs,
- To predict the future behavior of a repository and quantify the degree of risk.

The objective sought by the last of these categories of research has proved to be most elusive. It is clear that our ability to realistically predict the future behavior of a repository is severely constrained by the absence of accurate data describing many important events and processes. The repository site may indeed be chosen because of the absence of such events or processes in its prior history. For many aspects of a repository safety analysis, no realistic data is available at all, and only "conservative" calculations, which attempt to place upper bounds on potential adverse consequences, are possible. The need for more accurate data is obviated if a bounding calculation predicts no significant hazard.

Particularly scarce are reliable models and data from which the probabilities of various scenarios could be calculated. Although research in the earth sciences may produce some additional information of this type, it is most unlikely that it will be possible to calculate the probabilities of any relatively complete set of scenarios. The predictive work in the earth sciences is, in fact, oriented to a more practicable objective, that of identifying the processes which must be evaluated to choose a site and assess its safety. Many of the most important scenarios rest on either conscious actions by future humans or errors in our present understanding of geology; the probabilities of such scenarios are by their nature not objectively quantifiable. As a result, attempts to produce a "risk curve" showing probability as a function of the level of consequences are unlikely to yield much more than a careful description of someone's personal opinion.

Most studies to date have attempted to evade the shortage of data by hypothesizing "generic" sites having typical characteristics. Such work has been useful to aid in the development of analysis methods and to provide information bearing on whether safe disposal of radioactive wastes is possible at all. At this point, however, it seems that little more can be learned about disposal safety from such generic studies; a better understanding will require further examination of individual sites.

### B. What has been learned

Calculations of releases of radioactivity from a repository (other than those assuming either extraordinarily improbable catastrophes or intrusion by humans) have yielded doses to individuals ranging from near background to many orders of magnitude less. This is true even of analyses which assume unlikely failures of elements of the containment system. In view of the atmosphere of overestimation which pervades these studies (especially those giving results near background), it seems safe to conclude that a carefully sited and designed repository is quite unlikely to expose anyone but an intruder to more than a very small dose of radioactivity. To be sure, these small doses might be received by large populations over long periods of time, possibly resulting in from one to several hundred deaths over a period of several million years. Whether the possibility of such population exposures is acceptable to society is beyond the scope of this paper.

One thing that has not emerged from the work to date is any clear evidence for the superiority of any type of rock over any other as a medium in which to place a repository. Salt and granite have been studied most, and each of these has advantages and disadvantages with respect to the other. The few studies of shale, clay, and basalt have been no more decisive.

Studies of repository safety rely heavily on mathematical models to provide predictions of future behavior. Because some of the phenomena involved are not well understood, the concept of mathematical prediction of repository behavior has been questioned (Varanini, 1979; Cohen, 1980b). The method most commonly used to model the poorly understood phenomena is to make bounding assumptions. Cohen (1977; 1980a) employs physical analogs in another effort to avoid these difficulties. His work tends to confirm the general impression that even those detailed modeling studies which predict only small doses overestimate the likely consequences. The detailed models are valuable in delineating what information is necessary for selecting repository sites and designing containment systems.

### C. Where we go from here

The lessons of past studies can be used to direct future work aimed at better understanding the dangers of waste repositories.

The major change in direction from past to future work should be a shift from generic to site-specific studies. Within these studies, an attempt to model more of the barriers in the repository system would be desirable. Simple models which capture the essentials of each barrier's behavior will probably suffice. Even a very crude model which underestimates a barrier's contribution to safety will yield better results than would be obtained by ignoring the barrier altogether. At the same time, it may be possible, without significant loss of accuracy, to simplify some of the very complex models that have been used in such areas as contaminant transport in groundwater.

Within this framework, basic research should be performed on some of the less well understood phenomena which figure prominently in the safety analyses. These include leaching, sorption, and flow in fractured media, each of which may be affected by interactions with other components of the repository system.

The above discussion concentrates on improving the estimation of consequences; a safety analysis must also address the scenarios which may occur. These will vary in importance from site to site. Those who attempt safety studies would be well advised to avoid attempting to comprehensively quantify probabilities. There are a variety of better approaches to the issue of probability, including worst-case analysis and consequence analysis of a range of scenarios coupled with descriptive discussion of the scenarios' probabilities.

Perhaps the principal problem which remains is to obtain sufficient data at specific sites to allow an analysis of safety. In order to facilitate this, it is often suggested that sites should be located in as geologically simple an area as possible. Nevertheless, obtaining the desired information will not be a simple task and may require innovative techniques in exploratory geology.

Past work has delineated the potential and limitations of repository safety analysis, and has indicated roughly the magnitude of the danger to be expected. The principal task remaining is to apply the methods and results of this field, both those already developed and those yet to come, to the analysis of the particular sites at which the construction of repositories will be proposed.

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[10]Most of the technical reports listed here may be purchased from the National Technical Information Service. Kärnbränslesäkerhet reports are available from Box 5864, 10248 Stockholm, Sweden or (in microfiche) from the INIS clearinghouse. Electric Power Research Institute reports are available from Box 50490, Palo Alto, California 94303.


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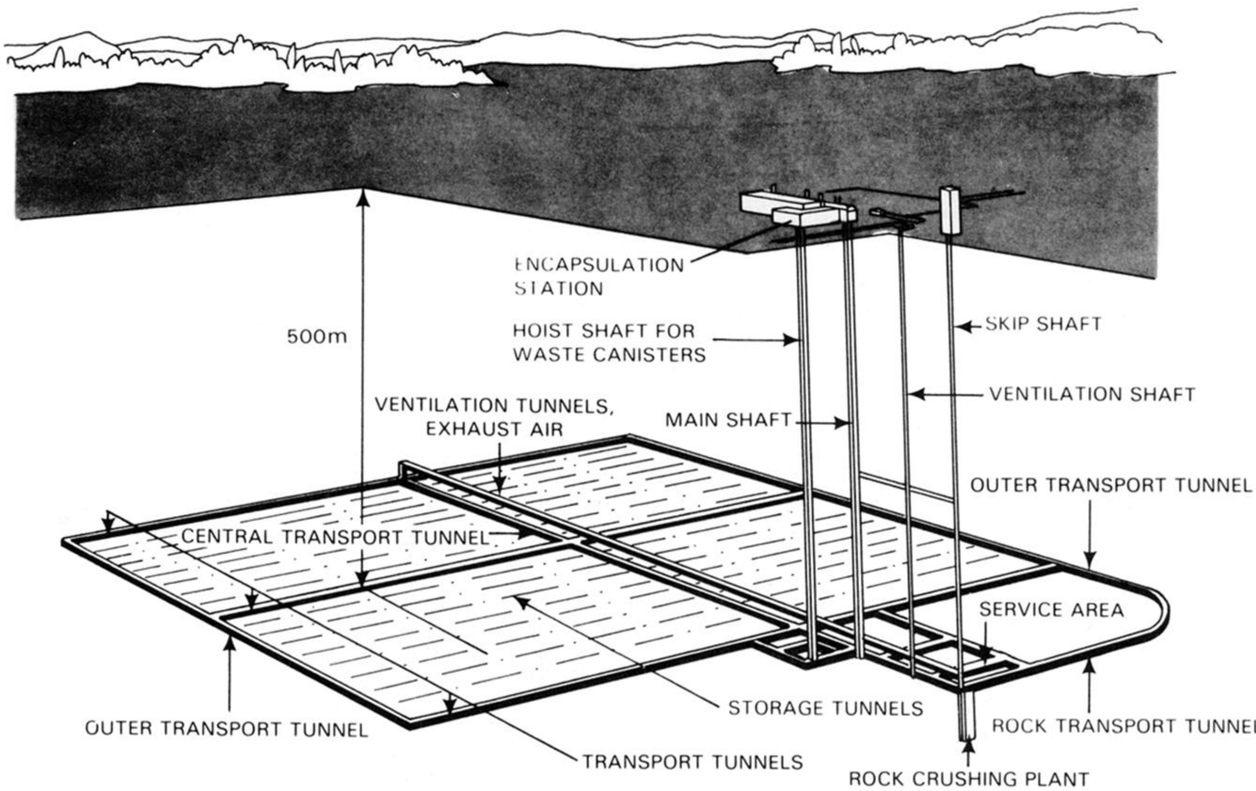


FIG. 4. Artist's conception of a nuclear waste repository (adapted from KBS, 1978b).

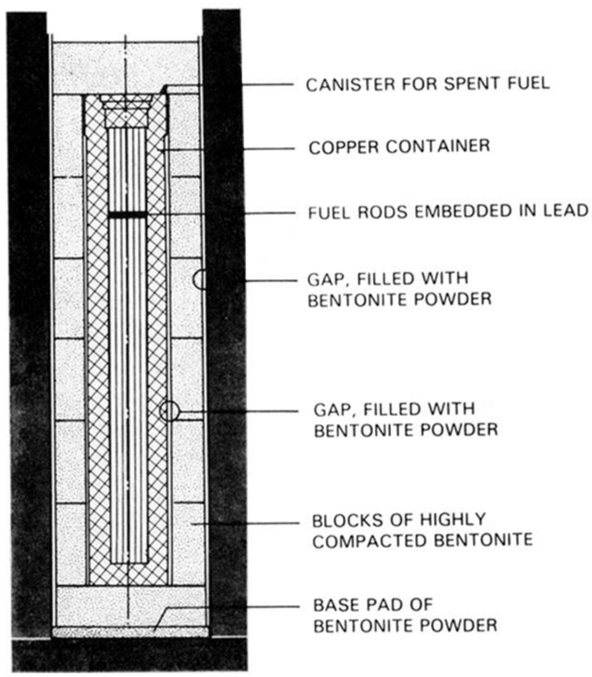


FIG. 5. Waste package and storage hole (KBS, 1978b).