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Comparing Risks from Low-Level Radioactive Waste Disposal on Land and in the Ocean: A Review of Agreements/Statutes, Scenarios, Processing/Packaging/Disposal Technologies, Models, and Decision Analysis Methods

COMPARING RISKS FROM LOW-LEVEL RADIOACTIVE WASTE DISPOSAL ON LAND AND IN THE OCEAN: A REVIEW OF AGREEMENTS/STATUS,SCENARIOS, PROCESSING/PACKAGING/DISPOSAL TECHNOLOGIES, MODELS, AND DECISION ANALYSIS METHODS

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Prepared for the Office of Solid Waste and Emergency Response
Office of Radiation Programs, U. S. Environmental Protection Agency, Washington, DC

BIOMEDICAL AND ENVIRONMENTAL ASSESSMENT GROUP
DEPARTMENT OF APPLIED SCIENCE
BROOKHAVEN NATIONAL LABORATORY
ASSOCIATED UNIVERSITIES, INC.

Under Contract No. DE-AC02-76CH00016 with the
U. S. Department of Energy

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ABSTRACT

Large volumes of low-level radioactive waste (LLW) are either in temporary storage or unexcavated from Superfund and U.S. Department of Energy Defense Facility sites. This waste must be isolated from the public for very long periods of time because of its long half-life. Because of the need to manage this waste, questions have been raised about the hazards associated with the disposal of these materials on land or in the ocean. Similarly, the U.S. as a signatory to the London Dumping Convention is now engaged in international discussion about the comparative risk of disposing of LLW on land and in the ocean. In support of these national and international activities, the U.S. Environmental Protection Agency is examining scientific and technical questions related to the comparative health and environmental risks of ocean- and land-based disposal of LLW.

In support of these efforts, this report gives background information on: The history of LLW disposal in the U.S.; agreements, statutes and regulations for the disposal of LLW; disposal scenarios and alternative treatment options for LLW; methods and models which could be used to assess and compare risks associated with land and ocean options for LLW disposal; technical and methodological issues associated with comparing risks of different options; and, roles of decision making approaches in comparing risks across media.

In any assessment effort, the minimal set of health risks to be examined should include routine and accidental radiation exposures to workers and the public from transport (land and ocean), disposal, and post-closure releases of radionuclides. Potentially important contributors to health and environmental risks which have not been fully evaluated include waste processing and event probabilities. Since waste processing, packaging and disposal requirements for land and ocean disposal can differ, their contribution to the overall risk can vary by medium. Human and natural processes that precipitate accident-initiated releases are also important determinants of risk and must be evaluated. Ideally, models selected for analysis should have completed some validation exercises and be capable of producing explicit estimates of model uncertainty. In practice, however, most models have not undergone complete validation efforts and are not now able to produce explicit estimates of output uncertainty.

It is clear, that the general public is concerned about a much broader range of characteristics and potential risks of the LLW disposal options than just health risks. These especially include socio-economic risks and the equity of the distribution of risks and benefits in space and time. Most LLW risk assessment models do not normally include these important considerations, and few of the models reviewed could be modified to include any of them with modest effort. This is a serious deficiency in current capabilities that will undermine the credibility of results in the eyes of the general public. When this information becomes available, there are decision aiding methods for incorporating a broad range of characteristics, potential risks, and stakeholder values into quantitative comparisons of alternatives. Development of necessary data and application of these approaches can permit decision-makers to make better and more informed decisions.

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1 INTRODUCTION

Large volumes of low-level radioactive waste (LLW) are either in temporary storage or unexcavated from Superfund and U.S. Department of Energy Defense Facility sites. This waste must be isolated from the public for very long periods of time because of its long half-life. Because of the need to manage this waste, questions have been raised about the hazards associated with the disposal of this material on land or in the ocean. Similarly, the U.S. as a signatory to the London Dumping Convention is now engaged in international discussion about the comparative risk of disposing of LLW on land and in the ocean. In support of these national and international activities, the U.S. Environmental Protection Agency (EPA) is examining scientific and technical questions related to the comparative health and environmental risks of the ocean- and land-based disposal of LLW. In support of these efforts, this report gives background information on technical and methodological issues associated with comparing risks among the different options. Results presented are based on an exploratory effort to identify the current state-of-knowledge and important existing gaps which should be evaluated before comprehensive and equitable multimedia risk assessments are prepared. In this context, this report includes:

(1) A brief review of the history of LLW disposal in the U.S.

(2) A review of agreements, statutes and regulations for the disposal of LLW.

(3) An identification of disposal scenarios and alternative treatment options for LLW.

(4) An identification, characterization and analysis of methods and models which could be used to assess and compare risks associated with land based and ocean options for LLW disposal.

(5) Role of decision making approaches in comparing risks across media.

The information presented in this report on models for risk assessment purposes is based on model documentation, publications, and previous experiences of the authors. Time and resources allocated for this effort did not permit full evaluation of the computer codes, nor were test runs of the models made.

2 HISTORICAL BACKGROUND[a]

LLW is generated during the routine operation of nuclear power plants; from biomedical applications such as nuclear medicine therapy and diagnostic techniques at hospitals and medical facilities; scientific research at universities and other research and development institutions; industrial isotope production; and from defense-related operations. It usually contains small amounts of radioactive materials mixed in larger volumes of nonradioactive materials.

During the early days of research using radioactive materials (from just after World War I to World War II), LLW was either burned, buried in shallow trenches, or diluted and released to sewer systems at the point of generation. Limited knowledge of the health effects of radiation and environmental transport of pollutants contributed to the lack of attention paid to waste disposal issues. Beginning with the Manhattan Project to develop nuclear weapons, the volume of radioactive materials and diversity of isotopes generated as waste by-products grew tremendously. As a result, the Atomic Energy Commission (AEC) created new disposal sites at federal facilities located in Hanford, Washington; Idaho Falls, Idaho; Los Alamos, New Mexico; Oak Ridge, Tennessee; Savannah River, Georgia; and at the Nevada Test Site, Nevada. Disposal techniques consisted mainly of shallow land burial in which long, shallow trenches were dug, filled with waste, and backfilled with soil. These facilities were primarily developed to handle defense-related wastes, but as the nuclear power industry began to expand, commercial wastes were also accepted at some sites. This continued until 1962 when the AEC licensed private companies to operate LLW disposal sites. The first commercial sites licensed were at Beatty, Nevada, and shortly thereafter at Maxey Flats, Kentucky.

In addition to the expanded use of shallow-land burial techniques, the AEC initiated disposal of LLW at sea in 1946. Originally carried out by the U.S. Navy, waste containers (55-gallon drums) were filled with LLW materials and with concrete to provide a cap and increase density to ensure sinking; approximately 107,000 waste containers comprising 4×10^{15} Becquerels (Bq) were placed into the Atlantic and Pacific Oceans. Ocean disposal was

a Information presented in this chapter is extracted from Burns and Briner, 1980; Holcomb, 1982; and Parker, 1988.

extensively pursued until 1962, when the introduction of commercial land-based disposal facilities made it economically unfeasible; operations were completely curtailed in 1970. European countries have also used the ocean for waste disposal. Since 1949, they have disposed of an estimated total mass of 100,000 metric tons containing 3 x 10^{16} Bq in the Northwest Atlantic Ocean.

In the U.S., LLW is currently being disposed of on land only. In 1982, over 75,000 m^3 of LLW was disposed of in commercially licensed shallow-land burial disposal sites. Six such sites have been in operation. In addition to Beatty and Maxey Flats, they include: Barnwell, South Carolina; Hanford, Washington; Sheffield, Illinois; and West Valley, New York. Maxey Flats, West Valley and Sheffield are now closed - the first two due to technical and environmental problems and the latter because it was filled to capacity. Because the number of commercial sites is so limited and the volume of LLW has increased greatly in the last decade, there is a need for new commercial sites. In addition, dwindling disposal capacity at existing sites has made existing host states increasingly reluctant to accept wastes generated outside their own borders.

3 REGULATORY BACKGROUND

Since current LLW disposal practices date back more than 40 years, both disposal techniques and concerns over environmental effects have varied considerably. Consequently, laws and regulations governing the disposal of LLW have also changed. Consideration of current and pending legislative constraints may play a key role in the selection of new LLW disposal options. Applicable international agreements and federal statutes/regulations for ocean dumping, as well as federal and state statutes/regulations governing land-based disposal are summarized in this section. A more complete review of agreements, statutes and regulations for the disposal of LLW is included in Appendix A.

Since the onset of ocean disposal in 1946, growing worldwide concern for preserving the condition of the seas and newly acquired knowledge on the effects of ionizing radiation on human health and safety have led to a dynamic regulatory environment. International concern was initially focused at the first U.N. Conference on the Law of the Sea (UNCLOS I), held in Geneva in 1958. This conference enacted the first international law of the sea, which is still in force today. In 1972 the London Dumping Convention (LDC) developed the most comprehensive set of international regulations for marine pollution by dumping. The requlations were ratified by some 50 countries, including the major maritime nations. Due to concern over uncertainties in environmental impacts, and widespread social protests, ocean disposal was temporarily suspended in 1983. Resolution (28):10 to the LDC, approved in 1986, called for a voluntary moratorium on the use of the ocean for disposal of LLW until certain scientific and technical matters, as well as social, political and economic issues are addressed. An LDC sponsored Intergovernmental Panel of Experts on Radioactive Waste Disposal at Sea has established two working groups that are currently investigating these issues. Figure 1 traces significant international milestones and agreements for disposal of LLW at sea.

In addition to its support of international ocean disposal agreements, the U.S. has developed a parallel set of laws and regulations. Following passage of the National Environmental Policy Act (NEPA) in 1970 which set forth general environmental policy, Congress enacted a series of specific environmental laws including the Marine Protection, Research and Sanctuaries

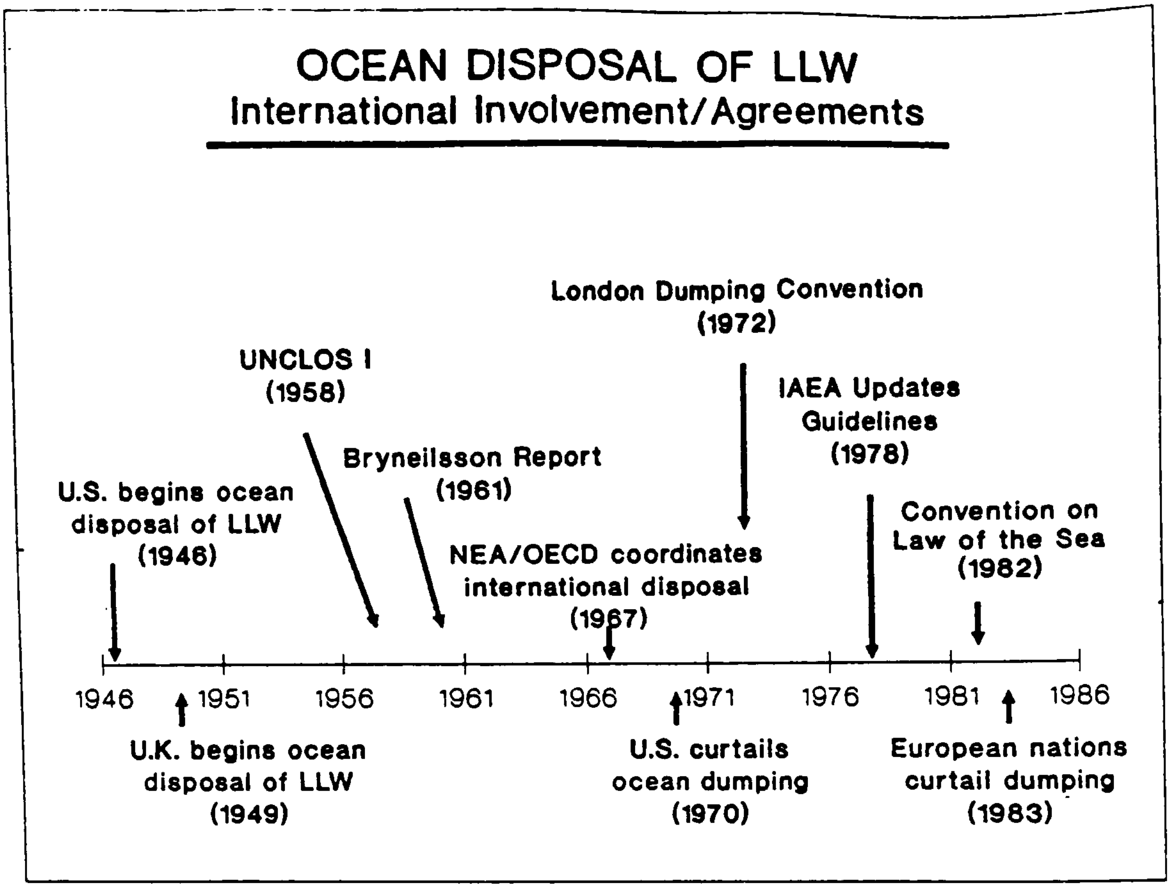


Figure 1. Time-line indicating significant international milestones and agreements for ocean disposal of low-level radioactive waste.

Act (MPRSA) of 1972. This act empowered EPA to oversee adherence to the policy guidelines of the MPRSA and issue regulatory criteria for ocean disposal permits. In January 1983, an amendment to MPRSA imposed a two-year moratorium on ocean disposal of LLW and a more stringent, supplementary set of permit requirements following the moratorium. For a LLW ocean disposal permit, it must be demonstrated that:

- the proposed dumping is necessary to conduct research either i) on new technology related to ocean dumping, or ii) to determine the degree to which the dumping of such substance will degrade the marine environment;
- the scale of the proposed dumping is limited to the smallest amount of such material and the shortest duration of time that is necessary to fulfill the purposes of the research, such that the dumping will have minimal adverse impact upon human health, welfare, and amenities, and the marine environment, ecological systems, economic potentialities, and other legitimate uses;
- after consultation with the Secretary of Commerce, the potential benefits of such research will outweigh any such adverse impact; and
- the proposed dumping will be preceded by appropriate baseline monitoring studies of the proposed dump site and its surrounding environment.

Other requirements under the 1983 amendment include submission of a Radioactive Material Disposal Impact Assessment and final approval by a joint resolution of Congress. Past U.S. ocean disposal practices and U.S. laws and regulations governing such practices are highlighted in Figure 2.

Policy and regulatory responsibilities for land-based disposal of LLW are covered by the Atomic Energy Act of 1954 (AEA) and the LLRWPA of 1980 and their amendments. The former establishes the Nuclear Regulatory Commission (NRC) as the licensing and regulatory body for commercial land-based radioactive waste disposal and provides states regulatory power under authority granted by NRC. Defense-related LLW generated by federal facilities and their contractors are the responsibility of the U.S.

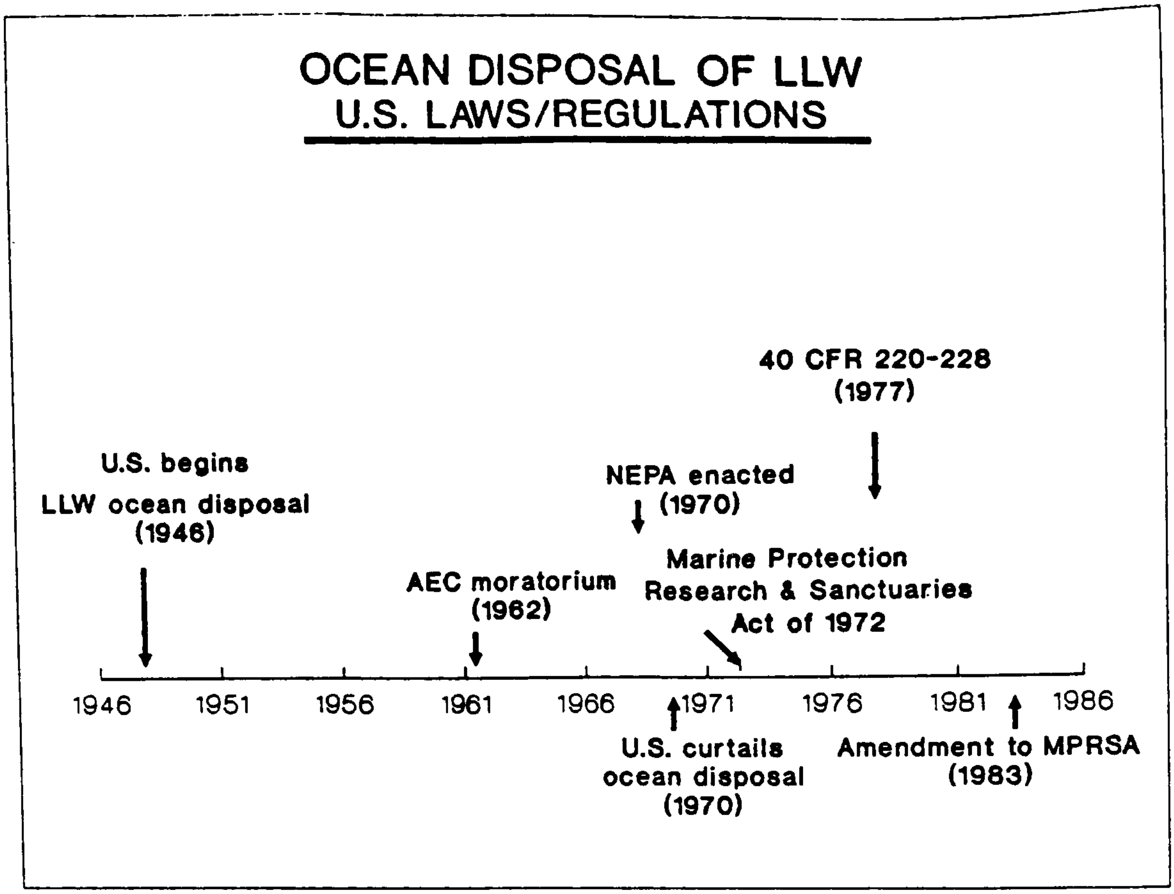


Figure 2. Time-line highlights past U.S. ocean disposal practices and U.S laws/regulations governing such practices.

Department of Energy (DOE). EPA is responsible for protecting public health and safety resulting from exposure to radiation from either commercial or defense-related disposal operations. NRC has issued a comprehensive regulation [NRC, 1982] governing licensing procedures, performance objectives, and technical requirements for land-based LLW disposal. DOE has promulgated an internal order [U.S. Department of Energy, 1988] covering disposal operations under its jurisdiction. EPA has issued an Advanced Notice of Proposed Rulemaking for LLW disposal standards that would establish allowable exposure limits, define a "below regulatory concern - BRC" waste classification, and set groundwater protection standards.[U.S. Environmental Protection Agency. 1983] Recognizing that the combination of these factors could lead to a severe disposal crisis in the near future, Congress mandated that states assume responsibility for disposal of their own LLW. The Low-Level Radioactive Waste Policy Act of 1980 (LLRWPA) as amended in 1985 [U.S. Congress, 1980] required that individual states or groups of states that form specific compacts must develop new disposal sites for their LLW by 1993 or face stiff financial and other penalties. In spite of these Congressional efforts, progress towards implementation of the LLRWPA has been slow. The primary reasons for the 1985 amendments to the LLRWPA were to extend the original deadline for implementation by five years, provide incentives for those states that do comply, and penalties for those that do not. Siting of new disposal facilities continues to be a serious problem for the states and regional compacts. In addition to the difficulties of locating potential areas that meet technical and environmental criteria (e.g., climate, hydrology, demographics, location), public acceptance of waste disposal sites within local communities has been hard to attain.

4 RISK ASSESSMENT ISSUES

4.1 Nature of the Problem

The objective of risk assessment modeling is to predict future quantities or concentrations of radioactivity in different environmental media, and to estimate from these the final dose to man from which health hazards can be calculated. The process of assessing the risks of different LLW disposal options can be divided into two basic steps:

(1) Scenario characterization;

(2) Consequence modeling.

The following sections expand on these steps.

4.2 Scenario Characterization

The first task establishes the conceptual bounds of the analysis. It involves identification and quantification of phenomena which could initiate release of radionuclides and or influence rates at which releases occur. In developing risk estimates of LLW disposal options, careful consideration must be given to setting technical and natural boundaries of the system to be characterized. If the boundaries are too small, important contributors to risk may be missed. Similarly, if the boundaries are too large, excessive time and effort could be spent evaluating issues of only secondary importance. Several major issues associated with the setting of appropriate boundaries for analysis are discussed below.

4.2.1 Scenario Definition

In scenario definition, identification of the activities of interest is of primary concern. Activities are defined here as a set of actions associated with the handling, processing, transport and disposal of waste. In this context, boundaries for analysis of risk generally must include all activities ranging from waste generation to the post-closure stage at the disposal site. To the extent that these options differ among the alternatives, a complete life-cycle analysis is essential if risk estimates are to be comparable.

Events may occur for each activity that could result in the release of radioactive materials. Table 1 gives a list of sample phenomena which are potentially relevant to scenario analysis for land disposal of LLW. These phenomena or events can be divided into three major categories; human activities (e.g., construction, drilling for mineral resources), natural processes (e.g., erosion and flooding), and waste and disposal site processes (gas generation or mechanical disturbance of soils or rocks at the disposal site.[International Atomic Energy Agency, 1984]

Sets of combined activities and events (phenomena) that could contribute to release of radionuclides from a disposal system and result in human exposure can be defined as a scenario. Although system characterization is very important to developing conceptual models for risk assessment, scenario development is equally important for characterizing the different hazards associated with the disposal operation as well as for showing regulatory compliance.

Selection of appropriate time scales is important because of the long half-lives of many radionuclides as well as the time scales in which various events may occur. Time scales of relevance to the LLW disposal option include:

(1) The assumed duration of dumping.

(2) The half-lives of the radioisotopes.

(3) The time-scales associated with physical transport processes.

(4) The time-scales associated with biological processes.

(5) The time-scales over which events occur.

(6) The integration times for assessing dose to man, including both annual and lifetime.

Selection of appropriate spatial resolution for analysis should be based on such important considerations as the media into which materials are released and the time-scales of interest. The geographic scales of interest for routine releases from the ocean disposal option will be orders of magnitude larger than that of the land-based option. For accidents, however,

Table 1. Phenomena potentially relevant to scenario analysis for shallow ground repositories [International Atomic Energy Agency, 1984a].

HUMAN ACTIVITIES

Improper design and operation

- Chemical liquid waste disposal
- Draining system obstruction
- Improper waste equipment
- Top cover failure

Future intrusion

- Construction activities
- Farming
- Groundwater exploitation
- Habitation
- Salvage
- Re-use of disposed material
- Archaeology

NATURAL PROCESSES AND EVENTS

Biological intrusion

- Animals
- Plants

Faulting seismicity

Fluid interactions

- Erosion
- Flooding
- Fluctuations in the water-table
- Groundwater flow
- Seepage water

Weathering

- Deterioration with time
- Freezing/thawing
- Wetting/drying

WASTE AND REPOSITORY PROCESSES

- Gas generation
- Waste and soil compaction
- Waste soil interaction

the differences may not always be as pronounced, especially if releases to the atmosphere from the land-based option must be modeled.

Careful consideration must be given to modeling exposures via different pathways (see Section 4.5) to critical population groups (CPG) and to the committed dose. In each case, such estimates must be prepared for public and occupational health impact assessment. Finally, occupational health impacts arising from non-radiologic events must also be evaluated.

4.2.2 Inventory/Source Term

LLW is defined as radioactive waste containing source materials, special nuclear materials, or by-product materials that are acceptable for disposal in an NRC licensed land facility.[U.S. Nuclear Regulatory Commission, 1982] Specifically excluded in this definition are high-level waste, spent nuclear fuel, by-product material specified as uranium or thorium tailings and waste, and transuranic (TRU) wastes [waste material contaminated with alpha-emitting radionuclides with atomic numbers greater than 92 and half-lives greater than 20 years in concentrations exceeding 100 nCi/g (3700 Bq/g)]. However, retrievable inventories of TRU stored since 1970 are currently being sampled to characterize these wastes. It is estimated that 38 percent (35,800 m^3) may be designated as LLW.[Oak Ridge National Laboratory, 1988]

Commercial nuclear power plants generate about 50% by volume of the waste shipped to commercial disposal sites; another 10% is produced by commercial nuclear fuel processors (UF_6 conversion, uranium enrichment and fuel fabrication plants). Routine wastes consist of spent resins, evaporator bottoms, filter sludges, dry compressible waste and contaminated plant hardware. Periodic disposal of small amounts of high-activity irradiated components is considered non-routine waste. Utilities account for about 50% of NRC Class A commercial LLW and about 90% of the commercial Class B and Class C waste.

The remainder of commercial LLW is produced by industry and institutions, including research laboratories, hospitals and medical laboratories, non-DOE governmental facilities, and universities. Industrial and institutional wastes may be biomedical (animal carcasses, tissue samples) or non-biological (compacted trash, absorbed liquids, contaminated hardware).

More than 17,000 licenses authorizing handling and use of radioactive materials have been issued by the NRC or by Agreement States.

Low-level waste is also generated by DOE facilities through uranium enrichment operations, the naval nuclear propulsion program, and defense and research and development activities. By volume, DOE LLW consists mainly of dry solids and decontamination debris; it is disposed of at DOE sites.

Tables 2 and 3 show volumes and activity of LLW from the commercial and DOE sectors from 1984-1987. Table 4 lists the principal radionuclides in the waste produced by each of these sectors.[EG&G, 1987; Oak Ridge National Laboratory, 1988]

LLW also results from environmental remedial action projects. Programs at DOE sites include the Formerly Utilized Sites Remedial Action Project (FUSRAP) and the DOE Environmental Restoration (ER) and Defense Decontamination and Decommissioning (D&D) Programs. Former DOE sites now classified and administered as civilian projects are the responsibility of the Surplus Facilities Management Project (SFMP). Projected volumes and characteristics of waste from these activities are currently being estimated. In general they are expected to be high volume and low specific activity LLW.

4.2.3 Waste Processing Technology

Both physical and chemical properties of the waste are important criteria used in the selection of appropriate waste processing technology. Basic physical properties of the as-generated waste stream (e.g., solid vs. liquid) influence the decision on which treatment option is chosen and the stability of the waste after disposal. For example, combustible solids that are incinerated with resultant ash solidified in cement or other binder will be more stable than similar waste that is simply compacted and shipped for disposal. Chemical properties also affect selection of treatment options (e.g., compatibility of waste and binder materials) and can have a significant impact on disposal performance parameters including leachability of radionuclides from the waste form and sorption characteristics of the trench, unsaturated and saturated zone soils. Chemical composition and ionic form can affect waste solubility and precipitation reactions in groundwater. Presence of chelating agents used in cleaning and decontamination processes can alter sorption interactions between waste and soil. Volume reduction

Table 2. Quantities (m^3) of LLW generated in the U. S. by source type, 1984-1987 [Oak Ridge National Laboratory, 1988].

Source	1984	1985	1986	1987
Commercial	69374	75909	51112	52233
Utility	42787	43260	29301	26602
Institutional	4398	4712	3780	5187
Industrial	22189	27937	18031	20444
DOE/Defense	90600	121200	97000	99500

Table 3. Activity levels (10^3 Ci)[1] of LLW generated in the U.S. by source type, 1984-1987 [Oak Ridge National Laboratory, 1988].

Source	1984	1985	1986	1987
Commercial	600.9	749.0	233.7	269.6
Utility	441.3	582.5	170.5	219.8
Institutional	--	8.1	5.0	7.1
Industrial	--	158.3	58.2	42.7
DOE/Defense	2053.0	1009.0	772.0	2750.0

[1] 1 Ci = 3.7×10^{10} Bq.

Table 4. Typical radionuclides in low-level waste [Oak Ridge National Laboratory, 1988].

Source	Radionuclides*
Commercial Utility	Co-58, Cr-51, Mn-54, Cs-134, Zn-65, Cs-137, Co-60, H-3, Ni-63, Fe-55, I-131
Institutional	H-3, C-14, I-125, P-32, S-35, I-131, Cs-137, B-137m, Cr-51, U-238, Co-60, Mo-90, Fe-55, Ir-192, Na-22
Industrial	H-3, P-32, Cs-137, Ba-137m, S-35, Co-60, U-238, Th-232, Ta-182, C-14, Ir-192, Sr-90, Y-90
DOE/Defense	U-238, Th-234, Pa-234m, Pu-241, Co-58, Mn-54, Cs-137, Ba-137m, Ce-144, Pr-144

*Listed in order by concentration or contribution to total activity.

Sources: ORNL 1988, EG&G 1987.

techniques can affect physical and chemical composition of the waste and increase activity concentrations to the extent that waste classifications may be subject to change. For example, large volumes of low concentration contaminated liquids can be reduced by evaporation leaving a more highly concentrated sludge residue (evaporator bottoms). The advantages inherent in volume reduction must be weighed against potential waste-binder compatibility problems and increased exposures associated with higher radioactivity levels, introduced by the secondary waste stream.

Characteristics of treated waste must also be considered in the performance assessment. For example, if wastes are encapsulated in a solid matrix, does the process physically bind the waste (as in thermoplastic materials such as bitumen or thermosetting polymers such as vinyl-ester styrene); or is a chemical bond formed (as in the hydration reaction of cement) that reduces radionuclide mobility? Leaching properties of cementitious waste forms are highly isotope- and species-dependent, whereas those of solidification materials such as bitumen are not. For those isotopes that are chemically bound, to what extent and under what conditions are the reactions reversible? In addition to decreasing radionuclide mobility, solidification of LLW improves disposal site stability by reducing the potential for slumping of backfill materials caused by biodegradation. Thus, solidified waste form performance is an important parameter in projecting potential health impacts from land disposal of LLW. Properties that can impact waste form structural stability include compressive strength, and resistance to freeze-thaw cycling and biodegradation damage. Treated and/or solidified waste is generally contained in a waste package prior to disposal. Package behavior (i.e., corrosion resistance) can also affect overall disposal site performance. However, the most common packaging material (mild steel drums) corrode relatively quickly, and consequently some assessments do not take credit for the expected useful life of packaging containers.

Many types of LLW treatment and packaging options are currently in use. Potential treatment options for primary waste streams that are either currently in use or are being developed are summarized in Table 5.[Triglio, 1981] Solidification and packaging options for primary and secondary waste streams are included in Table 6.

Table 5. Potential LLW treatment options [Triglio, 1981].

Primary waste stream	Treatment process	Secondary waste stream product
SOLIDS		
Combustibles:		
	Controlled air incineration	non-combustible, highly refractory oxide
	Cyclone incineration	non-combustible, highly refractory oxide
	Fluidized-bed incineration	non-combustible, refractory oxide
	Rotary kiln incineration	non-combustible, refractory oxide
	Agitated hearth incineration	non-combustible, refractory oxide
	Controlled pyrolysis	non-combustible, refractory oxide
	Molten salt combustion	non-combustible salt-ash or an oxide if salt is leached
	Acid digestion	non-combustible sulfates and oxides
Non-Combustibles:		
	Compaction	volume reduced solids
	Shredding	volume reduced solids
	Melting-casting (discarded equipment)	radioactive ingot and process slag
	Dissolution	acidic slurries
	Decontamination (contaminated equipment)	decon solutions (e.g., alkaline permanganate, mineral and organic acids, detergents, and chelates)
LIQUIDS		
	Filtration	sludge, spent filtration media (e.g., sand, diatomaceous earth, carbon, cartridges, pre-coat cartridges)
	Centrifugation	concentrated sludge
	Ion exchange	spent ion exchange resins (powdered or bead)
	Membrane technology	moderately concentrated blowdown effluents
	Evaporation	highly concentrated condensate slurries
	Calcination	dry solid, non-fused oxides

Table 6. Potential LLW solidification and packaging alternatives.

Technology/Material	Description
Solidification:	
Cement	hydration reaction with aqueous waste forms solid matrix
Bitumen	thermoplastic polymer; melted, mixed with waste to form homogeneous mixture; cooled to form solid
Polymer Concrete (e.g., vinyl-ester styrene)	liquid monomer emulsified with waste, polymerized by chemical initiators to form solid
Glass	high temperature process to calcine and/or incinerate waste and incorporate in glass matrix (currently R & D for HLW only)
Polymer-modified Gypsum Cement	hydrated modified $CaSO_4$ cement mixed with waste to form solid
Polyethylene	thermoplastic polymer; melted, mixed with waste to from homogeneous mixture; cooled to form solid
Packaging:	
Mild steel	55 gal. drums or large volume liners; suitable for solidified or compacted waste; limited expected life (<50 yrs)
Wood/cardboard boxes	Suitable only for low activity (Class A) dry solid waste; biodegradable, subject to rapid degradation in soil
High-integrity containers	Specially designed alloy or polymeric containers as per NRC specifications; minimum lifetime design goal of 300 yrs

4.2.4 Waste Disposal Options - Land

Passage of the LLRWPAA, implementation of NRC's disposal site performance criteria and the development of new regional and state disposal facilities have created interest in alternative [(i.e., other than conventional shallow land burial (SLB)] disposal technologies.[U.S. Nuclear Regulatory Commission, 1982; U.S. Congress, 1986] Since many regional compacts and individual states have already banned the use of SLB, it is clear that alternative disposal methods will play a major role in the planning of future land-based disposal of LLW. The ability to address alternative disposal options is therefore an important element in selecting a performance assessment model.

The LLRWPAA required the NRC, in consultation with the States and other interested parties, to identify methods for the disposal of LLW other than SLB (Table 7), and to establish and publish technical guidance regarding licensing of facilities that use such methods. In their Branch Technical Position Statement on Licensing Alternative Methods of Disposal for LLW, NRC defined alternative disposal methods as "disposal facility designs or disposal concepts which incorporate engineered barriers or structures, or otherwise differ from the past and present methods of near-surface land disposal of LLW by shallow land burial."[U.S. Nuclear Regulatory Commission, 1986]

SLB disposal facilities generally consist of long, unlined trenches about 15 m wide and 10 m deep. After excavation, trenches are filled with waste containers (either stacked or randomly placed), backfilled to the surface, capped with a mound of soil, and seeded to prevent erosion. Radionuclide migration is slowed only by natural processes such as the action of the sloped cap to divert excessive precipitation and the sorptive capacity of indigenous elements in the unsaturated and saturated zones. Several modifications to basic SLB technology have been proposed that also rely on natural barriers to isolate radionuclides from the accessible environment. These include small, unlined trenches; unlined augers and slit trenches which provide a large ratio of length to diameter or width to reduce surface area exposed to water infiltration and decrease potential for plant, animal or human intrusion; and intermediate depth disposal (15 - 30 m) which features a thicker cap to reduce permeability, erosion, and the possibility of

Table 7. Potential alternatives to conventional shallow land disposal of LLW.

Modified shallow land burial:

Natural barriers
- Small, unlined trench
- Unlined auger
- Slit trench

Engineered barriers
- Concrete-lined trench
- Concrete-lined slit trench

Alternative methods:

- Above-ground vault disposal
- Below-ground vault disposal
- Modular concrete canister disposal
- Intermediate depth disposal
- Hydrofracture
- Deep well injection
- Deep geological disposal (mined cavity)
- Earth-mounded engineered bunkers
- Lined shafts or boreholes
- Caissons or pipes

intrusion. Other SLB modifications have been suggested that incorporate engineered barriers such as improved cap designs using less permeable materials (e.g., clay or treated concrete); backfill materials with high sorptive capacity (e.g., natural zeolites) and trench liners of concrete or other materials. During the operational phase, active controls such as leachate collection and treatment systems may be used, but after closure it is assumed that only passive systems are operable.

Alternative disposal technologies rely on engineered structures or on a combination of natural and engineered barriers. They are designed to reduce contaminant migration, provide increased isolation of waste, and improve long-term stability of the disposal site. Numerous alternative concepts have been proposed including: above-ground vaults, below-ground vaults, modular concrete canisters, earth-mounded concrete bunkers, lined shafts or boreholes, caissons or pipes, concrete walled trenches, hydrofracture, deep well injection, and shallow land burial disposal (e.g., mined cavities). Many of these concepts are based on disposal techniques currently used or planned in other countries. Some designs have been proposed or are in use for temporary storage of LLW by agencies in the U.S. and elsewhere. The current status of five alternative methods is summarized in Table 8.

Below-ground vaults can be constructed of masonry blocks, reinforced formed or sprayed concrete, fabricated metal or polymers molded in situ. This design provides a barrier to both unauthorized and inadvertent intrusion, good structural integrity, and protection against exposure of the waste due to erosion.

Above-ground vaults can be constructed from the same materials as below-ground vaults and differ principally by the fact that they are located at or above grade. Advantages include potential retrievability for site remediation, reduced susceptibility to flooding and a greater degree of freedom in siting (since site geology is less critical). Disadvantages include increased vulnerability to unauthorized access and greater exposure to erosion.

Modular concrete cannisters consist of large metal or concrete over-packs that are filled with LLW packages, void spaces filled with cement grout or sorptive material and then sealed. Canisters can be placed in

Table 8. Status of five LLW alternative disposal methods [Bennett et al., 1984].

Disposal method	Status[a]	Location
Below-ground Vaults	S	Canada, Chalk River National Laboratory (CRNL), Ontario; shallow vaults
	D	UK, Drigg; below ground, shallow vault
	S	Canada, WNRE, Manitoba; shallow vaults
	S	USA, Oak Ridge National Laboratory (ORNL), Tennessee; shallow vaults
Above-ground Vaults	S	Canada, Ontario Hydro, Bruce Site, Ontario
	S	Canada, New Brunswick Electric Power Commission Pt Leprau Site, New Brunswick
Earth-Mounded Concrete Bunkers	S	Canada, Hydro Quebec Gentilly Site, Quebec
	S	Canada, CRNL, Ontario
	S	Canada, WNRE, Manitoba
	D	France, Centre de la Manche Site
Mined Cavity	D	Sweden, Low-level and Intermediate-Level radioactive wastes
	R	W.Germany, Gorlebon; boreholes in bedded salt mine floor
	R	USA, Department of Energy (DOE)
	R	USA, Tennessee Valley Authority (TVA)
	S,D	W.Germany, Asse Salt Mine (radwaste facility)
	S,D,	W.Germany, Herrfa-Neurode Potassium mine (hazardous waste facility)
Augered Holes Disposal Test	R	USA, DOE, Nevada, Greater Confinement
	R	W.Germany, Gorlebon, boreholes in bedded salt mine floor
	R	Canada, AECL, boreholes in glacial till
	S	USA, ORNL, Tennessee
	S	USA, Los Alamos National Laboratory, New Mexico
	S	Canada, Ontario Hydro, Bruce Site, Ontario; "tileholes"
	S	Canada, CRNL, Ontario; "tileholes"

[a]R - Research; S - Storage; D - Disposal.

conventional SLB sites, modified SLB sites or in combination with any other engineered disposal system as part of a multi-barrier approach.

Earth-mounded concrete bunkers currently used in France for LLW and intermediate-level waste disposal are a hybrid design using both above- and below-ground construction together with an earthen cap similar to SLB designs. Higher activity waste is placed in the concrete below-ground vault and back-filled, while lower activity waste is stacked above grade and is covered with a low permeability earthen cap after internal voids are filled.

Lined shafts/boreholes, caissons and pipes are modifications of the auger disposal concept described above. Addition of engineered materials provides improved structural integrity and reduced waste-soil interaction. Concrete walled trenches are similar to below-ground vaults but do not incorporate engineered materials for floor or cap construction (these components resemble SLB design).

Hydrofracture is a system that was used for LLW disposal by DOE at Oak Ridge, Tennessee. A waste-cementitious grout mixture was injected under high pressure into shale, causing it to fracture and provide locations within the host rock for the mixture to solidify. Deep-well injection involves pumping liquid waste into favorable geological media, isolated from pathways to the accessible environment. Due to the highly site-specific geological requirements associated with these methods and technical difficulties that have been experienced with pilot scale facilities, these methods are not considered likely candidates for LLW disposal. Existing limestone or bedded salt mine cavities have been used for low-level, high-level, and hazardous waste disposal in West Germany, Canada, Sweden, and, for research and development purposes, in the U.S. Deep geological isolation of waste has many advantages (e.g., isolation, reduced likelihood of intrusion, proven long-term stability).

Performance assessment considerations affected by disposal technology include selection of appropriate transport and exposure pathways, influence on infiltration and leach rates, prediction of containment failure time and extent, and impact on intruder scenarios. With the exception of modeling the failure of engineered structures and impacts on intruder scenarios, these additional considerations can be based on engineering experience and empirical data. Assessment of long-term performance of engineered barriers in the disposal environment and resulting impact on waste immobilization, however, requires analysis of interaction between materials, waste components, and soil geochemistry and how these factors affect structural integrity, water infiltration and leachate exfiltration rates. Most disposal site performance assessment models do not handle these analyses.

Selection of LLW treatment options is closely linked to available disposal options. Historically, the initial impetus for LLW volume reduction can be traced to rising disposal costs driven by the limited capacity of existing disposal facilities and the difficulties associated with siting and licensing new facilities. More recently, in an attempt to improve the environmental performance of land-based disposal sites, federal regulations imposed waste acceptance criteria on site operators.[U.S. Nuclear Regulatory Commission, 1982] These criteria encourage the practice of segregating waste according to hazard level by implementing regulations based on waste classification, and require generators to stabilize more hazardous waste streams by a combination of treatment and packaging.

Since treatment and packaging of waste can have a major impact on the mobility of radionuclides in the environment, these parameters must be examined in the context of overall disposal site performance. The extent to which treatment/packaging parameters are addressed and the manner in which they are handled in performance assessment models can affect the accuracy and level of uncertainty of exposure projections. It should be noted that many of the available LLW treatment options result in secondary waste streams that require further treatment, solidification, or packaging prior to disposal. Properties of waste in its pre-disposal form can vary significantly from those of as-generated waste. Accordingly, the source terms used in disposal site performance assessment models should reflect the properties of treated, packaged waste, as delivered for disposal.

4.2.5 Waste Disposal Options -- Ocean

The LDC represented the first comprehensive international effort to regulate marine pollution caused by dumping. The LDC classified wastes in three categories: (i) those prohibited from ocean disposal, including high-level radioactive waste; (ii) those that may be disposed under controlled conditions, requiring issuance of special permits (including LLW); and, (iii) those that may be disposed under general permit requirements. Operational control requirements were developed, with a strong emphasis placed on siting issues. Specific site selection criteria [International Atomic Energy Agency, 1978] include:

- The chance of recovering the waste by processes such as trawling shall be minimized;
- Dumping shall be restricted to those areas of the oceans between latitudes 50°N and 50°S. The area shall have an average water depth greater than 4000 m. Recognizing that variations in sea-bed topography do exist, this restriction should not be interpreted to exclude those sites within which there are localized areas with water depths of 3600 m;
- Sites should be located clear of continental margins and open sea islands, and not in marginal or inland seas. Nor should they be situated in known areas of natural phenomena, for example volcanic activity, that would make the site unsuitable for dumping;
- The area must be free from known undersea cables currently in use;
- Areas shall be avoided that have potential seabed resources which may be exploited either directly by mining or by the harvest of marine products, or indirectly (e.g., spawning) as feeding grounds for marine organisms important to man;
- The number of disposal sites should be strictly limited;
- The area must be suitable for the convenient conduct of the dumping operation and so far as possible shall be chosen to avoid the risk of collision with other traffic during maneuvering and undue navigational difficulties. The area chosen should be covered by electronic navigational aids.
- The dumping site shall be defined by precise coordinates. In order to ensure a reasonable operational flexibility, it should have an area as small as practicable, but no larger than 10^4 km^2.

Many of the siting requirements described above (e.g., minimum depth, limiting number of sites) represented improvements over past practices. By developing a coordinated international ocean disposal siting strategy and implementing controls on the location and characteristics of new disposal sites, the LDC aimed to minimize health and environmental consequences from future sea disposal operations.

In support of the objectives of the LDC, the Organization for Economic Cooperation and Development (OECD) Council established a Multilateral Consultation and Surveillance Mechanism for Sea Dumping of Radioactive Waste. One of the responsibilities of this advisory body is to assess the suitability of new and existing ocean disposal sites. According to Article 2(a)(iii) of the OECD Council Decision, site assessments must be conducted at least every five years.[Organization for Economic Cooperation and Development, 1977] As part of this effort, the OECD Nuclear Energy Agency (NEA) published a review of the Northeast Atlantic site which: (i) characterizes its physical, geological and biological aspects, (ii) reviews operational factors such as inventories, waste composition, packaging, etc., (iii) uses the generic International Atomic Energy Agency (IAEA) ocean disposal model to estimate International Council for Radiologic Protection (ICRP) exposure doses, and (iv) reviews compliance with the provisions established by the LDC.[Nuclear Energy Agency, 1980]

Under authority granted by the MPRSA [U.S. Congress, 1972], EPA has also published a set of ocean disposal site selection criteria.[U.S. Environmental Protection Agency, 1977] General criteria restrict dumping of materials into the ocean to only those areas that minimize interference with other activities in the marine environment such as existing fisheries or shellfisheries, and regions of heavy navigation. Sites should be: (i) located beyond the continental shelf in areas such that temporary perturbations in water quality and environmental effects return to normal ambient conditions before reaching any beach, shoreline, marine sanctuary, or known fishery/shellfishery. and (ii) be limited in size to facilitate monitoring and identification of adverse effects. Additional, specific site selection criteria require consideration of the following factors:

- Geographical position, depth of bottom water, bottom topography, and distance from coast;

- Location in relation to breeding, spawning, nursery, feeding, or passage areas of living resources in adult or juvenile phases;
- Location in relation to beaches and other amenity areas;
- Types and quantities of wastes proposed to be disposed of, and proposed methods of release, including methods of packing the waste, if any;
- Feasibility of surveillance and monitoring;
- Dispersal, horizontal transport and vertical mixing characteristics of the area, including prevailing current direction, and velocity, if any;
- Existence and effects of current and previous discharges and dumping in the area (including cumulative effects);
- Interference with shipping, fishing, recreation, mineral extraction, desalination, fish and shellfish culture, areas of special scientific importance and other legitimate uses of the ocean;
- The existing water quality and ecology of the site as determined by available data or by trend assessment or baseline surveys;
- Potentiality for the development or recruitment of nuisance species in the disposal site;
- Existence at or in close proximity to the site of any significant natural or cultural features of historical importance.

Another significant change in ocean disposal policy introduced by the LDC was in the area of design and performance criteria for disposal packaging. Previous policy assumed that dispersion and dilution of radioactive contaminants is sufficient to prevent adverse health and environmental impacts. Current policy emphasizes containment of radionuclides to the extent practically achievable. In their publication on guidelines for LLW ocean disposal packaging, NEA recommends: [Nuclear Energy Agency, 1979]

> The packages should be designed to ensure containment of the waste during their handling, transportation, dumping at a water depth corresponding to the dumping site in use but at any rate

not less than 4000 m, descent to and impact upon the sea floor, and to minimize to the extent reasonably achievable subsequent release of radionuclides.

Specifically, NEA guidelines require radioactive waste be either in a solid or a solidified form that is not permanently buoyant with a specific gravity of at least 1.2 to ensure waste descends to and remains on the seabed. NEA design basis recommendations for ocean disposal packages that address containers, package contents, the package as a whole, and pressure equalization devices are summarized in Table 9.

U.S. regulations issued by EPA [U.S. Environmental Protection Agency, 1977] go beyond NEA Guidelines in requiring containment of radioactive contaminants. For example, 40 CFR 227.7 requires all LLW be containerized and meet the following conditions:

- the materials to be disposed of decay, decompose or radiodecay to environmentally innocuous materials within the life expectancy of the containers and/or their inert matrix;
- materials to be dumped are present in such quantities and are of such nature that only short-term localized adverse effects will occur should the containers rupture at any time;
- containers are dumped at depths and locations where they will cause no threat to navigation, fishing, shorelines, or beaches.

A recent EPA report recommends a number of specific waste package performance criteria in support of existing regulations.[Colombo and Fuhrmann, 1988] These packaging criteria are based on a multibarrier approach consisting of components to contain and isolate radioactive elements from the accessible environment. The containment system consists of the solidified waste form and its container. Together these components make up the waste package, and each component contributes to its overall performance. For example, the container protects the waste form from erosion and degradation while the waste form provides structural integrity for the package under the hydrostatic pressure of disposal. A container lifetime of 200 years or 10 half-lives of the longest lived nuclide (whichever is less) is proposed. The isolation system includes the waste package in conjunction

Table 9. NEA design basis for ocean disposal packages [Nuclear Energy Agency, 1979].

Container

Containers should be designed to meet the following functions:

- act as a receptacle for receiving conditioned radioactive waste at the time of preparation of package;
- provide physical integrity against impact and/or damage during handling and transport, taking into account the severity of expected conditions at sea;
- provide a barrier to prevent the spread of radioactive contamination;
- contribute to shielding and reduction of radiation levels at outer surface of package;
- facilitate handling operations by its shape and/or configuration.

Container should be of good quality material, (e.g., metal or concrete) and of suitable size, taking into account weight limitations. Containers may be lined to provide strength, shielding or protection from corrosive materials.

Content

Conditioned radioactive waste when placed in the container should become an integral part of the sea dumping package. In this way, if the containers are damaged or deteriorate, release and spreading of radioactive materials would be slowed. Waste may be solidified with or without internal reinforcement, or may be packaged as an assembly of several components if void spaces are minimized.

Packaging

Minimum packaging requirements include:

- Specific gravity of ≥ 1.2. If materials with lower specific gravity are contained, they should be conditioned to prevent floating to the surface if container integrity is breached;
- Ability to withstand external sea pressure exerted during descent to sea floor. If not strong to resist deformation, pressure equalization device should be installed;
- Container should be closed with a suitable cap of proper material and dimensions that is an integral part of the package and contributes to structural stability;
- Package should provide inherent shielding so that radiation levels are kept within acceptable limits;
- Package should be strong enough to resist damage from handling and transport operations. Should be compatible with lifting and handling equipment.
- Package should be strong enough to remain intact upon impact with seabed and for a period of time thereafter to minimize to the extent reasonably achievable the radioactivity that might be released.

Pressure Equalization

Pressure equalization devices may consist of discs, plugs, seals, one-way or no-return valves or other devices which are activated or ruptured by a pressure differential across them, so long as they permit no escape of material from the container nor result in increased radiation exposure levels.

Tubes, valves and rupture devices should be designed to prevent ingress of water during storage and transport. They should be positioned to prevent them from being damaged during storage, handling, transport and dumping. Vent tubes should be oriented to minimize external radiation. If voids are not interconnected, a means of connection or appropriate number of pressure equalization devices should be provided.

with natural barriers such as sediments and the residence time of water at the depth of disposal. For example, a recent study of the adsorption capacity of sediment collected from the Atlantic 3,800 m disposal site indicated that 90 % of the Cs-137 is adsorbed under well-mixed conditions at 5 °C. In addition, modeling studies of dissolved tracers released at the seabed indicate that residence time for the western basin of the North Atlantic at 5000 m is about 110 years. Waste package performance criteria proposed for use by EPA in evaluating permit applications for ocean disposal of radioactive waste are presented in Table 10.

4.3 Consequence Modeling

The second task consists of developing or applying mathematical models to calculate environmental transport, human dosimetry and response, and integrating each scenario's consequence models into an overall, coherent analysis. These models provide a basis for rough assessment which can be expressed as mathematical equations that can be solved directly by conventional mathematical or analytical methods. Models are, however, very idealized representations of the natural environment which cannot include all processes that are important. Thus, their value is that they can identify distances and times over which concentrations vary, they may be useful in establishing the largest concentrations that could occur, and they may be useful guides in finding the most important prcesses by comparing the results of including one or another in the calculations.[Brookhaven National Laboratory, 1986]

Because of the complexity of these evaluations, the calculations are often done in a modularized manner, where each module describes a unique process, event, or environment. The modular approach to risk assessment is shown in Figure 3. In this context, the following modules are usually included for land disposal:

- Source-term (i.e., inventory and radionuclide release);
- Groundwater regional and local flow;
- Radionuclide transport in geosphere and biosphere;
- Human dose commitments, and effects.

Table 10. EPA proposed waste package performance criteria for ocean disposal of LLW [Colombo and Fuhramm, 1988].

Criterion	Specification
Package should have adequate density to ensure sinking to the seabed.	Specific gravity of waste package should not be < 1.2.
Package should be designed to remain intact upon impact with sea surface and seabed.	Package should maintain integrity on impact with sea and ocean floor at velocities of 10 m/s
Container should be capable of maintaining its contents until nuclides environment decay to acceptable limits.	Waste container should have expected life-time in deep-sea of 200 yrs or 10 half-lives of longest lived nuclide, whichever is less.
Liquid radioactive waste should be immobilized by suitable solidification gents.	Liquid wastes should be solidified to form a homogeneous, monolithic, free-standing solid containing no more than 0.5% free or unbound liquid by volume of waste form.
Buoyant material should be excluded or treated to preclude its movement or separation from waste form.	Buoyant materials should be treated to form a homogeneous free-standing mono-lithic solid having a spec. gravity ≥ 1.2.
Waste package should be able to withstand hydrostatic pressure encountered during and after descent to seabed.	Triaxial compressive strength (or 4 times uniaxial compressive strength) should be 25% greater than pressure encountered at disposal depth (125 kg/cm^2 uniaxial comp. strength for disposal at 4000m)
Leach rate of waste form should be as low as reasonably achievable.	Leach rate should be $\leq$ regulatory guidelines as measured by ANS 16.1 Leach Test for leaching in seawater.
Particulate waste should be rendered non-dipersible.	Particulate waste (ash, powders, etc) should be immobilized by a suitable solidification agent to form a homogeneous monolithic free-standing solid.
Free radioactive gases should be prohibited from ocean disposal	No radioactive gaseous waste should be accepted for ocean disposal unless they have been immobilized into stable waste forms such that waste package pressure does not exceed atmospheric pressure.

Table 10. cont.

Mixed wastes, which contain hazardous constituent should not be disposed of at a LLW ocean disposal site.	Wastes that contain constituents prohibited, as other than contaminants in 40 CFR 227.6 should not be disposed at a LLW ocean disposal site.
Waste should be physically and chemically compatible with the solidification agent.	Waste forms should retain their structural stability after immersion in seawater for 180 days.

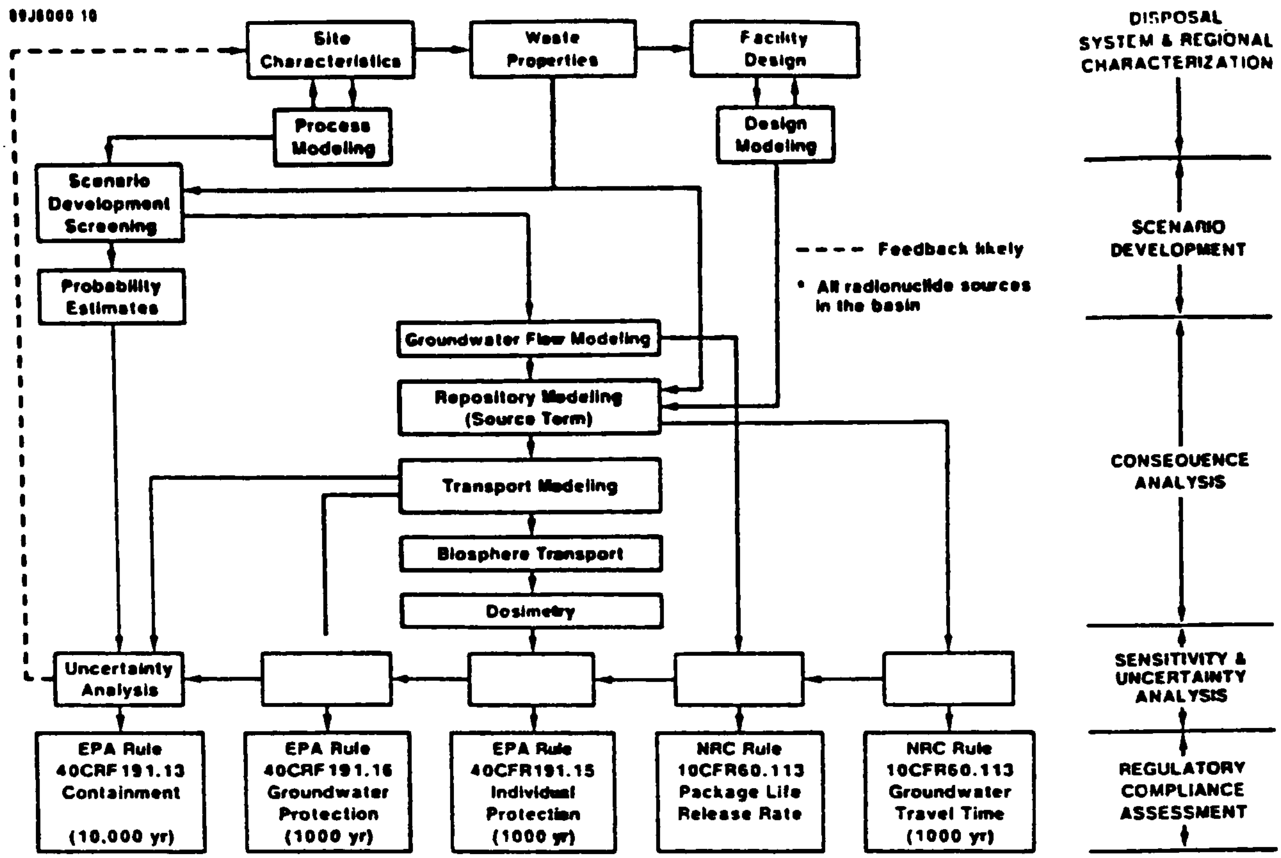


Figure 3. Modules used for land-based risk assessment purposes.

Similarly, for ocean disposal options, these modules are:

- Source-term (i.e., inventory and radionuclide release);
- Basin-scale ocean circulation;
- Regional-scale ocean circulation;
- Radionuclide transport;
- Human dose commitments, and effects.

Because of EPA and NRC regulatory criteria, much of the compliance evaluation efforts use results generated before the last module, human dose commitments. For example, the containment requirements included in EPA's proposed 40 CFR 191.13 use results of radionuclide transport in the geosphere to calculate discharge to the accessible environment. For this reason, a modular approach to radionuclide transport for the geosphere and biosphere is convenient.

In land-based option analyses, the primary pathways (Table 11) that can move the material from its buried position to some altered position or state where it is accessible to man are water and wind. Most of the attempts to model this situation have given a greater emphasis to waterborne movement. This is perhaps due to the earlier attempts to model movement from the deep sites suitable for the disposal of high-level waste. For that case, waterborne movement is the only real possibility, if we neglect volcanic activity.

In this context, regional groundwater flow modules simulate groundwater flow fields in a large region about the controlled area of the repository. The region should extend to natural boundaries or far enough that boundary conditions do not effect transport calculations on the time scale of interest. The regional flow fields then determine boundary conditions for the smaller scale model, the local groundwater flow model which then simulates flow in the repository-controlled area. The local model determines boundary conditions for the land disposal model if groundwater enters the site. Compliance with the NRC engineered barrier requirements can then be evaluated. If the scenario involves transient flow conditions, a coupled groundwater flow and transport module may be required. Then compliance with the proposed EPA containment and groundwater protection requirements can be

Table 11. Pathways and modes of exposure from the land disposal option [International Atomic Energy Agency, 1984a].

TRANSPORT PATHWAYS

Hydrologic
- Leaching
- Deep seepage[a]
- Groundwater
- Surface water
- Direct human consequences (ingestion, immersion)
- Indirect consequences (through fool chain /ingestion)

Atmospheric
- Trench erosion
- Surface contamination
- Suspension
- Deposition
- Direct human consequences (inhalation, immersion)
- Indirect consequences (through food chain/ingestion)

Food Chain
- Drinking water
- Crops
- Meat
- Milk
- Aquatic foods

EXPOSURE SCENARIOS

Operations Phase
- Routine release
- Accidental release

Post-Closure Phase
- routine release
- Accidental release[b]
- Groundwater
- Leachate accumulation
- Intruder
  - Drilling
  - Construction
  - Discovery
  - Agriculture
- Exposed waste

[a]Covers exposure to the public, not disposal site workers.
[b]Refers to post-closure impacts resulting from operational spills.

evaluated. To evaluate compliance with the proposed EPA individual protection requirements requires the last two modules, biosphere transport and dosimetry.

A great deal of effort has been spent in considering the rate at which various radionuclides can be leached from land-disposed wastes, and then transferred to ground water. Once part of the groundwater system, radionuclides are transported at various rates depending upon their partitioning between the aqueous and solid phases. Eventually, dose can be delivered to man by drinking contaminated potable water, the use of contaminated water for irrigation with the subsequent direct contamination of food crops or the indirect contamination of food crops via soil contamination, or the consumption of aquatic foods that have accumulated the radioactive contaminants.

Another pathway of great significance is that of surface erosion. Over geologic time, the erosion of soil by this process is dramatic and its effects are easily seen in undisturbed landscapes. After any overburden is removed, the radioactive contaminants can be removed as well, and the dose-to-man pathways would be the same as those above. In this case, however, the source of the contaminated water would be surface rather than ground. The amount of material transported by this water erosional pathway is strongly dependent upon episodic events, such as severe rainstorms.

Despite such well known occurrences as the "dust bowl" of the 1930s, the erosional power of the wind is not often recognized and treated in models. In areas of the western United States with low rainfall, the erosional effects of wind are readily seen. Often, it is not possible to separate the erosional effects of wind and rain, but both can be effective in soil removal, especially during episodic events. Material that is removed by wind will be deposited, either on the ocean surface, directly on food crops, or on soil. In the latter case, subsequent contamination of foodstuffs may occur. While the material is suspended, it can also be inhaled by man. And, when the material is deposited, it will be on the surface and can deliver dose via the external gamma-exposure pathway.

In the ocean, human dose commitments are usually the performance measure, but it is still convenient to separate radionuclide transport from human dose commitment and effects modules. In the ocean, the basin, e.g., N.

Atlantic, replaces the regional scale groundwater flow model. The continental shelf is the natural boundary. The northern and southern boundaries must be open to allow Norwegian Sea Water to feed the deep water formation in the northeast at the Iceland overflow and to simulate the connection with the equatorial circulation and the Antarctic.

Risk assessments for ocean disposal options involve linking codes of different scales to trace the evolution of a release from a deep bottom source and to estimate resulting human dose commitments and effects. Required codes include ocean circulation modules for several scales, a simplified transport module such as a box model that uses a simulated circulation as input, biological pathways, human dose commitment, and effects models. Risk assessment of ocean disposal options requires a systems approach to assimilate observed data and known features of the ocean into model simulations of the dispersal of radionuclides from deep bottom sources to local, basin or global scales. For ocean disposal, an example system would comprise source, regional, basin, box model (transport/particle), and pathways/dose/effects modules. Risk assessments for ocean options are site specific so meshes for this hierarchical series of model scales must be generated for each case. Given a specific site and a mesh for the bottom boundary layer, regional, and basin scales for that site, the ocean circulation can be simulated. A source is introduced and dispersal through at least the basin and perhaps the global ocean must be computed. Particle and radionuclide transport equations can be coupled with hydrodynamic equations, but this approach is computationally difficult. To make the problem tractable, previous risk assessment programs have used so-called box models as an interim fix while better numerical approaches were developed. A box model is a highly coarsened version of the basin finite-difference grid with nested boxes over the site to account for the bottom boundary layer and regional scales. Box models require ocean circulation as an input. The credibility of the resulting radionuclide concentrations in each box is completely dependent on the input circulation.

In the ocean, the exposure pathways to man (Table 12) may include: consumption of surface water fish, mid-depth water fish and deepwater fish; consumption of seaweed, molluscs, crustacea, and plankton; consumption of salt and desalinated seawater; inhalation of suspended airborne sediments and

Table 12. Pathways and modes of exposure for ocean disposal [International Atomic Energy Agency, 1984b].

Pathway	Mode of exposure
Actual pathways	
Surface fish consumption Mid-depth fish consumption Crustacea consumption Mollusc consumption Seaweed consumption Salt consumption Desalinated sea water consumption	Ingestion
Suspended airborne sediments Marine aerosols	Inhalation
Boating Swimming Beach sediments	External irradiation
Deep-sea mining	External irradiation/inhalation
Hypothetical pathways	
Deep-sea consumption Plankton consumption	Ingestion

marine aerosols; external irradiation during sailing, swimming and sunbathing at a beach; external radiation as a result of mining of minerals from the seabed. Some of these pathways are regarded as unimportant because they do not exist now or are unlikely to result in meaureable doses.

In the process of estimating the risks associated with different options, there are other important analytic issues which must be also examined. These include explicit description and estimation of model uncertainty and matching of model sophistication to the decision problem and to the natural attributes of the system to be modeled. Since any estimate produced by a model contains inherent uncertainty due to various sources including misspecification or oversimplification of models and simple lack of knowledge, characterization of the size and distributional form of the estimate uncertainty is critical. Of the appropriate techniques for producing uncertainty estimates, Monte Carlo analysis has been widely used for three major reasons. First, Monte Carlo analysis creates a mapping from input to output that can be studied by a variety of techniques (e.g., scatterplots, distribution functions, regression analysis, partial correlation analysis) for essentially any input or output variable. Unlike differential analysis and response surface methodology, this mapping does not smooth and obscure discontinuities and transitions between regimes of behavior, i.e., modules. Second, Monte Carlo analysis can accommodate large uncertainties and discontinuities that occur between linked codes. Although the analysis is complicated by these factors, it is superior to other techniques when such complications exist.[Helton et al., 1988] Third, Monte Carlo sampling can include variables with wide ranges and incorporate correlations between variables. The ability to adapt Monte Carlo algorithms to the larger land- and ocean-based models requires further exploration.

Finally questions about model complexity must be addressed. There is much debate surrounding this issue. In this context, Starmer et al. [1988] from the NRC state:

> "Modeling must be defensible. The most suitable model will be that which is consistent with the modeling objectives and easiest to use considering the complexity of the system and the data which can practically be obtained from the site and intended facility. The model should be verified for appropriateness of application,

validity of assumptions, accuracy of algorithms, and representativeness of input data. A critical consideration for the user will be the adequacy of the data available and uncertainty associated with the data. Generally, more complex models require more abundant and detailed data, while less sophisticated models rely more on simplifying assumptions and more generalized data. Where data is inadequate for complex models, there may be a temptation to use approximations based on assumptions. In this case, a more complex model provides no more support than does a simple systems model."

5 MODEL EVALUATION

5.1 Model Identification, Selection and Evaluation

Mathematical models are used to characterize health risks associated with LLW disposal on land and in the ocean. Calculating such estimates requires evaluation of package contents, decay and release; transport and dispersion through the environment; human exposures via food chains; calculation of doses to workers and the public; and, estimation of health effects. In this context, a wide range of models have been developed to cover individual components of this process, for example, air transport and dispersion, groundwater flow or surface transport. Others integrate two or more areas, e.g., air or water transport combined with food chain, human dosimetry and response models.

Eventually, a compatible set of pathway models must be used to evaluate health hazards from LLW. Since there is a very large number of single component and integrated models used for radiation-related risk assessments (see for example [EG&G, 1985]), the first task of this effort was to identify a subset of models to be reviewed. Actual selection of models was based on the combined judgment and experience of the research team and the EPA project staff. Strong candidates were those models that have been used previously by EPA, DOE, NRC, NEA/OECD and IAEA. Models selected for review by this process are listed in Table 13.

The model evaluation process itself, began with establishment of formal evaluation criteria (Table 14). The criteria can be divided into three major groups: Administrative, Technical, and Scientific. Administrative criteria include such points as availability of documentation and computer requirements. Technical issues of concern focus on such points as peer review and availability of results from sensitivity analyses. Finally, the last criterion - Scientific includes such important considerations as validation of model outputs against actual field data. The criteria are applicable to all models and were developed to provide a standardized evaluation approach across models and media. The criteria are qualitative and are not meant for quantitative scoring of models.

The reviews were based on model documentation, publications, and previous experience with these or similar models. Time and resources did not

Table 13. Models for evaluating health hazards associated with land- and ocean-based disposal.

Media	Model	Sponsor
Land	PRESTO	U.S. Environmental Protection Agency
Land	IMPACTS	U.S. Nuclear Regulatory Commission
Land	BARRIER	Electric Power Research Institute
Land	GEOTOX	U.S. Department of Defense
Ocean	MARK A	Nuclear Energy Agency
Ocean	MARINRAD	Nuclear Energy Agency
Ocean	Bryan-Semtner-Cox	National Oceanographic and Atmospheric Administration
Ocean	Sandia Ocean Modeling System	U.S. Department of Energy
Ocean	Holland	National Science Foundation and Office of Naval Research
Ocean	Harvard	Office of Naval Research
Ocean	NRPB91-Box	U.K. National Radiological Protection Board

Table 14. Model evaluation criteria for performance assessment models to compare health hazards from land and ocean disposal of low-level radioactive wastes.

Administrative:

Documentation - Is documentation available? Does it include sufficient information for implementing and reviewing code?

Hardware Requirements - Is model designed to run on a mainframe or desktop computer?

Application - How long does it take to run a single problem? Can it be run in a batch-oriented mode?

Level of Expertise Required - What level of expertise is needed to implement the model?

Technical:

Peer Review - Has model undergone independent peer review?

Verification - Have models undergone testing to verify mathematical computations?

Uncertainty - Does the model propagate input parameter uncertainty to calculate resulting uncertainty in the calculated outputs?

Sensitivity - Have sensitivity analyses been performed to determine the relative importance of individual input parameters on the overall assessment?

Required Input - Are input data readily available? Are data generic or site specific? Can model parameters be modified by the user? Is code structured in modular subroutines so that pieces of the code can be updated if improvements become available?

Output - How are output data presented? Does the model estimate human health effects or exposed dose; cumulative or maximum effects? What time frame is covered; what incremental time steps are used?

Source Term - Does the model accommodate a representative inventory of waste types and isotopes? Are data fixed or supplied by user input?

Scenarios - Are both accident and routine release scenarios included? Do the scenarios sufficiently define potential event?

Relationship to Regulatory Standards - Are model outputs directly relevant to existing regulations?

Scientific

Theory - Are theoretical bases for each model component based on state-of-the-art information?

Validation - Has model been validated, i.e., have performance predictions been compared with actual disposal site data?

Treatment of Radioactive Decay Products - Is the production of radioactive daughter products considered in the source term?

Underlying Assumptions - Are assumptions explicitly stated, complete (i.e., adequately define the problem), and credible?

Pathways - Does the model adequately represent all credible pathways to human exposure?

Dose Conversion and Dose Response - Does the model incorporate dose conversion and dose response algorithms? What models are used to estimate human exposures and response?

permit full evaluation of the computer codes, nor were test runs of the models made. Presented below are brief summaries of the completed model reviews. More complete details on each model are contained in Appendixes B (Land) and C (Ocean).

5.2 Model Reviews - Land

5.2.1 PRESTO

PRESTO-EPA (Prediction of Radiation Effects from Shallow Trench Operations) [Fields et al., 1987] is a suite of computer models developed for EPA to evaluate possible health effects from shallow land burial disposal of LLW. It's original purpose was assessment of impacts from varying shallow land burial disposal scenarios to assist in the development of environmental standards. Individual models in the PRESTO-EPA family include:

PRESTO-EPA-POP	Estimates cumulative population health effects to local and regional basin populations from land disposal of LLW by shallow methods; long-term analyses are modeled (generally to 10,000 years) [Fields et al., 1987].
PRESTO-EPA-DEEP	Estimates cumulative population health effects to local and regional basin populations from land disposal of LLW by deep methods [Rogers and Hung, 1987].
PRESTO-EPA-CPG	Estimates maximum annual whole-body dose to a critical population group (CPG) from land disposal of LLW by shallow or deep methods; dose in maximum year is determined [Rogers and Hung, 1987b].
PRESTO-EPA-BRC	Estimates cumulative population health effects to local and regional basin populations from less restrictive disposal of BRC wastes by sanitary landfill and incineration methods [Rogers and Hung, 1987c].
PATHRAE-EPA	Estimates annual whole-body doses to a critical population group from less restrictive disposal of BRC wastes by sanitary landfill and incineration method [Rogers and Hung, 1987d].

Since PRESTO-EPA-POP was developed first and serves as the basis for other codes in the PRESTO family, it was the focus of this review. PRESTO-EPA-POP assesses radionuclide transport, resultant exposures, and health impacts of a LLW disposal site on a static local population for 1000 years after closure and on the general population residing in the downstream regional basin for an additional 9000 years. The model simulates leaching of nuclides from a waste form, hydrological, hydrogeological, and biological transport, resultant human exposures, and finally assessment of potential human health effects. Exposure scenarios treated by the model include normal release (leaching, spills during operations), human intrusion, and site farming/reclamation. Environmental pathways considered include ground water transport, over-land water flow, erosion, surface water dilution, resuspension, atmospheric transport, deposition, inhalation, and ingestion of contaminated foods and water. Individual and population doses are calculated, as well as doses to intruders and farmers. Cumulative health effects (deaths from cancer) are calculated for the population over the 1000 year period using a life-table approach. Model performance predictions, however, have not been compared with acutual shallow land disposal site data.

5.2.2 BARRIER

BARRIER [Shuman et al.,1988] was developed for the Electric Power Research Institute (EPRI) to assist the nuclear power industry meet federal and state disposal site performance assessment requirements and expedite licensing of new disposal facilities. It is an integrated model that estimates: groundwater flow through a disposal facility; radionuclide release; long-term performance and degradation of concrete barriers used in various engineered storage disposal designs (e.g., below-ground vault, above-ground vault, modular concrete canister disposal and earth mounded concrete bunker); transport through an aquifer to the accessible environment; and doses to the CPG. Projected performance of engineered disposal options were compared to a shallow land burial base-case. Inclusion of a module that simulates the performance of concrete structures is a unique feature of this code. Prediction of engineered barrier performance is a significant addition to the capabilities of performance assessment modeling, especially in light of the increasing emphasis on alternative disposal technologies. At the same time, however, lack of empirical data on concrete behavior over long time periods introduces new uncertainties in the calculation that must be addressed. Barrier has not undergone formal peer review. Model performance has not been compared with actual disposal site performance data.

5.2.3 IMPACTS

IMPACTS [Oztunali et al., 1986] was initially developed for the NRC, to assist in preparation of shallow land disposal regulations for low-level radioactive waste.[U.S. Nuclear Regulatory Commission, 1982] It is an integrated performance assessment model for comparing potential health impacts from various land disposal options on a generic basis. Users are cautioned against using the methodology for a site-specific application, where site-specific models, inventories, disposal options, and environmental parameters would be required to accurately simulate conditions. Further caution is advised in interpreting the absolute magnitude of results. Rather, the model was intended to provide a relative estimate for comparing potential benefits and costs of a number of potential disposal options. Users provide information on combination of waste streams to be considered and regions where they are generated and then select specific waste processing scenarios, the environmental setting of disposal site, and the particular combination of disposal technologies to be used. The model calculates effective dose equivalents (mrems/yr) for 9 organs plus effective whole body equivalent for each exposure scenario and waste classification. For chronic exposure scenarios, estimates are given in varying time increments from 20 to 20,000 years. In addition to IMPACTS, users may select the following subroutines: INVERSE calculates acceptable nuclide total activity and/or concentration limits for disposal; ECONOMY calculates transportation and routine operational radiological impacts as well as disposal cost estimates; INTRUDE analyzes radiological impacts to an inadvertent intruder as a function of time; and VOLUMES calculates and updates region and waste stream dependent annual volume projections. A separate code (CLASIFY) is used to classify the waste streams and organize input data for use by IMPACTS. Model performance predictions have not been compared with actual disposal site performance data.

5.2.4 GEOTOX

GEOTOX [McKone, 1981; McKone and Layton, 1986] is a set of programs used to calculate time-varying chemical concentrations in multiple environmental media (e.g., soil and ground water) and to estimate potential human exposures. The chemical transport component of this model uses landscape data and physiochemical properties to determine the distribution and concentration of chemicals among compartments such as air, water and soil. Environmental concentrations are linked to human exposures and health effects using an exposure model that accounts for intake through inhalation, consumption of food and water, and dermal absorption. GEOTOX is intended for use in public health and environmental risk assessment and risk management efforts, especially for screening and ranking chemicals according to their potential risks. In this context, GEOTOX was originally developed for ranking potential health risks associated with toxic metals and radionuclides in the global environment. Recently, the model has been extended to handle organic chemicals. The GEOTOX program was tested and debugged as part of the development program. This included verification of the mathematical computations. GEOTOX offers the user the option of performing a Monte Carlo analysis, but this option is not available in Version 1.2. Sensitivity analyses can be performed. Model performance predictions have not been compared with actual disposal site performance data.

5.3 Model Reviews - Ocean

Ocean disposal risk assessment requires implementation of regional (i.e., Harvard, Holland, and Sandia Ocean Modeling Systems), basin (i.e., Byran-Semtner-Cox and Sandia Ocean Modeling System), and box (i.e., MARK A, NRPB, and MARINRAD) circulation models. Human dose commitment and effects models are usually independent of the ocean circulation codes and can be often linked with any suitable set of codes. Thus, circulation models which are critical to any ocean disposal risk assessment are discussed below.

5.3.1 Byran-Semtner-Cox

The Bryan-Semtner-Cox (BSC) model [Cox, 1984] is acclaimed as "the principal tool for modeling ocean circulation in irregular domains having realistic coastlines and bottom topography".[Semtner and Chervin, 1988] However, the BSC has been used by English, French, and the German scientists in connection with the SWG-POTG project, and all but the Germans dropped it in favor of other models, and the others questioned the validity of the German results. The BSC does not address boundary layer phenomena, which are important in environmental questions. It is fully coupled to the "free-stream" submodel, which models the overlying ocean. The Bryan-Semtner-Cox model has not been shown to converge as resolution increases. Such demonstration is a necessary part of verification.

5.3.2 Sandia Ocean Modeling System

Sandia Ocean Modeling System (SOMS) [Marrietta and Simmons, 1988] was initially developed for the U.S. DOE, under the Subseabed Waste Disposal Program, to help evaluate deep ocean circulations. SOMS is designed specifically to address bottom boundary layer flows over realistic topography, but is applicable to the whole range of geophysical flows, from small lakes to large oceans. SOMS has been applied to both high-level subseabed [Marrietta and Simmons, 1988] and low-level sea dumping [Nyffeler and Simmons, 1989]. SOMS has also been applied to the circulation of the North Atlantic Ocean, continental shelf phenomena, and flow around a seamount. SOMS is the only deep ocean model designed to address in detail the dynamics of boundary layers over realistic topography; is far less dissipative, more accurate, and runs ten times faster for a given resolution than the world standard BSC model; and is the first ocean model to clearly demonstrate convergence in a realistic geophysical prototype problem, which is a fundamental requirement in verifying a model.

5.3.3 Holland

The Holland model has had some noted success in modeling ocean flows.[Schmitz and Holland, 1986; Holland and Schmitz, 1985] It is limited to free-ocean calculations. Due to its vorticity-streamfunction formulation, it is inapplicable to flow over realistic topography, which must be addressed by primitive equations, or to turbulent boundary layer flows. The main motivation for using the Holland model is that its numerical efficiency is aided by not having to resolve internal waves in time. However, this is largely compensated by needing to solve Poisson equations, one for each layer at each time step. Important data on model convergence used to determine model validation have not been published.

5.3.4 Harvard

The Harvard model [Miller et al., 1983] is part of the ocean prediction forecasting system presently being put into operation by the U.S. Navy. It's use is limited to free-ocean calculations with strongly stratified thermal structures. Due to its vorticity-streamfunction formulation, it is inapplicable to flow over realistic topography, which must be addressed by primitive equations, or to turbulent boundary layer flows. The Harvard model has some sound theory behind it, and can be used as a teaching and research tool in spite of its practical limitations (although its value as a research tool is limited). The Harvard model was the first model used for ocean forecasting and as such has compared favorably with observed field data by several investigators.

5.3.5 NRPB91

NRPB91 has been used for risk assessments of high-level waste disposal in the subseabed and assessments of low-level sea dumping at the North-East Atlantic NEA site. NRPB91 is a coarse-grid box model which calculates radionuclide transport based on an ocean circulation that was subjectively assembled from an extensive literature review. It is not coupled with a dynamical circulation model although like all box models, it could be driven by simulated circulations. The compartment model covers the area of the Atlantic Ocean from 50 S to 65 N and uses observed isopycnal surfaces to define the vertical box structure since mixing and movement in the ocean are believed to occur principally along isopycnal surfaces. Exchanges with other oceans are also included since radionuclides entering the Atlantic Ocean will eventually disperse throughout the worlds oceans. Box models like NRPB91 have not been able to adequately reproduce ocean tracer, heat and salinity distributions.

5.3.6 MARINRAD

MARINRAD was a participant code in a model comparison study conducted by the NEA CRESP Modeling Task Group.[Mobbs et al., 1986] It has has been used for risk assessments of high-level waste disposal in the subseabed of both the Atlantic and Pacific and assessments of low-level sea dumping at the North-East Atlantic NEA site. MARINRAD is a coarse-grid box model which calculates radionuclide transport based on an ocean circulation that was subjectively assembled from historical literature. It is not coupled with a dynamical circulation model although like all box models, it could be driven by simulated circulations. If simulated circulations are used, interbox transports must be hand-calcuated and entered. Interbox transports for nested boxes which must be site-specific have to be obtained from subjective expert estimates. Like other box models, MARINRAD outputs have not compared favorably with any ocean tracer distribution.

5.3.7 MARK A

Mark A [Robinson and Marietta, 1985; Marrietta and Simmons, 1988; de Marsily et al., 1989] has been used for risk assessments of high-level waste disposal in the subseabed for two different locations. The Mark A code was developed by an NEA working group comprised of national experts from the CEC, FRG, France, Switzerland, U.K, and the U.S. It has been exercised and intercompared at Sandia, Ecoles des Mines de Paris, and the CEC-Ispra. The Mark-A is a coarse-grid box model which calculates radionuclide transport using an ocean circulation as input. Interbox transports must be obtained via a subsidiary calculation from a high resolution simulation of the basin circulation. Interbox transports for nested boxes which must be site-specific have to be otained from local eddy-resolving simulations of the site region. Like other box models, Mark A has not been shown to compare favorably with ocean tracer distributions. Further work is required to reproduce both natural and introduced tracers in the ocean.

6 COMPARING RISKS OF THE LAND AND OCEAN DISPOSAL OPTIONS

Comparing the relative risks of land and ocean disposal of LLW is exceedingly complicated, not only because there are many important considerations having high uncertainty, but also because they are extremely different in their characteristics and impacts. This chapter outlines a framework for bridging the gap between available information and assessment capabilities, and needed information and capabilities. It focuses on ways to expand risk assessments to include a broader range of risks and characteristics that are important for comparing the land and ocean disposal options. It describes how quantitative estimates of different risks can be combined in a common metric by which the options can be compared directly in spite of their differences in characteristics. It includes discussion of the impacts that are addressed in some way by existing analytical models, identifies gaps in the available knowledge and modeling capabilities, and discusses how capabilities could be expanded. The intent of the chapter is to provide the necessary background for planning the next stage in the comparison of the LLW disposal options.

6.1 Characteristics of Concern

Branch, et al. describe the setting in which the relative risks of LLW disposal options must be evaluated (Figure 4) and the kinds of characteristics and risks that should be included in the evaluation.[Branch et al., 1987] The basic elements are:

- The physical environment,
- Natural ecosystems,
- Human systems,
- The evaluation process and characteristics of the implementation, and
- Outside influences.

The authors envision that Figure 4 is recursive, essentially rolled into a cylinder matched at the box for the physical environment, to represent evolving risks and changing responses of the system over time.

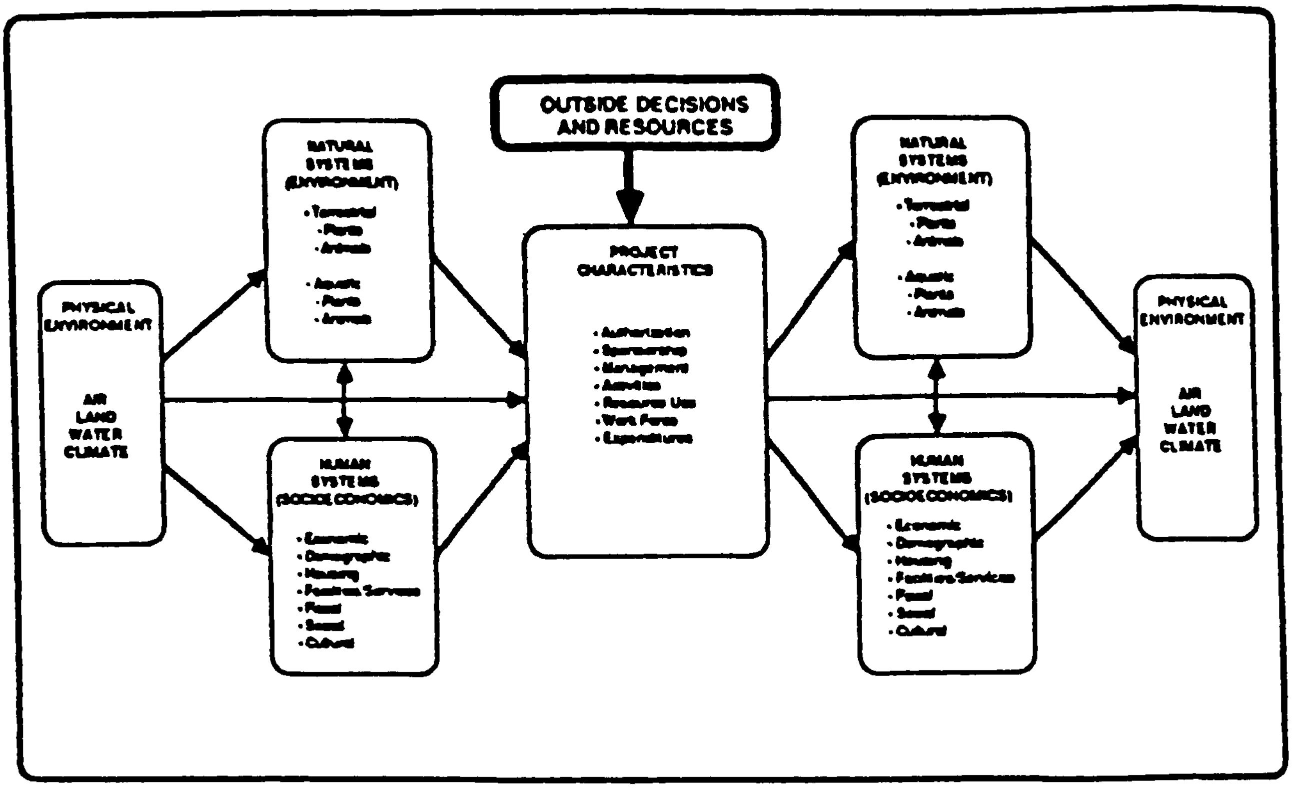


Figure 4. Integrated environmental/socioeconomic impact assessment model [Branch et al., 1986].

Historically, risk assessments of LLW disposal have emphasized human health and safety. But it is now becoming increasingly clear that fears about radiation health and safety can produce ripples that span a much broader range of possible impacts, and the magnitudes of the effects of fear are nearly independent of estimated levels of health risks from radiation exposure. In particular, fear of consequences of radiation exposure causes people to change their behavior with respect to potential sources of exposure, and the economic and social impacts of that change in behavior can be profound. So in many cases, it may be that the change in behavior produced by fear of health risk is important, not the risk, itself.

As a result, potential radiation exposure has two kinds of impacts, which we call risk-induced and perception-induced. Risk-induced impacts arise directly out of "real" risks of "real" exposure (a finite probability of an effective exposure); these include health effects, costs of mitigation measures, and costs of dealing with accidents. Perception-induced impacts are indirect, arising from peoples' behavior in response to <u>fear</u> of exposure; these mostly include a broad range of avoidance behaviors, with associated costs, that are not justified by "real" risks of exposure. Note that some costs and some behaviors are justified by "real" risks of exposure; how much and what behaviors is not clear. The division between risk-induced and perception-induced impacts depends on individual values and attitudes toward risk, which are discussed in more detail below. Both risk-induced and perception-induced impacts are important in risk assessment.

At least two kinds of importance must be considered when comparing risks of alternatives:

- <u>Importance to relative risks.</u> A characteristic is important if it is sufficiently different among alternatives, and of sufficient concern, that leaving it out would change the risk ranking of alternatives; and
- <u>Importance to the compleltness of the risk evaluation.</u> An important function of a comparison of risks is to quantify their absolute magnitudes to determine whether or not they are acceptable. So an impact or characteristic can be important if leaving it out would make the assessment incomplete, regardless of its contribution to

differences in risks of alternatives. This creates an impression of ignoring or hiding something bad, which creates stress and conflict.

An example of the second might be something such as transport of LLW by truck, the overall risks of which could be about the same for land and ocean disposal. Seen from the national perspective, all persons along all routes are equal, so it might not seem important to the comparison. Seen from an individual perspective, each person will be along a transport route or not -- a large difference that matters to those involved. Ignoring risks of transportation in such a comparison is sure to create hostility, regardless of the magnitudes and distributions of differences among the alternatives.

Slovic et al. identify 19 characteristics by which people judge the relative seriousness of risks.[Slovic et al., 1978] They are listed in Table 15 with a description of differences in the land and ocean disposal options with respect to these characteristics. Basically, people perceive risks to be greater, and fear them more, if they are involuntary, catastrophic, not personally controllable, inequitable in distribution, unfamiliar, highly complex, have delayed and uncertain effects, cause fatalities, and arise from unnecessary technologies. Slovic et al. combine these characteristic into two risk-perception "factors" -- dreadness and unfamiliarity.[Slovic et al., 1978]

Risks of nuclear technologies have most of these frightening characteristics and have extremely high levels of dreadness and unfamiliarity. As a result, the general public believes risks of nuclear technologies to be much greater than are estimated by knowledgeable professionals.[Slovic et al., 1978; Slovic et al., 1980; Covello, 1983] Even when they understand the levels of risk, people often fear them more than similar levels of risk from other, nonradiological sources; they think that it is in some way more "serious."

Three important reasons for the large differences with respect to perception of risks of radiation exposure are:

- Differences in Knowledge. Unfamiliarity with the details of radiation health effects, lack of information on normal levels of radiation exposure, and miscellaneous misinformation and disinformation cause many to over-estimate the magnitudes of the risks. This is exacerbated by the special attention radiation health risks receive in the news media relative to risks from other sources.[Greenburg et al., 1989] Most people, including experts, are over-confident in the quality of their estimates of risks and are reluctant to change their minds, even in the face of contradictory evidence.[Slovic and Fischhoff, 1980]

- Distrust. The public lacks confidence in the government and its hired authorities, which causes them to assume that the "official" estimates of experts are self-serving and excessively optimistic.[National Academy of Sciences, 1984]
- Fear. There is a high level of aversion to the effects of radiation exposure, even when the levels of risk are well understood.[Slovic et al., 1980]

These three reasons are linked in that the unknown can produce a high level of fear and decrease confidence in the assurances of authorities, or distrust can reduce attempts to obtain knowledge from "untrustworthy" sources or reduce understanding by deflecting the distrustful to "more trustworthy" misinformation or disinformation.

It is common for risk experts to attribute differences between their assessments of risks and public perceptions of risks to ill-informed, "irrational" evaluations on the part of the public. Risk assessors attempt to make their evaluations scientifically rational, which they believe to be "correct" for society at large. And they assume that the public would agree with their evaluations if only they understood them. But this is equivalent to stating, "Anyone who disagrees with me is ignorant," a judgment that is seldom correct. It is now becoming increasingly clear that public evaluations of risk are not necessarily irrational; rather they are made from a different perspective on the significance of hazards and the role that risk should play in individual and public decision making.[Freudenburg, 1987] While it is true that the general public may often be uninformed or misinformed, they also often disagree with experts when properly informed. The disagreement arises from differences in values. The significance of values to risk perception and decision making is discussed in more detail below.

6.2 Quantitative Results

To understand how risks of land and sea disposal of LLW differ, we must be able to quantify the characteristics of concern. The first step in quantification is to find a unit of measure. For example, if we wish to characterize length quantitatively, we use feet or meters as the measure. If we wish to characterize human health effects, however, we find difficulties. Simple measures like number of cases of disease or person-days of disability may be insufficient to characterize important differences. There are many gradations of disability, and different diseases have different implications in cost, human suffering, and social consequences. A more detailed, multi-faceted measure seems needed, yet the ability to distinguish differences or to even understand the meaning of the measure can be lost in the morass of detail.

Choosing an adequate method to quantify effects of concern is thus an important part of the process of measuring relative risk of land and sea disposal. To clarify this process we distinguish between the characteristics of concern, discussed in the previous section, and the measures used to quantify each characteristic. These are tabulated in Table 15, along with comments describing short-comings, possible additions, sources of information, and other clarifying information. Given the complexity and subjective nature of many of the important characteristics, success in quantifying them varies. Table 16 includes a ranking of the state of the measure, in terms of current capabilities to estimate risks of land and ocean disposal options, from "currently quantified" to "no current way to quantify." Table 17 summarizes the current capabilities to estimate risks of the land and ocean disposal options by four major categories of impact, and provides recommendations on how these can be expanded.

The quantitative measures available for comparing the land and ocean disposal alternatives arise from:

- Site plans and engineering design calculations describing the size, operating conditions, and requirements of the necessary facilities and transportation systems,
- Information on average occupational illness and accidents per person-hour in related industries,

Table 15. Important characteristics determining individual evaluation of seriousness of risks.

Characteristic	Comments
Voluntariness of risk	Exposures associated with land and ocean disposal are both involuntary.
Immediacy of effect	An accidental release in transport or packaging of waste could lead to immediate doses to the public for both land disposal and the land or near-land component of ocean disposal. Once the waste is in place, leakage from the waste package in the oceans would not reach man for hundreds of thousands of years. For land disposal, depending on local conditions, exposure could begin in tens of years.
Individual knowledge about risk	People especially fear radiation because they cannot tell when they are being exposed. This applies to land and ocean disposal, but exposure from ocean disposal might be from seafood anywhere in the world. Specific wells around a contaminated land disposal site could be identified and monitored, making risks more visible.
Level of scientific knowledge	In general, the health risks of radiation doses are well understood by specialists, but not by the public.
Level of personal control	People have little control over risks from either option, although the more easily identified area at risk in the land option makes monitoring more feasible. Also, people can participate in the site selection process for the land operation, which may make them feel they have more control.
Old or new risk	People consider radiation to be a new risk, and therefore more serious. Length of experience with land and ocean disposal of LLW is about equal (40 years).
Chronic or catastrophic risk	In neither option are the risks catastrophic. Even accidents leading to exposure of large numbers of persons would not lead to catastrophic health effects, since the levels of activity of the waste are so low and the effects would be diluted in time and space.
Common or dread risk	Radiation is clearly not a risk people have learned to think about reasonably and calmly. It is a dread risk.

Table 15. cont.

Characteristic	Comments
Fatality	The low probability of exposure and the low level of exposure were there an accident are such that the likelihood of significant consequences is small. Cancer is the consequence of special concern, and the fatality rate for cancers is currently fairly high. Since the probability of exposure extends over hundreds or even thousands of years, we can expect that the fatality rate for cancers will decrease significantly, perhaps even approaching zero.
Preventability of risk	The chance of an accident can be reduced considerably by many control measures, but it can never be reduced to zero. In this context, the differences between processing and transportation requirements for the land and ocean options may produce different types of accidents with different characteristics with respect to preventability.
Controllability of damage	In some cases (e.g., a massive fire in a waste packaging or processing plant), control may be difficult. Since these kinds of accidents occur in the early stages of the waste system, retrievability of waste packages may be required. It is much easier to retrieve waste packages from land sites.
Number of persons at risk	Many more persons are at risk of contamination in the ocean option because of potential worldwide exposure through the marine food chain. But dilution in the ocean is so great that the individual level of risk is exceedingly small, much less than that in the land option.
Threat to future generations	Both options threaten future generations.
Personal threat	The degree of feeling personally threatened is much greater for those living near a land disposal site. A similar, although perhaps lower level of personal threat will exist along transport routes for both options, and in the area around the port for the ocean option. It is doubtful that any rational person would feel a personal threat from contaminated seafood at the time a decision to resume ocean disposal is made. If elevated radiation levels were reported in seafood some time later, then individuals might feel personally threatened.

Table 15. cont.

Characteristic	Comments
Equitability of distribution of benefits	Benefits include military defense, electric power, and medical diagnostics and care. In general, people near a land disposal site feel they are taking the risks while others are gaining most, if not all of the benefits. Equitably is better defined for ocean disposal as that among nations, rather than individuals. Less-developed countries having no nuclear weapons, no nuclear power, and only limited access to nuclear medicine feel they get none of the benefits and are either placed at some small risk or suffer some loss of their share of ocean resources.
Threat of global catastrophe	Neither land nor ocean disposal of low-level waste threatens global catastrophe. Some might argue the case for the ocean, but no argument can be made for land.
Observability of damage-producing processes	Monitoring is relatively easy on land and may provide early warning of contamination. Monitoring at sea is technically more complex and much more expensive.
Increasing or decreasing risk	Since ocean disposal has stopped temporarily, that risk is static. The risk from land is increasing as more waste is disposed on land and new regional and state disposal facilities are planned throughout the country. The moratorium on sea disposal, combined with increasing difficulty in obtaining approval for land sites, continues to build pressure in the system.
Reducibility of risk	Technology exists for reducing risk to virtually nothing for both options, but costs would be unreasonable compared with reductions in risk achieved. The costs would be equally prohibitive for both options.

Table 16. Impacts, characteristics, and risks of concern for comparing land versus ocean disposal of LLW.

Characteristics	Measures	State	Comments
	HEALTH RISKS		
Public Health and Safety			
Death, illness, injury, stress			
Radiological,	Person-rem exposure,	1	
	cancers	1	
nonradiological	Cases	2	Traffic accidents could be added from data in site plans
Magnitude, uncertainty	Expected value,	1	
	variance,	4	The uncertainty of most modelling results is essentially unknown
	maximum individual,	1	
	worst case,	1	
Location	Local,	1	
	national,	2	
	international	1	
Timing	Routine, accidental,	1	
	current and future	1	
Occupational Health and Safety			
Death, illness, injury			
Radiological,	Person-rem exposure,	1	
	cancers	1	
nonradiological	Cases	1	
Magnitude, uncertainty	Expected value,	1	
	variance,	2	Variance is included in available statistics
	maximum individual,	2	
	worst case	1	
Location	Local	1	Production of materials and equipment is not included.

Table 16. cont.

Characteristics	Measures	State	Comments
Timing	Current routine and	1	
	accidental	1	
	ENVIRONMENTAL IMPACTS		
Air, water, land			
Local, national,			
international	Concentration	2	The significance of concentration is often not well understood
Managed ecosystems	Productivity,	2	land use effects
	robustness	4	
Natural ecosystems	Reproduction,	4	
	productivity,	3	land use effects
	stability	4	
	ECONOMIC IMPACTS		
<u>Risk-Induced Costs</u>			
To U.S. government for	Dollars,		
normal operations	opportunity cost,	2	Cost of alternatives
	conservatisms	4	Difficult to separate
accident mitigation	Reduction of magnitude of		
	potential impacts	4	Difficult, what base?
recoverability	Cost of recovery	2	
accidents	Cleanup,	3	
	compensation	3	
To local area	Dollars		
Neighborhood/Town for			
normal operations	Cost of infrastructure,	2	
	need for facilities,	2	
	emergency preparedness	4	Needs are related to public perception
accidents	Emergency response	4	

Table 16. cont.

Characteristics	Measures	State	Comments
To individuals for normal operation and accidents	Dollars		
Home	Real estate value,	4	Much research is required in this area to separate differences between facilities with and without radiation risks and the associated perception-induced effects
	taxes	4	
Livelihood	Business,	4	
	investments,	4	
	income	4	
Cost of living	Prices,	4	
	taxes,	4	
	medical care	4	
To other countries	Dollars	4	
<u>Perception-Induced Costs</u>			
To U.S. government for	Dollars		
normal operations	Conservatisms	3	
accidents	Excess cleanup,	3	
	compensation	4	
To region, state			
Stigma	Dollars	4	
To local area for	Dollars		
normal operations,	Excess emergency preparedness	4	
accidents	Excess emergency response,	4	Even with significant amounts of research, only a few effects of accidents can be expected to be quantified and only with high uncertainty
	loss of business,	4	
	loss of products,	4	
	loss of markets	4	
stigma	Loss of markets,	4	Public perception has a huge influence here; some details are essentially unpredictable
	outmigration	4	

Table 16. cont.

Characteristics	Measures	State	Comments
To individuals for	Dollars		Even with significant amounts of research,
normal operation,		4	only a few effects of accidents can be
accidents, stigma		4	expected to be quantified and only with high
Home	Real estate value, taxes	4	uncertainty
Livelihood	Business and income	4	Public perception has a huge influence here;
Cost of living	Prices,	4	some details are essentially unpredictable
	taxes,	2	:
	medical care (stress)	4	:
			:
To other countries	Dollars	4	:
<u>Asthetic Impacts (linked to property value)</u>			
Appearance,	Intrusion on important views	4	
Noise,	Decibels	4	
Traffic	Vehicles per day	2	
Pollution	Concentration	2	
<u>Benefits</u>			
To U.S.	Opportunity cost of waste disposal ($)	2	
To local area	Jobs, $ business,	2	From site plans
	taxes, compensation	2	
	in dollars and	2	
	infrastructure	2	
To individuals	Compensation in dollars	2	From site plans
	infrastructure, and	2	
	taxes	2	
To other countries	?	4	

Table 16. cont.

Characteristics	Measures	State	Comments
	SOCIAL IMPACTS		
National			
Government programs	Number, dollars	1	
International politics	Number of governments, see Equity	2	
Regional			
Employment	In-migration	1	See economic impacts
Archaeology		4	From site plans
Local			:
Culture and lifestyle	Number of outsiders	2	:
Social system and infrastructure (boom town)	In-migration, new facilities	2 2	: From site plans
Power structure		5	
Stigma	Out-migration, community pride and identity	4 5	: : :

Equity - Distribution of Costs and Benefits

Benefits and costs	Ratios and distributions	2	:
Local: long-range, national and international		2 3	: : Useful methods of quantifying equity must
Producers and nonproducers of LLW		3	be developed
Current and future generations		2	: :

Table 16. cont.

Characteristics	Measures	State	Comments
Desirable Characteristics of the Decision Process			
Acceptance and involvement of public as legitimate partner	:	:	:
Planning and performance evaluation	:	:	:
Listening to public concerns	No measures other than presence or absence	4	Perhaps presence of desirable characteristics is sufficient
Honesty, frankness, openness	:	:	:
Coordination and collaboration	:	:	:
Meeting needs of news media	:	:	:
Communication skill	:	:	:
Symbolic Meaning			
Radioactive garbage dump	No measures	5	
Inviolability of oceans		5	
Local control - NIMBY		5	
Atomic bombs, cancer, mutants	Same for both alternatives	5	
Information Available and Individual Knowledge			
DOE response is similar for both alternatives. Success in reaching individuals is different.			

KEY: (1) Currently quantified; (2) Quantify with small additional effort; (3) Quantify with large modelling effort; (4) Quantify with more research; (5) No current way to quantify.

Table 17. Impacts, characteristics, and risks of concern for comparing land versus ocean disposal of LLW: Summary and Recommendations.

Impacts and Characteristics	Quantity of measurement	Recommendations
Health risks	Human exposure to radionuclides is well covered with respect to magnitude, distribution, and timing. Occupational accidents are the only nonradiological health risks included.	Non-radiological risks to the public could be included. The international distribution of radiological risks could be specified in more detail.
Environmental impacts	Current models do not include quantitative estimates of environmental impacts. Some general information might be provided in site plans.	There are currently no methods for making quantitative estimates of environmental impacts at the required level except for simple measures like acres consumed.
Economic impacts	The only information on economic impacts currently provided is in site plans.	More research and modeling is required to facilitate estimation of economic impacts other than costs. Health-related system costs could be broken out and displayed separately to increase understanding of their contribution to the whole.
Social impacts	The only information on social impacts currently provided (if any) is in site plans. Mostly this is related to estimates of employment.	Measures of equity of distributions of costs and benefits could be added. More detailed estimates require more research and modeling. Characteristics of decision processes that are important to public acceptance of risky facilities should be included.

- Models to estimate radiation exposure-dose relationships, and
- Computer models designed to estimate human exposure to radionuclides released from the facilities under routine conditions and in accidents.

Thus potential characteristics related to human health are well covered (Tables 15, 16). The quality of the information, however, is not uniform. In general, the uncertainty of the estimates increases with distance and time from the source of radionuclides released to the environment, mostly because of random variability of natural processes and relative success in incorporating it in models.

Measures that could be generated with relatively small changes in data requirements or computer codes are mostly:

- Those that can be extracted directly or with modest effort from site plans, engineering designs (provided they exist), and readily available statistics; and
- Those that are related to intermediate physical changes estimated by existing computer models but not currently displayed as part of the results.

The first includes such things as traffic volume and accidents, local employment and business, in-migration, needs for additional capacity in the infrastructure (schools, roads, police, hospitals, etc.). These are either required for engineering design calculations or can be estimated using statistics on unit requirements appropriate to the area (e.g., students per employee, teachers per student, classroom space per student).

The second type of measure includes anything related to environmental concentrations of radionuclides in air, water and soil, which are calculated internally in order to estimate human exposure, but not necessarily displayed in the results. Different spatial disaggregations of human exposure, for example, can be produced by apportioning results with appropriate coefficients or adjusting the spatial units of the models. Impacts on natural environments can be estimated to the extent that they are known to be proportional to levels of anything that is included in the models.

Given the extremely low concentrations of radionuclides that are expected, it is possible that one of the largest impacts on the natural environment will be from physical disruption during construction and operation of facilities (not related to concentrations) and to changes in human behavior with respect to low levels of radionuclide concentrations (not directly proportional to concentrations). For example, changes in magnitude, timing, and spatial distribution of fishing effort as a result of fears of contamination could affect survival patterns and reproductive success in international marine fisheries. The effect could be positive or negative and would be specific to the actions taken and the characteristics of the fisheries, especially the distribution and timing of changes with respect to the breeding cycle. Although the responses of fish populations might be predictable using available information, the responses of people are not.

Distributional equity is a special case among characteristics that could be measured with small additional effort. On a coarse scale, equity can be described as ratios between things people care about. We might calculate, for example, the spatial distribution of the benefit:cost ratio, ratios of local effects to long-range effects, current effects to future effects, or exposures to producing nations and nonproducing nations. The difficulty in such a case is not how to measure equity, but what is the meaning of the measure provided. In part, this is a matter of experience and interpretation -- understanding the relationship of the measure to the real world, and in part is a matter of values -- the relative importance of specific levels of the measure to the comparison.

6.3 Qualitative Results

Some impacts and characteristics are readily available in descriptive form, but are not easily quantified for inclusion in an analytical comparison. Aesthetic characteristics are of this type, as are many kinds of impacts on natural and managed environmental systems. Characterization of the process is another, although if all could agree on the ideal characteristics and the amounts of each, then it would be relatively easy to construct an index of success in meeting these ideals.

Often, however, qualitative results are provided not because it is impossible to create quantitative measures, but because the effort required is excessive compared to the available resources and the quality and significance of the results obtained.

Lack of quantification is only a problem for those characteristics that are important to an analytical comparison -- that is, those that will make a difference in the ranking of alternatives, or those that must be included to demonstrate the acceptability of risks. All of the quantitative methods for making comparisons (Appendix D) require that the attributes of the alternatives be expressed as numbers of some kind. They need not necessarily be such straightforward numbers as persons or dollars or picocuries per liter; they can be indices that express the magnitudes of the attributes indirectly. But they must at a minimum be the numbers 0 and 1, expressing presence or absence of a specified amount (unacceptable, harmful, illegal, etc.) of an attribute. Methods of quantifying attributes are discussed in more detail below.

6.4 Gaps in Available Quantitative Information

Gaps in quantitative information are apparent in the inability of the measures listed in Table 16 to adequately describe the characteristics of concern; in some cases there is no ability to quantitatively characterize the concern. Table 16 thus comments on the current state-of-the-art of quantifying and assessing specific risks of the land and ocean disposal options. The list of risks is long and the gaps in available information and capabilities are large. Many of these gaps could be filled with additional research and modeling effort. Gaps in the four major categories of impact are discussed in greater detail in the four subsections below.

In considering information gaps, one must keep in mind that there will always be gaps between what we know, and what we would like to know. Human resources are never sufficient to satisfy all needs. The key is to rank the relative importance of the gaps and focus on filling only the most important gaps. A form of quantitative analysis, usually termed "value of future information," can aid in determining the ranking, but the principle factors in ranking are perceptions of the relative importance of various kinds of risk and impact. This varies among different population groups and often between analysts and the general public. Treatment of these value considerations is covered in Section 4.6.

6.4.1 Health Risks

Human exposures to radionuclides are well covered in the models with respect to magnitude, distribution, and timing. More work to reduce and characterize the uncertainty of these estimates would be useful.

An argument could be made that more detail should be provided on the international distribution of exposures from ocean disposal. But a parallel argument can be made that current modeling capability is inadequate to support more detailed estimates because the uncertainties are simply too high. The models now estimate total global exposure. With effort, they might be disaggregated to estimate exposure at a smaller scale, perhaps at the level of continents or even nations, but only at the expense of a huge decrease in the accuracy and precision of the smaller-scale results. The relative merits of smaller-scaled information of especially high uncertainty are not clear. Some would argue that it can increase concerns without providing more useful information for evaluating relative risks.

Occupational accidents are the only nonradiological health effects that have been included in LLW risk assessments. They are not included in existing models. Risks of transportation accidents could be added easily, but their only real significance is in helping to maintain perspective; transportation risks are normally much larger than radiological risks. No other nonradiological health effects are likely to be of significance.

6.4.2 Environmental Impacts

The only quantitative information on environmental impacts or risks produced by current models is physical contamination of groundwater, surface water, and selected terrestrial and aquatic foodchains. Extension to the environmental or ecological significance of that contamination is confined to the resulting exposure to humans.

Some general land-use measures like acres required and fuels consumed are available in site plans. As a first-order analysis, we can assume that the ecosystems of areas covered by land- or ocean-based facilities will effectively be replaced by other organisms. What might replace them and the significance of that replacement to the surrounding environmental system is site-specific. Capability exists to model some impacts at the local scale; mostly these are expected to be too small to cause concern, so long as no endangered species, special environments, or other unique, site-specific features are involved.

6.4.3 Economic Impacts

The most significant gaps with respect to comparing risks of land and ocean disposal of LLW are in potential economic impacts, especially those that directly affect individuals -- home and livelihood (real estate value, employment, sales of local products, etc.). LLW assessment models in use produce no information on economic impacts. Some information may be provided in site plans and environmental assessments on employment in construction and operation of facilities that is either useful directly or can be used to estimate direct economic impacts. Little information is available related specifically to facilities involving radioactive materials as distinct from those involving other hazardous materials or undesirable characteristics.

Generating useful information on economic impacts of accidents will require some research and modeling. Few data are available on economic effects of accidents involving radiation, because few such accidents have occurred. For LLW, the accidents that have occurred have been relatively minor, with few producing release of radionuclides to the environment, and those being such small releases that economic impacts were insignificant.[U.S. Council for Energy Awareness, 1988]

The risk-induced impacts of an accident can be expected to be similar to those of other accidents involving hazardous substances requiring special care, such as chemical spills or fires, for which some data are available.[Lee et al., 1989] Mostly these are costs of cleanup, evacuation, and loss of business and income. No LLW accidents have involved more than costs of cleanup and repair of equipment.[U.S. Council for Energy Awareness, 1988] Computer models are available that estimate direct, risk-induced costs of a reactor accident, but these are not applicable to LLW.[Lee et al., 1989] Almost nothing is available on perception-induced economic impacts of a radiation accident. The only radiation accident in the United States for which economic impact data are available is the Three Mile Island nuclear power plant accident, the nature and scale of which is completely different from any accident possible in the LLW disposal system, with the possible exception of a large-scale fire. Devastating perception-induced economic impacts are reported for the accident in Goiania, Brazil, in which cesium-137 found during a metal-scrapping operation was spread about as children played with it.[Patterson et al., 1988]

We can expect that the perception-induced economic impacts of an accident in the LLW disposal system will not be significantly related to actual levels of risk, but instead will be linked to the time required for cleanup and the handling of the accident by the news media. Mostly this is related to the length of time it is treated as a potentially serious problem by the media, which is also related to cleanup time. The attention span of the news media is normally short, even for serious problems.[Greenburg et al., 1989]

6.4.4 Social Impacts

The only information on social impacts of the LLW disposal system is that related to employment during construction and operation of facilities. Land-based facilities are not large enough to produce "boom-town" effects, but a large port facility might be if nearby towns lack sufficient capacity to accommodate additional workers and their families without strain.

Information on social impacts of radiation accidents is mostly confined to general discussion and qualitative descriptions; those from the Three Mile Island accident are an example.[Lee et al., 1989] There are few data.

Although current models do not deal with the question of equity of distributions of costs and benefits, they could produce useful information with modest additional effort. Mostly this would be in the form of maps of costs and benefits and ratios of good and bad characteristics, which might include:

- Local:national:international radiation exposures,
- Exposures to producers and nonproducers of LLW,
- Exposures to current and future generations,
- Compensation to receiving communities,
- Costs to sending communities, or
- Economic cost:benefit ratios - local, national, and international.

Table 16 lists characteristics of the planning process that are important to public perception of the process and the government, and that can produce important social impacts related to public responses to government actions and associated turmoil.[Covello and Allen, 1988] These characteristics are not easily quantified other than by presence or absence, but that is probably enough. If planners deliberately seek to incorporate these characteristics into the process, then the important differences will be related to the local versus international scale of the land and ocean disposal options and the differences in approach that would be required for each.

6.5 Methods of Filling Gaps

The methods available for filling gaps in information include:

- Modification of computer codes to produce intermediate calculations that are not now included in results,
- Addition of coefficients to modify existing measures in models or computer codes,
- Creation of surrogate measures or indices that capture, quantitatively, relative magnitudes of differences that cannot be quantified directly.
- Research to produce information required for existing models and computer codes,
- New modeling based on existing information,
- Research on which to base new models, and
- Elicitation of expert opinions.

Some gaps can be filled relatively easily once it is determined that they are important. But in general, gaps are results of inadequacies of some kind. Often it is no more than an a perception on the part of analysts that the cost of filling the gap exceeds the direct importance of the information to the comparison. In such cases, careful attention must be given to the importance of the information to the process to ensure that analysts are not accused of hiding important problems.

Ability to modify models or add coefficients to existing computer codes depends mostly on an individual's familiarity with the code which, in turn, is affected by the quality of the code. Unless a code is well organized and documented, it is extremely difficult for someone other than its creator to make changes with confidence that they will work as intended and not adversely affect other parts of the code. For models of even modest complexity that are not well organized and documented, it is often considered easier to start anew than to try to understand the code well enough to make necessary changes. In many cases, modifications to codes are best left to persons already familiar with them for other reasons.

Expert opinion is used to clarify data of low quality or in the absence of applicable data.[Morgan et al., 1981] Expert opinion may also be used to guide the design of studies to obtain necessary data. Elicitation of expert judgment is a more formal approach than what is called "engineering judgment." The basic difference between expert judgment and elicitation of expert opinion is in the specificity of the results. In elicitations, the experts are identified and qualified, an attempt is made to use experts who span the range of accepted opinion, and elicitation methods are highly formalized, with great pains taken to obtain an understanding of the causes and magnitudes of uncertainties in their estimates. Results are usually expressed as subjective probability distributions over the range of values the experts consider possible. Because of its emphasis on quantifying uncertainty, elicitation of expert opinion can often be better than use of averages of unspecified uncertainty, which tend to make small differences among alternatives seem more important than they really are.

6.6 Relative Risk Evaluation

The decision-aiding theories and models appropriate for making the kind of complex, multiattribute analyses needed to compare risks of the land and ocean disposal alternatives are discussed in Appendix D. These methods require calculation of a "goodness (or badness) score" that combines the necessary information on risks into a single measure that can be used for direct comparisons of alternatives with different characteristics. Most involve the sum of the products of measured levels of attributes times value coefficients expressing the relative importance of each attribute to the comparison. In theory, this application is straightforward; in practice, it is not. Problems can arise from decisions on selection of attributes to be included, determining how to measure them, the quality of the resulting measure, determining whose values should be represented in the coefficients, and measuring those values. This section stresses the significance of values to such a comparison.

6.6.1 Values

In the sense used here, values are measures of the relative importance of characteristics and the tradeoffs that can be made among them. They can be explicit or implicit, and they can be for individuals or representative of groups.

Explicit values are stated subjective relationships used in an analysis. Since we are concerned with quantitative comparisons, explicit values must be quantitative and refer to measured levels of attributes.

Implicit values are those incorporated in the (usually) unstated assumptions used in selecting the objectives of an assessment, the attributes to be quantified, ways of quantifying or describing them, and ways of combining them. Selecting for analysis only those measurements that are currently available, for example, contains an implicit value judgment that the value of the time and cost of obtaining more information exceeds the value of the contribution of that information to the analysis. But this implicit judgment is normally made by default by someone "doing the best he can with the available information," without much evaluation of its implications. Any quantitative analysis contains many such implicit value judgments.

If attributes have a recognized common metric, such as dollar cost, then no separate values need be placed on them; their dollar costs can be compared directly. This is done in cost:benefit analysis (see Appendix D). But if the attributes are measured in incommensurable units -- those that cannot be compared directly with a common metric -- then analysts must use value judgments about how important each is relative to the others and how much of one should be given up in order to get some of another. This is the familiar "apples and oranges" problem. Of all the components of a quantitative comparison of characteristics of the land and ocean disposal alternatives, determination of values will be the most troublesome.

What is required is a numerical "weight" for each attribute expressing how important a difference of one unit of measurement of that attribute should be and the tradeoffs analysts can make among the alternatives. These weights cannot be established in isolation; they are specific to the problem, to the units in which the attributes are measured, and to the person or persons whose values are involved. One cannot, for example, simply state that "human health is more important than environmental quality." This implies that any health impact is worse than any environmental impact, which at the extreme is clearly not the case. Instead, what is intended by that statement is, "To me, the human health effects I expect are more important than the environmental impacts I expect," where the expected amounts are specific enough to support the judgment. To establish a weight, analysts must be able to go a step further and specify that X specific human health effects are equal in importance to Y specific environmental impacts. Or even more difficult, they must specify that a given probability of X specific human health impacts is equal to Y specific environmental impacts. Mechanisms for assisting in making these difficult judgments of relative importance are outlined in Appendix D.

A large part of the uncertainty involved in comparing land and ocean disposal of LLW arises from a need to evaluate the relative importance of many different potential impacts that are incommensurable (Table 16). These two alternatives are different in the kinds and magnitudes of potential impacts they entail. Land-based approaches mostly threaten local water supplies and ocean-based approaches mostly threaten international fisheries, for example. One especially important difference is the distribution of risks of contamination between those who benefit directly from the activities producing the LLW and those who do not, a question of equity. Risks to producing nations should clearly be evaluated differently from risks to nonproducing nations, even in cases where they have a common metric (e.g., human exposure).

The uncertainty from incommensurables becomes very large when the full range of characteristics of concern is included and the special concerns of all persons affected are accounted for. When analysts are faced with dealing with a large number of incommensurables intuitively, the normal response is to eliminate those that are seen as relatively less significant and evaluate only a manageable subset that is considered unavoidably significant. If the full range of incommensurables is to be included in a comparison, then it is necessary to deal separately with each and to devise ways of quantifying their relative importance by some common metric. It is specifically this application for which analytical decisionmaking methods were devised (Appendix D).

6.6.2 Individual Values

For any particular person, there is clearly a hierarchy of values for various kinds of characteristics, but it is not always clear what that hierarchy is. In general, people care more about things that are close to them in space, time, and personal relationship:

- Here, there, somewhere.
- Home, neighborhood, town, country.
- Now, soon, sometime, a long time.
- Me and mine, you, them, someone.
- Food, clothing, shelter, health, income, lifestyle, etc.

But even this generalization varies depending on the person and the characteristic evaluated. Most people would place their childrens' welfare above their own, especially with respect to health issues, but not all and not in all things (e.g., conflicts between children and career are relatively common). So the specifics of individual value systems are highly personal and not readily predictable.

They are, however, quantifiable. The two basic approaches to quantifying these values are through questions designed to elicit expressed preferences (e.g., "How much of this are you willing to trade for that?") and statistical analyses designed to quantify preferences implied by observed behaviors (e.g., average housing prices are lower close to an undesirable facility).

The situation is further complicated by differences in perspectives between individuals protecting things they value personally, and perhaps engaging in strategic bias and gamesmanship in an attempt to influence outcomes in their favor, and government analysts and decisionmakers representing society, but who also have personal values that influence their views of what societal values should be.

6.6.3 Values in Conflict

Whose values should be used in social decision making (e.g., Congress, environmental groups or others) to make better and more informed decisions? If all agree on appropriate weights for attributes, then there would be no conflict in evaluating alternatives. But it is clear that there are large differences in values, not only among individuals, but also between individuals (including decision makers) and the society as a whole. This is most particularly true for risk of exposure to radiation, which has a special, and especially sensitive, place in individual perceptions.[Slovic et al., 1978] Large differences produce conflict.

Payne and Williams identify three sources of conflict affecting individual values related to radioactive waste management:[Payne and Williams, 1985]

- The social history of development of nuclear technology,
- Differing value orientations, and
- Differing perceptions of acceptable risks.

The social history is important because of fear associated with nuclear weapons, which is transferred in the minds of lay persons to all nuclear technologies. Most people incorrectly think about all nuclear technologies as though they were potential bombs. In addition, people do not readily change these attitudes, especially when presented with conflicting information, but also when presented with convincing information that is contrary to strongly-held beliefs.

Differing value orientations arise from four basic areas:

- Participatory democracy,
- Stewardship,
- Environmentalism, and
- Equity.

People, groups, and communities want to maintain control over their own destinies. Conflict arises when they feel excluded from decisionmaking processes affecting them directly. They also feel that common resources (land, water, air) under government stewardship should be used carefully and productively for the common benefit. Environmentalism, in particular, emphasizes conservation and preservation of common natural resources that should be maintained without harm for the common benefit, present and future. This, in part, arises from a sense of a need for equity in the distribution of costs and benefits in all things, but especially in use of common resources. Because of differences in orientation with respect to these values between decisionmakers at the national level, who think in terms of benefits to the society as a whole, and local communities directly affected by their decisions, who think in terms of their own lives, potential for conflict is high. Neither is necessarily wrong or selfishly motivated. They simply have different perspectives on what is important.

6.6.4 Social Amplification of Risk and Effects on Values

Differences in risk perception between "experts" and the general public are well documented. They arise from differences in knowledge and experience that produce different assumptions yielding different conclusions. In general, people overestimate the frequency of catastrophic and sensational hazards with which they have little experience, and they especially fear hazards that are involuntary, not under personal control, inequitable in distribution of costs and benefits (not fair), and highly complex and unfamiliar.

The result of these differences is that the general public believes the risks of exposure to radiation to be greater and more serious than those of other, more familiar risks, and they overestimate the likelihood of accidents producing exposures, so they do not accept the numbers calculated by "experts" that are in conflict with their preformed beliefs.

Kasperson, et al. outline the sources of special public concern about the risks of radiation exposure, which they call "social amplification of risk."[Kasperson et al., 1987] The roots of social amplification of risk lie in experience, either direct or indirect through information from friends and the media. Direct experience with risky activities can be reassuring, as in driving automobiles, or alarming, as in tornados or floods. Limited experience with dramatic accidents tends to make them more memorable, which heightens perception of risk. But repeated experience can also afford greater perspective and capability for avoiding risk, as in occupational accidents.

The role of the news media in amplification of risk is critical. The key attributes of information that cause amplification are volume, level of dispute, and extent of exaggeration. All of these are under the control of the news media and are used to maximum advantage to generate sales.[Greenburg et al., 1989] Misinformation and distortion are common.[Combs and Slovic, 1978;Freimuth et al., 1984] High volumes of information mobilize latent fears about a particular risk. Debates among "experts" increase uncertainty. Erroneous information, usually exaggeration to increase their value to the news, increases memorability and fear without basis in fact.

Public amplification of the relative importance of catastrophic accidents means that decision makers must make some conscious judgment about how to deal with their significance in a decision. Consider, for example, the comparison shown in Figure 5, in which two alternatives differ in the probability distribution of human exposure to radionuclides. The distributions have the same expected value, but one alternative has risks concentrated in the medium range (perhaps routine emissions with no likelihood of a severe accident) while the other has risks spread over a wide range, mostly at very low levels, but including a reasonable possibility of a catastrophic accident. With respect to expected exposure, these alternatives are the same. With respect to the worst possible outcome, the first alternative is superior. With respect to the most likely outcome, the second alternative is superior. How should these different measures of impact be valued?

Mathematically, if the two distributions have the same expected value and the more catastrophic outcomes are considered more important per unit exposure, then a distribution having a higher probability of catastrophic accidents will always be more important than one having a lower probability. But if they do not have the same expected value, then a judgment must be made of the relative importance per unit exposure of small, frequent exposures to many versus large, infrequent exposures to a few. This judgment can be an

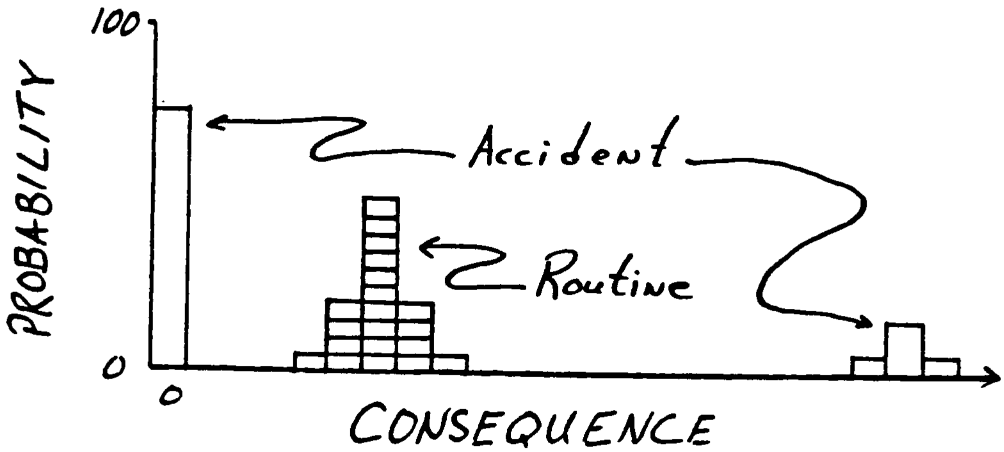


Figure 5. Hypothetical distributions of risks of human exposure, one with only routine low-level risks and another with only higher-level risks of accidents.

important source of conflict between decision makers with a national perspective, for whom all exposures are much the same per unit of measurement, and the general public, for whom catastrophic accidents are especially feared.

6.6.5 Incorporating Stake-holder Values

Establishing values is an important stage in the comparison of the land and ocean disposal options. These include not only specific values for measured attributes, but also implicit and explicit values, judgments, and tradeoffs imbedded in decisions on what attributes should be included in the analysis and what methods should be used to evaluate them. Exceedingly important here is the question of whose values should be used.

Appendix D discusses some theories of social decision making and problems with applying them in the real world. Treatments of values are fundamentally different in these approaches. Cost-benefit analysis uses monetary value as established by market forces or other estimates of "willingness to pay," which are assumed to represent aggregate social value (no decision maker). Decision analysis uses the values of one person (or a group of like mind) representing himself, his organization, or some larger group of stakeholders, including society as a whole (one decision maker). Social welfare theory attempts to provide a rational synthesize of the preferences of all stakeholders through some aggregate social welfare function (many decision makers).

If the outcome of the comparison is mostly of political significance, then it is useful to maintain separate value functions for the different groups of stakeholders involved, either by measuring them for small groups or selecting a representative to measure for larger groups. The results obtained using such value functions reveal the significance of differences of opinion on the outcome of a comparison. In fact, the weights can even be constructs representing hypothetical points of view with respect to important attributes.[Radioactive Waste Division of the Department of Environment, 1986] This approach has the advantage of not requiring specific measurements of value functions to complete a comparison, and the disadvantage that the results do not represent the values of real people, which may not be captured well by the constructs.

6.7 Designing a Quantitative Comparison

All analytical methods for comparing alternatives involve some basic steps that separate planning functions, information on characteristics of alternatives, and the values and preferences used to evaluate them. The steps usually include most of the following, in whatever order is convenient.

- Define objectives
- Identify alternatives and structure the analysis
- Define performance measures
- Identify important characteristics
- Quantify characteristics
- Specify values, preferences, and permissible tradeoffs
- Evaluate alternatives
- Evaluate sensitivity and value of additional information.

The objective of the land versus ocean comparison is predetermined -- quantify the relative risks to provide an information base for decisionmaking.

Alternatives to be evaluated must be determined by analysts and decision makers. The land option can be represented as a system of existing and planned sites, a representative site, or a hypothetical generic site, with alternative or representative treatment and disposal techniques as required. Only one ocean site is currently in use, but alternative treatment and disposal techniques can be included. Needs to analyze a broad range of alternatives within each option depend, in part, on the nature and magnitude of the differences between them. In a general comparison of land and ocean disposal, if the differences between alternatives within options are small compared to the differences between options, then little is gained by expanding the list of alternatives considered. This can occur if there are large differences between the options with respect to relative magnitudes of risks or with respect to the relative importance (value) of risks of different kinds. The more the options tend to overlap with respect to magnitudes and values of risks, the greater is the need to include all available alternatives within them.

Analysts should keep in mind that risks of alternatives are variable functions of cost. The more money is spent on protective measures, the smaller will be the magnitudes of the risks. This distinction is universally ignored in risk assessments, which are normally based on fixed engineering designs specifically intended to meet some predetermined set of criteria, and which are thereafter taken to be immutable. We forget that we can always spend more money to reduce specific risks if it is justified.

Important characteristics and impacts that might be included in a comparison are outlined in Table 16. Constraints on time and resources restrict the comparison to some subset of the total that includes the information currently available plus whatever other information is considered sufficiently important to justify additional expenditures of time and resources. Importance should be evaluated not only with respect to the comparison, itself, but also with respect to the process and effects on public opinion. This comparison involves tradeoffs under a finite budget between relative importance of information, the cost of getting it, and its likely effect on the outcome. Table 18 outlines some of the objectives and criteria that might be used in the selection.

This stage of an analytical comparison is seldom quantified, and selections are mostly based on direct experience, knowledge of other, similar problems, and intuition. Many implicit value judgments must be made in the absence of specific, quantitative information related to the alternatives under evaluation. It would be useful with respect to public opinion if analysts could provide some kind of rationale for the selections made at this stage, particularly if something the general public cares a lot about is left out (e.g., economic impacts). There is a tendency, however, to be as unspecific as possible at this stage to avoid providing ammunition to critics early in the process.

Specifying values poses two problems -- whose values are to be used and how to measure them. Because of the political sensitivity of the land versus ocean comparison, it would appear that maintaining separate value functions for the various stakeholder groups would provide the most useful information. Not only might it demonstrate the overall effect of different value systems on the rankings of alternatives and options, but also it might assist in

Table 18. Rationales for selecting characteristics and impacts for inclusion in the multimedia comparison,

Objective	Criterion
Maximize feasibility	Potential for producing obstructive responses
Minimize fear	Public perception of relative risks
Minimize health risk	Potential Health effects
Minimize cost	Cost-related characteristics, including cost of obstructive behavior caused by objections to leaving out other important information
Minimize total impact	Experts estimates of magnitudes and importance of impacts
Social optimization	Societal preferences for general impacts
Promote acceptance of a particular alternative	Favorability ratio
Provide appearance of objectivity	Level of controversy
Support organizational goals	What are the organizational goals?

showing the specific differences in risks and values that actually affect the comparison. These indicate candidates for more detailed attention.

7 CONCLUSIONS

Methods and models are available to compare some health and some environmental risks associated with the ocean- and land-based disposal of LLW. If only health risk estimates are prepared, the minimal set of risks to be examined should include routine and accidental radiation exposures to workers and the public from transport (land and ocean), disposal, and post-closure releases of radionuclides; health consequences for maximally exposed or critical population groups, and the estimated collective dose to workers and the public for each defined scenario. Since waste processing, packaging and disposal requirements for land and ocean disposal can differ, their contribution to the overall risk can vary by medium; these have not yet been fully evaulated.

Human and natural processes that precipitate accident-initiated releases are also important determinants of risk and must be examined. Some of these, such as human/animal intruder scenarios or climatic events (e.g., a 100 year flood) may only relate to one medium, while others, such as earthquake activity, apply to both media. Differences in exposure scenarios can have a large impact on the overall risk.

On land, the geographic scales of interest for routine releases will generally be confined within the boundaries of the ground water system. In contrast, the ocean option will require analysis at more global scales (i.e., oceans) because of the nature of the circulation of the deep ocean. Because of extensive computational requirements to model ocean circulations at such a wide scale, all available dynamical models are too large to be directly incorporated into a systems analysis model for full uncertainty analysis. Thus, dynamical models should be used in performing "most realistic" ocean transport calculations for risk. For efficiency in systems studies, calculations such as these are replaced, albeit crudely, by box model calculations. It is tempting, in box modeling, to increase the number of grid points so as to produce greater resolution and/or accuracy, and to some extent, this can be done. However, it is a dubious practice because errors in the underlying model parameters usually increase as the resolution of problems increase.

In principle, the models selected for analysis should have completed some validation exercises and be capable of producing explicit estimates of model uncertainty. In practice, however, it appears that most available pathway models have not undergone rigorous validation efforts or are not capable of producing explicit estimates of output uncertainty. Similarly, while estimates produced by models containing complex descriptions of physical phenomena may appear to be more credible or accurate, this is not always true. In any case, continued efforts to validate the predictions of performance assessment models against actual field data are critical to reduce uncertainty.

It is clear that the general public is concerned about a much broader range of characteristics and potential risks of the LLW disposal options than just health risks. These especially include socio-economic risks and the equity of the distribution of risks and benefits in space and time. None of the available LLW risk assessment models include these important considerations, and few of the models could be modified to include any of them with modest effort. This is a serious deficiency in current capabilities that will undermine the credibility of results in the eyes of the general public. When this information becomes available, there are decision aiding methods for incorporating a broad range of characteristics, potential risks, and stakeholder values into quantitative comparisons of alternatives. Development of necessary data and application of these approaches can permit decision-makers to make better and more informed decisions.

8 REFERENCES

Bennett, R.D., et al., 1984. Alternative Methods for Disposal of Low-Level Radioactive Wastes, NUREG/CR- 33774, Vol.1, Geotechnical Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Branch, K.M., C. Cluett, and T.L. Page, 1987. "Integrating Environmental and Socioeconomic Assessment," Vol. 2, pp. 85-90, in R.G. Post, ed., Waste Management '87. Proceedings of the Symposium on Waste Management, Tucson, AZ, March 1-5, 1986, American Nuclear Society, LaGrange Park, IL.

Brookhaven National Laboratory - LLW Deep Sea Group, 1986. Technologies for and Impacts of Low-Level Radioactive Waste in the Deep Sea, Draft report prepared for the Office of Technology Assessment, U.S. Congress, Washington, DC.

Burns, M.E., and W.H. Briner, 1988. "Setting the Stage," in M.E. Burns, ed., Low-Level Radioactive Waste Regulation - Science, Politics, and Fear, Lewis Publishers, Inc., Chelsea, MI.

Colombo, P. and M. Fuhrmann, 1988. Waste Package Performance Criteria for Deepsea Disposal of Low-Level Radioactive Wastes, EPA 520/1-88-009, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington DC.

Combs, B., and P. Slovic, 1978. "Newspaper Coverage of Causes of Death," Journalism Quarterly 56:837-843.

Covello, V., and F. Allen, 1988. Seven Cardinal Rules of Risk Communication. U.S. Environmental Protection Agency, Washington, DC.

Covello, V.T, 1983. "The Perceptions of Technological Risks: A Literature Review," Technol. Forecast. Social Change, 23:285.

Cox,M.D., 1984. A Primitive Equation 3-Dimensional Model of the Ocean, Technical Report No. 1, Geophysical Fluid Dynamics Laboratory, Princeton, NJ.

de Marsily, G. et al., 1989. Feasibility of Disposal of High-Level Radioactive Wastes into the Seabed, Volume 2, Radiological Assessment, Nuclear Energy Agency, Paris.

EG&G, 1985. Low-Level Waste Disposal Performance Assessment Model Review, EGG-LLW-8083, Idaho National Engineering Laboratory, Idaho Falls, Idaho.

EG&G, 1987. The 1986 State-by-State Assessment of Low-Level Radioactive Wastes Received at Commercial Disposal Sites, DOE/LLW-66T, Idaho National Engineering Laboratory, Idaho Falls, Idaho.

Fields, D.E., et al, 1987. PRESTO-EPA-POP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, EPA 520/1-87-024 (2 Volumes), Office of Radiation Programs, U.S. Environmental Protection Agency, Washington DC.

Freimuth, V.S., R.H. Greenburg, J. DeWitt, and R. Romano, 1984. "Covering Cancer: Newspapers and the Public Interest," Journal of Communications 34:62-73.

Freudenburg. W.R, 1987. "Rationality and Irrationality in Estimating the Risks of Nuclear Waste Disposal," in Waste Management '87. Proceedings of the Symposium on Waste Management. Tucson. AZ. March 1-5. 1987. 2:109-115, American Nuclear Society, LaGrange Park, IL.

Greenburg, M.R., D.B. Sachsman, P.M. Sandman, and K.L. Salomone, 1989. "Network Evening News Coverage of Environmental Risk," Risk Analysis 9:119-126.

Helton et al., 1988. "Integration of NUREG-1150 Analyses: Calculation of Risk and Propagation of Uncertainties," in Proc. Fifteenth Water Reactor Safety Research Information Meeting.

Holcomb, W.F., 1982. "A History of Ocean Disposal of Packaged Low-Level Radioactive Waste," Nuclear Safety 22(2).

Holland, W.R. and W.J. Schmitz, 1985. "Zonal Penetration Scale of Model Midlatitude Jets," J. Phys. Oceanogr. 1859-1875.

International Atomic Energy Agency, 1978. Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter. The IAEA Revised Definition and Recommendations of 1978 Concerning Radioactive Wastes and Other Radioactive Matter. Referred to Annex I and II of the Convention. Information Circular INFCIRC/205/Add.1/Rev.1, Vienna, Austria.

International Atomic Energy Agency, 1984a. Safety Series No. 64 - Safety Analysis Methodologies for Radioactive Waste Repositories in Shallow Ground, Vienna, Austria.

International Atomic Energy Agency, 1984b. Safety Series No. 66 - The Oceanographic and Radiological Basis for the Definition of High-Level Wastes Unsuitable for Dumping at Sea, Vienna, Austria.

Kasperson, R.E., J. Emel, R. Goble,, C. Hohenemser, J.X. Kasperson, and R. Renn, 1987. "Radioactive Wastes and the Social Amplification of Risk," in R.G. Post, ed., Waste Management '87. Proceedings of the Symposium on Waste Management. Tucson. AZ. March 1-5. 1986, 2:85-90, American Nuclear Society. LaGrange Park, IL.

Lee, J.C., M.D. Rowe, C.A. Grimshaw, P.D. Kalb, and C.G. Miles, 1989. Economic Impacts of Accidents in the High-Level Nuclear Waste Disposal System: A Review of Methods and Data. Draft report, Brookhaven National Laboratory. Upton, NY.

Marietta M.G. and W.F. Simmons, 1988. Feasibility of Disposal of High-Level Radioactive Wastes into the Seabed: Dispersal of Radionuclides in the Oceans: Models. Data Sets. and Regional Descriptions. SAND87-0753, Sandia National Laboratories, Albuquerque, New Mexico.

McKone, T.E. and D.W. Layton, 1986. "Screening the Potential Risks of Toxic Substances Using a Multimedia Compartment Model: Estimation of Human Exposure". Regulatory Toxicology and Pharmacology 6:359-380.

McKone, T.E., 1981. Chemical Cycles and Health Risks of Some Crustal Nuclides. Ph.D. Dissertation, University of California, Los Angeles.

Miller, R.N., A.R. Robinson, and D.B Haidvogel, 1983. "A Baroclinic Quasigeostrophic Open Ocean Model," J. Comp. Phys. 50:38-70.

Mobbs S., et al., 1986. A Preliminary Comparison of Models for the Dispersion of Radionuclides Released into the Deep Ocean. Report No. NRPB-R194, Chilton, Didcot, Oxon, United Kingdom.

Morgan, M.G., M. Henrion, and S.C. Morris, 1981. Expert Judgments for Policy Analysis, BNL-51358, Brookhaven National Laboratory, Upton, NY.

National Academy of Sciences Panel on Social and Economic Aspects of Radioactive Waste Management, 1984. Social and Economic Aspects of Radioactive Waste Disposal: Considerations for Institutional Management, National Academy Press, Washington, DC.

Nuclear Energy Agency, 1979. Guidelines for Sea Dumping Packages of Radioactive Waste, Nuclear Energy Agency, Organization for Economic Cooperation and Development, Paris, France.

Nuclear Energy Agency, 1980. Review of the Continued Suitability of the Dumping Site for Radioactive Waste in the North-East Atlantic, Nuclear Energy Agency, Organization for Economic Cooperation and Development, Paris, France.

Oak Ridge National Laboratory, 1988. Integrated Data Base for 1988: Spent Fuel and Radioactive Waste Inventories, Projections, and Characteristics, DOE/RW-0006, Rev. 4, Oak Ridge National Laboratory, Oak Ridge, TN.

Organization for Economic Cooperation and Development, 1977. Decision of the OECD Council of 22nd July, 1977, Paris, France.

Oztunali, O.I., et al., 1986. Update of Part 61: Impacts Analysis Methodology, NUREG/CR-4370, (2 Volumes) U.S. Nuclear Regulatory Commission, Washington, DC.

Parker, F.L., 1988. "Low-Level Radioactive Waste Disposal," in M.E. Burns, ed., Low-Level Radioactive Waste Regulation - Science, Politics, and Fear, Lewis Publishers, Inc., Chelsea, MI.

Patterson, J.S, 1988. "From Perception to Reality: The Goiania Socioeconomic Impact Model," In R.G. Post, ed., Waste Management '88, Proceedings of the Symposium on Waste Management, Tucson, AZ, February 26 to March 3, 1988, American Nuclear Society, LaGrange Park, IL.

Payne, B.A., and R.G. Williams, 1985. "Conflict, Public Communication, and Radioactive Waste Management," In R.G. Post, ed., Waste Management '85, Proceedings of the Symposium on Waste Management, Tucson, AZ, March 24-28, 1985, 3:103-106, American Nuclear Society, LaGrange Park, IL.

Radioactive Waste (Professional) Division, 1986. Assessment of Best Practicable Environmental Options (BPEO's) for Management of Low- and Intermediate-level Solid Radioactive Wastes, Department of Environment, United Kingdom.

Robinson, A.R. and M.G. Marietta, 1985. Research, Progress and the Mark-A Box Model for Physical, Biological and Chemical Transports, SAND84-0646, Sandia National Laboratories, Albuquerque, New Mexico.

Rogers, V., and C. Hung, 1987. PATHRAE-EPA: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, EPA 520/1-87-028, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington DC.

Rogers, V., and C. Hung, 1987. PRESTO-EPA-CPG: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, EPA 520/1-87-026, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington DC.

Rogers, V., and C. Hung, 1987. PRESTO-EPA-DEEP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, EPA 520/1-87-025, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington DC.

Rogers, V., and C. Hung, 1987. PRESTO-EPA-BRC: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, EPA 520/1-87-027, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington DC.

Schmitz, W.J. and W.R. Holland, 1986. "Observed and Modeled Mesoscale Variability Near the Gulf Stream and Kuroshio Extension," J. Geophys. Res. 91:9624-9638.

Semtner, A.J. and R.M. Chervin, 1988. "A Simulation of the Global Ocean Circulation with Resolved Eddies," J. Geophys. Res. 93:15502-15522

Shuman, R., et al., 1988. Performance Assessment for Low-Level Waste Disposal Facilities, NP-5745, Prepared for Electric Power Research Institute, by Rogers and Associates Engineering Corp., Salt Lake City, UT.

Slovic, P. and B. Fischhoff, 1980. "How Safe is Safe Enough." In R.C. Schwing and W.A. Albers, eds., Risk Assessment: How Safe Is Safe Enough?, pp. 181-216, Plenum Press, NY.

Slovic, P., B. Fischhoff, S. Lichtenstein, S. Read, and B. Bombs, 1978. "How Safe Is Safe Enough? A Psychometric Study of Attitudes Toward Technological Risks and Benefits," Policy Sci. 9:127.

Slovic, P., B. Fishhoff, and S. Lichtenstein, 1980. "Facts and Fears: Understanding Perceived Risk." In R.C. Schwing and W.A. Albers, eds. Risk Assessment: How Safe is Safe Enough? pp. 181-216 Plenum Press, NY.

Starmer R.J., L.G. Deering and M.F. Weber, 1988. "Performance Assessment Strategy for Low-Level Waste Disposal Sites," Tenth Annual DOE Low-Level Waste Management Conference, August 30-September 1, Denver, Colorado.

Triglio, G., 1981. Volume Reduction Techniques in Low-Level Radioactive Waste Management, Prepared for U.S. Nuclear Regulatory Commission, Division of Waste Management, by Teknekron, Inc., Berkeley, CA.

U.S. Congress, 1972. Marine Protection, Research, and Sanctuaries Act of 1972, Enacted by P.L.92-532, 86 Stat. 1052, 33 U.S.C. 1401 et seq., and 16 U.S.C. 1431 et seq..

U.S. Congress, 1980. Low-Level Radioactive Waste Policy Act of 1980, P.L. 96-573, as amended by P.L. 99-240.

U.S. Congress, 1986. Low-Level Radioactive Waste Policy Act Amendments of 1985, P.L. 99-240 (H.R. 1083).

U.S. Council for Energy Awareness, 1988. Transporting Low Level Radioactive Waste: Questions and Answers, Washington, DC. October, 1988.

U.S. Department of Energy, 1988. Order 5820.2A, Chapter III, Management of Low-Level Waste, September 26, 1988.

U.S. Environmental Protection Agency, 1977. "40 CFR 227 - Criteria for the Evaluation of Permit Applications for Ocean Disposal of Materials," 42 FR 2476, January 11, 1977.

U.S. Environmental Protection Agency, 1977. "40 CFR 228, Criteria for the Management of Disposal Sites for Ocean Dumping," 42 FR 2482, January 11, 1977.

U.S. Environmental Protection Agency, 1983. "Environmental Radiation Protection Standards for Low-Level Radioactive Waste Disposal: Advanced Notice of Proposed Rulemaking," 48 FR 39563, August 31, 1983.

U.S. Nuclear Regulatory Commission, 1982. "10 CFR 61 - Licensing Requirements for Land Disposal of Radioactive Waste," 47 FR 57446, December 27, 1982.

U.S. Nuclear Regulatory Commission, 1986. "10 CFR 61 Branch Technical Position Statement on Licensing of Alternative Methods of Disposal for Low-Level Radioactive Waste," 51 FR 7806, March 6, 1986.

APPENDIX A

A REVIEW OF AGREEMENTS, STATUES, AND REGULATIONS FOR THE DISPOSAL OF LOW-LEVEL RADIOACTIVE WASTE

1 INTRODUCTION

Low-level radioactive waste (LLW) disposal practices date back more than forty years, over which time both techniques and concerns over environmental effects have varied considerably. Consequently, laws and regulations governing the disposal of LLW have changed. Consideration of current and pending legislative constraints may play a key role in the selection of new LLW disposal options. Applicable international agreements and federal statutes/regulations for ocean dumping, as well as federal and state statutes/regulations governing land-based disposal are reviewed in this report.

2 REGULATIONS FOR OCEAN DISPOSAL OF LOW-LEVEL RADIOACTIVE WASTE

2.1 Background

Disposal of low-level radioactive waste (LLW) in the ocean was first initiated by the United States in 1946. Between 1946 and 1970, the U.S. placed approximately 107,000 waste containers comprising about 4.3×10^{15} Becquerels (Bq) in the Atlantic and Pacific Oceans. Most of this waste was dumped prior to 1962; an Atomic Energy Commission moratorium on ocean disposal licenses significantly reduced the practice (only about 350 packages estimated at 8.5×10^{12} Bq were dumped between 1963-1970) until it was curtailed completely in 1970. The United Kingdom began ocean disposal of LLW in 1949, and was joined by other European nations in 1967 when a cooperative program was organized by the Nuclear Energy Agency (NEA). Since 1949, an estimated total mass of 100,000 metric tons of packaged LLW accounting for about 3×10^{16} Bq has been disposed by European nations in the Northeast Atlantic Ocean. Due to concerns over uncertainties in environmental impacts and widespread social protests, European ocean disposal was suspended in 1983.

Increased awareness of environmental pollution in general, and concern over potential hazards associated with radioactive waste in particular, has led to the promulgation of federal laws and regulations and international agreements pertaining to the disposal of LLW at sea. Since the onset of LLW ocean disposal, the regulatory environment has been a dynamic one, influenced by: (i) a growing scientific understanding of the nature of waste package performance, radionuclide transport mechanisms, and biological/environmental effects and, (ii) a changing social and political climate in the areas of nuclear power and environmental affairs.

2.2 International Laws/Agreements

International concern over potential environmental impacts resulting from ocean disposal of radioactive wastes was initially addressed at the first U.N. Conference on the Law of the Sea (UNCLOS I), held in Geneva in 1958. This conference provided the first codification of the international law of the sea, still in force today. Although unanimity among participating nations over the advisability of ocean disposal was not reached, several general principles were incorporated in Article 25 of the Geneva Convention on the High Seas that laid the groundwork for future accords [1]:

- Every State shall take measures to prevent pollution of the seas from the dumping of radioactive wastes, taking into account any standards and regulations which may be formulated by the competent international organizations.

- All States shall cooperate with the competent international organizations in taking measures for the prevention of pollution of the seas or air space above, resulting from any activities with radioactive materials or other harmful agents.

The need for international research, supervision and control of waste disposal was further recognized in the recommendation to the International Atomic Energy Agency (IAEA) and other organizations to [2]:

> . . .pursue whatever studies and take whatever action is necessary to assist States in controlling the discharge or release of radioactive materials to the sea, in promulgating standards, and in drawing up internationally acceptable regulations to prevent pollution of the sea by radioactive materials in amounts which would adversely affect man and his marine resources.

Following the recommendations of UNCLOS I, the IAEA assembled a panel of experts on ocean disposal of radioactive wastes which issued a report of its findings in 1961. The resultant document (sometimes referred to as the Brynielsson Report) [3] recommended a ban on high-level radioactive waste disposal at sea and that low-level waste disposal be conducted under controlled and specified conditions, on a site-specific basis. It also proposed that wastes be certified and internationally registered, that dump-sites be designated, and that operational procedures for disposal be formulated.

In 1967 the NEA of the Organization for Economic Cooperation and Development (OECD) agreed to coordinate international disposal of LLW at sea. It set up the Multilateral Consultation and Surveillance Mechanism for Sea Dumping of Radioactive Waste to provide a disposal system at a Northeast Atlantic site. Guidelines on packaging and disposal methods were issued in conjunction with the IAEA recommendations. [4,5] The NEA also called for a one-year notification requirement prior to disposal operations, an internationally supervised surveillance program, and a system of permanent records to be kept by the International Maritime Organization (IMO). A coordinated research program to carry out a thorough radiological assessment of the site was initiated in 1980 at the suggestion of the U.S. This Coordinated Research and Environmental Surveillance Program (CRESP), still ongoing, contributed significantly to the 1985 site suitability review. These reviews are required every five years.

The general obligation of nations to preserve the marine environment was reaffirmed at the United Nations Conference on the Human Environment in Stockholm, June 1972. The Stockholm conference was the impetus for the Convention on the Prevention of Marine Pollution by Dumping of Wastes and

Other Matter (better known as the London Dumping Convention, LDC) in November of 1972. The LDC is recognized as the most comprehensive international regulation of marine pollution by dumping undertaken to date [6]. Some fifty nations, including the major maritime countries, are now Contracting Parties to the Convention. Three categories of waste are defined under the LDC: (i) materials prohibited from dumping, including high-level wastes as defined by the IAEA, and listed in Annex I, (ii) materials requiring a special permit for dumping, including low-level wastes, listed in Annex II, and, (iii) all other materials requiring a prior general permit. Permits are issued by appropriate national authorities following the guidance provided in Annex III. Factors to be considered for issuance of permits include waste characteristics (e.g., type and quantity of contained activity, radioactive half-life, toxicity, bioaccumulation, dose-response); disposal site characteristics (e.g., location, depth); disposal methods (e.g., waste packaging, disposal density); effects on other uses of the ocean (e.g., fishing, navigation, other industrial uses, amenities); and availability of alternative land-based disposal options. The LDC also authorized the IAEA to formulate recommendations on low-level waste dumping and requires the reporting of permits (issued by national authorities) to the IMO.

Responding to the LDC, the IAEA first formulated its recommendations in 1974 and revised them in 1978 [7]. These recommendations call for detailed environmental and ecological assessments to be submitted to IMO with each permit application. The assessment and permit procedure are designed to ensure that ocean disposal of low-level waste will involve no unacceptable degree of hazard to human health, harm to living resources and marine life, damage to amenities, or interference with other legitimate uses of the sea. To this end, limits on allowable releases of radioactivity at a given site and for a finite ocean basin are specified (summarized in Table I) and the need for suitable packaging to optimize containment is emphasized. Specific site selection criteria are given as follows:

- Sites should be located between 50°S and 50°N latitudes to avoid sources of bottom water (characterized by strong vertical mixing) and areas of high biological productivity.

- Depth should be $\geq$ 4000 m (biological, chemical, physical and topographical gradients are generally low, bottom water circulation is slow, and organic carbon in the sediments tends to be low).

- Sites should be away from continental margins (to avoid regions of high biological productivity, and active resource exploration/exploitation), and areas where geologic hazards (e.g., submarine slides, volcanos and earthquakes) reduce stability.

- Areas encompassing potential seabed resources, trans-oceanic cables, commercial fishing trawlers, or areas that are difficult to navigate, should be avoided.

- Designated sites should be as small as possible with a maximum size of 10,000 km^2. Precise site coordinates are required and navigational aids for relocation are desirable.

- The number of disposal sites should be strictly limited.

- Features such as submarine canyons that can adversely impact the rate of exchange between deep and surface waters near the continental shelf should be avoided.

- Sites should be selected for convenience of disposal operations and to minimize navigational traffic.

- Bottom current shear stress should not exceed critical erosional shear stress to prevent high rates of resuspension and erosion of sediments.

- Monitoring of the area near the disposal site should be carried out to examine waste package and site performance.

Another international treaty that could affect ocean disposal of radioactive waste is the Convention on the Law of the Sea (LOS). The U.S. has thus far declined to sign the agreement, but it has been signed by 132 nations and ratified by 11. If 60 nations ratify the document, the LOS treaty will govern participants' international maritime affairs. Although the LOS convention does not directly address the issue of radioactive waste disposal, it does strongly emphasize protection of the marine environment. Article 194 declares, for example, that "States shall take ...all measures...necessary to prevent, reduce and control pollution of the marine environment from any source..." Articles 232 and 235 hold a nation financially responsible for damages caused by pollution from its sources.

The U.S. policy as negotiated by the Department of State (DOS) has been that any substantive decisions with respect to ocean disposal of low-level radioactive waste must be based on sound technical and scientific grounds. In 1986, Resolution (28):10 to the LDC called for a voluntary moratorium until certain scientific/technical, as well as social, political and economic issues were addressed. To accomplish this, in 1987, an Intergovernmental Panel of Experts on Radioactive Waste Disposal at Sea was formed, consisting of two working groups: Working Group I is addressing social, political and economic questions, and Working Group II is addressing scientific and technical issues.

The U.S. is participating in the deliberations of both working groups. However, particular focus has been given to assisting the international organizations charged with resolving the scientific and technical questions identified in Working Group II. A key question to be addressed internationally is the scientific and technical issues related to the comparative ocean and land-based options for disposal of LLW and the associated costs and risks. The IAEA was recently asked to address these issues pursuant to its status as the internationally recognized cognizant authority. Additional details of current IAEA activities and U.S. involvement in comparing risks of land-based and ocean disposal options are included in Appendix A.

2.3 Federal Laws/Regulations

In the U.S., the National Environmental Policy Act (NEPA) enacted in 1970 was designed to focus attention on environmental impacts of federal projects and encourage greater public participation in environmental affairs. Its stated purpose was to:

> declare a national policy which will encourage productive and enjoyable harmony between man and his environment; to promote efforts which will prevent or eliminate damage to the environment and biosphere and stimulate the health and welfare of man; to enrich the understanding of the ecological systems and natural resources important to the Nation.

In keeping with environmental policy set forth by NEPA, a number of specific environmental laws were passed that cover air quality (Clean Air Act), water quality (Federal Water Pollution Control Act, Safe Drinking Water Act, and Marine Protection, Research and Sanctuaries Act), and toxic materials/waste disposal (Toxic Substances Control Act, Resource Conservation and Recovery Act, and Comprehensive Environmental Response, Compensation and Liability Act).

The Marine Protection, Research and Sanctuaries Act of 1972 (MPRSA) recognized that unregulated dumping of radioactive and non-radioactive materials into the ocean can endanger human health, welfare, and amenities, and the marine environment, ecosystems, and resources [8]. It establishes as U.S. policy to:

> regulate the dumping of all types of materials into ocean waters and to prevent or strictly limit the dumping into ocean waters of any material which would adversely affect human health, welfare, or amenities, or the marine environment, ecological systems or economic penalties.

Dumping of high-level radioactive waste and radiological, chemical, or biological warfare agents are prohibited. The Environmental Protection Agency (EPA) is empowered to issue waste dumping permits within the policy guidelines of the Act. Minimum criteria for evaluation of dumping permit applications as specified in Section 102 of MPRSA are listed in Table A-2.

A number of regulations pursuant to MPRSA have been issued by EPA. EPA Ocean Dumping Regulations of 1977, found in 40 CFR 220 establish the scope of these regulations and define five categories of ocean dumping permits [9]:

- General permits are issued for dumping of certain materials that have minimal adverse environmental impact and are generally disposed of in small quantities or for specific classes of materials that must be disposed of in emergency situations.

- Special permits, valid for three years, are issued for materials that meet 40 CFR 227 criteria (see below).

- Emergency permits are issued for materials specified in 40 CFR 227.6 that are present in concentrations above trace levels, provided that an imminent threat to human health and safety is demonstrated and no feasible alternative options exist.

- Interim permits were issued for some wastes prior to April, 1978 during the regulatory phase-in period.

- Research permits are issued on a short-term basis (18 months) for disposal of materials associated with a research project whose merits outweigh potential environmental or other damage.

EPA permit application requirements and evaluation criteria are contained in 40 CFR 221 and 227, respectively [10,11]. These requirements include demonstrating compliance with environmental impact criteria (Subpart B), absence of acceptable alternatives (Subpart C), and lack of adverse effects on esthetic, recreational, or economic resources (Subparts D and E). LLW must be containerized (40 CFR 227.7) and meet the following conditions (40 CFR 227.11):

- the materials to be disposed of decay, decompose or radiodecay to environmentally innocuous materials within the life expectancy of the containers and/or their inert matrix;

- materials to be dumped are present in such quantities and are of such nature that only short-term localized adverse effects will occur should the containers rupture at any time;

- containers are dumped at depths and locations where they will cause no threat to navigation, fishing, shorelines, or beaches.

Criteria for managing waste disposal sites in the ocean are contained in 40 CFR 228 [12].

A January 1983 amendment to MPRSA imposed a two-year moratorium on ocean disposal of low-level radioactive wastes and a more stringent, supplementary set of permit requirements following the moratorium [13]. For a LLW ocean disposal permit, it must be demonstrated that:

- the proposed dumping is necessary to conduct research either i) on new technology related to ocean dumping, or ii) to determine the degree to which the dumping of such substance will degrade the marine environment;

- the scale of the proposed dumping is limited to the smallest amount of such material and the shortest duration of time that is necessary to fulfill the purposes of the research, such that the dumping will have minimal adverse impact upon human health, welfare, and amenities, and the marine environment, ecological systems, economic potentialities, and other legitimate uses;

- after consultation with the Secretary of Commerce, the potential benefits of such research will outweigh any such adverse impact; and

- the proposed dumping will be preceded by appropriate baseline monitoring studies of the proposed dump site and its surrounding environment.

In addition to complying with other requirements, a Radioactive Material Disposal Impact Assessment (RMDIA) must be included with the permit application. Information to be included in the assessment is listed in Table A-3. Under the amended law, final approval for LLW ocean disposal permits is also required by a joint resolution of Congress. EPA is currently developing updated criteria for ocean disposal of LLW to be included in the Agency's revisions to the 1977 Ocean Disposal Regulations.

Another recently enacted amendment to MPRSA known as the "Ocean Dumping Ban Act of 1988" bans dumping of sewage sludge and industrial waste into ocean waters after December 31, 1991. The amendment defines industrial waste as: "any solid, semisolid, or liquid waste generated by a manufacturing or processing plant, other than an excluded material." It is not yet clear whether this definition will be interpreted to include low-level radioactive waste. Previous congressional actions involving LLW, however, contained explicit reference to it as such, indicating that the ban on dumping industrial waste was probably not intended to cover LLW.

3 REGULATIONS FOR LAND-BASED DISPOSAL OF LLW

3.1 Federal Laws and Regulations

Regulations and responsibilities for land-based disposal of LLW are outlined in two federal laws: (i) the Atomic Energy Act of 1954, as amended, 42 U.S.C. 2011 et seq., and (ii) the Low-Level Radioactive Waste Policy Act of 1980 (P.L. 96-573) as amended by the Low-Level Radioactive Waste Policy Act Amendments of 1985 (P.L. 99-240).

The Atomic Energy Act of 1954 (AEA), as amended by the Energy Reorganization Act of 1974, empowers the Nuclear Regulatory Commission (NRC) to license and regulate commercial LLW disposal and grants the Department of Energy responsibility for LLW generated by defense-related activities. It also provides states the opportunity to license and regulate disposal practices by authorizing special agreements between NRC and individual states. Under Section 274.b of the Atomic Energy Act, a state can assume licensing and regulatory authority and become an "Agreement State" if it agrees to enact and uphold statutes comparable to existing NRC regulations.

State involvement in LLW disposal policy was expanded by the Low-Level Radioactive Waste Policy Act of 1980 (LLRWPA) and its amendments of 1985 (LLRWPAA), which mandated state responsibility for developing new land-based disposal capacity. Under the LLRWPA, individual states or groups of states that form compacts are responsible for disposal of all LLW generated within their borders, except for waste produced by federal facilities. The LLRWPAA, which superceded the 1980 law, went even further in spelling out specific incentives for implementation. After a phase-in period ending in 1992, states or compacts can refuse to accept waste generated elsewhere, or may impose severe economic penalties if they choose to accept waste from other regions/states. Thus the major thrust of the LLRWPAA was to decentralize shallow land burial and "encourage" the development of additional disposal capacity. In addition to its main provisions the LLRWPAA:

- allows the NRC to intercede and grant emergency access to a regional disposal facility if an immediate and serious threat to public health and safety or security occurs that cannot be mitigated by alternative means;

- requires generators of LLW to implement volume reduction to the maximum extent practicable or face economic penalties;

- directs the Department of Energy (DOE) to: (i) provide states/compacts with technical assistance in the areas of site selection, alternative technologies for disposal, volume reduction and management techniques to maximize disposal capacity, transportation issues, health and safety considerations for storage, shipment, and disposal, and development of a computerized data base to monitor LLW disposal management; and (ii) to provide financial assistance through 1993 to implement the Act;

- requires the DOE Secretary to prepare yearly progress reports to Congress on status of LLW disposal;

- requires NRC, in consultation with the States and other interested parties, to: (i) identify alternative methods for LLW disposal, (ii) develop application requirements for alternative methods, and (iii) publish technical criteria for licensing of alternative methods;

- directs NRC to establish criteria for evaluating LLW that contains radionuclides in sufficiently low concentrations to be considered "below regulatory concern".

Under authority of the Atomic Energy Act of 1954, NRC implemented a comprehensive set of LLW shallow land disposal regulations in 1983 (10 CFR 61) aimed both at waste generators and burial site operators [14]. Major portions of these requirements are reviewed below.

Responding to needs and requests of the public, Congress, industry, NRC and other federal agencies, the purpose of 10 CFR 61 was to provide licensing procedures, performance objectives and technical requirements for licensing of LLW disposal sites on a nationally consistent basis. It specifically addresses procedural and technical requirements of near-surface disposal (i.e., $\leq$ 30 m) and expressly does not apply to other methods such as burial below 30 m or ocean dumping. Objectives of 10 CFR 61 include insuring protection of the public from radioactive releases, protection of individuals from inadvertent intrusion at the site after closure, radiological protection for operators, and long-term site stability.

The regulatory approach taken to achieve these objectives is to maintain stability of both waste packages and disposal site. Since leaching of radionuclides by natural precipitation is the main environmental pathway leading to potential human exposures (via ground water contamination), stability is critical. For waste packages, physical integrity is important to prevent slumping or collapse of the disposal trench that can lead to water intrusion, and to maintain minimum surface areas subject to leaching; chemical stability reduces potential for interactions that can enhance leachability. Degradation of site stability (e.g., by subsidence) can seriously alter important site characteristics such as permeability and drainage. Recognizing that radioactive hazards are both isotope- and concentration-specific, NRC developed a waste classification scheme that establishes three categories of waste:

- Class A: Minimally contaminated waste which because of its physical form (e.g., trash and other bio-degradables) is generally segregated from other waste at disposal;

- Class B: Contain higher concentrations of short-lived radionuclides, must meet more rigorous stability requirements;

- Class C: Contain significant concentrations of long-lived radionuclides, thus must meet stability criteria and additional measures against inadvertent intrusion.

Wastes are classified based on consideration of specific concentrations of long- and short-lived isotopes, as shown in Table A-4. If concentrations exceed the maximum specified for Class C, the waste is considered unsuitable for disposal by shallow-land burial. For mixtures of radionuclides, classification is determined by summing ratios of concentrations/limits for each isotope as described in 10 CFR 61.55 (a) (7). Wastes containing radionuclides other than those listed in Table A-4 are considered Class A. Disposal requirements vary by classification, but all wastes must meet the following minimum criteria designed to facilitate safe handling:

- Waste must not be packaged for disposal in cardboard or fiberboard boxes.

- Liquid waste must be solidified or, if Class A, may be packaged in sufficient absorbent material to absorb twice the volume of the liquid.

- Solid waste containing liquid shall contain as little free standing and noncorrosive liquid as is reasonable achievable, but in no case shall the liquid exceed 1 % of the volume.

- Waste must not be readily capable of detonation or of explosive decomposition or reaction at normal pressures and temperatures, or of explosive reaction with water.

- Waste must not contain, or be capable of generating, quantities of toxic gases, vapors, or fumes harmful to persons transporting, handling, or disposing of the waste. This does not apply to radioactive gases packaged as described below.

- Waste must not be pyrophoric. Pyrophoric materials contained in waste shall be treated, prepared, and packaged to be nonflammable.

- Waste in a gaseous form must be packaged at a pressure that does not exceed 1.5 atm at 20°C. Total activity must not exceed 100 Ci per container.

- Waste containing hazardous, biological pathogenic, or infectious material must be treated to reduce to the maximum extent practicable the potential hazard from the non-radiological materials.

In addition to these minimum requirements, Class B and C wastes must also meet three criteria intended to maintain waste and site stability and reduce exposure to an inadvertent intruder by ensuring waste forms are still identifiable. These are:

- Waste must maintain structural stability under expected disposal conditions such as weight of overburden and compaction equipment, presence of moisture, and microbial activity, and internal factors such as radiation effects and chemical changes. Structural stability can be provided by the waste itself, processing to a stable form, or placing the waste in a disposal container or structure that provides stability after disposal.

- Class B and C liquid wastes must be converted into a form that contains as little free standing and noncorrosive liquid as is reasonable achievable, but in no case can the liquid exceed 1% of the volume of the waste when in a disposal container designed to ensure stability, or 0.5% of the volume of the waste for waste processed to a stable form.

- Void spaces within waste and between waste and its package must be reduced to the extent practicable.

Further clarification on stability requirements was issued by NRC in the form of a Branch Technical Position on Waste Form [15], which identified a series of waste form performance tests that could be used to demonstrate long-term stability.

In addition to waste form standards, 10 CFR 61 establishes a set of technical requirements for land disposal facilities. These include specifications for: (i) preliminary site suitability for shallow land burial that cover geology, hydrology, population, etc.; (ii) site design, covering features to protect against water intrusion and ensure stability after closure; (iii) operation and closure, such as segregation of Class A wastes, placement of Class C wastes to guard against inadvertent intrusion, filling of voids, etc.; and (iv) environmental monitoring before, during, and after disposal operations are complete. Radiation exposure standards for the general public included in 10 CFR 61.41 limit releases to the environment by all pathways, such that annual average dose equivalents do not exceed 25 mrem to the whole body, 75 mrem to the thyroid, and 25 mrem to any other organ.

Department of Energy policy and regulations concerning LLW treatment and disposal are covered in the recently amended DOE Order 5820.2A. [16] Similar to 10 CFR 61, this order establishes a maximum effective dose equivalent of 25 mrem/yr to any member of the public from external exposure to radioactive materials released from DOE LLW disposal facilities, while "reasonable efforts" must be made to maintain releases as low as reasonably achievable (ALARA). Atmospheric releases must meet requirements for hazardous air emissions in 40 CFR 61. Specific requirements for waste performance assessment, generation, minimization, characterization, acceptance criteria, treatment, storage, shipment, and disposal are included. In addition, criteria for disposal site selection, design, operations, closure, and monitoring are discussed.

In August 1983, EPA issued an Advanced Notice of Proposed Rulemaking (ANPRM) stating the agency's intention to develop generally applicable standards for LLW disposal and requesting public comment on its form and content. [17,18]. The ANPRM consists of two parts: (i) 40 CFR 193 encompasses LLW treatment and disposal at DOE and commercial facilities, and (ii) 40 CFR 764 which deals with NARM (naturally occurring and accelerator produced radioactive waste) under authority granted by the Toxic Substances Control Act. NARM wastes are not covered by existing NRC or DOE rules.

40 CFR 193 consists of three subparts that address allowable exposures to the public from LLW management, storage, and disposal facilities, and extend groundwater protection standards to encompass these activities. Subpart A establishes environmental standards for management, processing and storage and Subpart B presents environmental standards for land disposal. Both require that activities be conducted in such a manner that combined, no member of the public in the general environment shall receive an annual effective dose equivalent of more than 25 mrem from all routes of exposure. Subpart A also defines the category of "below regulatory concern" (BRC) for LLW that contain sufficiently low concentrations of radioactivity that their disposal, in combination with all other BRC waste streams, would not expose any member of the public to an annual effective dose equivalent of more than 4 mrem in any year. These wastes are exempt from disposal requirements in 40 CFR 193. Subpart B requires: (i) credit for active institutional controls can only be taken for 100 years after disposal in order to meet exposure standards ; (ii) danger warnings must be posted in the most permanent way practicable to discourage inadvertent intrusion; (iii) sites may not be located in areas where there is a reasonable potential for future exploration of resources; and (iv) monitoring after disposal is required without jeopardizing integrity of isolation. Subpart C proposes groundwater protection requirements for facilities regulated under Subparts A or B. Two options are proposed and differ only with respect to Class II groundwater. The first option stipulates that disposal of radioactive waste cannot result in: (i) any increase in levels of radioactivity for all Class I groundwaters, or (ii) any increase in the levels of radioactivity for Class II groundwaters from high yield aquifers such that an individual can receive more than 4 mrem annual effective dose equivalent by drinking two liters per day of affected groundwater, while the remainder of Class II groundwaters are protected within the 25 mrem/yr limits of Subparts A and B. Class III groundwaters are also protected within the 25 mrem/yr limits of Subparts A and B unless such groundwater is highly interconnected to a higher class of groundwater, in which case the more restrictive standards are extended to the

Class III groundwater. The second groundwater protection option proposed would apply the 4 mrem/yr limit to all Class II groundwater while Class I and Class III groundwater would be protected as in the first option.

Standards for management and disposal of NARM wastes are contained under Subchapter R (Toxic Substances Control Act regulations), 40 CFR 764. Appendix A of the regulation contains a classification system based on the approach used in Appendix A of 10 CFR 61. NARM wastes greater than 2 nCi/gm (with the exception of certain consumer items) would be disposed in an AEA regulated LLW disposal facility and meet shipping manifest requirements given in Appendix B of the regulation.

3.2 State Regulations

By statutory authority of the Agreement State Amendment to the AEA [19], states may assume certain regulatory authority from NRC over radioactive by-products, source materials, and small quantities of special nuclear materials. Under this agreement, a state regulates institutional and industrial radioactive waste generators and commercial LLW disposal sites, but does not have jurisdiction over nuclear power plants or federal facilities. Alternatively, an NRC policy statement issued in 1981 allows states to license and regulate LLW disposal facilities only, by seeking to become a Limited Agreement State [20]. In either case, to be eligible a state must pass enabling legislation and have an adequate program (compatible with NRC regulations) to protect the public health, safety and the environment. States are also required to have staff with training and expertise in the areas of radiation protection, State law, regulations and procedures. Twenty-nine states are currently classified as Agreement States. These are listed in Table A-5.

LLW disposal activities that can be regulated by states include site selection, leasing/contracting of site to an operator, facility licensing, operations and closure, and long-term care following closure. State responsibilities vary depending on whether the state is a host state, i.e., provides disposal capabilities within its borders, and whether it is an Agreement State, a limited Agreement State, or a non-agreement state. Host state responsibilities are summarized in Table A-6 by disposal activity and type/status of state agreement [21].

The AEA as amended in 1959 requires agreement states to meet or exceed federal standards in regulating LLW disposal. [22] Some states have gone beyond federal standards by expressly prohibiting certain disposal options in the relevant enabling legislation. For example, six of the nine regional compacts (representing 28 states) and four unaffiliated states have all passed measures banning the use of shallow-land burial for LLW disposal [23]. The New York State Low-Level Radioactive Waste Management Act includes the following passage concerning allowable disposal options [24]:

> ...the Department [of Environmental Conservation] shall publish draft regulations which specify the criteria for siting permanent disposal facilities and shall promulgate final regulations... specific to the types of disposal methods which may be employed at a permanent disposal site and shall include criteria for: (a)

> above ground, engineered, monitored disposal; (b) underground mined repository disposal; and (c) where practicable, other disposal methods for which there are applicable regulations *but in no event including shallow land burial* [emphasis added].

In this case, shallow land burial is specifically defined as "emplacement of low-level radioactive waste in or within the upper 30 meters of the surface of the earth in trenches, holes, or other excavations in which only soil provides structural integrity, a barrier to migration of low-level radioactive waste from or subsurface water into such excavation, or in a manner that fails to allow during the institutional control period for monitoring and control of releases of radioactivity." In other words, near-surface disposal will be allowed only if certain engineering controls are incorporated to maintain structural integrity, minimize water intrusion and leaching, and monitor site performance.

Table A-1. Radioactive waste release limits for ocean disposal

Radioactive Waste Type	Release Rate Limits (Bq/yr) Single Dumpsite	Finite Ocean Basin
alpha emitters[a]	3.7×10^{15}	3.7×10^{15}
beta and gamma[b] ($t_{1/2} \geq 0.5$ yr)	3.7×10^{17}	3.7×10^{18}
beta and gamma[c] ($t_{1/2} \leq 0.5$ yr)	3.7×10^{21}	3.7×10^{22}

[a] Limited to 3.7×10^{14} Bq/yr for ^{226}Ra and supported ^{210}Po.
[b] Excluding ^{3}H. Includes beta and gamma emitters of unknown $t_{1/2}$.
[c] Including ^{3}H.
Source: Reference 5

Table A-2. Minimum criteria for evaluation of ocean disposal permits [a]

- Need for proposed dumping.
- Effect of dumping on human health and welfare, including economic, esthetic, and recreational values.
- Effect of dumping on fisheries resources, plankton, fish, shellfish, wildlife, shore lines and beaches.
- Effect of dumping on marine ecosystems, particularly with respect to: i) the transfer, concentration, and dispersion of such material and its by-products through biological, physical and chemical processes, ii) potential changes in marine ecosystem diversity, productivity, and stability, and iii) species and community population dynamics.
- Persistence and permanence of the effects of dumping.
- Effect of dumping particular volumes and concentrations of materials.
- Appropriate locations and methods of disposal or recycling, including land-based alternatives and probable impact of requiring use of alternate locations or methods upon considerations affecting the public interest.
- Effect on alternate uses of oceans such as scientific study, fishing, and other living resource exploitation, and nonliving resource exploitation.
- In designating recommended sites, EPA shall utilize wherever feasible, locations beyond the Continental Shelf.

[a] Adapted from Reference 6

Table A-3. Minimum requirements for Radioactive Material Disposal Impact Assessment [a]

- Listing of all radioactive materials in each container to be disposed, the number of containers to be dumped, the structural diagrams of each container, the number of curies of each material in each container, and the exposure levels (in rems) at the inside and outside of each container;

- An analysis of the environmental impact of the proposed action on human health and welfare and marine life, at the site where the applicant desires to dispose of the material;

- Any adverse environmental effects at the site which cannot be avoided should the proposal be implemented;

- An analysis of the resulting environmental and economic conditions if the containers fail to contain the radioactive waste material when initially deposited at the specific site;

- A plan for the removal of the disposed nuclear material if the container leaks or decomposes;

- A determination by each affected State whether the proposed action is consistent with its approved Coastal Zone Management Program;

- An analysis of the economic impact upon other users of marine resources;

- Alternatives to the proposed action;

- Comments and results of consultation with State officials and public hearings held in the coastal States that are nearest to the affected areas;

- A comprehensive monitoring plan to be carried out by the applicant to determine the full effect of the disposal on the marine environment, living resources, or human health. The plan shall include, but not be limited to, exterior monitoring of container radiation, taking of water, sediment, fish and benthic animal samples, and the acquisition of such other information as the [EPA] Administrator may require;

- Any other information the [EPA] Administrator may require in order to determine the full effects of such disposal.

[a] Source: Reference 11

Table A-4. NRC waste classification scheme contained in 10 CFR 61.55 [a]

Radionuclide	Maximum concentrations, Ci/m^3 Class A	Class B	Class C
Long-lived isotopes			
C-14	0.8	-	8.
C-14 (in activated metal)	8.	-	80.
Ni-59 (in activated metal)	22.	-	220.
Nb-94 (in activated metal)	0.02	-	0.2
Tc-99 (in activated metal)	0.3	-	3.
I-129	0.008	-	0.08
Alpha TRUs [b]	10. [c]	-	100. [c]
Pu-241	350. [c]	-	3500. [c]
Cm-242	2,000. [c]	-	20,000. [c]
Short-lived isotopes			
Total of all nuclides with $T_{1/2}$ < 5 yrs	700.	d	d
H-3	40.	d	d
Co-60	700.	d	d
Ni-63	3.5	70.0	700.
Ni-63 (in activated metal)	35.	700.0	7,000.
Sr-90	0.04	150.	7,000.
Cs-137	1.	33.	4,600.

[a] Based on Tables 1 and 2 and Section 61.55 in Ref. 12.
[b] Alpha emitting transuranic elements with $T_{1/2}$ > 5 yrs.
[c] Concentration in nCi/g.
[d] No limits established for these nuclides in Class B or C wastes. Practical considerations (e.g., external radiation, heat generation) will limit concentrations.
These wastes are Class B unless concentrations of other nuclides in table make them Class C.

Table A-5. Agreement States as of October 1988 [a]

Alabama
Arizona
Arkansas
California
Colorado
Florida
Georgia
Idaho
Illinois
Iowa
Kansas
Kentucky
Louisiana
Maryland
Mississippi
Nebraska
Nevada
New Hampshire
New Mexico
New York
North Carolina
North Dakota
Oregon
Rhode Island
South Carolina
Tennessee
Texas
Utah
Washington

[a] Source: Reference 25

Table A-6. State regulatory responsibilities for LLW disposal

Site Selection -

All Host States can:
- influence siting through zoning laws, land use and siting criteria and procedures.
- approve or reject a new disposal site.

Lease/Contract for Disposal Facility -

All Host States can:
- select operator.
- require operator conform to license; enforce compliance.
- set additional requirements for records, reporting systems, security, buffer zone, and closure etc.
- as landlord, exercise oversight and management responsibility, within radiological and safety guidelines.
- require payment of local business taxes, payment in lieu of State taxes, incentives to State and local communities, fees and surcharges, financial arrangements for closure, extended care, insurance, surety (consistent with and in addition to 10 CFR 61).

Licensing of Facilities -

Agreement States can:
- license treatment, storage and disposal facilities that are not part of facilities licensed only by NRC (reactors and federal facilities).
- regulate operations, monitoring, radiation control, performance objectives, site design, assurances of adequate financing for operations, closure, and stabilization (compatible with NRC regulations).
- when provided in State law, conduct environmental reviews necessary for licensing.

Limited Agreement States can:
- license disposal facilities that are not part of those licensed only by NRC. (NRC licenses treatment and storage facilities)
- regulate disposal site operations, monitoring, radiation control, performance objectives, site design, assurance of adequate financing for operations, closure, and stabilization.
- when provided in State law, conduct environmental reviews necessary for licensing.

Non-Agreement States can:
- provide review and input to NRC licensing process particularly under NEPA, and can be party to NRC licence hearings if held; conduct State and local hearings. NRC is responsible for licensing and regulation of disposal facilities (specifies site operations, monitoring, radiation control, performance objectives etc.)

Table A-6. cont.

Operations and Closure of Facilities -

Agreement States can:

- regulate, monitor, inspect and enforce regulations for treatment, storage and disposal facilities.

Limited Agreement States can:

- regulate, monitor, inspect and enforce regulations for disposal facilities.
- pursuant to agreement with NRC (under section 274i of the AEA, inspect NRC licensed facilities, notify NRC of violations, and enforce requirements applicable under state law. NRC regulates, monitors, inspects, and enforces regulations for treatment and storage facilities.

Non-Agreement States can:

- pursuant to agreement with NRC (under section 274i of AEA), inspect NRC licensed facilities, notify NRC of violations, and enforce requirements applicable under state law. NRC regulates, monitors, inspects, and enforces regulations for treatment and storage facilities.

Long-Term Care -

All Host States can:

- carry out long-term institutional care responsibilities under terms of license and operator's lease or contract; or
- request consideration by the federal government to carry out long-term care of the facility.

REFERENCES

1. United Nations Conference on the Law of the Sea (UNCLOS), Convention on the High Seas, 13 U.S.T. 2312, TIAS No. 5200, 450 U.N.T.S. 82, entered into force September 30, 1962, U.N. Conference on the Law of the Sea, Geneva, 1958.

2. United Nations Conference on the Law of the Sea (UNCLOS), Resolutions Adopted by the U.N. Conference on the Law of the Sea, Pollution of the High Seas by Radioactive Materials, 450 U.N.T.S. 58, U.N. Conference on the Law of the Sea, April 27, 1958.

3. International Atomic Energy Agency, Radioactive Waste Disposal into the Sea, Safety Series No.5, International Atomic Energy Agency, Vienna, 1961.

4. Nuclear Energy Agency, Guidelines for Sea Dumping Packages of Radioactive Waste, Revised Version, NEA, Organization for Economic Cooperation and Development, Paris, April 1979.

5. Nuclear Energy Agency, Recommended Operational Procedures for Sea Dumping of Radioactive Waste, NEA, Organization for Economic Cooperation and Development, Paris, April 1979.

6. Nuclear Energy Agency, Seabed Disposal of High-Level Radioactive Waste - A Status Report on the NEA Coordinated Research Programme, NEA, Organization for Economic Cooperation and Development, Paris, 1984.

7. International Atomic Energy Agency, Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter. The IAEA Revised Definition and Recommendations of 1978 Concerning Radioactive Wastes and Other Radioactive Matter, Referred to Annex I and II of the Convention. Information Circular INFCIRC/205/Add.1/Rev.1, International Atomic Energy Agency, Vienna, 1978.

8. Marine Protection, Research, and Sanctuaries Act of 1972, Enacted by P.L.92-532, 86 Stat. 1052, 33 U.S.C. 1401 et seq., and 16 U.S.C. 1431 et seq., October 23,1972.

9. 40 CFR 220.3, Categories of Permits, 42 FR 2468, January 11,1977.

10. 40 CFR 221, Applications for Ocean Dumping Permits Under Section 102 of the Act, 42 FR 2470, January 11, 1977.

11. 40 CFR 227, Criteria for the Evaluation of Permit Applications for Ocean Disposal of Materials, 42 FR 2476, January 11, 1977.

12. 40 CFR 228, Criteria for the Management of Disposal Sites for Ocean Dumping, 42 FR 2482, January 11, 1977.

13. Amendment to MPRSA contained in P.L. 97-424, January 6, 1983.

14. 10 CFR 61, Licensing Requirements for Land Disposal of Radioactive Waste, 47 FR 57446, December 27, 1982.

15. U.S. Nuclear Regulatory Commission, Branch Technical Position on Waste Form, U.S. NRC, Washington, DC, May 1983.

16. U.S. Department of Energy, Order 5820.2A, Chapter III, Management of Low-Level Waste, September 26, 1988.

17. U.S. Environmental Protection Agency, Environmental Radiation Protection Standards for Low-Level Radioactive Waste Disposal: Advanced Notice of Proposed Rulemaking, 48 FR 39563, August 31, 1983.

18. Gruhlke, J.M., F.L. Galpin, W.F. Holcomb, and M.S. Bandrowski, USEPA's Proposed Environmental Standards for the Management and Land Disposal of LLW and NARM Waste, Presented at Waste Management '89, Tucson, AZ, February 26 - March 2, 1989.

19. Agreement State Amendment to Atomic Energy Act, contained in P.L. 86-373.

20. U.S. Nuclear Regulatory Commission, Criteria for Guidance of States and NRC in Discontinuance of NRC Regulatory Authority and Assumption Thereof by States Through Agreement, 46 FR 7540, January 23,1981.

21. Brenneman, F.N., and S.N. Salomon, The Role of the State in the Regulation of Low-Level Radioactive Waste, NUREG-0962, U.S. Nuclear Regulatory Commission, Washington, DC, March, 1983.

22. Burns, M.E., and W.H Briner, Setting the Stage, Chapter in Low-Level Radioactive Waste Regulation: Science, Politics, and Fear, M.E. Burns, ed., Lewis Publishers, Chelsea, MI, 1988.

23. Salomon, S.N., U.S. NRC Office of State Programs, Internal NRC Memo Re: Current Status of Each State in Providing Disposal of LLW, August 16, 1988.

24. New York State Low-Level Radioactive Waste Management Act, Laws of New York, Chapter 673, July 26, 1986.

25. Salomon, S.N., U.S. NRC Office of State Programs, personal communication, October 12,1988.

Summary of Current IAEA Activities Comparing Land-Based and Ocean Disposal of LLW

The LDC Intergovernmental Panel of Experts on Radioactive Waste Disposal at Sea (IGPRAD) is currently investigating scientific/technical, as well as social, political and economic issues relating to ocean disposal of LLW. The IAEA was asked, as the internationally recognized cognizant authority, to develop a framework for comparison of risks from land and ocean disposal. Current IAEA activities/schedule and U.S. involvement are reviewed in this appendix.

IAEA Safety Series 65 will be used as the main reference in reviewing the comparative assessments made to date. The following studies will also be considered, two of which have been provided by the United States:

1. Assessment of best practicable environmental options (BPEO's) for management of low- and intermediate-level solid/solidified radioactive wastes. DOE-UK/March 1986.
2. Long-term management of the existing radioactive wastes and residues at the Niagara Falls storage site. DOE-USA/April 1986.
3. Disposal of decommissioned defueled naval submarine reactor plants. Department of Navy-USA/May 1984.
4. Comparative environmental and safety assessment of four generic disposal options for the Surrey low-level radioactive wastes. AECL-Canada/April 1986.
5. Studie naar de mogelijkheden voor de verwijdering von uit Nederland ofkomstig laag - en middelactief vast afval anders dan door storten in de Atlantische Oceaan. CHVRA-Netherlands/March 1983.

The second and third documents were submitted for consideration by the U.S. in 1987 at the first meeting of the LDC IGPRAD. Two additional documents were submitted for consideration in September 1988, at the second IGPRAD panel meeting, one of which refers to the study currently being conducted by EPA. These are:

1. A U.S. study of risk and uncertainty comparisons associated with the disposal of materials containing elevated levels of naturally occurring radionuclides. (The first phase of this study should be available in April 1989.)
2. A comparative assessment by Sweden of land disposal and ocean dumping of nuclear power wastes.

The projected IAEA schedule for completion of tasks is as follows:

Completion Date	Task
August - September 1988	Collection and analysis of documents. Preliminary review - IAEA Secretariat.
March 1969	Consultants' meeting in Vienna to prepare working document (2-3 experts).
July or September 1989	Consultant's meeting (or possibly Advisory Group meeting) in Vienna to review and improve working document (6-8 experts).
Autumn 1989	Technical editing by Secretariat.
December 1989	Submit for publication (IAEA Technical Report Series or Technical Document).
September 1990	Submit report to the London Dumping Convention.

The U.S. plans to provide expertise to the consultant/advisory group deliberations.

APPENDIX B

MODEL REVIEWS - LAND

1 IDENTIFICATION

1.1 *Name:*	PRESTO-EPA-POP
1.2 *Prepared By:*	Oak Ridge National Laboratory, U.S. Environmental Protection Agency, Rogers and Associates Engineering Co.
1.3 *Prepared For:*	U.S. Environmental Protection Agency, Office of Radiation Programs
1.4 *Report Title:*	PRESTO-EPA-POP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code (Vol. 1: Methodology Manual; Vol. 2: Users Manual)
1.5 *Report Number:*	EPA 520/1-87-024-1 and EPA 520/1-87-024-2
1.6 *Report Date:*	December 1987
1.7 *Availability:*	U.S. Environmental Protection Agency
1.8 *Purpose and Scope:*	Evaluate possible health effects from shallow land burial disposal of LLW to assist in the development of environmental standards

2 SUMMARY OF FINDINGS

PRESTO-EPA (Prediction of Radiation Effects from Shallow Trench Operations) is a suite of computer models developed for EPA to evaluate possible health effects from shallow land burial disposal of LLW. It's original objective was the assessment of impacts from varying shallow land burial disposal scenarios to assist in the development of environmental standards. Since PRESTO-EPA-POP was developed first and serves as the basis for other codes in the PRESTO family, it is the focus of this review. PRESTO-EPA-POP assesses radionuclide transport, resultant exposures, and health impacts of a LLW disposal site on a static local population for 1000 years after closure and on the general population residing in the downstream regional basin for an additional 9000 years. The model simulates leaching of nuclides from a waste form, hydrological, hydrogeological, and biological transport, resultant human exposures, and finally assessment of potential human health effects. Exposure scenarios treated by the model include normal release (leaching, spills during operations), human intrusion, and site farming/reclamation. Environmental pathways considered include ground water transport, over-land water flow, erosion, surface water dilution, resuspension, atmospheric transport, deposition, inhalation, and ingestion of contaminated foods and water. Individual and population doses are calculated, as well as doses to intruders and farmers. Cumulative health effects (deaths from cancer) are calculated for the population over the 1000 year period using a life-table approach.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

Separate documentation is available for each of the individual models in the PRESTO suite of models. The two volumes issued for PRESTO-EPA-POP cover theory and background (Volume 1) and specific implementation data along with sample input and output and a listing of the code (Volume 2). Additional information for all of the PRESTO models is contained in EPA's Background Information Document for the Draft Environmental Impact Statement for Proposed Rules [EPA 520/1-87-012-1].

3.2 Hardware Requirements

PRESTO-EPA-POP is written in FORTRAN IV for implementation on an IBM 3081 computer or comparable system, using 850K bytes of memory. Users of non-IBM systems may have to modify the job control language, NAMELIST inputs, and other program segments where character manipulations are used.

3.3 Application

PRESTO is an integrated program designed to run in batch mode. Time required to execute depends on the scope of the problem. For example, evaluation of local and regional basin health effects considering a total of 31 nuclides over a period of 10,000 years takes approximately 7 minutes, whereas the same problem for only 10 nuclides requires less than 2 minutes.

3.4 Level of Expertise Required

In spite of its size and complex calculational abilities, execution of this program appears relatively straightforward. The well documented Users Manual which contains an overview of the code, sections on input and output, an example problem with printout, and a listing of the code, is especially helpful in this regard. Organization of output data is also designed to assist inexperienced users.

4 TECHNICAL CRITERIA

4.1 Peer Review

The Radiation Advisory Committee of EPA's Science Advisory Board (SAB) conducted a peer review of the Background Information Document to the Draft Environmental Impact Statement for Proposed Rules (3/13/85). This document contains a description of the theory and methodolgy used in PRESTO to predict disposal site performance. The SAB found it to be "a reasonable presentation of the potential sources and risks associated with the disposal of low-level radioactive wastes." Deficiencies were cited in the explanation of how the assessment will be used and in descriptions of data and calculational uncertainties. The SAB felt that the 10,000 year time frame was unrealistically long and disagreed with some of the methods and presentation used in the dose response section. Recommendations for additional research were presented. EPA's latest version of the Background Information Document

(6/88) incorporates some of these comments, but a response to the peer review has not been published.

4.2 Verification

External verification was performed for an April 1983 version of PRESTO-EPA by Inter Systems, Inc. (ISI). A quality assurance audit was prepared to check the coding of equations, compare the logic flow with that presented in the documentation, and verify the correctness of calculations and results. All parts of PRESTO-EPA were checked except for the dose conversion/response calculations contained in DARTAB and its subroutines. Discrepencies between the code and documentation and problems with unit conversions, logic flow, and mass balance were identified by ISI. Appropriate changes were incoporated in the code and approved by EPA so that PRESTO-EPA "now performs the calculations as described in the documentation package including ISI's revisions."

4.3 Uncertainty

The issue of uncertainty in risk estimates is not addressed in the model documentation, but is discussed for the PRESTO suite of models in the Background Information Document. Uncertainties are divided into 5 components: (i) source term concentration, (ii) nuclide geosphere transport, (iii) food chain tranport, (iv) human organ dosimetry, and (v) health effects conversion factors. Due to limited time and budget, quantitative analysis of uncertainty is limited to nuclide transport in the geosphere (uncertainties in the other 4 components are discussed qualitatively). Instead of Monte Carlo or other simulation method, uncertainty is estimated using a method proposed by Hung which calculates joint probability density distribution for two successive random variables. A number of simplifying assumptions are made including: (i) a quasi-steady-state approximation that assumes cap failure reaches maximum at the start of the analysis and remains constant, (ii) conversion of a dynamic, numerical ground water model to a steady-state model that can be solved analytically, and (iii) consideration of a humid permeable site only (worst case assumption.) Values for an arbitrary distribution of probability density for each of the parameters in the anlytical ground water equation are estimated based on engineeering judgment. Input parameters that are varied include: trench material, soil, and aquifer Kds, degree of trench cap failure, trench to aquifer distance, site to river basin distance, ground water velocity, and percolation velocity in host soil. Some of the input parameters were not assigned probability density distributions because the expected standard deviations were small enough to treat them deterministically, or the probability density distribution was calculated from other random input parameters.

Uncertainties calculated for geosphere transport were combined with qualitative uncertainty estimates for the other four components using the Theorem of Variance for the joint distribution of random variables. Assuming each component probability density distribution varies log-normally, this method estimates total uncertainty as the sum of the standard deviations of the components.

4.4 *Sensitivity*

Discussion of sensitivity analyses is also included in the Background Information Document. Sensitivity analyses were performed to test the relative impact of single parameters and scenarios consisting of a set of assumed variables. In total, 54 sensitivity test runs, analyzing 30 input parameters were conducted for PRESTO-EPA-POP. An additional 10 test runs were performed to evaluate the health effect conversion factor used by most versions of PRESTO-EPA. Because the measure of impact for PRESTO-EPA-POP is cumulative health effects integrated over a long period of time, few parameters were found to have a significant impact beyond short-term effects when varied. A number of input parameters affected near-term (i.e., <1000 yrs) health effects to the local population, but these effects are but a small fraction of the total cumulative health effects. Input parameters that were judged to be relatively sensitive were those that affected structural stability of the trench cap and leaching out of the trench. The isotope-specific health effects conversion factors were found to be very sensitive to river-flow-to-population ratio, fish consumption rate, and fish bioaccumulation factors.

4.5 *Required Input*

Four sets of input and control data are required by PRESTO-EPA-POP: (i) site- and nuclide-specific data to calculate nuclide concentrations in the environmental transport parts of the program, (ii) subroutine DARTAB control options for processing exposure data, dosimetric data and tabulation of output, (iii) hydrogeologic and meteorologic data for subroutine INFILtration, and (iv) dosimetric and health effects data used by DARTAB. The first three sets of input are supplied by the user; the fourth is generated by the program RADRISK and a copy of this data file is needed for execution of PRESTO-EPA-POP. Site- and nuclide-specific data consist of about 150 variables that describe the physical and hydrogeological characteristics of the dispoal site, data for the biological pathways, and the radionuclide inventory, as well as define the release/exposure scenario for each run. A partial sample of input requirements is included in Table B-1.

4.6 *Output*

Program output is intended to be self-explanatory and does not assume that the user is familiar with code structure. Definitions and descriptive comments are included, where appropriate. Intermediate tabulations are given in addition to final results. Output is organized into 12 sub-sections as summarized below:

1. Summary of Input Data - With the exception of subroutine INFIL, input data are printed as they are read.

2. Input Data Organization - Data are organized and summarized according to data type, transport sub-system or pathway to facilitate review.

3. Radionuclide Summary Tables - Three summary tables are presented that specify initial inventories (in the trench, surrounding soil,

stream, and atmosphere); decay constants; solubility constants; chemical distribution coefficients (K_ds) for the surface soil, trench, underlying soil and aquifer; and nuclide-specific food chain parameters.

4. INFIL Input/Output - Input data and results for the subroutine INFIL.

5. Unit Response Calulations - Nuclide-specific annual transport and soil loss calculations used for each simulation year.

6. Annual Summary Tables - Hydrologic and tranport variables such as status of trench cap, water depth in trench, and trench inventory. as well as nuclide concentrations in key pathways are reported for user-specified years.

7. Radionuclide Concentration Tables - Average concentration, maximum concentration, and year of maximum concentration are given for well and stream.

8. Radionuclide Exposure Tables - Population intake of nuclides, percent attributable to drinking water, nuclide-specific exposure, maximum dose by exposure route, and year of maximum dose are included.

9. DARTAB Control Information - Included are subroutine DARTAB (dose conversion) control data such as critical organs for consideration, nuclide uptake and clearance data, organ dose weighting factors, etc.

10. DARTAB Dose Tables - Individual and collective dose rates by radiation type, organ,and nuclide are presented.

11. DARTAB Fatal Cancer Risk Tables - Individual and collective fatal cancer risk and genetic risks are summarized.

12. Residual Radioactivity Released and Health Effects - Presents the quantity of nuclides released to the basin each millenium during the 10,000 year simultation, aggregated total release of each nuclide, and basin population health effects by nuclide.

Sample output data from PRESTO-EPA-POP are given in Table B-2.

4.7 *Source Term*

PRESTO-EPA-POP can handle up to 40 user-specified radionuclides. Physical and chemical properties of individual waste streams are not considered directly. Rather, these properties are treated generically for the total waste inventory when the user chooses one of five possible leaching options.

4.8 *Scenarios*

Scenarios that are considered include: normal disposal site operations, spillage of waste during operations (with residual contamination remaining on soil surface at closure), resident intruder on site after closure, farming of the site, and eroded trench cap with subsequent atmospheric contamination via suspension of waste/soil mixture. Normal operations cover post-closure impacts to local population assuming ingestion of off-site water and foods and inhalation of downwind air. User can modify scenario by specifying parameters such as population size, location and distance to well, degree of trench cap failure, resuspension rate, etc. Two scenarios that are not included in analysis are: major changes in meteorology or mining of trench contents. Neither of these are routinely considered in assessment of LLW disposal (they are considered in HLW repository performance projections) since probability of occurrence is low especially in the relatively short time periods of interest. The code is structured to consider one scenario/run.

4.9 *Relationship to Regulatory Standards*

Model output includes an estimate of annual mean effective dose equivalents which is compatible with the proposed standards contained in 40 CFR 193. The current standard contained in NRC's 10 CFR 61 does not use the organ weighting factor associated with effective dose equivalent. Federal regulations also specify allowable radionuclide concentrations in air and water. Radionuclide concentration data for surface soil, surface water, atmosphere at spill, atmosphere downwind, and well water are presented for each radionuclide at user-specified time intervals.

5 SCIENTIFIC CRITERIA

5.1 *Theory*

The assessment is performed in two phases: the initial stage calculates health effects to the local population for up to 1000 years and the second stage calculates health effects to the regional basin population over 10,000 years. After 1000 years the local community is assumed to be incorporated in regional basin community, simplifying calculations. The analysis begins after closure. Specified inventories are adjusted by the program to account for decay during operations phase. Time to container failure is user specified.

Complex physical and chemical interactions between nuclides and geologic media in the leaching and groundwater subroutines are treated using the distribution coefficient (K_d). Different K_d values are applied for soil, soil/waste mixture, sub-trench soil, and aquifer.

Leaching options are calculated by submodule LEAOPT using one of five user-specified calculation methods: K_d with waste in total contact with water or partially immersed (wetted fraction is calculated as the ratio of maximum water depth to trench depth); solubility factor to estimate maximum concentrations of nuclides in water, also as either total contact or wetted fraction; and finally, a user-specified average annual fractional release. Fraction release is used for solidified waste. Solubility limits are used when Kd values are not known. Average annual fractional release of the total inventory is specified and the program calculates adsorption effects inside and outside of the waste form before release from trench.

Container effects are accounted for by multiplying concentration in the trench water by a time dependent container fracture factor (CFF). Initially CFF = 0 when conatainers are intact and then increases over time to unity when all containers have failed. Apparantly CFF is the fraction of all containers that have completely failed, not a measure of extent of failure.

Groundwater flow below the trench is vertical to the aquifer and horizontal through the aquifer to a well. Groundwater in the vertical reach is assumed to be saturated or partially saturated. Degree of saturation determines water velocity and retardation factors. Nuclide transport in the aquifer is evaluated using Hung's groundwater transport model which uses an approximate solution to the basic transport equation to avoid extensive computer time needed for numerical solutions. The groundwater model initially neglects the effect of radionuclide transport by dispersion and subsequently compensates for this effect by applying a health effects conversion factor. Calculated concentrations in well water are averaged over the length of the simulation and are used by food chain and human exposure modules of the code for drinking water and cattle feed pathways. Trench waters that overflow can result in overland flow to nearby surface water bodies, percolation down to the aquifer, or sorption on the soil leading to potential resuspension and air transport or delayed downward migration.

Atmospheric transport is handled by a "simplified, compact algorithm suitable for those sites where the population is concentrated into a single, small community." Gaussian transport module called DWNWND is incorporated as a subroutine. For complex population distributions, an externally computed X/Q ratio (e.g., using AIRDOS-EPA) should be used to explicitly specify population and wind rose distribution. Atmospheric source strength is the sum of wind-driven suspension and mechanical disturbance during user specified time interval.

Average concentrations of each nuclide over the assessment period in environmental media, such as well water or the atmoshpere, are used to calculate radionuclide concentrations in foodstuffs. Foodstuff concentrations and average human ingestion and breathing rates are utilized to calculate annual average radionuclide intake/individual in the local population by ingestion and inhalation. These intake data are then used to estimate dose rate and health effects.

5.2 Validation

Model performance predictions have not been compared with actual shallow land disposal site data.

5.3 Treatment of Radioactive Decay Products

To maintain simplicity and reduce computer time and expense, PRESTO does not calculate in-growth of daughter products. In cases where the major dose contribution is from external exposure to short-lived progeny in equilibrium with parent radionuclides and accounting for daughters can significantly impact results, daughter product activity can be externally estimated and added to the initial trench inventory for appropriate nuclides.

5.4 *Underlying Assumptions*

As in any comparative risk model, many simplifying assumptions are required to perform the assessment. Because of the number of assumptions and the magnitude of uncertainties inherent in such an analysis, EPA strongly cautions the user against misuse or misinterpretation of results. For example, results will not be valid if modeling assumptions are not appropriate or significant input parameters such as inventories, site meteorology, hydrology, geology, or demographics, are improperly assigned. The underlying assumptions in each of these areas are explicitly described in the documentation.

5.5 *Pathways*

Hydrologic, atmospheric, and food chain transport pathways are considered. Hydrologic tranport is divided into two main pathways: (i) precipitation initiated leaching carries radionuclides through trench, vertical soil column, and aquifer to a well or stream, and (ii) operational spillage and/or overflow from trench contaminates surface soil, and radionuclides are carried either to an aquifer via deep seepage where ultimately they enter a well or stream, or to a surface water body via leaching. Resultant exposure pathways are either direct via drinking contaminated water, or indirect via dthe food chain (irrigation, plant uptake, and ingestion of contaminated plants, animals, and milk). Atmospheric transport is triggered by suspension of contaminated surface soil or trench soil that has been exposed by erosion. Resultant exposure pathways include direct exposure by immersion and inhalation of respirable contamination, or indirect exposure via the food chain (deposited contamination and ingestion of crops contaminated with deposited radionuclides). Hydrologic and atmospheric environmental transport pathways are illustrated in Figures B-1 and B-2. Other exposure pathways considered include an inadvertent intruder living at the site after closure and off-site exposures due to suspension of particles from on-site farming activities.

5.6 *Dose Conversion and Dose Response*

Dose and health effect calculations are made within the subroutine DARTAB. Three calculations and tabulations for dose rate and dose are performed: (i) dose rate to an individual at a selected location, (ii) dose rate to a mean or average individual, and (iii) collective population dose rate. A weighting factor for the various organs (developed by EPA) was used to combine dose rates in a manner similar to the approach recommended by ICRP 30.

Table B-1. Sample input data for PRESTO-EPA-POP

```
 IPATH=30AS,SAND
1                              PRESTO - A MODEL FOR PREDICTING THE MIGRATION OF RADIOACTIVE WASTES
                                         FROM SHALLOW TRENCH BURIAL SITES
0
                   SITE AREA ..... POP RUN ....    FACILITY CHARACTER


      AAA CONTROL INFORMATION AAA
0          THE BURIAL SITE IS LOCATED AT  SITE AREA
           THE SIMULATION WILL RUN FOR 1000 YEARS AND WILL INCLUDE 33 NUCLIDES
           LEACHING OPTION NUMBER 2 WILL BE USED
           IN YEAR   1  0.20 OF THE CAP WILL BE ASSUMED TO FAIL
                THIS WILL CONTINUE UNTIL 0.30 HAS FAILED IN YEAR  200
           CAP MAY ALSO FAIL BY SURFACE EROSION
           LENGTH OF VERTICAL SATURATED ZONE WILL BE SET TO THE TRENCH TO AQUIFER DISTANCE
           POPULATION INDICATOR IS   1
                GENERAL POPULATION EXPOSURE WILL BE USED TO CALCULATE HEALTH EFFECTS
            0.000 OF IRRIGATION WATER WILL BE GOTTEN FROM WELL
            0.500 OF DRINKING WATER FOR ANIMALS WILL BE GOTTEN FROM WELL
            1.000 OF DRINKING WATER FOR HUMANS WILL BE GOTTEN FROM WELL
            0.000 OF IRRIGATION WATER WILL BE GOTTEN FROM STREAM
            0.000 OF DRINKING WATER FOR ANIMALS WILL BE GOTTEN FROM STREAM
            0.000 OF DRINKING WATER FOR HUMANS WILL BE GOTTEN FROM STREAM
1     AAA TRENCH INFORMATION AAA

           THE TRENCH HAS AN AREA OF  0.5000E+00 SQUARE METERS AND A DEPTH OF  0.2600E+01 METERS
           TRENCH POROSITY IS     0.25
           ANNUAL INFILTRATION FOR THE WATERSHED IS 0.4300 METERS


      AAA AQUIFER INFORMATION AAA

           THE GROUND WATER HAS A VELOCITY OF      27.800 METERS PER YEAR
           TRENCH TO AQUIFER DISTANCE IS     7.3 METERS
           TRENCH TO WELL DISTANCE IS  1610.00 METERS
           WELL TO STREAM DISTANCE IS 3220.00METERS
           THE AQUIFER THICKNESS IS     30.50 METERS
           THE AQUIFER DISPERSION ANGLE IS   0.3000 RADIANS
           POROSITY OF THE AQUIFER REGION IS  0.39000
           POROSITY BENEATH THE TRENCH IS  0.35000
           PERMEABILITY BENEATH THE TRENCH IS    2.200 METERS/YEAR


1     AAA ATMOSPHERIC INFORMATION AAA

           SOURCE HEIGHT IS    1.0 METERS
           VELOCITY OF GRAVITATION FALL IS   0.01 METERS/SECOND
           WIND VELOCITY IS   2.01 METERS/SECOND
           DEPOSITION VELOCITY IS   0.10 METERS/SECOND
           GAUGE DISTANCE FROM SOURCE IS 17700.00 METERS
           LID HEIGHT IS   300.00 METERS
           HOSKER ROUGHNESS FACTOR IS   0.01
           TYPE OF STABILITY FORMULATION IS  1
           STABILITY CLASS IS  4
           FRACTION OF TIME WIND BLOWS TOWARD POPULATION IS  0.484000
           RESUSPENSION FACTOR PARAMETERS         0.1000E-05     -0.1500E+00       0.1000E-10
           FROM YEAR    0 TO YEAR    0 THE RESUSPENSION RATE DUE TO MECHANICAL DISTURBANCES WILL BE  4.0000E-04
                THIS WILL OCCUR DURING  2.40 OF EACH YEAR


1     AAA NUCLIDE INFORMATION AAA

                                              INFORMATION ON INDIVIDUAL NUCLIDES
```

NUCLIDE	AMT IN TRENCH CI	SPILLAGE CI	STREAM AMT CI	AIR CONCEN CI/M**3	DECAY CONST 1/Y	SOLUBILITY CONST G/ML	DECAY CORRECTION FACTOR
H-3	6.1017E-01	6.1017E-03	0.0000E+00	0.0000E+00	5.6500E-02	0.0000E+00	6.1633E-01
C-14	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	1.2100E-04	0.0000E+00	1.0000E+00
CR-51	5.5223E+00	5.5223E-02	0.0000E+00	0.0000E+00	9.1400E+00	0.0000E+00	5.5781E+00
MN-54	9.4212E-02	9.4212E-04	0.0000E+00	0.0000E+00	8.0900E-01	0.0000E+00	9.5164E-02
FE-55	2.1833E-01	2.1833E-03	0.0000E+00	0.0000E+00	2.5700E-01	0.0000E+00	2.2054E-01
CO-58	1.3281E-01	1.3281E-03	0.0000E+00	0.0000E+00	3.5500E+00	1.0000E+02	1.3415E-01
NI-59	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	8.6600E-06	0.0000E+00	1.0000E+00
CO-60	3.7227E-01	3.7227E-03	0.0000E+00	0.0000E+00	1.3200E-01	1.0000E+02	3.7603E-01
NI-63	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	7.5300E-03	0.0000E+00	1.0000E+00
SR-90	7.9438E-01	7.9438E-03	0.0000E+00	0.0000E+00	2.4700E-02	0.0000E+00	8.0241E-01
NB-94	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	3.4700E-05	0.0000E+00	1.0000E+00
TC-99	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	3.2500E-06	0.0000E+00	1.0000E+00
RU-106	1.0341E-01	1.0341E-03	0.0000E+00	0.0000E+00	6.3900E-01	0.0000E+00	1.0445E-01
SB-125	2.2343E-01	2.2343E-03	0.0000E+00	0.0000E+00	2.5000E-01	0.0000E+00	2.2569E-01
I-129	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	4.0800E-08	0.0000E+00	1.0000E+00
CS-134	1.7488E-01	1.7488E-03	0.0000E+00	0.0000E+00	3.3600E-01	3.0000E+02	1.7665E-01
CS-135	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	2.3000E-07	0.0000E+00	1.0000E+00
CS-137	8.0204E-01	8.0204E-03	0.0000E+00	0.0000E+00	2.3100E-02	3.0000E+02	8.1014E-01
CE-144	8.9684E-02	8.9684E-04	0.0000E+00	0.0000E+00	8.9000E-01	0.0000E+00	9.0590E-02
EU-154	6.7688E-01	6.7630E-03	0.0000E+00	0.0000E+00	4.3300E-02	0.0000E+00	6.8372E-01
U-234	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	2.8300E-06	0.0000E+00	1.0000E+00
U-235	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	9.8500E-10	0.0000E+00	1.0000E+00
NP-237	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	3.3000E-07	0.0000E+00	1.0000E+00
U-238	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	1.5500E-10	0.0000E+00	1.0000E+00
PU-238	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	7.9000E-03	0.0000E+00	1.0000E+00
PU-239	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	2.8700E-05	0.0000E+00	1.0000E+00
PU-240	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	1.0600E-04	0.0000E+00	1.0000E+00
PU-241	6.2929E-01	6.2929E-03	0.0000E+00	0.0000E+00	5.2500E-02	0.0000E+00	6.3565E-01
AM-241	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	1.5100E-03	0.0000E+00	1.0000E+00
PU-242	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	1.8300E-06	0.0000E+00	1.0000E+00
AM-243	9.9000E-01	9.9000E-03	0.0000E+00	0.0000E+00	9.4000E-05	0.0000E+00	1.0000E+00
CM-243	8.1193E-01	8.1193E-03	0.0000E+00	0.0000E+00	2.1700E-02	0.0000E+00	8.2014E-01
CM-244	6.9870E-01	6.9870E-03	0.0000E+00	0.0000E+00	3.9400E-02	0.0000E+00	7.0576E-01

Table B-1. (cont.) Sample input data for PRESTO-EPA-POP

DISTRIBUTION COEFFICIENTS ML/G

NUCLIDE	SURFACE	TRENCH	VERTICAL	AQUIFER
H-3	1.00E-07	1.00E-02	1.00E-02	1.00E-02
C-14	1.00E-02	1.00E-02	1.00E-02	1.00E-02
CR-51	2.00E+02	2.00E+02	2.00E+02	2.00E+02
MN-54	1.50E+02	1.50E+02	1.50E+02	1.50E+02
FE-55	6.00E+03	6.00E+03	6.00E+03	6.00E+03
CO-58	5.50E+01	5.50E+01	5.50E+01	5.50E+01
NI-59	1.50E+02	1.50E+02	1.50E+02	1.50E+02
CO-60	5.50E+01	5.50E+01	5.50E+01	5.50E+01
NI-63	1.50E+02	1.50E+02	1.50E+02	1.50E+02
SR-90	1.50E+02	1.50E+02	1.50E+02	2.00E+01
NB-94	3.50E+02	3.50E+02	3.50E+02	3.50E+02
TC-99	5.00E-01	5.00E-01	5.00E-01	5.00E-01
RU-106	2.20E+02	2.20E+02	2.20E+02	2.20E+02
SB-125	4.50E+01	4.50E+01	4.50E+01	4.50E+01
I-129	3.00E+00	3.00E+00	3.00E+00	5.00E-01
CS-134	1.00E+03	1.00E+03	1.00E+03	5.00E+02
CS-135	1.00E+03	1.00E+03	1.00E+03	5.00E+02

```
*** SURFACE INFORMATION ***

     PARAMETERS FOR UNIVERSAL LOSS EQUATION
          RAINFALL              250.00
          ERODIBILITY             0.23
          STEEPNESS-SLOPE         0.27
          COVER                   0.30
          EROSION CONTROL         0.30
          DELIVERY RATIO          1.00
     SOIL POROSITY IS  0.39000
     SOIL BULK DENSITY IS  1.60000 G/CC
     RUNOFF FRACTION IS  0.29000

               STREAM FLOW RATE IS  3.5700E+05 CUBIC METERS PER YEAR
     CROSS SLOPE EXTENT OF SPILLAGE IS   1003.00 METERS
     ACTIVE SOIL DEPTH IS       0.10 METERS
     AVERAGE DOWN SLOPE DISTANCE TO STREAM IS    100.00 METERS
1   *** AIR-FOODCHAIN INFORMATION ***

     AGRICULTURAL PRODUCTIVITY FOR GRASS        0.67 KG/M**2
     AGRICULTURAL PRODUCTIVITY FOR VEGETATION        0.65 KG/M**2
     SURFACE DENSITY FOR SOIL      240.00 KG/M**2
     WEATHER DECAY CONSTANT         0.00 1/HOURS
     PERIOD PASTURE GRASS EXPOSURE GROWING SEASON      720.00 HOURS
     PERIOD CROP/VEGETATION EXPOSURE GROWING SEASON    1440.00 HOURS
     PERIOD BETWEEN HARVEST PASTURE GRASS AND INGESTION BY ANIMAL        0.00 HOURS
     PERIOD BETWEEN STORED FEED AND INGESTION BY ANIMAL     2160.00 HOURS
     PERIOD BETWEEN HARVEST LEAFY VEGETABLES AND INGESTION BY MAN(M.I.E.)       24.00 HOURS
     PERIOD BETWEEN HARVEST PRODUCE AND INGESTION BY MAN(M.I.E.)    1440.00 HOURS
     PERIOD BETWEEN HARVEST LEAFY VEG AND INGESTION BY MAN(G.P.E.)     336.00 HOURS
     PERIOD BETWEEN HARVEST PRODUCE AND INGESTION BY MAN(G.P.E.)    336.00 HOURS
     FRACTION OF YEAR ANIMALS GRAZE ON PASTURE          1.00
     FRACTION OF DAILY FEED THAT IS FRESH GRASS          0.83
     AMOUNT OF FEED CONSUMED BY CATTLE          50.00 KG
     AMOUNT OF FEED CONSUMED BY GOATS            6.00 KG
     TRANSPORT TIME FEED-MILL-RECEPTOR FOR M.I.E.        48.00 HOURS
     TRANSPORT TIME FEED-MILL-RECEPTOR FOR G.P.E.        96.00 HOURS
     TIME FROM SLAUGHTER OF MEAT TO CONSUMPTION     480.00 HOURS
     ABSOLUTE HUMIDITY OF THE ATMOSPHERE          9.90 G/M**3
     FRACTIONAL EQUILIBRIUM RATIO FOR C-14        1.00
1    *** WATER-FOODCHAIN INFORMATION ***

     FRACTION OF YEAR CROPS ARE IRRIGATED         0.40
     IRRIGATION RATE        0.015 L/(M**2-H)
     AMOUNT OF WATER CONSUMED BY COWS          60.00 L/D
     AMOUNT OF WATER CONSUMED BY GOATS          8.00 L/D
     AMOUNT OF WATER CONSUMED BY BEEF CATTLE         50.00 L/D

*** HUMAN INGESTION AND INHALATION RATE INFORMATION ***

     ANNUAL INTAKE OF LEAFY VEG        2.00 KILOGRAMS PER YEAR
     ANNUAL INTAKE OF PRODUCE     18.00 KILOGRAMS PER YEAR
     ANNUAL INTAKE OF COW'S MILK      11.00 LITERS PER YEAR
     ANNUAL INTAKE OF GOAT'S MILK      0.00 LITERS PER YEAR
     ANNUAL INTAKE OF MEAT       9.00 KILOGRAMS PER YEAR
     ANNUAL INTAKE OF DRINKING WATER  370.00 LITERS PER YEAR
     ANNUAL INHALATION RATE OF AIR  8000.00 CUBIC METERS PER YEAR
     A POPULATION OF          25. WILL BE CONSIDERED
```

Table B-2. Sample output data for PRESTO-EPA-POP

HEALTH EFFECTS RESULTING FROM RESIDUAL RADIOACTIVITY RELEASED IN 1000 YEARS

NUCLIDE	RESIDUAL ACTIVITY	CONVERSION FACTOR	HEALTH EFFECTS
H-3	2.4601E-03	0.0000E+00	0.0000E+00
C-14	9.6049E-01	0.0000E+00	0.0000E+00
CR-51	5.7943E-08	0.0000E+00	0.0000E+00
MN-54	9.4963E-06	0.0000E+00	0.0000E+00
FE-55	2.4504E-06	0.0000E+00	0.0000E+00
CO-58	1.3099E-06	0.0000E+00	0.0000E+00
NI-59	4.1560E-03	0.0000E+00	0.0000E+00
CO-60	5.6022E-04	0.0000E+00	0.0000E+00
NI-63	3.3200E-03	0.0000E+00	0.0000E+00
SR-90	1.8456E-03	0.0000E+00	0.0000E+00
NB-94	4.1431E-03	0.0000E+00	0.0000E+00
TC-99	5.2446E-01	0.0000E+00	0.0000E+00
RU-106	9.0382E-06	0.0000E+00	0.0000E+00
SB-125	2.3476E-04	0.0000E+00	0.0000E+00
I-129	4.0010E-02	0.0000E+00	0.0000E+00
CS-134	8.5406E-06	0.0000E+00	0.0000E+00
CS-135	4.1125E-03	0.0000E+00	0.0000E+00
CS-137	5.6075E-04	0.0000E+00	0.0000E+00
CE-144	1.1240E-06	0.0000E+00	0.0000E+00
EU-154	7.3340E-05	0.0000E+00	0.0000E+00
U-234	4.1419E-03	0.0000E+00	0.0000E+00
U-235	4.1439E-03	0.0000E+00	0.0000E+00
NP-237	4.1996E-03	0.0000E+00	0.0000E+00
U-238	4.1439E-03	0.0000E+00	0.0000E+00
PU-238	5.9967E-04	0.0000E+00	0.0000E+00
PU-239	3.0277E-03	0.0000E+00	0.0000E+00
PU-240	2.9386E-03	0.0000E+00	0.0000E+00
PU-241	6.4080E-05	0.0000E+00	0.0000E+00
AM-241	1.2274E-04	0.0000E+00	0.0000E+00
PU-242	3.0596E-03	0.0000E+00	0.0000E+00
AM-243	2.2566E-04	0.0000E+00	0.0000E+00
CM-243	2.0718E-04	0.0000E+00	0.0000E+00
CM-244	1.0014E-04	0.0000E+00	0.0000E+00

TABLE FOR ACCOUNTING MODEL -- YEAR 1000

COLUMN 1: INDIVIDUAL CANCER RISK (UNITLESS)
COLUMN 2: POPULATION DOSE (PERSON-REM/YEAR)
COLUMN 3: COLLECTIVE CANCER RISK (DEATHS/YEAR)
COLUMN 4: GENETIC RISK (EFFECTS/YEAR)
COLUMN 5: R-ACTIVITY PUMPED OUT THE WELL 1ST 1000 YEARS (CI)
COLUMN 6: R-ACTIVITY PUMPED OUT THE STREAM 1ST 1000 YEARS (CI)
COLUMN 7: R-ACTIVITY RELEASED TO ATMOSPHERE 1ST 1000 YEARS (CI)
COLUMN 8: R-ACTIVITY RELEASED DOWNSTREAM 1ST 1000 YEARS (CI)
COLUMN 9: R-ACTIVITY RELEASED DOWNSTREAM LAST 9000 YEARS (CI)
COLUMN 10: R-ACTIVITY IN TRENCH AFTER 10000 YEARS ASSUMING PERFECT CONTAINMENT (CI)

	1	2	3	4	5	6	7	8	9	10
H-3	5.3663E-12	6.7506E-09	1.8961E-12	7.1733E-13	1.6686E-06	0.0000E+00	1.3492E-07	2.4601E-03	3.4116E-27	0.0000E+00
C-14	6.8240E-08	8.5945E-05	2.4111E-08	4.4872E-09	1.1868E-03	0.0000E+00	2.3101E-07	9.6049E-01	9.2666E-04	2.9522E-01
CR-51	2.6372E-24	3.3214E-21	9.3176E-25	3.9532E-25	0.0000E+00	0.0000E+00	6.5909E-08	5.7943E-08	0.0000E+00	0.0000E+00
MN-54	1.6628E-18	2.0942E-15	5.8749E-19	2.5655E-19	0.0000E+00	0.0000E+00	5.3645E-06	9.4963E-06	0.0000E+00	0.0000E+00
FE-55	2.7432E-20	3.4549E-17	9.6923E-21	2.5168E-21	0.0000E+00	0.0000E+00	2.5954E-05	2.4504E-06	0.0000E+00	0.0000E+00
CO-58	6.2927E-21	7.9253E-18	2.2234E-21	1.0292E-21	0.0000E+00	0.0000E+00	4.0227E-07	1.3099E-06	0.000[illegible]E+00	0.0000E+00
NI-59	5.8589E-18	7.3791E-15	2.0701E-18	2.9964E-19	0.0000E+00	0.0000E+00	1.6599E-04	4.1560E-03	1.274[illegible]E-01	9.0707E-01
CO-60	3.1822E-16	4.0079E-13	1.1244E-16	4.8614E-17	0.0000E+00	0.0000E+00	4.6919E-05	5.6022E-04	0.0000E+00	0.0000E+00
NI-63	1.2525E-19	1.5775E-16	4.4255E-20	1.2694E-20	0.0000E+00	0.0000E+00	1.6384E-04	3.3200E-03	2.9503E-13	1.9646E-33
SR-90	2.8587E-18	3.6004E-15	1.0100E-18	1.4442E-20	0.0000E+00	0.0000E+00	1.2784E-04	1.8456E-03	2.5130E-27	0.0000E+00
NB-94	5.3753E-14	6.7699E-11	1.8992E-14	8.2990E-15	0.0000E+00	0.0000E+00	1.7103E-04	4.1431E-03	1.4089E-01	6.9974E-01
TC-99	8.1199E-08	1.0227E-04	2.0690E-08	1.3592E-09	1.1355E-03	0.0000E+00	1.0836E-05	5.2446E-01	4.7142E-01	9.5834E-01
RU-106	2.7097E-19	3.4127E-16	9.5739E-20	3.8767E-21	0.0000E+00	0.0000E+00	6.8681E-06	9.0382E-06	0.0000E+00	0.0000E+00
SB-125	1.5439E-17	1.9445E-14	5.4551E-18	2.3686E-18	0.0000E+00	0.0000E+00	2.3321E-05	2.3476E-04	0.0000E+00	0.0000E+00
I-129	1.7046E-06	2.1468E-03	6.0226E-07	1.3381E-09	5.3212E-04	0.0000E+00	4.9001E-05	4.0810E-02	9.5014E-01	9.8960E-01
CS-134	3.60[illegible]E-17	4.5451E-14	1.2751E-17	5.4954E-18	0.0000E+00	0.0000E+00	1.8546E-05	8.5406E-06	0.0000E+00	0.0000E+00
CS-135	1.6940E-17	2.1335E-14	5.9853E-18	2.4679E-18	0.0000E+00	0.0000E+00	1.7707E-04	4.1125E-03	1.9767E-01	9.8773E-01
CS-137	1.0777E-19	1.3574E-16	3.0079E-20	1.4181E-20	0.0000E+00	0.0000E+00	1.3537E-04	5.6075E-04	4.8146E-16	0.0000E+00
CE-144	1.5012E-19	1.8907E-16	5.3041E-20	4.6217E-21	0.0000E+00	0.0000E+00	4.7999E-06	1.1240E-06	0.0000E+00	0.0000E+00
EU-154	1.3887E-15	1.7409E-12	4.9064E-16	2.1276E-16	0.0000E+00	0.0000E+00	1.1102E-04	7.3340E-05	3.5431E-24	0.0000E+00
U-234	7.2312E-17	1.3766E-13	2.5550E-17	3.3962E-18	0.0000E+00	0.0000E+00	1.7559E-04	4.1419E-03	1.9083E-01	9.6238E-01
U-235	8.9114E-15	1.1223E-11	3.1486E-15	1.2508E-15	0.0000E+00	0.0000E+00	1.7559E-04	4.1439E-03	1.9439E-01	9.0999E-01
NP-237	2.2047E-16	2.7767E-13	7.7896E-17	5.1427E-18	0.0000E+00	0.0000E+00	6.8405E-05	4.1996E-03	9.9348E-01	9.8674E-01
U-238	6.1397E-17	1.1898E-13	2.1693E-17	2.6221E-18	0.0000E+00	0.0000E+00	1.7559E-04	4.1439E-03	1.9439E-01	9.9000E-01
PU-238	5.8969E-16	7.4269E-13	2.0835E-16	1.1823E-17	0.0000E+00	0.0000E+00	1.7376E-04	5.9967E-04	5.8051E-08	4.8571E-35
PU-239	5.6600E-16	7.1396E-13	2.0029E-16	1.0203E-17	0.0000E+00	0.0000E+00	1.8455E-04	3.0277E-03	1.5960E-01	7.4301E-01
PU-240	5.8977E-16	7.4279E-13	2.0838E-16	1.2314E-17	0.0000E+00	0.0000E+00	1.8421E-04	2.9386E-03	9.5971E-02	3.4299E-01
PU-241	2.7374E-18	3.4476E-15	9.6719E-19	8.9020E-20	0.0000E+00	0.0000E+00	1.0162E-04	6.4080E-05	2.6451E-28	0.0000E+00
AM-241	1.6366E-15	2.0613E-12	5.7827E-16	1.9770E-16	0.0000E+00	0.0000E+00	1.8372E-04	1.2274E-04	3.9856E-05	2.7402E-07
PU-242	5.5708E-16	7.0162E-13	1.9683E-16	1.1333E-17	0.0000E+00	0.0000E+00	1.8466E-04	3.0596E-03	1.9134E-01	9.7305E-01
AM-243	5.2253E-15	6.5811E-12	1.0462E-15	7.4610E-16	0.0000E+00	0.0000E+00	1.9236E-04	2.2566E-04	2.0385E-02	3.8672E-01
CM-243	7.2015E-16	9.0700E-13	2.5445E-16	7.7511E-17	0.0000E+00	0.0000E+00	1.3824E-04	2.0718E-04	1.8771E-14	0.0000E+00
CM-244	2.3068E-16	2.9053E-13	8.1505E-17	1.4220E-17	0.0000E+00	0.0000E+00	1.1531E-04	1.0014E-04	1.8643E-22	0.0000E+00
TOTALS:	1.8540E-06	2.3350E-03	6.5506E-07	7.1852E-09						

0 MESSAGE SUMMARY: MESSAGE NUMBER - COUNT
0 208 511 OR OVER

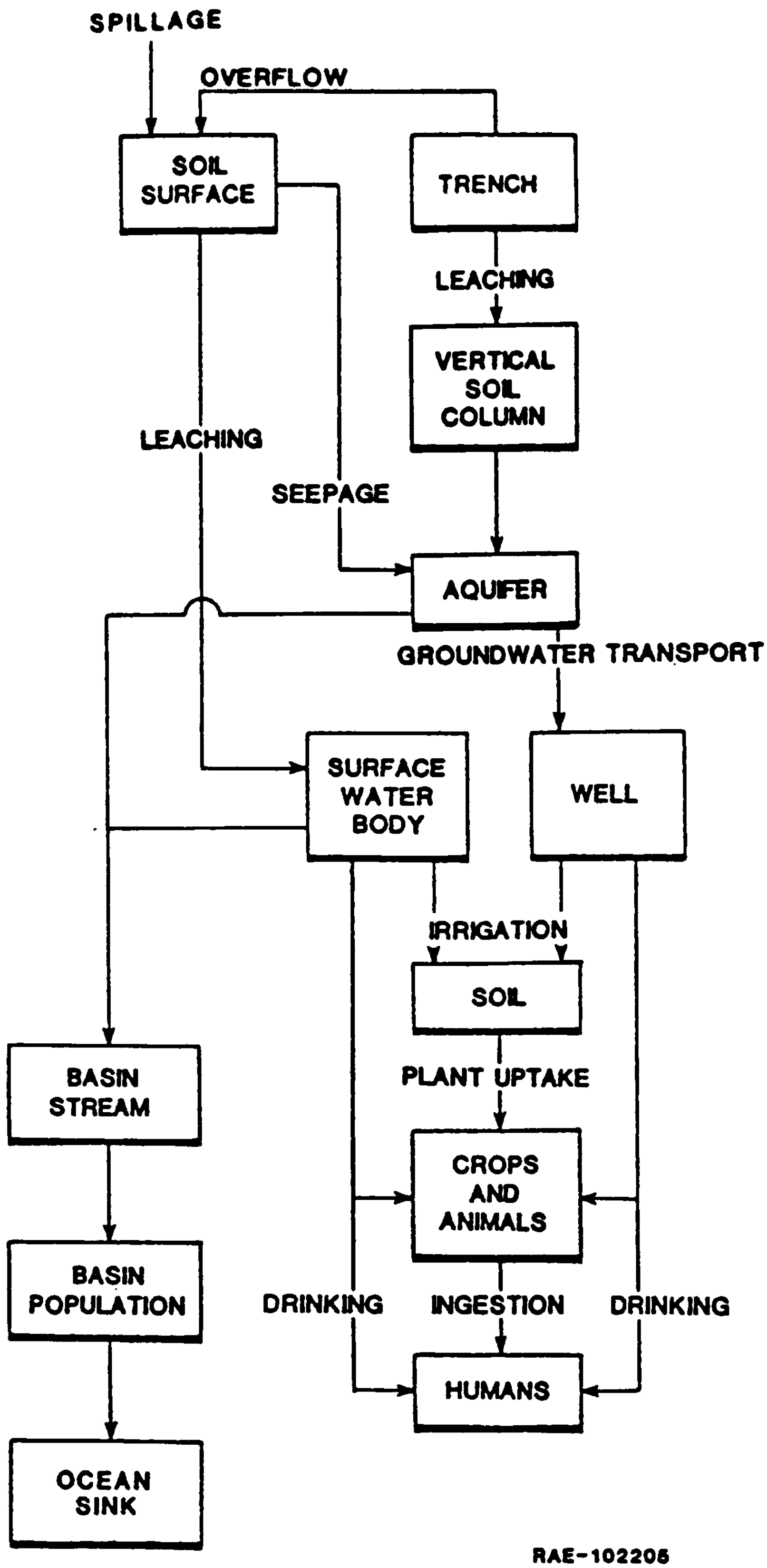


Figure B-1. Hydrologic environmental pathways in PRESTO-EPA-POP

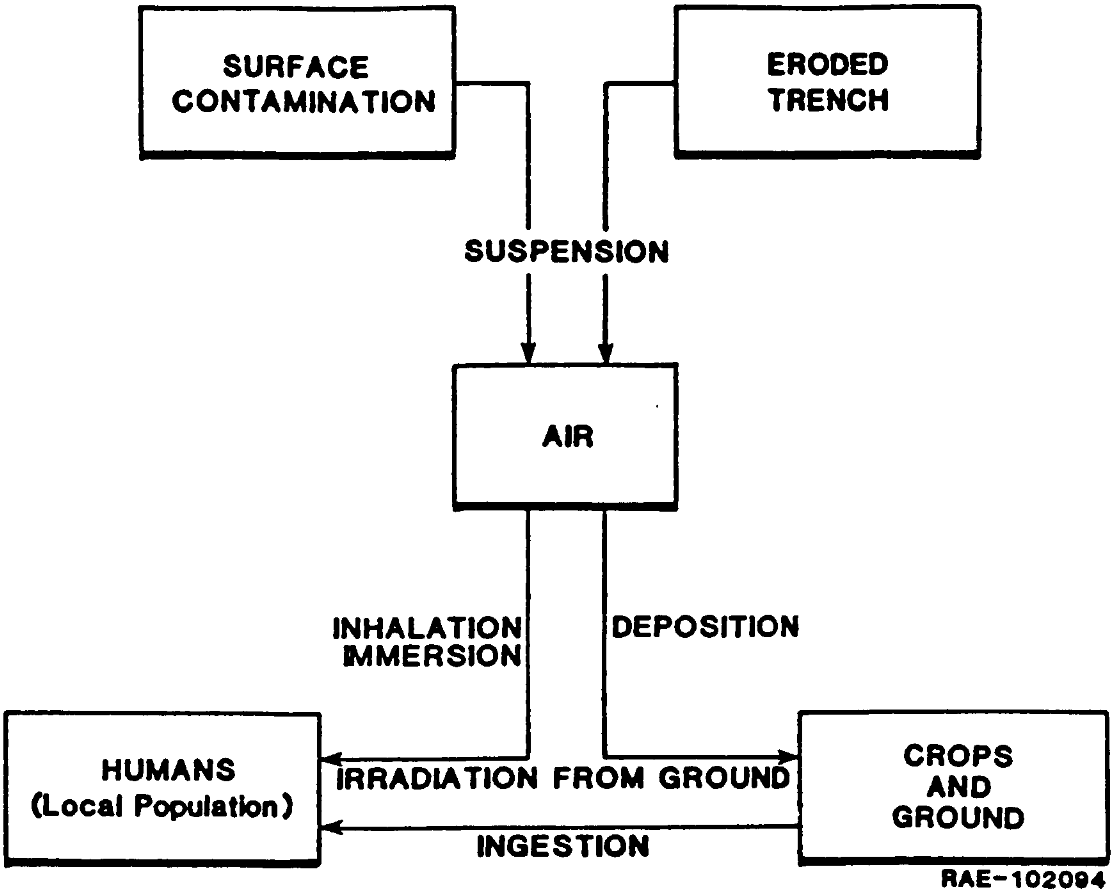


Figure B-2. Atmospheric environmental pathways in PRESTO-EPA-POP

1 IDENTIFICATION

1.1 Name:	BARRIER
1.2 Prepared By:	Rogers and Associates Engineering Corp.
1.3 Prepared For:	Electric Power Research Institute
1.4 Report Title:	Performance Assessment for Low-Level Waste Disposal Facilities
1.5 Report Number:	EPRI NP-5745M; EPRI NP-5745SP
1.6 Report Date:	April 1988
1.7 Availability:	EPRI Research Reports Center, Box 50490, Palo Alto, CA, 94303
1.8 Purpose and Scope:	Estimate the long-term performance and degradation of concrete barriers used in various engineered storage disposal designs compared to a shallow land burial base-case.

2 SUMMARY OF FINDINGS

BARRIER was developed for the Electric Power Research Institute (EPRI) to assist the nuclear power industry meet federal and state disposal site performance assessment requirements and expedite licensing of new disposal facilities. It is an integrated model that estimates: groundwater flow through a disposal facility; radionuclide release; long-term performance and degradation of concrete barriers used in various engineered storage disposal designs (e.g., below-ground vault, above-ground vault, modular concrete canister disposal and earth mounded concrete bunker); transport through an aquifer to the accessible environment; and doses to the critical population group (CPG). Projected performance of engineered disposal options were compared to a shallow land burial base-case.

Inclusion of a module that simulates the performance of concrete structures is a unique feature of this code. Prediction of engineered barrier performance is a significant addition to the capabilities of performance assessment modeling, especially in light of the increasing emphasis on alternative disposal technologies. At the same time, however, lack of empirical data on concrete behavior over long periods of time introduces new uncertainties in the calculation that need to be addressed.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

Documentation for BARRIER is contained in the interim report published by EPRI. Two versions have been issued: NP-5745M presents a qualitative summary of the assessment model and its findings; NP-5745SP is intended for limited distribution and provides additional detailed data and information on concrete degradation/failure mechanisms. Neither document contains a code listing or the information typically found in a user's guide. If BARRIER becomes available for general use, additional documentation would be needed to implement.

3.2 Hardware Requirements

The code occupies about 4000 lines of FORTRAN code and was written for implementation on a Micro Vax II computer.

3.3 Application

BARRIER is an integrated code that contains subroutines to model groundwater unsaturated flow, radionuclide leaching and transport, well-water concentration, food chain and uptake, human dose, and engineered barrier performance. An inital data set contains required data and parameters to run the code to completion. Total time for execution is dependent on complexity of the unsaturated flow calculations, ranging from about 5 minutes for a simplified problem to several hours for a complex one.

3.4 Level of Expertise Required

Not discussed in the documentation.

4 TECHNICAL CRITERIA

4.1 Peer Review

BARRIER has not undergone a formal peer review.

4.2 Verification

EG & G Idaho has a licensing agreement with EPRI to implement BARRIER and has provided comments that have resulted in code modifications.

4.3 Uncertainty

Uncertainty analysis was performed to quantify the impact of parameter variability on model output. Selected parameters relating to the performance of concrete structures were examined using a statistical variation methodology originally designed for radionuclide transport from disposal facilities. RANDIS (Random Distribution) code, which uses a Monte Carlo method of statistical sampling was modified for this purpose. Probability distributions for projected structural failure of engineered barriers were generated by specifying a range for each of the relevant variables, along with the type of distribution (uniform, normal, log uniform, or log normal). Concrete structural characteristics such as compressive strength, percentage of calcium aluminate, and bulk modulus, as well as groundwater concentrations of potentially destructive species such as chloride, magnesium, sulfate, and oxygen were examined. Although not included in this study, application of uncertainty methodology could be extended to examine parameter uncertainty in the other modules of the code (e.g., groundwater flow, contaminant leaching and transport, food chain, dose assessment).

4.4 *Sensitivity*

Sensitivity analyses were performed on selected aspects of concrete structural properties (compressive strength, water-to-cement ratio), diffusion coefficients of chemical species (CaOH, O_2, SO_4), and environmental characteristics (freeze-thaw cycles, maximum thermal gradient). Relative impact on the time to onset of structural cracking and time to total cracking failure were measured by varying each parameter within a prescribed range (e.g., $\pm$ 10%) while keeping other factors constant. Sensitivity of these parameters was examined for each of the alternative disposal options at both humid and arid sites.

4.5 *Required Input*

Since the available documentation is not a user's guide, specific input requirements such as particular data files and user specified parameters are not discussed. A general discussion of the types of input used by BARRIER is included, however. Four disposal options are examined: i) shallow land burial (SLB), ii) modular concrete canister disposal (MCC), iii) below-ground vault disposal (BGV), and iv) above-ground vault (AGV). Each set of options is examined at both a humid site (data from Charleston, SC) and an arid site (data from El Paso, TX).

Radioactive waste inventory used in the model includes 27 radionuclides (listed in Table B-3). Wastes in Class A are handled separately from Classes B and C wastes, but further characterization of waste composition (such as physical or chemical composition) is not included. For example, the code uses a single value of K_d for each isotope, ignoring possible differences based on waste composition or solidification medium. All solidified waste is assumed to be in a concrete matrix. Although separate input data for each disposal site are presented for some parameters, only one set of soil K_ds is reported. It is not clear whether these correspond to the South Carolina site (silty loam soil) or the Texas site (sandy clay soil). Since soil distribution coefficients are isotope and soil specific, separate data are required for each site. Data on aquifer velocities, depths, and dispersivities were taken from humid permeable site characterization in the EPA LLW Draft Background Information Document. Eight freeze-thaw cycles are assumed for humid site, none for arid. Freeze-thaw cycling is a significant parameter in the concrete degradation simulation.

4.6 *Output*

Calculated output from BARRIER includes peak dose, year peak dose occurs, dominant nuclide contributing to peak dose, and lifetime dose to members of CPG.

4.7 *Source Term*

Waste inventories are the same as for RAE study for DOE on LLW Conceptual Designs. 27 isotopes are included, 11 of which are TRU. Quantities of each isotope are specified in four inventory categories: Class

A packaged in steel drums, Class A packaged in steel liners, Class B & C packaged in steel drums, and Class B & C packaged in steel liners. No distinction is made to account for waste type or physicochemical properties other than Class A vs. Classes B & C (NRC categories).

4.8 Scenarios

A simple exposure scenario is used that assumes a well, located at the site boundary, is used for direct consumption by humans and animals, and for irrigation of crops. Doses to the CPG result from ingestion of contaminated water, crops, meat, and milk. Intruder and exposed waste scenarios are not considered. Although engineered barriers would probably reduce the likelihood of occurrence for both of these exposure scenarios in the near-term, after eventual failure of concrete structures the probability of occurrence would be comparable to a shallow land burial site. The example problem does not address worker exposures but a separate program called Worker Occupational Radiation Dose Assessment Code for the Disposal of Low-Level Waste (WORDRAE) is included in the appendix.

4.9 Relationship to Regulatory Standards

Output of projected effective dose equivalents (in terms of millirem/year) is consistent with EPA's proposed exposure standards for the public contained in 40 CFR 193. The current standard contained in NRC's 10 CFR 61 does not use the organ weighting factor associated with effective dose equivalent. Federal regulations also specify allowable radionuclide concentrations in air and water. These are calculated and used in the dose calculations, but are not reported in BARRIER output.

5 SCIENTIFIC CRITERIA

5.1 Theory

The methodology used in this integrated performance assessment code is organized into four categories:

- Unsaturated groundwater flow analysis
- Concrete degradation and failure analysis
- Contaminant leaching and transport
- Dose projections

UNSAT-H is used to simulate one-dimensional water flow in unsaturated soils and sediments. This mechanistic model based on Darcy's Law, can simulate movement of water in liquid and vapor phases in response to precipitation and irrigation, plant water extraction, and deep drainage. Flow through cracked concrete is described by an equation that is similar to that characterizing bulk flow, with fracture spacing hydraulically equivalent to granular porous media.

Concrete degradation and cracking over time are simulated by subroutines that estimate: (i) thermal and static loading moments, (ii) $Ca(OH)_2$ leaching, (iii) SO_4 attack, (iv) freeze-thaw degradation, (v) corrosion of reinforcing steel, and (vi) cracking moment and ultimate strength.

Contaminant leaching is via one of four leaching options: (i) constant leach rate, (ii) nuclide-specific leach rate, (iii) diffusion leaching (migration through a solid waste form), and (iv) dissolution or K_d leaching (movement of nuclides in unsolidified waste or deteriorated waste forms where porosity is high). Data for the first two options are user specified; the latter are dynamic parameters calculated by the program. As solidified waste form or concrete structures degrade, migration increases due to enhanced water movement and leaching by dissolution eventually dominates. Similar to other contaminant groundwater pathway codes (e.g., PATHRAE), effective groundwater migration speed is described as the quotient of the groundwater flow velocity and contaminent retardation factor As the facility degrades and cracking develops, hydraulic resistance decreases and flow is enhanced. For transport through fractured media, it is assumed that contaminants travel to external surface in one year, then proceed through bulk media at normal rates.

Dose conversion is discussed in Section 5.6.

5.2 Validation

Model performance predictions have not been compared with actual disposal site performance.

5.3 Treatment of Radioactive Decay Products

Treatment of radioactive decay products isnot considered in BARRIER.

5.4 Underlying Assumptions

Assumptions used in the analysis including disposal facility design and performance characteristics; climate, geology, and hydrology of the generic sites; and waste and package parameters are clearly stated.

5.5 Pathways

Transport/exposure pathways are all linked by groundwater and surface water flow. Unsaturated groundwater flow is modeled (using UNSAT, described above) as the driving force for disposal site degradation and radionuclide release. Vertical flow of leachate is expressed as the quotient of groundwater velocity and a contaminant retardation factor that takes into consideration soil density, porosity and distribution coefficient (K_d). Radionuclide transport through engineered barriers is either via flow in fractured porous media (upon barrier failure) or advection through intact concrete matrix. Horizontal transport through the saturated zone to the accessible environment is handled using the same algorithm used in the PRESTO family of codes, accounting for aquifer velocity, porosity, and depth, disposal site area, dispersion angle of contaminated plume, and distance to well. Exposure is by ingestion of contaminated well water, crops, milk, and meat. Other transport/exposure pathways (e.g., direct exposure to inadvertent intruder, resuspension via erosion or flooding with concomitant air and/or surface water transport), are not addressed.

5.6 Dose Conversion and Dose Response

Dose conversion methodology is the same as that used in PRESTO family of performance assessment codes (e.g., PATHRAE and PRESTO-EPA). PRESTO calculates effective whole body dose equivalent as per ICRP 30. Calculations assume individual: (i) is situated 100 m from site boundary and uses contaminated well water for drinking, irrigation of food crops, and as a water source for animals, and (ii) consumes contaminated quantities of vegetables, milk, and meat. Dose to the CPG is calculated as annual dose rate and cumulative lifetime dose. Dose response is not calculated.

Table B-3. Radioactive waste inventories used in BARRIER

Nuclide	Class A Inventory (Ci)		Class B & C Inventory (Ci)	
	Steel Drums[a]	Steel Liners	Steel Drums[a]	Steel Liners
H-3	5.7E+02	2.8E+03	3.8E+05	1.4E+03
C-14	2.8E+01	1.5E+02	3.3E+02	9.7E+01
Na-22	2.3E+00	3.0E+03	2.0E-03	7.3E+03
Fe-55	1.5E+01	1.5E+04	1.5E-01	2.9E+05
Co-58	4.0E+00	8.6E+03	0.0E+00	1.2E+05
Co-60	5.8E+01	1.7E+04	9.4E-06	2.2E+05
Ni-59	6.2E-03	1.2E+01	0.0E+00	1.8E+02
Ni-63	1.0E+00	1.8E+03	2.4E-03	2.6E+04
Sr-90	1.9E+01	6.8E+01	3.2E-01	1.6E+03
Nb-94	3.6E-05	2.2E-01	0.0E+00	1.6E+00
Tc-99	4.5E-05	2.1E-01	8.1E-07	8.8E-01
I-129	2.1E-08	5.8E-02	6.7E-10	2.3E-01
Cs-134	5.5E-13	5.4E+03	0.0E+00	2.2E+04
Cs-135	2.5E-05	2.1E-01	8.4E-07	8.8E-01
Cs-137	2.8E+01	5.7E+03	1.5E-02	2.3E+04
Ba-137m	2.8E+01	5.7E+03	1.5E-02	2.3E+04
U-235	4.6E-01	5.2E-02	5.1E-02	3.6E-01
U-238	2.1E+00	2.2E-01	2.9E-01	1.1E-01
Np-237	3.1E-14	1.2E-06	9.9E-16	2.7E-06
Pu-238	1.1E-05	6.5E+00	3.7E-07	1.4E+01
Pu-239	3.2E-06	4.0E+00	1.0E-07	8.3E+00
Pu-241	4.1E-04	1.8E+02	1.3E-05	3.8E+02
Pu-242	5.5E-09	8.8E-03	1.8E-10	1.8E-02
Am-241	9.7E-03	9.1E+00	2.0E-08	1.9E+01
Am-243	7.2E-08	2.6E-01	2.3E-09	5.0E-01
Cm-243	1.7E-08	3.9E-03	5.3E-10	7.9E-03
Cm-244	9.5E-06	3.8E+00	3.1E-07	7.5E+00

[a] These wastes were considered to be unpackaged for the modular concrete canister disposal facility, reflecting compaction of the drums within the canisters.

1 IDENTIFICATION

1.1 Name:	Part 61 Impacts Analysis Methodology
1.2 Prepared By:	Envirosphere Co., and U.S. Nuclear Regulatory Commission
1.3 Prepared For:	U.S. Nuclear Regulatory Commission
1.4 Report Title:	Update of Part 61 Impacts Analysis Methodology
1.5 Report Number:	NUREG/CR-4370, Vol. 1 and 2
1.6 Report Date:	January 1986
1.7 Availability:	Documentation: U.S. NRC; U.S. Government Printing Office; National Technical Information Center Software: Radiation Shielding Information Center, Oak Ridge, TN
1.8 Purpose and Scope:	Integrated performance assessment code for comparative analysis of various land-based disposal options for low-level radioactive waste.

Note: The full title of this performance assessment code is Update of Part 61 Impacts Analysis Methodology. For the purposes of this discussion it will be referred to simply as IMPACTS.

2 SUMMARY OF FINDINGS

IMPACTS was initially developed for the U.S. NRC, to assist in the preparation of shallow land disposal regulations for low-level radioactive waste (10 CFR 61). It is an integrated performance assessment model for comparing potential health impacts from various land disposal options on a generic basis. The user is cautioned in using the methodology for a site-specific application, where site-specific models, inventories, disposal options, and environmental parameters would be required to accurately simulate conditions. Further caution is advised in interpreting the absolute magnitude of results. Rather, the model was intended as a relative comparison to estimate potential benefits and costs from a number of potential disposal options.

User provides information on combination of waste streams to be considered and regions where they are generated and then selects specific waste processing scenarios, the environmental setting of disposal site, and the particular combination of disposal technologies to be used. Output gives calculated effective dose equivalents (mrems/yr) for 9 organs plus effective whole body equivalent for each exposure scenario and waste classification. For chronic exposure scenarios, data are given in varying time increments from 20 to 20,000 years. In addition to IMPACTS, user may select the following subroutines: INVERSE calculates acceptable nuclide total activity and/or concentration limits for disposal; ECONOMY calculates transportation and routine operational radiological impacts as well as disposal cost estimates; INTRUDE analyzes radiological impacts to an inadvertent intruder as a function of time; and VOLUMES calculates and updates region and waste stream dependent annual volume projections. A separate code (CLASIFY) is used to classify the waste streams and organize input data for use by IMPACTS. Code is written for implementation on an IBM PC.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

Comprehensive documentation for IMPACTS is available in Volumes 1 and 2 of NUREG/CR-4370. Volume 1 presents an overview of the method, describes the theory and algorithms for implementation, and provides background in the areas of waste inventories, processing and disposal options, and pathway dose conversion factors. Volume 2 includes a listing of the codes and data files used by the model and presents several example problems. In addition, the software package includes a README file which contains information on which input files are needed to run each subroutine, along with expected run times.

3.2 Hardware Requirements

IBM PC (or compatible) with 640 KB, hard disk drive, and 8087 math coprocessor.

3.3 Application

Six separate subroutines, listed below along with expected run times, are executed independently. CLASIFY and INVERSE are operated in batch mode; others are interactive.

CLASIFY:	Classifies waste streams in four classes	(~10 min)
IMPACTS:	Various impact measures	(~23 min)
INVERSE:	Activity or concentration limits	(~20 min)
ECONOMY:	Costs of disposal	(~12 min)
INTRUDE:	Impacts of an intruder	(~26 min)
VOLUMES:	Waste stream annual volume	(time not given)

3.4 Level of Expertise Required

Although written for implementation on a personal computer, IMPACTS is not "user friendly". Several characteristics of the program structure combine to complicate its use. First, almost all of the parameters that define the problem to be examined (e.g., disposal region and technology, waste characterization, type of waste processing, waste form behavior) are assigned index values. The user must therefore frequently refer back to the series of tables that define index acronyms and provide numerical index values that correspond with appropriate responses in order to supply needed information. These index values are contained in input control data files. More significantly, the programs do not contain interactive subroutines to create control data files, so user must either: (i) edit the example control data files supplied with the software to alter the problem, or, (ii) create programs to write new control data files. The former can be accomplished without a great deal of computing experience, but the process is time consuming, especially if multiple runs are needed.

4 TECHNICAL CRITERIA

4.1 *Peer Review*

The original Part 61 Impacts Analysis Methodology (NRC 1981) written for the Environmental Impact Statement on 10 CFR 61 was included in the Low-Level Waste Disposal Performance Assessment Model Review (EG&G 1985), which compared exposure scenarios and pathways, transport pathways and parameters, and dosimetry methods, among several land-based disposal performance models. Model capabilities and parameters were identified, but this report does not constitute a comprehensive peer review. In addition, many significant changes in the updated methodology reduce the usefulness of EG&G's review of IMPACTS. No other independent peer reviews were found.

4.2 *Verification*

The code was tested and de-bugged at the Radiation Shielding Information Center, but verification of mathematical computations was apparently not performed.

4.3 *Uncertainty*

No uncertainty analysis is reported in the documentation.

4.4 *Sensitivity*

No sensitivity analysis is reported in the documentation.

4.5 *Required Input*

IMPACTS requires up to nine input files (summarized below). These include data base files and output files from several codes that must be run before IMPACTS (CLASIFY and VOLUMES) to consolidate some of the input data. CLASIFY classifies waste into regulatory categories based on radionuclide characteristics and activities and requires the following data files: LIMITS.DAT which contains information representing specific nuclide/solubility option; WASCAR.DAT which contains user-specified records representing specific waste streams; CLACON.DAT which specifies manner in which waste streams are considered, and whether disposal technology can stabilize waste streams. CLASIFY combines the multitude of alternatives presented by these data files into one file (CLAOUT.DAT) for use by IMPACTS. VOLUMES projects future waste volumes based on specific generating characteristics (from VRATES.DAT input file) and information on nuclear power reactors (from REACTR.DAT input file) and supplies the output file VOLUME.DAT for IMPACTS. Because of this modular structure, data subject to change such as reactor and waste generation data can be easily updated.

IMPCON.DAT Input command file which specifies problem to be considered including facility, schedule and disposal configuration parameters.

FUNDCF.DAT	Fundamental dose conversion factors and other radiological specific information such as isotope solubility and half-life, pathway uptake coefficients, etc.
ENVIRO.DAT	Environmental parameters of 4 reference sites and region-dependent pathway uptake parameters.
INPUTS.DAT	Specifies waste streams to be considered.
CLAOUT.DAT	Output from CLASIFY (see discussion above).
DISTEC.DAT	Disposal technology parameters.
LIMITS.DAT	Nuclide concentration and activity limits.
VOLUME.DAT	Output from VOLUMES (see discussion above).
METALS.DAT	Activated metals information, where applicable.

Necessary data files are provided with the code. Disposal site data are generic - "typical" values are provided for four regional disposal sites (Northeast, Southeast, Midwest, and Southwest). A summary of input data used in IMPACTS is given in Table B-4.

4.6 Output

Output of IMPACTS is stored in IMPOUT.DAT. which provides a summary of specified parameters used in calculations and calculated effective dose equivalents (mrems/yr) for 9 organs plus whole body equivalent for each exposure scenario (intruder, exposed waste, leachate accumulation, operational accidents, and groundwater) and waste classification. For chronic exposure scenarios, data are given in varying time increments from 20 to 20,000 years. Information on individual radionuclides (e.g., concentrations, relative contribution to organ and total dose) is not provided. Human health effects are not calculated. Sample output from IMPACTS illustrating the groundwater scenario (boundary well) for Classes A, B, and C waste is included in Table B-5.

4.7 Source Term

IMPACTS accepts a wide range of radionuclides (100 radionuclide and solubility combinations) and waste types (148 waste streams). Radionuclides are listed in Table B-6. Waste streams include those from: nuclear power plants, fuel fabrication, reprocessing and decommissioning activities, other fuel cycle facilities, institutional and industrial generators, and other non-fuel cycle generators. Source term characterization in the current version of IMPACTS represents an improvement over the original program which considered 23 radionuclides and a total of 37 waste types.

Consideration of such a large number of isotopes and waste types can be extremely unwieldy for a program to process. Thus, the range of potential options was bounded by consolidation into six waste spectra that denote the collective volume and other properties of all waste streams after processing by a set of selected waste treatment options. Each spectrum corresponds to a general level of waste performance (e.g., structural stability, resistance to dispersion) that result from application of processing options.

4.8 Scenarios

Exposure scenarios are considered for operations and post-closure phases. During operations, exposures resulting from routine operations and two accident scenarios (drop and breakage of waste package; release by fire) are included. Package drop scenario assumes airborne material may be transported offsite and calculates offsite exposures. Onsite impacts are not calculated but algorithm is provided so user can modify program. Similar approach is used for fire scenario, assuming waste is either combustible or is mixed with combustible waste material. Two approaches are used to examine routine exposures: one based primarily on number of shipments received, and a more involved method that accounts for different types of waste packaging and emplacement modes.

Post-disposal impacts are considered in four categories:

(1) Intruder scenarios - inadvertent human intrusion into waste disposal area.

(2) Groundwater scenarios - nuclide migration resulting in human exposures via groundwater pathway

(3) Leachate accumulation - leachate percolation and accumulation with disposal cells

(4) Exposed waste scenarios - waste is uncovered at surface and dispersed via air or water.

Intruder scenarios (and pre-closure operations) are acute dose exposure events and are termed "concentration scenarios" because they "depend upon the concentrations of the radionuclides within the waste streams considered". These are contrasted with the remaining three chronic exposure categories which are termed "total activity scenarios" since they "depend upon total radionuclide inventory and waste volume disposed at the disposal site."

Intruder scenarios include: (i) drilling (prior to construction); (ii) construction (basement is dug for a dwelling); (iii) discovery (same as construction except that intruder discovers waste and abandons site); (iv) agricultural (lives in house constructed on site and consumes food grown in contaminated soil).

Groundwater scenarios trace nuclide migration to four biota access locations downstream: (i) well at boundary of disposal area, (ii) well outside disposal buffer zone, (iii) well located between disposal facility and surface hydrologic boundary, and (iv) stream at surface hydrologic boundary. Annual dose rate (mrem/yr) during the 50th year of exposure is calculated as the product of radionuclide concentration, interaction factors and pathway dose conversion factors.

If the disposal site contains a liner or soil of low permeability beneath the waste, groundwater migration calculations are complicated due to potential accumulation of leachate. This "bathtub" effect which increases leach rate has been observed at some existing sites. Three leachate accumulation scenarios are addressed: (i) accumulated leachate is continuously pumped and treated prior to release, (ii) accumulated leachate overflows to nearby stream and enters accessible environment, and (iii) accumulated leachate is collected (batch mode) and treated by evaporation. However, the user must determine if the leachate accumulation scenario is appropriate by estimating whether leachate will exit the disposal cell at a rate less than the water infiltrating in. If this is not the case, then leachate flows through the trench and the groundwater migration scenario should be selected instead.

Exposure of earthen cover by inadvertent intruder or wind/surface water erosion lead to four exposed waste scenarios: (i) intruder-air, (ii) intruder-water, (iii) erosion-air, and (iv) erosion-water. Erosion scenarios assume a conservative time delay of 2000 years prior to exposure. Wind transport of eroded material use separate airborne mobilization rates for each of the four generic sites. A single, conservative surface water mobilization rate (greater than annual rate for Appalachian region) is used. In addition, a conservative dilution factor is used.

4.9 Relationship to Regulatory Standards

Output of calculated whole-body effective dose equivalent in terms of millirem/year is consistent with proposed federal radiation standards contained in 40 CFR 193. The current standard contained in NRC's 10 CFR 61 does not use the organ weighting factor associated with effective dose equivalent. Federal regulations also specify allowable radionuclide concentrations in air and water. These are calculated and used in the dose calculations, but are not reported in output from IMPACTS.

5 SCIENTIFIC CRITERIA

5.1 Theory

Given a number of waste streams with specific radionuclide concentrations and specified disposal and environmental conditions, IMPACTS integrates release/transport/exposure pathways to calculate radiological impacts, i.e., annual exposures. This is accomplished using the following general equation:

$$H_{ij} = {}_{n}C_{ijn} \times I_{ijn} \, PDCF_{n}$$

where H_{ij} is the dose rate to the individual (mrem/yr) resulting from waste stream (i) in waste class (j), where the impacts are summed over all individual radionuclides (n) in the waste, and where:

C_{ijn} = the concentration of radionuclide (n) in the (i)th waste stream in the (j)th waste class considered;

I_{ijn} - an interaction factor relating the concentration of the (n)th radionuclide in the (i)th waste stream in the (j)th waste class to the concentration of the radionuclide at the biota access location; and

$PDCF_n$ - the pathway dose conversion factor for that radionuclide (mrem/yr per Ci/m^3

The interaction factor, I_{ijn}, represents the fraction of the original nuclide source term that is transported through all environmental transport media. It, in turn, is the product of factors that account for radionuclide decay, disposal site design characteristics, physical and chemical characteristics of the waste, and environmental parameters. Each of these transport factors is determined through a series of scenario-specific calculations. For example, the interaction factor for groundwater scenarios considers the effects of: (i) rainwater percolation, (ii) radionuclide release to leachate, and (iii) migration reduction (including effects of site geometry, dilution, nuclide-soil interaction, groundwater travel time etc.).

The one-dimensional groundwater transport model used in IMPACTS assumes flow through unsaturated and saturated zones, each of which is stationary, homogeneous and isotropic, and fluid moving through these zones is incompressible and of constant viscosity. Migration through the unsaturated zone is treated separately from the saturated zone (these factors were combined in the original version of IMPACTS).

For unsaturated flow, the migration reduction factor accounts for radionuclide decay, distance between the waste and the saturated zone, flow speed, and retardation coefficient (dispersion is neglected to simplify calculations). Retardation coefficients reflect the ion exchange capability of soils and represent the ratio of the radionuclide velocity in the soil to the groundwater velocity. Five nuclide-specific retardation coefficient options are available that span the range of values for diverse types of geochemical environments, from sandy soils with low cation exchange capacity to high-clay soils with high cation exchange capacity. Default values for retardation coefficients are assigned for the regional generic sites, but these can be modified if specific conditions warrant (e.g., presence of chelating agents that reduce retardation effects.) Unsaturated zone thickness and groundwater travel speeds for each site are also given.

For saturated flow, the migration reduction factor accounts for the time lapse from initiation of the scenario to exposure, groundwater travel time, retardation coefficient of the aquifer, source duration time (treated as a fixed "slug" of activity that decreases linearly until depletion), and waste stream characteristics. To simplify the calculation, the disposal site is divided into 10 equal sectors and it is assumed that each sector releases 1/10 of the total activity as a point source. Impacts from radionuclide migration from each sector are then summed. Reference groundwater parameters such as travel speeds, dispersivity, and distance to surface water for each of the four generic sites are provided.

Pathway dose conversion factors ($PDCF_n$) are the product of a pathway usage factor (specific to the combination of nuclide, pathway, and scenario) and the fundamental dose conversion factor (specific to the combination of nuclide, pathway, and organ that receives the dose).

5.2 Validation

Model performance predictions have not been compared with actual disposal site performance.

5.3 Treatment of Radioactive Decay Products

Production of radionuclide chain daughter products is considered in the program. Concentrations of daughter products from four natural decay chains (U-238, U-235, Th-232, and Np-237) and artificial nuclides that merge with natural chains after several decay steps (e.g., Am-241, Pu-239) are calculated. These effects are automatically applied to calculations involving exposures to individual inadvertent intruders, off-site exposures to populations due to intrusion, and hypothetical impacts from erosion. Fractional ingrowth of daughter products are multiplied by respective pathway dose conversion factors and added to the pathway dose conversion factor for the parent nuclide. Since it is a gas, effects of radon-222 ingrowth are considered separately for inadvertent intruder scenarios.

5.4 Underlying Assumptions

As in any performance assessment model that attempts to simulate actual conditions and events, IMPACTS contains a number of significant simplifying assumptions. The documentation appears to do a good job alerting the reader to these underlying assumptions throughout the text. A separate section devoted to model limitations is included. The major limiting factor inherent in the analysis to which the user is duly cautioned, is its generic approach. Simplifying assumptions necessary for a comparative analysis of alternatives are not appropriate for analyses of a specific waste type, site or disposal technology. For example, despite the wide range of waste types and disposal options considered in the program, the combination of specific characteristics of a given waste stream, treatment/solidification technology, disposal site and environmental parameters, etc. is not necessarily encompassed. In the context of a comparative assessment, however, such limitations are not a significant liability.

5.5 Pathways

A number of exposure/transport pathway combinations are used by IMPACTS, based on the particular scenario simulated. Transport pathways are listed below, grouped by exposure modes:

<u>Ground and surface water modes</u>: bioaccumulation, saturated and unsaturated zone groundwater transport (including hydrospheric dispersion due to mixing, and sorption), and ingestion of water.

<u>Air exposure modes</u>: atmospheric dispersion, atmospheric deposition, resuspension, and inhalation.

Biological exposure modes: atmospheric deposition, irrigation, plant uptake, weathering by wind and/or water erosion, ingestion of contaminated vegetation, ingestion of contaminated animals.

5.6 Dose Conversion and Dose Response

Five sets of dose conversion factors are incorporated that cover internal exposure by ingestion and inhalation (50-yr committed dose, mrem/pCi ingested or inhaled), and external exposure from a volume source, area source, or immersion in uniformly contaminated air. Each set is radionuclide-specific (data for 53 radionuclides are included) and are given in terms of nine organs (whole body, red bone marrow, bone surface, liver, thyroid, kidney, lung, stomach wall, lower large intestine) plus the effective whole body equivalent, calculated using ICRP-26 and ICRP-30 methodology. Internal exposure dose conversion factors are based on data from ICRP-30; external dose conversion factors are based on data from Kocher (NUREG/CR-1918, August 1981). Dose response is not calculated.

Table B-4. Sample input data for IMPACTS

```
IMPACTS RUN OF REGION 4 + SOURCES
IR  =  4  OVFL=  0  IBUF=  0  NBRN=  0  NBES=  0
ICLS=    1  IOBS=    5  IINS=  100
IBEG= 1991  IEND= 2020  ILFE=   30

COMBINATION INDICES ARE:    1   4   4   4   4   1

MINIMUM DEPTHS ARE:    2.0   2.0   2.0   5.0  10.0  10.0

DISPOSAL CONFIGURATION

 NO ID IU IT IC IE IB IX IS    EFF        SEF        DPT        DTK        VOLE       AREA
  1  4  1  1  1  1  1  3  1 5.25E+00 6.90E-01 2.00E+00 7.00E+00 3.53E+03 6.00E+02
  2  3  2  2  2  1  2  2  1 1.14E+01 8.80E-01 2.00E+00 1.30E+01 6.55E+04 5.4UE+03
  3  3  2  2  2  1  2  2  1 1.14E+01 8.80E-01 2.00E+00 1.30E+01 6.55E+04 5.40E+03
  4  3  2  2  2  1  2  2  1 1.14E+01 8.80E-01 2.00E+00 1.30E+01 6.55E+04 5.40E+03
  5  3  2  2  2  1  2  2  1 1.14E+01 8.80E-01 2.00E+00 1.30E+01 6.55E+04 5.40E+03
  6 10  6  6  1  2  3  1  1 5.70E+00 4.40E-01 2.00E+00 5.70E+00 5.03E+03 8.60E+02

INPUTS STREAMS: NAME, REGN, BKLG, FRAC - OLD NSTR =  215
P-IXRESIN  4 0 1.00 P-CONCLIQ  4 0 1.00 P-FSLUDGE  4 0 1.00 P-FCARTRG  4 0 1.0U
B-IXRESIN  4 0 1.00 B-CONCLIQ  4 0 1.00 B-FSLUDGE  4 0 1.00 P-COTRASH  4 0 1.00
P-NCTRASH  4 0 1.00 B-COTRASH  4 0 1.00 B-NCTRASH  4 0 1.00 L-NFRCOMP  4 0 1.00
L-DECONRS  4 0 1.00 F-PROCESS  4 0 1.00 F-COTRASH  4 0 1.00 F-NCTRASH  4 0 1.00
U-PROCESS  4 0 1.00 L-PUDECON  4 0 1.00 L-BURNUPS  4 0 1.00 I-[illegible]TRASH  4 0 1.00
I+COTRASH  4 0 1.00 I-ABSLIQD  4 0 1.00 I+ABSLIQD  4 0 1.00 I-[illegible]IQSCVL  4 0 1.00
I+LIQSCVL  4 0 1.00 I-BIOWAST  4 0 1.00 I+BIOWAST  4 0 1.00 N-SSTRASH  4 0 1.00
N+SSTRASH  4 0 1.00 N-SSWASTE  4 0 1.00 N-LOTRASH  4 0 1.00 N+LOTRASH  4 0 1.00
N-LOWASTE  4 0 1.00 N-ISOPROD  4 0 1.00 N-ISOTRSH  4 0 1.00 N-SORMFG1  4 0 1.00
N-SORMFG2  4 0 1.00 N-SORMFG3  4 0 1.00 N-SORMFG4  4 0 1.00 N-NECOTRA  4 0 1.00
N-NEABLIQ  4 0 1.00 N-NESOLIQ  4 0 1.00 N-NEVIALS  4 0 1.00 N-NENCGLS  4 0 1.00
N-NEWOTAL  4 0 1.00 N-NETRGAS  4 0 1.00 N-NETRILI  4 0 1.00 N-NECARLI  4 0 1.00
N-MWTRASH  4 0 1.00 N-MWABLIQ  4 0 1.00 N-MWSOLIQ  4 0 1.00 N-MWWASTE  4 0 1.00
N-TRIPLAT  4 0 1.00 N-TRITGAS  4 0 1.00 N-TRISCNT  4 0 1.00 N-TRILIQD  4 0 1.00
N-TRITRSH  4 0 1.00 N-TRIFOIL  4 0 1.00 N-HIGHACT  4 0 1.00 N-TRITSOR  4 0 1.00
N-CARBSOR  4 0 1.00 N-COBLSOR  4 0 1.00 N-NICKSOR  4 0 1.00 N-STROSOR  4 0 1.00
N-CESISOR  4 0 1.00 N-PLU8SOR  4 0 1.00 N-PLU9SOR  4 0 1.00 N-AMERSOR  4 0 1.00
N-PUBESOR  4 0 1.00 N-AMBESOR  4 0 1.00 N-RANEEDS  4 0 1.00 N-RACELLS  4 0 1.00
N-RAPLAQU  4 0 1.00 N-RANPAPP  4 0 1.00 N-RABESOR  4 0 1.00 N-RAMISCL  4 0 1.00
N-CARBSOR  1 0 1.00 N-COBLSOR  1 0 1.00 N-NICKSOR  1 0 1.00 N-STROSOR  1 0 1.00
N-CESISOR  1 0 1.00 N-PLU8SOR  1 0 1.00 N-PLU9SOR  1 0 1.00 N-AMERSOR  1 0 1.00
N-PUBESOR  1 0 1.00 N-AMBESOR  1 0 1.00 N-RANEEDS  1 0 1.00 N-RACELLS  1 0 1.00
N-RAPLAQU  1 0 1.00 N-RANPAPP  1 0 1.00 N-RABESOR  1 0 1.00 N-RAMISCL  1 0 1.00
N-CARBSOR  2 0 1.00 N-COBLSOR  2 0 1.00 N-NICKSOR  2 0 1.00 N-STROSOR  2 0 1.00
N-CESISOR  2 0 1.00 N-PLU8SOR  2 0 1.00 N-PLU9SOR  2 0 1.00 N-AMERSOR  2 0 1.00
N-PUBESOR  2 0 1.00 N-AMBESOR  2 0 1.00 N-RANEEDS  2 0 1.00 N-RACELLS  2 0 1.00
N-RAPLAQU  2 0 1.00 N-RANPAPP  2 0 1.00 N-RABESOR  2 0 1.00 N-RAMISCL  2 0 1.00
N-CARBSOR  3 0 1.00 N-COBLSOR  3 0 1.00 N-NICKSOR  3 0 1.00 N-STROSOR  3 0 1.00
N-CESISOR  3 0 1.00 N-PLU8SOR  3 0 1.00 N-PLU9SOR  3 0 1.00 N-AMERSOR  3 0 1.00
N-PUBESOR  3 0 1.00 N-AMBESOR  3 0 1.00 N-RANEEDS  3 0 1.00 N-RACELLS  3 0 1.00
N-RAPLAQU  3 0 1.00 N-RANPAPP  3 0 1.00 N-RABESOR  3 0 1.00 N-RAMISCL  3 0 1.00
N-RARESIN  4 0 1.00 M-NAVYWET  4 0 1.00 M-NAVYDRY  4 0 1.00 P-DECORES  4 0 1.00
P-DEACINT  4 0 1.00 P-DEACVES  4 0 1.00 P-DEACTCO  4 0 1.00 P-DECONME  4 0 1.00
P-DECONCO  4 0 1.00 P-DETRASH  4 0 1.00 P-DERESIN  4 0 1.00 P-DEFILCR  4 0 1.00
P-DEEVAPB  4 0 1.00 B-DECORES  4 0 1.00 B-DEACINT  4 0 1.00 B-DEACVES  4 0 1.00
B-DEACTCO  4 0 1.00 B-DECONME  4 0 1.00 B-DECONCO  4 0 1.00 B-DETRASH  4 0 1.00
B-DERESIN  4 0 1.00 B-DEEVAPB  4 0 1.00 N-TRITSOR  1 0 1.00 N-TRITSOR  2 0 1.00
N-TRITSOR  3 0 1.00            0 0  .00            0 0  .00            0 0  .00
```

Table B-5. Key to symbols and acronyms used in IMPACTS.

Symbol	Property
AREA	Disposal cell area
BKLG	Years of backlog
DPT	Disposal cell depth
DTK	Disposal cell thickness
EFF	Volumetric disposal efficiency
FRAC	Fraction of waste volume shipped for disposal
IBEG	First year of operation
IBUF	Disposal facility buffer zone index
IC	Cover index
ICLS	Closure period
ID	Disposal technology
IE	Emplacement index
IEND	Last year of operation
IINS	Active institutional control period
ILFE	Operations period
IOBS	Observation period
IR	Region index
IS	Chemical segregation index
IT	Topmost waste
IU	Utilization index
IX	Compaction index
NBES	Scenario index
NBRN	Barnwell classification index
NSTR	Number of waste streams
OVFL	Overflow scenario index
REGN	Region
SEF	Surface disposal efficiency
VOLE	Disposal cell volume

Table B-6. Sample output from IMPACTS groundwater scenario (boundary well) by waste type.

CLASS = A

TIME	LUNGS	S. WALL	LLI WALL	T. BODY	KIDNEYS	LIVER	RED MAR	BONE	THYROID	ICRP
20 YR	7.28E-10	9.40E-10	1.25E-09	7.23E-10	7.45E-10	7.21E-10	7.19E-10	5.71E-10	7.21E-10	7.82E-10
40 YR	2.77E-07	1.96E-07	5.80E-07	8.53E-07	2.85E-07	3.02E-07	2.95E-07	3.22E-07	1.91E-03	5.74E-05
60 YR	3.11E-07	2.20E-07	6.51E-07	9.58E-07	3.20E-07	3.39E-07	3.31E-07	3.61E-07	2.14E-03	6.44E-05
80 YR	1.22E-05	1.69E-05	2.12E-05	2.80E-05	1.51E-05	1.74E-05	4.67E-05	9.67E-05	4.26E-03	1.49E-04
100 YR	1.36E-05	1.89E-05	2.35E-05	3.12E-05	1.68E-05	1.94E-05	5.22E-05	1.08E-04	4.30E-03	1.52E-04
120 YR	1.38E-05	1.91E-05	2.42E-05	3.21E-05	1.71E-05	1.97E-05	5.24E-05	1.08E-04	6.41E-03	2.16E-04
160 YR	2.67E-05	3.72E-05	4.65E-05	6.15E-05	3.31E-05	3.83E-05	1.03E-04	2.13E-04	8.55E-03	3.03E-04
200 YR	2.71E-05	3.76E-05	4.72E-05	6.26E-05	3.35E-05	3.88E-05	1.04E-04	2.14E-04	1.07E-02	3.67E-04
400 YR	6.52E-05	9.07E-05	1.13E-04	1.50E-04	8.08E-05	9.34E-05	2.51E-04	5.20E-04	2.14E-02	7.55E-04
600 YR	8.84E-05	1.24E-04	1.53E-04	2.03E-04	1.10E-04	1.27E-04	3.43E-04	7.12E-04	2.15E-02	7.99E-04
800 YR	1.22E-04	1.71E-04	2.10E-04	2.78E-04	1.51E-04	1.75E-04	4.76E-04	9.89E-04	2.16E-02	8.60E-04
1K YR	1.19E-04	1.68E-04	2.06E-04	2.72E-04	1.48E-04	1.72E-04	4.66E-04	9.68E-04	2.16E-02	8.55E-04
5K YR	7.47E-05	1.04E-04	1.29E-04	1.71E-04	9.25E-05	1.07E-04	2.88E-04	5.98E-04	2.15E-02	7.74E-04
10K YR	4.22E-05	5.79E-05	7.35E-05	9.80E-05	5.20E-05	6.00E-05	1.59E-04	3.28E-04	2.15E-02	7.15E-04
20K YR	1.47E-05	1.87E-05	2.62E-05	3.59E-05	1.77E-05	2.02E-05	4.97E-05	1.00E-04	2.14E-02	6.65E-04

CLASS = B

TIME	LUNGS	S. WALL	LLI WALL	T. BODY	KIDNEYS	LIVER	RED MAR	BONE	THYROID	ICRP
20 YR	1.59E-11	2.05E-11	2.71E-11	1.58E-11	1.62E-11	1.57E-11	1.57E-11	1.25E-11	1.57E-11	1.70E-11
40 YR	1.54E-09	1.13E-09	3.30E-09	4.68E-09	1.59E-09	1.69E-09	1.64E-09	1.79E-09	1.04E-05	3.13E-07
60 YR	1.72E-09	1.26E-09	3.70E-09	5.25E-09	1.78E-09	1.89E-09	1.84E-09	2.00E-09	1.17E-05	3.52E-07
80 YR	6.23E-09	6.51E-09	1.22E-08	1.68E-08	7.04E-09	7.83E-09	1.49E-08	2.74E-08	2.32E-05	7.02E-07
100 YR	6.60E-09	7.01E-09	1.28E-08	1.76E-08	7.49E-09	8.35E-09	1.62E-08	3.02E-08	2.34E-05	[illegible]
120 YR	8.28E-09	8.23E-09	1.64E-08	2.28E-08	9.22E-09	1.02E-08	1.80E-08	3.22E-08	3.48E-05	1.05E-06
160 YR	1.31E-08	1.39E-08	2.54E-08	3.50E-08	1.48E-08	1.65E-08	3.21E-08	5.97E-08	4.65E-05	1.41E-06
200 YR	1.48E-08	1.51E-08	2.91E-08	4.02E-08	1.66E-08	1.84E-08	3.40E-08	6.19E-08	5.81E-05	1.76E-06
400 YR	3.23E-08	3.41E-08	6.28E-08	8.66E-08	3.66E-08	4.08E-08	7.87E-08	1.46E-07	1.16E-04	3.53E-06
600 YR	3.80E-08	4.22E-08	7.26E-08	9.95E-08	4.37E-08	4.90E-08	1.01E-07	1.93E-07	1.17E-04	3.55E-06
800 YR	4.61E-08	5.38E-08	8.66E-08	1.18E-07	5.39E-08	6.08E-08	1.34E-07	2.61E-07	1.17E-04	3.56E-06
1K YR	4.55E-08	5.29E-08	8.55E-08	1.17E-07	5.31E-08	5.99E-08	1.31E-07	2.56E-07	1.17E-04	3.56E-06
5K YR	3.46E-08	3.73E-08	6.63E-08	9.19E-08	3.95E-08	4.41E-08	8.79E-08	1.65E-07	1.17E-04	3.54E-06
10K YR	2.66E-08	2.58E-08	5.21E-08	7.39E-08	2.95E-08	3.25E-08	5.62E-08	9.91E-08	1.17E-04	3.53E-06
20K YR	1.98E-08	1.59E-08	3.95E-08	5.86E-08	2.10E-08	2.25E-08	2.94E-08	4.33E-08	1.17E-04	3.51E-06

CLASS = C

TIME	LUNGS	S. WALL	LLI WALL	T. BODY	KIDNEYS	LIVER	RED MAR	BONE	THYROID	ICRP
20 YR	6.34E-14	8.19E-14	1.08E-13	6.29E-14	6.49E-14	6.28E-14	6.26E-14	4.97E-14	6.28E-14	6.81E-14
40 YR	2.43E-10	1.72E-10	5.09E-10	7.50E-10	2.50E-10	2.65E-10	2.59E-10	2.82E-10	1.68E-06	5.05E-08
60 YR	2.88E-10	2.04E-10	6.05E-10	8.89E-10	2.97E-10	3.15E-10	3.07E-10	3.35E-10	1.99E-06	5.99E-08
80 YR	1.49E-09	1.74E-09	2.76E-09	3.80E-09	1.74E-09	1.96E-09	4.36E-09	8.54E-09	3.69E-06	1.13E-07
100 YR	1.64E-09	1.92E-09	3.03E-09	4.17E-09	1.92E-09	2.17E-09	4.86E-09	9.52E-09	3.94E-06	1.20E-07
120 YR	1.87E-09	2.09E-09	3.52E-09	4.90E-09	2.16E-09	2.42E-09	5.10E-09	9.79E-09	5.57E-06	1.70E-07
160 YR	3.21E-09	3.78E-09	5.93E-09	8.16E-09	3.76E-09	4.25E-09	9.55E-09	1.88E-08	7.62E-06	2.33E-07
200 YR	3.48E-09	3.98E-09	6.50E-09	9.00E-09	4.05E-09	4.55E-09	9.86E-09	1.91E-08	9.47E-06	2.89E-07
400 YR	7.82E-09	9.22E-09	1.45E-08	1.99E-08	9.18E-09	1.04E-08	2.33E-08	4.58E-08	1.86E-05	5.68E-07
600 YR	9.79E-09	1.20E-08	1.78E-08	2.44E-08	1.16E-08	1.32E-08	3.11E-08	6.20E-08	1.87E-05	5.75E-07
800 YR	1.26E-08	1.60E-08	2.26E-08	3.07E-08	1.51E-08	1.73E-08	4.22E-08	8.51E-08	1.88E-05	5.82E-07
1K YR	1.24E-08	1.57E-08	2.23E-08	3.03E-08	1.49E-08	1.70E-08	4.15E-08	8.36E-08	1.88E-05	5.81E-07
5K YR	8.69E-09	1.04E-08	1.59E-08	2.19E-08	1.03E-08	1.16E-08	2.67E-08	5.29E-08	1.87E-05	5.74E-07
10K YR	5.96E-09	6.52E-09	1.11E-08	1.58E-08	6.85E-09	7.66E-09	1.59E-08	3.03E-08	1.87E-05	5.69E-07
20K YR	3.65E-09	3.20E-09	7.02E-09	1.05E-08	3.96E-09	4.31E-09	6.74E-09	1.12E-08	1.87E-05	5.65E-07

Table B-7. Radionuclide-solubility combinations considered by IMPACTS.

Nuclide	Solubilities	Half-Life (Years)	Nuclide	Solubilities	Half-Life (Years)
H-3	*[1]	1.23E+01[2]	Th-228	W,Y	1.91E+00
C-14	*	5.73E+03	Th-229	W,Y	7.34E+03
Na-22	D	2.62E+00	Th-230	W,Y	8.00E+04
Cl-36	D,W	3.08E+05	Th-232	W,Y	1.41E+10
Fe-55	W;Y	2.60E+00	Pa-231	W,Y	3.25E+04
Co-60	W,Y	5.26E+00	U-232	D,W,Y	7.20E+01
Ni-59	D,W	8.00E+04	U-233	D,W,Y	1.62E+05
Ni-63	D,W	9.20E+01	U-234	D,W,Y	2.47E+05
Sr-90	D,Y	2.81E+01	U-235	D,W,Y	7.10E+08
Nb-94	W,Y	2.00E+04	U-236	D,W,Y	2.39E+07
Tc-99	D,W	2.12E+05	U-238	D,W,Y	4.51E+09
Ru-106	Y	1.01E+00	Np-237	W,Y	2.14E+06
Ag-108m	D,W,Y	1.27E+02	Pu-236	W,Y	2.85E+00
Cd-109	D,W,Y	1.24E+00	Pu-238	W,Y	8.64E+01
Sn-126	D,W	1.05E+05	Pu-239	W,Y	2.44E+04
Sb-125	D,W	2.71E+00	Pu-240	W,Y	6.58E+03
I-129	D	1.17E+07	Pu-241	W,Y	1.32E+01
Cs-134	D	2.05E+00	Pu-242	W,Y	3.79E+05
Cs-135	D	3.00E+06	Pu-244	W,Y	7.60E+07
Cs-137	D	3.00E+01	Am-241	W,Y	4.58E+02
Eu-152	W	1.27E+01	Am-243	W,Y	7.95E+03
Eu-154	W	1.60E+01	Cm-242	W,Y	4.45E-01
Pb-210	W	2.04E+01	Cm-243	W,Y	3.20E+01
Rn-222	*	1.05E-02	Cm-244	W,Y	1.76E+01
Ra-226	W	1.60E+03	Cm-248	W,Y	4.70E+05
Ra-228	W	6.70E+00	Cf-252	W,Y	2.65E+00
Ac-227	W,Y	2.16E+01			

(1) Solubility: * - Not applicable, D - Day, W - Week, Y - Year
(2) Exponential Notation: 1.23E+01 = 1.23×10^1
(3) Radiation types: a - Alpha, b - Beta, g - Gamma

```
01: Number of landscape cells
08: Number of compartments in each cell
01: Output control, 01 for more detailed output otherwise 00
** Compartment names in 10 A6 format on the following record(s) **
 AIR   PMAIR BIOTA SOIL1 SOIL2 GWTR  SWTR  SDMT  LSTC  NXTC
** Landscape and problem title on the following record **
 Western Ecoregion
 1.00  e04:  1 area in km**2
 1.00  e03:  2 height of the air compartment (m)
 5.00  e-6:  3 humidity (kg/l)
 3.20  e01:  4 precipitation onto land  (cm/yr)
 0.30  e00:  5 precipitation onto surface water (cm/yr)
 1.26  e01:  6 total surface water runoff (cm/yr)
 1.10  e01:  7 land surface runoff (cm/yr)
 6.15  e01:  8 atmospheric dust load (micro-gm/m**3)
 3.34  e02:  9 deposition velocity of atmospheric particles m/d
 7.00  e05: 10 biota dry mass inventory (kg/km**2)
 9.00  e04: 11 biota dry mass production (kg/km**2/yr)
 3.30  e-1: 12 biota dry mass fraction
 1.88  e01: 13 evapotranspiration from soil (cm/yr)
 0.16  e00: 14 evaporation from surface water (cm/yr)
 1.60  e-1: 15 thickness of the A soil horizon (m)
 1.50  e00: 16 bulk density of the soil in the A horizon (kg/L)
 3.80  e-1: 17 water content of the soil in the A horizon (kg/L)
 7.00  e-2: 18 volumetric air content in the A horizon (L/L)
 1.19  e06: 19 mechanical erosion rate (kg/km**2/yr)
 0.68  e00: 20 irrigation from ground water (cm/yr)
 1.00  e01: 21 thickness of the B soil horizon (m)
 3.00  e-1: 22 water content of the soil in the B horizon (kg/L)
 1.70  e00: 23 bulk density of the soil in the B horizon (kg/L)
 5.00  e-2: 24 volumetric air content in the B horizon (L/L)
 1.10  e10: 25 groundwater inventory (kg/km**2)
 3.50  e-1: 26 porosity of rock in the ground water zone (L/L)
 1.70  e00: 27 density of rock in the ground water zone (kg/L)
 8.42  e-3: 28 fraction of the total surface area in surface water
 6.00  e00: 29 average depth of surface waters (m)
 8.90  e-3: 30 suspended sediment load in surface water (kg/L)
 3.83  e03: 31 deposition rate of suspended sediment (kg/m**2/yr)
 5.00  e-2: 32 thickness of the sediment layer (m)
 1.50  e00: 33 bulk density of the sediment layer (kg/L)
 0.20  e00: 34 porosity of the sediment zone
 3.83  e03: 35 resuspension rate from the sediment layer (kg/m**2/yr)
 2.83  e02: 36 ambient environmental temperature (k)
 1.00  e-2: 37 boundary layer thickness at air/soil interface (m)
 1.00  e-2: 38 boundary layer thickness at water/air interface (m)
 2.00  e-2: 39 boundary layer thickness at sediment/water interface (m)
 1.20  e-2: 40 fraction organic carbon in the upper soil zone
 2.00  e-3: 41 fraction organic carbon in the lower soil zone
 2.87  e-4: 42 fraction organic carbon in the groundwater zone
 2.00  e-2: 43 fraction organic carbon in the sediment zone
 1.00  e00: 44 wet deposition scavenging efficiency (default = 1)
 1.00  e00: 45 yearly average wind speed (m/s)
```

Figure B-3. Sample landscape data file (western ecoregion).

```
arsenic   :element or compound name
7.4900e01 :molecular weight
1         :type (1 for element, 0 or blank for organic compound)
1.00  e-6 :henry's law constant (torr/(mole/l)
3.171e-17 :diffusion coefficient in air (m**2/s)
3.171e-17 :diffusion coefficient in water (m**2/s)
1.00  e-1 :part. coeff., ksp   (ppm in plant dry mass)/(ppm total soil)
7.50  e01 :bioconcentration factor, bcf  (ppm in fish meat)/(ppm water)
4.00  e-2 :part. coeff., xkmd1 (mg/kg in meat)/(mg/kg dm in cattle diet)
4.00  e-2 :part. coeff., xkmd2 (mg/kg in milk)/(mg/kg dm in cattle diet)
1.300 e03 :part. coeff., xkds1 (ppm soil)/(ppm water) in upper soil
1.300 e03 :part. coeff., xkds2 (ppm soil)/(ppm water) in lower soil
1.300 e03 :part. coeff., xkdgw (ppm rock)/(ppm water) in groundwater
1.300 e03 :part. coeff., xkdsd (ppm solid)/(ppm water) in surface water
*** removal rate constants (1/day) in each compartment ***
1.10  e-4 8  :sdmt
                    end of list
*** transform rate constants (1/day) in each compartment ***
                    end of list
*** the source vector (values in mol/km**2 per day) ***
2.7397e-3 4  :soil1
                    end of list
benzene   :element or compound name
7.8120e01 :molecular weight
          :type (1 for element, 0 or blank for organic compound)
4.10  e03 :henry's law constant (torr/(mole/l)
4.80  e01 :organic carbon partition coefficient koc
4.98  e-6 :diffusion coefficient in air (m**2/s)
4.98 e-10 :diffusion coefficient in water (m**2/s)
0.00  e00 :part. coeff., ksp   (ppm in plant dry mass)/(ppm total soil)
7.50  e01 :bioconcentration factor, bcf  (ppm in fish meat)/(ppm water)
4.00  e-2 :part. coeff., xkfd1 (mg/kg in meat fat)/(mg/kg dm in cattle diet)
4.00  e-2 :part. coeff., xkfd2 (mg/kg in milk fat)/(mg/kg dm in cattle diet)
*** removal rate constants (1/day) in each compartment ***
2.08  e-2 1  :air
2.08  e-2 2  :pmair
1.10  e-4 8  :sdmt
                    end of list
*** transform rate constants (1/day) in each compartment ***
                    end of list
*** the source vector (values in mol/km**2 per day) ***
2.7397e-3 4  :soil1
                    end of list
```

Figure B-4. Sample Chemical data file (arsenic and benzene).

```
7.0    E01:  1  AMASS THE BODY MASS OF AN ADULT IN KG
1.70   E01:  2  CMASS THE BODY MASS OF A CHILD IN KG
1.80   E00:  3  AAREA THE SURFACE AREA OF AN ADULT IN M**2
0.68   E00:  4  CAREA THE SURFACE AREA OF A CHILD IN M**2
2.20   E04:  5  AINHL THE INHALATION BY AN ADULT IN L/DAY
1.00   E04:  6  CINHL THE INHALATION BY A CHILD IN L/DAY
2.0    E00:  7  ADRNK THE INGESTION OF WATER BY AN ADULT IN L/DAY
1.0    E00:  8  CDRNK THE INGESTION OF WATER BY A CHILD IN L/DAY
0.13   E00:  9  AVEG  THE INGESTION OF VEGETATION BY AN ADULT IN KG(DM)/DAY
0.15   E00: 10  CVEG  THE INGESTION OF VEGETATION BY A CHILD IN KG(DM)/DAY
0.30   E00: 11  AMILK THE INGESTION OF MILK BY AN ADULT IN L/DAY
0.50   E00: 12  CMILK THE INGESTION OF MILK BY A CHILD IN L/DAY
0.30   E00: 13  AMEAT THE INGESTION OF MEAT BY AN ADULT IN KG/DAY
0.10   E00: 14  CMEAT THE INGESTION OF MEAT BY A CHILD IN KG/DAY
6.50   E-3: 15  AFISH THE INGESTION OF FISH BY AN ADULT IN KG/DAY
2.00   E-4: 16  CFISH THE INGESTION OF FISH BY A CHILD IN KG/DAY
5.80   E-5: 17  ASOIL THE INGESTION OF SOIL BY AN ADULT IN KG/DAY
1.00   E-4: 18  CSOIL THE INGESTION OF SOIL BY A CHILD IN KG/DAY
```

Figure B-5. Sample input data file for the program EXPOSE.

```
 01: Control variable. When set to 00, initial inventories in all
*    compartments are set to 0.0 . When set to 01, initial inventories
*    are set at the steady-state values obtained from GEOTOX A.
 21: Total number of time steps (must be < or = 50)
** Time step values (in days)
**  listed in F11.1 format (values between colons are not read by the program)
  000000.0:-  1 -:        10.0:-  2 -:        20.0:-  3 -:        30.0:-  4 -:
     100.0:-  5 -:       150.0:-  6 -:       200.0:-  7 -:       250.0:-  8 -:
     300.0:-  9 -:       350.0:- 10 -:       400.0:- 11 -:       450.0:- 12 -:
     500.0:- 13 -:       550.0:- 14 -:       600.0:- 15 -:       650.0:- 16 -:
     700.0:- 17 -:       750.0:- 18 -:       800.0:- 19 -:       900.0:- 20 -:
    1000.0:- 21 -:
* The source data is used to construct a source of the following form:
*   SOURCE(i,t) = [(A + B*H(t,T1,T2)/(T1 - T2)]*AREA
*    where,
*       SOURCE = the time dependent source term for species i at
*                time t in mole/day,
*       A = the time-independent component of the source in mol/km**2 per day,
*       B = the total amount of material that is introduced over the
*           time T1-T2 in mol/km**2,
*       H(t,T1,T2) = function whose value is 1 when t is between T1 and
*                    T2 and 0 otherwise,
*       T1 = time in days when the "B" component of the source begins,
*       T2 = time in days when the "B" component of the source ends, and
*       AREA = the area of the landscape in km**2.
     000.0: Value of T1 in days
     100.0: Value of T2 in days
*Source constants in E10.3 format
* Chemical   "A" constants   "B" constants    Compartment    Compartment
*  number     (mole/day)        (mole)           number          name
------>01:-----> 2.74  e-3:-----> 1.37  e02:--------->04:-------> SOIL1:
      02         2.74  e-3        1.37  e00           04          SOIL1
      03         2.74  e-3        1.37  e-2           01          AIR
      04         2.74  e-3        1.37  e-2           01          AIR
```

Figure B-6. Sample time step and source term data file.

1 IDENTIFICATION

1.1 Name:	GEOTOX Version 1.2
1.2 Prepared by:	Thomas E. McKone, Environmental Sciences Division, Lawrence Livermore National Laboratory
1.3 Prepared for:	U.S. Army Biomedical Research and Development Laboratory (USABRDL)
1:4 Report Title:	GEOTOX Multimedia Compartment Model User's Guide
1.5 Report Number:	UCRL-15913
1.6 Report Date:	May 1987
1.7 Availability:	Documentation: U.S. Government Printing Office, National Technical Information Center Software: T.E. McKone or D.W. Layton, Environmental Sciences Division, Lawrence Livermore National Laboratory
1.8 Purpose and Scope:	Multimedia compartment screening model that simulates the transport and transformation of environmental contaminants and potential human exposure.

2 SUMMARY OF FINDINGS

GEOTOX is a set of programs used to calculate time-varying chemical concentrations in multiple environmental media (e.g., soil, ground water, etc.) and to estimate potential human exposures. The current version of GEOTOX performs two major tasks: (1) it predicts the transport and transformation of chemicals in a multimedia environment, and (2) it estimates human exposure. The chemical transport model uses landscape data and physicochemical properties to determine the distribution and concentration of chemicals among compartments such as air, water, and soil. Environmental concentrations are linked to human exposures and health effects using an exposure model that accounts for intake through inhalation, consumption of food and water, and dermal absorption. GEOTOX is intended for use in public health and environmental risk assessment and risk management -- particularly for the screening and ranking of chemicals according to the potential risks they pose.

The present version of GEOTOX is coded in Microsoft[R] FORTRAN and designed for the IBM Personal Computer (PC) and its family of successors and compatibles. GEOTOX was originally developed for ranking the potential health risks associated with toxic metals and radionuclides in the global environment (McKone, 1981). Recently, at the Lawrence Livermore National Laboratory, the model has been extended to handle organic chemicals (Layton et al., 1986). GEOTOX was originally implemented on mainframe systems. Because of this, most of the input must be entered in fixed-format data files. However, comments have been added to the input files to make this format easy to use.

The U.S. Army Biomedical Research and Development Laboratory (USABRDL) has been charged with the responsibility of developing environmental criteria for military materials such as propellants, pyrotechnics, and explosives as well as by-products associated with their demilitarization. USABRDL has funded Lawrence Livermore National Laboratory to prepare a health and environmental effects data-base assessment on technologies for demilitarizing conventional ordnance and the environmental by-products of those technologies. In the first phase of the project, explosives and associated by-products were ranked according to their potential health risks. The GEOTOX program addressed in this report was used to estimate the equilibrium partitioning of demilitarization by-products between different environmental compartments and to determine human exposures to the contaminants via different pathways (e.g., inhalation, ingestion, etc.).

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

The original version of GEOTOX was developed at the University of California at Los Angeles for ranking the potential risks of toxic metals and radionuclides in the global environment. A detailed description of this version of the model is provided in the dissertation entitled "Chemical Cycles and Health Risks of Some Toxic Crustal Nuclides" (McKone, 1981); and a summary and application of the original model is published in the journal Risk Analysis (McKone et al., 1983). At Lawrence Livermore National Laboratory, the model was modified to handle organic chemicals and to run on a personal computer. These modifications are described in the report "Demilitarization of Conventional Ordnance: Priorities for Data-Base Assessments of Environmental Contaminants," (Layton et al.,1986). This version of the model was referred to as GEOTOX version 1.0. The theory behind this version of the model was published in a peer review journal article (McKone and Layton, 1986). A user's guide for GEOTOX was prepared in 1987 (McKone et al., 1987). Additional discussions of the revised GEOTOX model (Version 1.0) are available in McKone (1986), McKone and Layton (1986b), McKone and Kastenberg (1986) and Layton and McKone (1988). The current version of the model is Version 1.2. Differences between Versions 1.0 and 1.2 are described in the user's guide supplement (McKone, 1988).

3.2 Hardware Requirements

GEOTOX requires an IBM PC, XT, AT, or compatible, running under MS-DOS[R] (version 2.0 or higher), with at least 150 kilobytes of free RAM. The ANSI device driver, supplied with DOS, is also requires. a printer with 8.5 inch or wider paper is recommended. The program does not require an 8087 math coprocessor, but runs faster if one is present.

3.3 Application

GEOTOX consists of three principal programs: GEOTOX-A, GEOTOX-B, and EXPOSE. Each of these programs is designed to handle a specific task in the sequence of calculations that translate the continuous or short-term addition of a chemical to a compartment (e.g., the upper-soil layer) into an estimate of human exposure. The GEOTOX batch file coordinates the input, execution, and the output of the GEOTOX modules.

GEOTOX-A calculates the transfer rate constants and finds the steady-state inventories of chemicals in an eight-compartment landscape. GEOTOX-B uses the matrix of transfer and transformation rate constants calculated by GEOTOX-A and sources defined as input to calculate time-varying inventories and concentrations in the eight compartments. GEOTOX-B calls a set of subroutines called GEARB to solve systems of first-order ordinary differential equations. GEARB selects its own time step size according to an internal convergence test. It only returns output information for the times specified by the user. EXPOSE uses the information from GEOTOX-A or GEOTOX-B to determine human exposure by inhalation, drinking water, biota ingestion, meat and dairy product ingestion, fish ingestion, soil ingestion, and dermal absorption. Using EXPOSE with input from GEOTOX-A results in calculations of steady-state exposure, while using EXPOSE with input from GEOTOX-B results in time-varying exposures.

3.4 Level of Expertise Required

The set of GEOTOX programs is controlled by a batch file which limits the need for the user to define input and output sets. Through the use of function keys this batch file allows the user to execute specific GEOTOX programs and view the output listings and graphic output of the programs. There is currently no "user friendly" system for preparing input to the GEOTOX programs. However, the program comes with several sample input files that have comments nested with each line to assist the user in tracing and modifying the inputs. Creating or modifying the input files requires an editor such EDLIN, but no editor is included with the program.

TECHNICAL CRITERIA

4.1 Peer Review

The GEOTOX model itself has not received extensive peer review, however the models and theory behind this model have been published in the peer review literature. A summary review and application of the original GEOTOX model is published in the journal Risk Analysis (McKone et al., 1982). The theory behind Version 1.0 was published in Regulatory Toxicology and Pharmacology (McKone and Layton, 1986).

4.2 Verification

The GEOTOX program was tested and debugged as part of the program development process at Lawrence Livermore National Laboratory including verification of mathematical computations.

4.3 Uncertainty

GEOTOX offers the user the option of performing a Monte Carlo analysis, but this option is not available in Version 1.2 and will not execute.

4.4 *Sensitivity*

The user does have the option of performing a first order differential sensitivity analysis using GEOTOX A and EXPOSE A by requesting option B from the list of three options that appears when the program GEOTOX A begins execution. This sensitivity analysis can only be applied to one chemical at a time.

The differential sensitivity analysis calculates the normalized partial derivative of the output chemical concentrations and human exposure with respect to each input parameter in both the landscape and chemical properties file. This derivative is calculated as follows:

$$\left|\frac{dF}{dX_i}\right| = a\ (F[X_i] - F[X_i*(1 - a)])/F \tag{1}$$

where $|dF/dX_i|$ is the normalized partial derivative of the output F with respect to the parameter X_i and a is small fraction (0.01). The differential sensitivity analysis allows one to make a quick estimate of the influence of small changes in each input parameter on the reference estimates of chemical concentration and human exposure.

4.5 *Required Inputs*

GEOTOX requires four classes of input data: landscape description, chemical description, human exposure data and source term data. These data are accessed by the program through three input files, which are detailed below.

Landscape Data. The GEOTOX program uses the protocol that all landscape input files end with the extension ".LND". All files with this extension are listed on the screen when the GEOTOX-A program requests a landscape file. There are four landscape files available in the data disk:

CALIF.LND, NORTHC.LND, SOEAST.LND, and WEST.LND.

which represent the California, northeast/central, southeastern, and western ecoregions of the United States. These files contain six setup records and 45 input parameters. Figure 1 provides a listing of the input file WEST.LND as an example of a landscape data file. Each entry line is divided into two sections. The entry value is listed first, followed by a colon and a description of the variable. This description is provided for the benefit of the user but not read by the program. Lines that begin with an asterisk are comment lines, which are not read by the program but must be present.

Chemical Data. GEOTOX uses the protocol that all chemical input files end with the extension ".CHM". all files with this extension are listed on the screen when the GEOTOX-A program requests a chemical file. There are fourteen chemical data files available on the data disk:

ARSENIC.CHM	DBCP.CHM	DBUTLPHT.CHM	EDB.CHM	LEAD.CHM
PCE.CHM	TCDD.CHM	TCE.CHM	TEN.CHM	TEST.CHM
TNT.CHM	VOCS.CHM	BENZENE.CHM	CADMIUM.CHM	

All but three of these contain a single chemical corresponding to the name of the file. The file TEN.CHM contains ten of the listed chemical species; the file VOCS.CHM contains three volatile compounds; and the file TEST.CHM contains arsenic and benzene data used for the sample problem. GEOTOX-A can handle up to 10 chemicals per set. However, GEOTOX-B can only handle 4 per set. Figure B-4 provides a listing of the input file TEST.CHM as an example of a chemical data file. The first record provides the name of the first chemical species in the file. The second record provides the molecular weight of this element or compound. The third record signals whether the chemical is an organic (0 or blank), or inorganic species (1). For an inorganic species, the next eleven records provide information on its environmental properties. For an organic species, the next eight records provide environmental data. There must be exactly eleven records for an inorganic species and exactly eight records for an organic species. The removal rate constants are stored in an array of length n (where n is the number of compartments). Only nonzero values need be specified. The rate constant (d^{-1}) and the corresponding compartment number are read in the same line. The removal rate constants are the rate constants for removal by radioactive decay and all physical and chemical transformations, including biodegradation, hydrolysis, etc. The input array is terminated by a record having ten blank columns at the beginning. The next array contains the transform rate constants, which reflect the rate (d^{-1}) at which a given chemical transforms to the next chemical on the list. The last array is the steady state source vector, which specifies the source (mole/km^2-d) and the compartment number. There must be a non-zero source term in at least one compartment for the program to run.

Exposure Data. The program EXPOSE reads input parameters for the exposure calculation (i.e. human body mass and ingestion rates) from the file EXPOSE.DAT, which can be modified by the user. Figure B-5 lists the contents of the file EXPOSE.DAT. In this file as in all GEOTOX input files, anything to the right of the colon is not used as input but is only for identification.

GEOTOX-B Input. The file TIME.DAT is the time step/source term input file for GEOTOX B. This file tells GEOTOX-B the time steps for the output file and for the exposure calculation and defines the time-dependent source term. TIME.DAT can have source constants listed for four chemicals even if less than four chemicals are being used in the calculation. Only the required records are read and the others are ignored. The first line of this file allows the user to accept the steady-state inventories from GEOTOX-A as the initial inventories or to set all compartment inventories to zero. The sample input file TIME.DAT is shown in Figure B-6.

4.6 Output

GEOTOX-A Output. GTXA.OUT is the output file from GEOTOX-A that is sent to the printer. The first page of output summarizes the distribution of mass among the eight compartments of the landscape cell and the exchange of solids and liquids between the compartments. Following this overview, the movement of each species identified in the chemical input file is allotted one page of output. A listing of the physical constants and partition coefficients is followed by a table of the rate constants that define the movement of the species in the eight-compartment landscape. Three rate constants mediate the movement of chemicals: alpha (diffusion), beta (liquid advection), and betas (solid advection). After the rate constants are calculated, the inflow and outflow of the chemical is listed for each compartment. Finally, the distribution of the chemical in each compartment is summarized. The final summary table provides an estimated inventory of the specific chemical in each of the eight compartments. Figure B-7 shows GEOTOX-A output for the sample problem.

GEOTOX-B Output. GTXB.OUT is the output file from GEOTOX-B that is sent to the printer. For each time step specified in TIME.DAT, GTXB.OUT lists the parameters that define the compartmental inventories for each chemical being considered. These parameters include the source rate, the inventory, the concentration, and the loss rate for each compartment.

EXPOSE Output. EXPOSA.OUT is the steady-state exposure file written by EXPOSE that is based on the compartment inventories produced by GEOTOX-A. For each chemical being considered, EXPOSA.OUT lists the adult, child, and lifetime average exposures via seven pathways. EXPOSB.OUT is the time-varying exposure file written by EXPOSE that is based on the compartment inventories produced by GEOTOX-B. For each chemical being considered, EXPOSA.OUT lists the adult, child, population average, and cumulative exposures via seven pathways, for each of the time steps specified in TIME.DAT as input to GEOTOX-B.

Graphic Output. Graphic output displays are available for the programs GEOTOX A and EXPOSE A. The graphic output for GEOTOX-A lists the distribution of each contaminant by compartment. Figure B-8 provides a sample of the graphic output from GEOTOX-A. In the EXPOSE graphic output, the chemical selected has its exposure distribution summarized by one of two histograms. The EXPOSE graphic output also provides an option to compare all chemicals in a given set on the basis of the ratio of the exposure estimated by GEOTOX relative to a reference exposure entered by the user. Figures B-9 through B-14 show sample graphic output from EXPOSE. GEOTOX-B produces an output file called GXBOUT.PRN in a format that allows it to be imported to Lotus 123(R) in order to create graphs of the output or perform other types of analyses available in the spread-sheet program. Figure B-15 shows a plot that was made using Lotus 123(R) and the data in GXBOUT.PRN for the sample problem.

4.7 Source Term

The source S(i,n,t) for the n^{th} chemical in compartment i at time t is defined by the expression:

$$S(i,n,t) = \{a + b/(t_2 - t_1) \times H(t,t_1,t_2)\}A \quad (1)$$

where:

a = source constant for the continuous source, mole/km^2-d;

b = source constant for the time-varying source, mole/km^2;

t_1, t_2 = time that define the interval during which a time-varying source is present, d;

$H(t,t_1,t_2)$ = a step function that is equal to 1 if t is between t_1 and t_2, and is equal to 0 otherwise, and

A = area of the landscape, km^2

GEOTOX-B uses Equation 1 to obtain the compartment inventories for situations in which the source term varies as a function of time.

4.8 Scenarios

GEOTOX is a tool for screening the potential risks of contaminants released as nonpoint sources to soils, air, ground water, and surface water. It can handle radionuclides, metals, volatile organic compounds, and other organic compounds in a single execution. GEOTOX serves as a screening model when it is used to calculate the exposure that results from the addition of each chemical to environmental compartments within a specified ecoregion.

It is widely recognized that of the roughly 70,000 chemicals now in commercial use, some can cause serious harm at low exposure levels. Nonetheless, relatively little is known about how most of these species behave in the environment. The very magnitude of this list precludes a detailed assessment of every chemical. However, with limited data, one can perform preliminary screening studies to identify the compounds that represent the most immediate hazards. This, indeed, is the goal of GEOTOX.

A multimedia compartment model can screen chemicals based on their inherent toxicity, persistence, and dilution. In this context, it can be viewed as a tool for comparing potential risks. The concept of risk applies here, because we are dealing with uncertainties in both exposure and health effects. McKone and Kastenberg (1986) have identified five criteria that make a multimedia model suitable for risk assessment or risk management: (1) able to handle organic chemicals, trace metals, and radionuclides; (2) fast enough to allow multiple runs for sensitivity studies; (3) able to link environmental concentrations with human exposure pathways; (4) able to handle dynamic and steady-state situations; and (5) flexible enough for easy alteration to simulate different types of environments. No model fully satisfies all these criteria, but the GEOTOX model was developed with these objectives in mind.

An important feature of the GEOTOX model is the ability to link estimated environmental concentrations with exposure pathways in order to project accumulated lifetime exposures within a population. GOETEX uses three primary exposure pathways -- inhalation, ingestion, and dermal absorption. The ingestion route is divided into water; fruits, grains, and vegetables; milk and meat; fish; and soil. Exposure is expressed as the daily contact in mg/kg of body weight of a given chemical with the lungs, gut wall, or skin surface. Exposure is averaged over three age categories--child (0 to 15 y), adult (15 to 70 y), and lifetime average.

5 SCIENTIFIC CRITERIA

5.1 Theory

Predicting the movement of a toxic substance in the environment involves the use of models that describe the partitioning of chemical species among the various environmental media, including air, water, soils, and sediments. One approach to this problem is the use of compartment models. Models of this type have been developed over the last decade for studying the global fate of toxic elements and radionuclides and for studying the transport and transformation of organic chemicals. Compartment models are often a reasonable starting point in environmental assessments because they cover all the primary media simultaneously. Multimedia models can be structured to describe complex systems with hundreds of state variables. However, the amount of information available for most chemicals restricts practical models to the consideration of one or more atmosphere compartments, two or three soil zones, and land biota, ground water, and sediment compartments used by the GEOTOX model.

The transport and transformation equations solved by GEOTOX have the general form:

$$\frac{dN(i,n,t)}{dX_i} = -L(i,n)N(i,n,t) - SUM_{j/i}^{j=1,m}\,[T(i \Longrightarrow j,n)N(i,n,t)] - T(i \Longrightarrow o,n)N(i,n,t) + K(i,n-1)N(i,n-1,t) + SUM_{j/i}^{j=1,m}\,[T(j \Longrightarrow i,n)N(j,n,t)] + S(i,n,t) \qquad (2)$$

where:

N(i,n,t) = time-varying inventory of species n in compartment i, moles;

L(i,n) = first-order rate constant for removal of species n from compartment i by chemical decompositon, etc., 1/d;

T(i==>j,n,t) = rate constant for the transfer of species n from compartment i to compartment j, 1/d;

K(i,n-1) = first-order rate constant for the transformation of species n-1 to species n within compartment i, 1/d;

T(i==>o,n) = rate constant for the transfer of species n from compartment i to a compartment outside of the landscape system, 1/d;

S(i,n,t) = source term for the introduction of species n into compartment i, mole/d; and

m = total number of compartments within the landscape system.

GEOTOX-A, the first module of GEOTOX, obtains the steady-state solution to Equation 2. When Equation 2 is written for m compartments and k chemicals, the following matrix equation is obtained:

$$\frac{d}{dt}\underline{n}(t) = [T]\underline{n}(t) + \underline{s}(t) \tag{3}$$

in which $\underline{n}$ and $\underline{s}$ are vectors of length M=mxk and [T] is a matrix of size M by M. A steady-state solution for this system of equations is obtained by defining a constant source vector

Results of the GEOTOX-A and GEOTOX-B modules are used in another module, termed EXPOSE, which estimates exposure levels for humans from a series of pathways. Exposure is expressed as daily intake per unit body weight averaged over the population. The general model used to calculate exposure has the form:

$$E = (a/BW)\ \mathrm{SUM}^{j=1,p}\ I_i(t)$$

where:

E = lifetime exposure, mg/kg-d;

BW = body weight, kg;

p = number of pathways; and

I_i = the daily intake by pathway i, mg/d based on the environmental concentrations estimated using GEOTOX-A or -B.

5.2 Validation

Model performance predictions have not been compared with actual disposal site performance.

5.3 Treatment of Decay Products

GEOTOX simulates all decay and transformation processes (radioactive decay, hydrolysis, photolysis, oxidation, biodegradation, sedimentation, and advective losses) as first-order removal processes. Each process is governed by its own decay constant, and is treated as irreversible. Any fraction of the contaminant which is transformed in a given compartment can be treated as a source for another contaminant in the simulation. This allows the treatment of radioactive decay chains or chemical transformation sequences.

5.4 Underlying Assumptions

GEOTOX lumps each component of the environment into a homogeneous subsystem or compartment that can exchange water, nutrients, and chemical contaminants with other adjacent compartments. This compartment system is used to simulate the behavior of a single substance or of two or more substances linked by transformation, such as a radionuclide and its progeny. A compartment is described by its total mass, total volume, solid-phase mass, liquid-phase mass, and gas-phase mass. Mass flows among compartments include solid-phase flows, such as dust suspension or deposition, and liquid-phase flows, such as surface run-off and groundwater recharge. The transport of individual chemical species among compartment occurs by diffusion and advection at the compartment boundaries. Each chemical species is assumed to be in chemical equilibrium among the phases within a single compartment. However, there is no requirement for equilibrium between adjacent compartments.

For example, in the upper soil layer; which contains solids, liquids, and gases; an organic chemical added to the soil distributes itself among these three phases such that it achieves chemical and physical equilibrium. Among the potential transport pathways from the upper-soil compartment are liquid advection (soil water run-off), solid-phase advection (erosion to surface water or dust stirred up and blown about), and diffusion from the soil gas phase into the lower atmosphere.

5.5 Exposure Pathways

In GEOTOX, the exposure function E is divided into a series of terms that relate sources to environmental concentrations and the concentrations to human contact:

$$E = \underset{i\,j}{\mathrm{SUM}}\ C_i(S)F6i_j \qquad \{i=\text{air,soil,water}\} \quad \{j=\text{inhalation,ingestion,dermal}\}$$

where $C_i(S)$ is the concentration of contaminant in environmental compartment i associated with the source S and F_{ij} is the pathway exposure factor (PEF) that relates this concentration to a level of human contact through pathway j. The PEF incorporates information on anatomy, physiology, diet, activity patterns, and environmental partitioning into a term that translates a unit concentration into a daily exposure in mg/kg-d via inhalation, ingestion, or dermal absorption.

5.6 Dose Conversion and Dose-Response

The GEOTOX model calculates lifetime equivalent daily exposure but does not explicitly calculate the risk associated with this exposure. It does provide the option of comparing exposures on the basis of a reference safe dose (or exposure).

6 REFERENCES

Layton, D.W., T.E. McKone, C.H. Hall, M.A. Nelson, and Y.E. Ricker (1986) Demilitarization of Conventional Ordnance: Priorities for Data Base Assessments of Environmental Contaminants, Lawrence Livermore National Laboratory, UCRL-15902.

Layton, D.W. and T.E. McKone (1988) "Multimedia Transport of Organic Contaminants and Exposure Modelling," Toxicology Letters (in press).

McKone, T.E. (1981) Chemical Cycles and Health Risks of Some Crustal Nuclides, Ph.D. dissertation, University of California, Los Angeles.

McKone, T.E. (1986) "Dioxin Risk Management at Times Beach, Missouri: An Evaluation," The Environmental Professional 8:13-24.

Mckone, T. E. (1988) GEOTOX Multimedia Compartment Model Users' Guide Supplement, Lawrence Livermore National Laboratory, UCID-21673.

McKone, T.E., and D.W. Layton (1986a) "Screening the Potential Risks of Toxic Substances Using a Multimedia Compartment Model," Reg. Tox. Pharm. 6:359-380.

McKone, T.E., and D.W. Layton (1986b) "Chemical Transport, Human exposure, and Health Risk: A Multimedia Approach," in J.S.Evans (ed), Environmental Risk Management, Is Analysis Useful?, Air Pollution Control Association, Pittsburgh.

McKone, T.E., W.E. Kastenberg, and D. Okrent (1983) "The Use of Landscape Chemical Cycles for Indexing the Health Risks of Toxic Elements and Radionuclides," Risk Analysis 3:189-205.

McKone, T.E., and W.E. Kastenberg (1986) "Application of Multimedia Transport Models to Risk Analysis," in Y.Cohen (ed), Pollutants in a Multimedia Environment, Plenum Press, New York.

stern Ecoregion

ANDSCAPE MASS DISTRIBUTION AND EXCHANGE

COMPARTMENT	GAS MASS +	LIQUID MASS +	SOLID MASS =	TOTAL MASS	VOLUME
	(kg)	(kg)	(kg)	(kg)	(liters)
AIR	1.200E+13	5.000E+07	0.000E+00	1.200E+13	1.000E+16
PMAIR	0.000E+00	0.000E+00	6.150E+05	6.150E+05	1.000E+16
BIOTA	5.120E+02	1.421E+10	7.000E+09	2.121E+10	2.121E+10
SOIL1	1.343E+08	6.080E+11	2.400E+12	3.008E+12	1.600E+12
SOIL2	6.000E+09	3.000E+13	1.700E+14	2.000E+14	1.000E+14
GWTR	0.000E+00	1.100E+14	5.343E+14	6.443E+14	3.143E+14
SWTR	1.434E+04	5.052E+11	4.496E+09	5.097E+11	5.052E+11
SDMT	0.000E+00	8.420E+08	6.315E+09	7.157E+09	4.210E+09

Mass Exchange of Solids Among the Compartments in kg/d :

Matrix of flows from top compartments to the left-hand compartments:

	AIR	PMAIR	BIOTA	SOIL1	SOIL2	GWTR	SWTR	SDMT	LSTC	TOTL INFLW
AIR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
PMAIR	0.000E+00	0.000E+00	0.000E+00	3.290E+05	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	3.290E+05
BIOTA	0.000E+00	0.000E+00	0.000E+00	2.466E+06	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	2.466E+06
SOIL1	0.000E+00	2.054E+05	2.466E+06	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	3.260E+07	3.527E+07
SOIL2	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
GWTR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
SWTR	0.000E+00	0.000E+00	0.000E+00	3.260E+07	0.000E+00	0.000E+00	0.000E+00	8.835E+08	0.000E+00	9.161E+08
SDMT	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	8.835E+08	0.000E+00	0.000E+00	8.835E+08
NXTC	0.000E+00	1.236E+05	0.000E+00	0.000E+00	0.000E+00	0.000E+00	3.260E+07	0.000E+00		
TOTL OTFLW	0.000E+00	3.290E+05	2.466E+06	3.540E+07	0.000E+00	0.000E+00	9.161E+08	8.835E+08		

Mass Exchange of Liquid Among the Compartments in kg/d :

Matrix of flows from top compartments to the left-hand compartments:

	AIR	PMAIR	BIOTA	SOIL1	SOIL2	GWTR	SWTR	SDMT	LSTC	TOTL INFLW
AIR	0.000E+00	0.000E+00	0.000E+00	5.151E+09	0.000E+00	0.000E+00	4.384E+07	0.000E+00	3.452E+09	8.647E+09
PMAIR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
BIOTA	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
SOIL1	8.767E+09	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.863E+08	0.000E+00	0.000E+00	0.000E+00	8.953E+09
SOIL2	0.000E+00	0.000E+00	0.000E+00	7.890E+08	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	7.890E+08
GWTR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	7.890E+08	0.000E+00	0.000E+00	0.000E+00	0.000E+00	7.890E+08
SWTR	8.219E+07	0.000E+00	0.000E+00	3.014E+09	0.000E+00	6.027E+08	0.000E+00	0.000E+00	0.000E+00	3.699E+09
SDMT	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
NXTC	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	3.452E+09	0.000E+00		
TOTL OTFLW	8.849E+09	0.000E+00	0.000E+00	8.953E+09	7.890E+08	7.890E+08	3.496E+09	0.000E+00		

Figure B-7. Sample output from GEOTOX A.

CHEMICAL PROPERTIES FOR arsenic

MOLECULAR WEIGHT... 7.490E+01 HENRY'S LAW CONSTANT... 1.000E-06 TORR/(MOLE/L)

AIR DIFFUSION CONSTANT... 3.171E-17 M**2/S WATER DIFFUSION CONSTANT... 3.171E-17 M**2/S

PARTITION COEFFICIENTS:

AIR/WATER.....	5.663E-11 KG/L	SOIL1/WATER...	1.300E+03
SOIL2/WATER...	1.300E+03	ROCK/GRDWTR...	1.300E+03
SEDMT/SFWTR...	1.300E+03	BIOTA/SOIL1...	1.000E-01
MEAT/DIET.....	4.000E-02	MILK/DIET.....	4.000E-02
FISH/WATER....	7.500E+01		

CALCULATION OF RATE CONSTANTS FOR arsenic

	ALPHA (1/day) +	FLOW (kg/day) x	BETAL (1/kg) +	SFLOW (kg/day) x	BETAS (1/kg) =	T(i=>j) (1/day)
AIR => SOIL1	3.042E-15	8.767E+09	1.978E-08	0.000E+00	0.000E+00	1.734E+02
AIR => SWTR	2.583E-17	8.219E+07	1.978E-08	0.000E+00	0.000E+00	1.625E+00
AIR => NXTC	2.010E-01	0.000E+00	1.978E-08	0.000E+00	0.000E+00	2.010E-01
PMAIR=> SOIL1	0.000E+00	0.000E+00	0.000E+00	2.054E+05	1.626E-06	3.340E-01
PMAIR=> NXTC	0.000E+00	0.000E+00	0.000E+00	1.236E+05	1.626E-06	2.010E-01
BIOTA=> SOIL1	0.000E+00	0.000E+00	0.000E+00	2.466E+06	1.429E-10	3.523E-04
SOIL1=> AIR	4.930E-23	5.151E+09	0.000E+00	0.000E+00	4.166E-13	4.930E-23
SOIL1=> PMAIR	0.000E+00	0.000E+00	3.205E-16	3.290E+05	4.166E-13	1.371E-07
SOIL1=> BIOTA	0.000E+00	0.000E+00	3.205E-16	2.466E+06	3.324E-14	8.197E-08
SOIL1=> SOIL2	0.000E+00	7.890E+08	3.205E-16	0.000E+00	4.166E-13	2.528E-07
SOIL1=> SWTR	0.000E+00	3.014E+09	3.205E-16	3.260E+07	4.166E-13	1.455E-05
SOIL2=> GWTR	0.000E+00	7.890E+08	4.524E-18	0.000E+00	5.882E-15	3.570E-09
GWTR => SOIL1	0.000E+00	1.863E+08	1.440E-18	0.000E+00	1.871E-15	2.682E-10
GWTR => SWTR	0.000E+00	6.027E+08	1.440E-18	0.000E+00	1.871E-15	8.676E-10
SWTR => AIR	2.057E-22	4.384E+07	0.000E+00	0.000E+00	2.047E-10	2.057E-22
SWTR => SDMT	1.816E-12	0.000E+00	1.575E-13	8.835E+08	2.047E-10	1.809E-01
SWTR => NXTC	0.000E+00	3.452E+09	1.575E-13	3.260E+07	2.047E-10	7.2[illegible]9E-03
SDMT => SWTR	1.405E-12	0.000E+00	1.218E-13	8.835E+08	1.583E-10	1.399E-01

COMPARTMENT	SOURCES (mol/day)			LOSSES (1/day)			
	direct	transform	last cell	transform	reaction +	transfer =	total
AIR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.752E+02	1.752E+02
PMAIR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	5.350E-01	5.350E-01
BIOTA	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	3.523E-04	3.523E-04
SOIL1	2.740E+01	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.502E-05	1.502E-05
SOIL2	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	3.570E-09	3.570E-09
GWTR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.136E-09	1.136E-09
SWTR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.881E-01	1.881E-01
SDMT	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.100E-04	1.399E-01	1.400E-01

ENVIRONMENTAL DISTRIBUTION FOR arsenic IN LANDSCAPE CELL 1

COMPARTMENT	SOURCES (mol/d)	INVENTORY (moles)	CONCENTRTN (mg/kg) totl	CONCENTRTN (mg/L) vol.	CONCENTRTN (mg/L) wtr	FRACTIONAL INVENTORY	FRACTION REMOVAL	LOSSES (mol/d
AIR	0.000E+00	5.255E-19	3.280E-27	3.936E-30	7.784E-22	9.636E-28	3.855E-21	1.056E-
PMAIR	0.000E+00	4.745E-01	5.779E-02	3.554E-12	0.000E+00	8.701E-10	3.481E-03	9.538E-
BIOTA	0.000E+00	4.310E+02	1.522E-03	1.522E-03	0.000E+00	7.903E-07	0.000E+00	0.000E+
SOIL1	2.740E+01	1.852E+06	4.612E-02	8.670E-02	4.445E-05	3.396E-03	0.000E+00	0.000E+
SOIL2	0.000E+00	1.312E+08	4.913E-02	9.826E-02	4.445E-05	2.406E-01	0.000E+00	0.000E+
GWTR	0.000E+00	4.123E+08	4.793E-02	9.826E-02	4.445E-05	7.560E-01	0.000E+00	0.000E+
SWTR	0.000E+00	3.710E+03	5.451E-04	5.500E-04	4.375E-05	6.802E-06	9.773E-01	2.677E+
SDMT	0.000E+00	4.792E+03	5.015E-02	8.526E-02	4.372E-05	8.787E-06	1.924E-02	5.271E-
TOTALS	2.740E+01	5.453E+08				1.000E+00	1.000E+00	2.740E+

Figure B-7. (continued).

CHEMICAL PROPERTIES FOR benzene

MOLECULAR WEIGHT .. 7.812E+01 HENRY'S LAW CONSTANT.. 4.100E+03 TORR/(MOLE/L)

AIR DIFFUSION CONSTANT... 4.980E-06 M**2/S WATER DIFFUSION CONSTANT... 4.980E-10 M**2/S

PARTITION COEFFICIENTS:

AIR/WATER	2.322E-01 KG/L	SOIL1/WATER...	5.760E-01
SOIL2/WATER...	9.600E-02	ROCK/GRDWTR...	1.378E-02
SEDMT/SFWTR...	9.600E-01	BIOTA/SOIL1...	8.681E+00
MEAT-FAT/DIET.	4.000E-02	MILK-FAT/DIET.	4.000E-02
FISH/WATER....	7.500E+01		

CALCULATION OF RATE CONSTANTS FOR benzene

	ALPHA (1/day) +	FLOW (kg/day) x	BETAL (1/kg) +	SFLOW (kg/day) x	BETAS (1/kg) =	T(i=>j) (1/day)
AIR => SOIL1	4.266E-02	8.767E+09	4.307E-16	0.000E+00	0.000E+00	4.267E-02
AIR => SWTR	7.800E-08	8.219E+07	4.307E-16	0.000E+00	0.000E+00	1.134E-07
AIR => NXTC	2.010E-01	0.000E+00	4.307E-16	0.000E+00	0.000E+00	2.010E-01
PMAIR=> SOIL1	0.000E+00	0.000E+00	0.000E+00	2.054E+05	1.626E-06	3.340E-01
PMAIR=> NXTC	0.000E+00	0.000E+00	0.000E+00	1.236E+05	1.626E-06	2.010E-01
BIOTA=> SOIL1	0.000E+00	0.000E+00	0.000E+00	2.466E+06	1.429E-10	3.523E-04
SOIL1=> AIR	4.913E+01	5.151E+09	0.000E+00	0.000E+00	2.857E-13	4.913E+01
SOIL1=> PMAIR	0.000E+00	0.000E+00	4.959E-13	3.290E+05	2.857E-13	9.399E-08
SOIL1=> BIOTA	0.000E+00	0.000E+00	4.959E-13	2.466E+06	2.886E-12	7.115E-06
SOIL1=> SOIL2	0.000E+00	7.890E+08	4.959E-13	0.000E+00	2.857E-13	3.913E-04
SOIL1=> SWTR	0.000E+00	3.014E+09	4.959E-13	3.260E+07	2.857E-13	1.504E-03
SOIL2=> GWTR	0.000E+00	7.890E+08	1.438E-14	0.000E+00	1.381E-15	1.135E-05
GWTR => SOIL1	0.000E+00	1.863E+08	8.521E-15	0.000E+00	1.174E-16	1.587E-06
GWTR => SWTR	0.000E+00	6.027E+08	8.521E-15	0.000E+00	1.174E-16	5.136E-06
SWTR => AIR	3.554E-04	4.384E+07	0.000E+00	0.000E+00	1.884E-12	3.554E-04
SWTR => SDMT	3.555E-04	0.000E+00	1.963E-12	8.835E+08	1.884E-12	2.020E-03
SWTR => NXTC	0.000E+00	3.452E+09	1.963E-12	3.260E+07	1.884E-12	6.837E-03
SDMT => SWTR	2.624E-02	0.000E+00	1.448E-10	8.835E+08	1.390E-10	1.491E-01

COMPARTMENT	SOURCES (mol/day)			LOSSES (1/day)			
	direct	transform	last cell	transform	reaction +	transfer =	total
AIR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	2.080E-02	2.437E-01	2.645E-01
PMAIR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	2.080E-02	5.350E-01	5.558E-01
BIOTA	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	3.523E-04	3.523E-04
SOIL1	2.740E+01	0.000E+00	0.000E+00	0.000E+00	0.000E+00	4.913E+01	4.913E+01
SOIL2	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.135E-05	1.135E-05
GWTR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	6.723E-06	6.723E-06
SWTR	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	9.212E-03	9.212E-03
SDMT	0.000E+00	0.000E+00	0.000E+00	0.000E+00	1.100E-04	1.491E-01	1.492E-01

ENVIRONMENTAL DISTRIBUTION FOR benzene IN LANDSCAPE CELL 1

COMPARTMENT	SOURCES (mol/d)	INVENTORY (moles)	CONCENTRTN (mg/kg) totl	CONCENTRTN (mg/L) vol.	CONCENTRTN (mg/L) wtr	FRACTIONAL INVENTORY	FRACTION REMOVAL	LOSSES (mol/d)
AIR	0.000E+00	1.235E+02	8.041E-07	9.649E-10	4.156E-09	6.641E-01	1.000E+00	2.740E+01
PMAIR	0.000E+00	1.124E-07	1.428E-08	8.784E-19	0.000E+00	6.046E-10	9.103E-10	2.494E-08
BIOTA	0.000E+00	1.343E-02	4.947E-08	4.947E-08	0.000E+00	7.222E-05	0.000E+00	0.000E+00
SOIL1	2.740E+01	6.650E-01	1.727E-08	3.247E-08	2.576E-08	3.575E-03	0.000E+00	0.000E+00
SOIL2	0.000E+00	2.293E+01	8.957E-09	1.791E-08	2.576E-08	1.233E-01	0.000E+00	0.000E+00
GWTR	0.000E+00	3.870E+01	4.693E-09	9.620E-09	2.576E-08	2.081E-01	0.000E+00	0.000E+00
SWTR	0.000E+00	1.686E-01	2.584E-08	2.607E-08	2.585E-08	9.064E-04	4.207E-05	1.153E-03
SDMT	0.000E+00	2.283E-03	2.492E-08	4.236E-08	2.583E-08	1.227E-05	9.166E-09	2.511E-07
TOTALS	2.740E+01	1.860E+02				1.000E+00	1.000E+00	2.740E+01

Figure B-7. (continued).

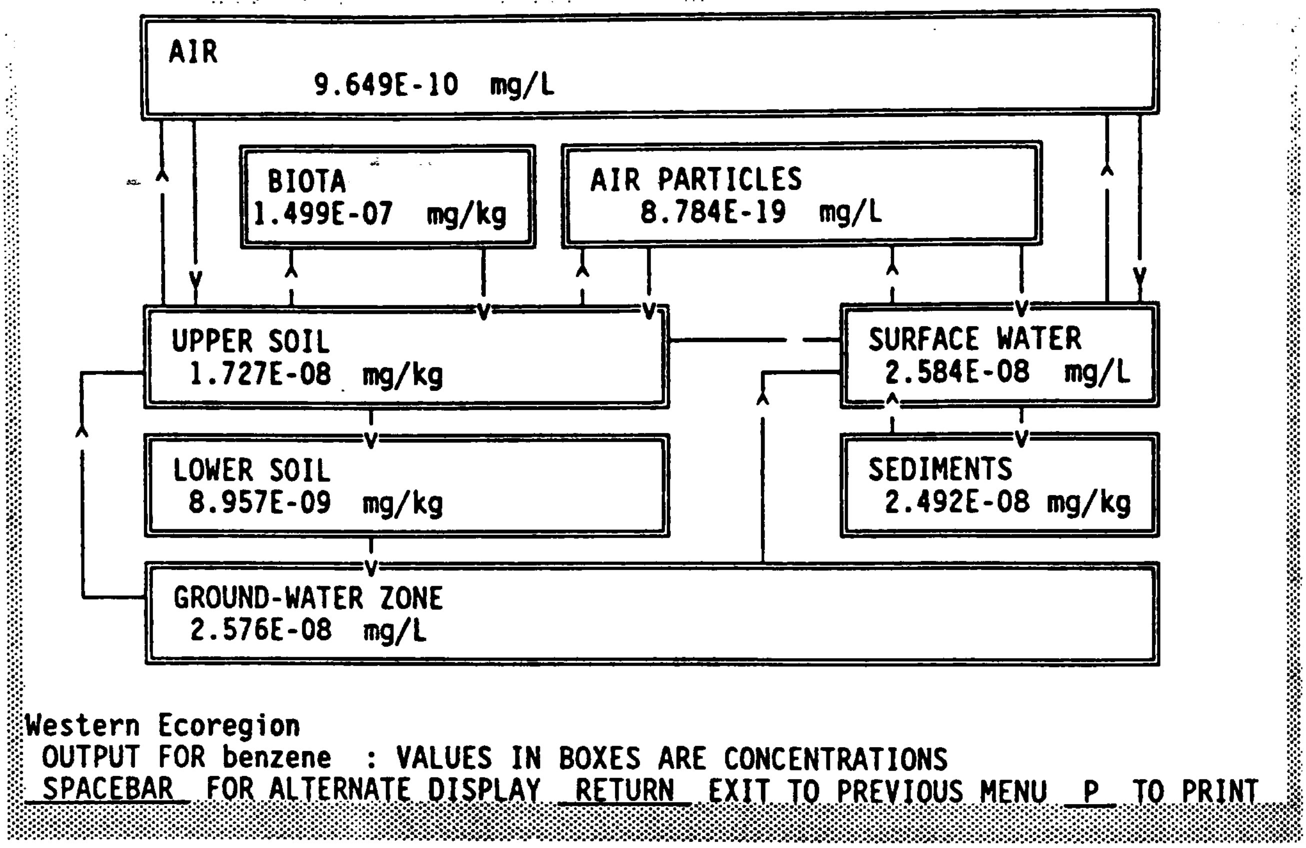


Figure B-8. Chemical distribution of benzene as displayed by the graphic-output routine for GEOTOX A.

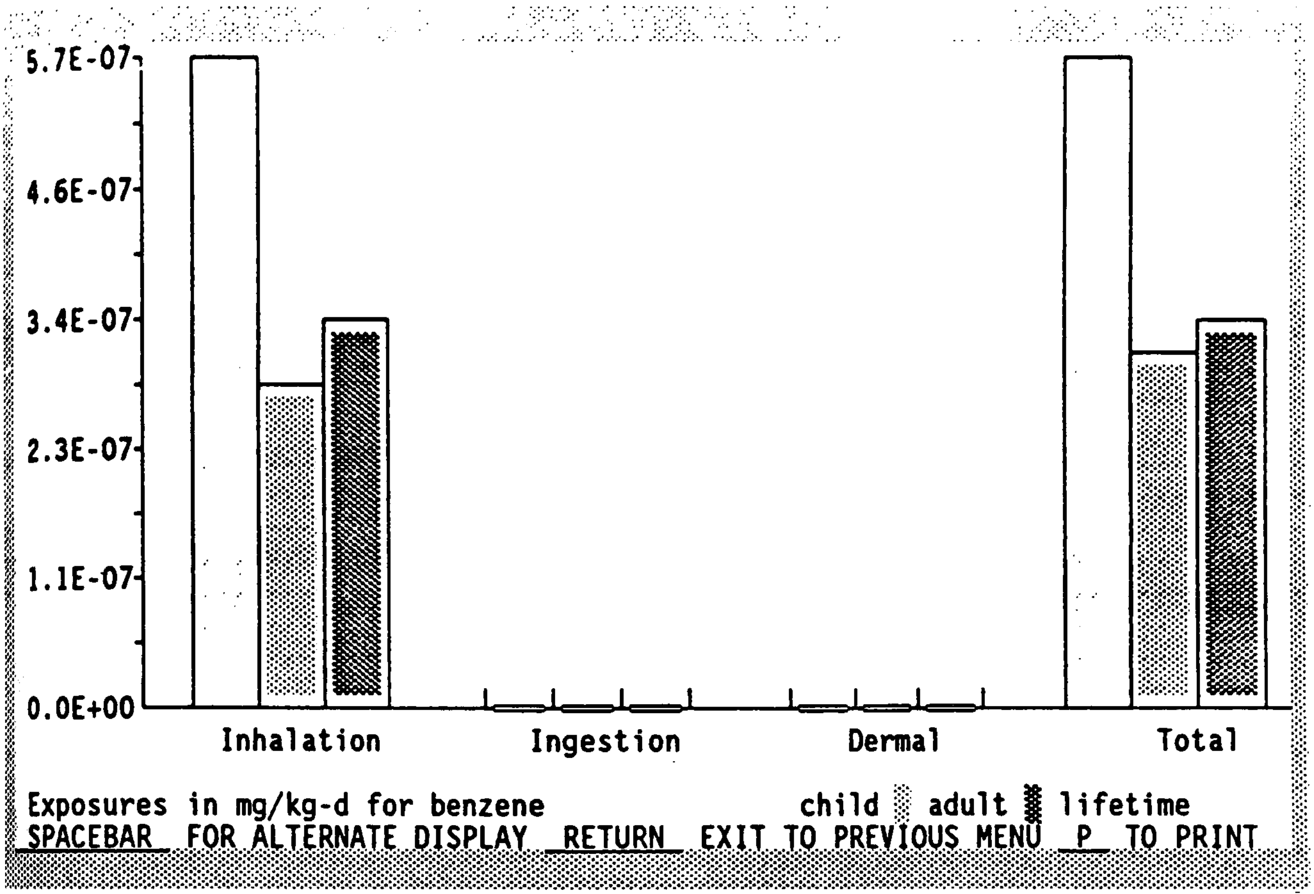


Figure B-9. Total exposure to benzene for the sample problem as displayed on the PC screen by the program EXPOSE.

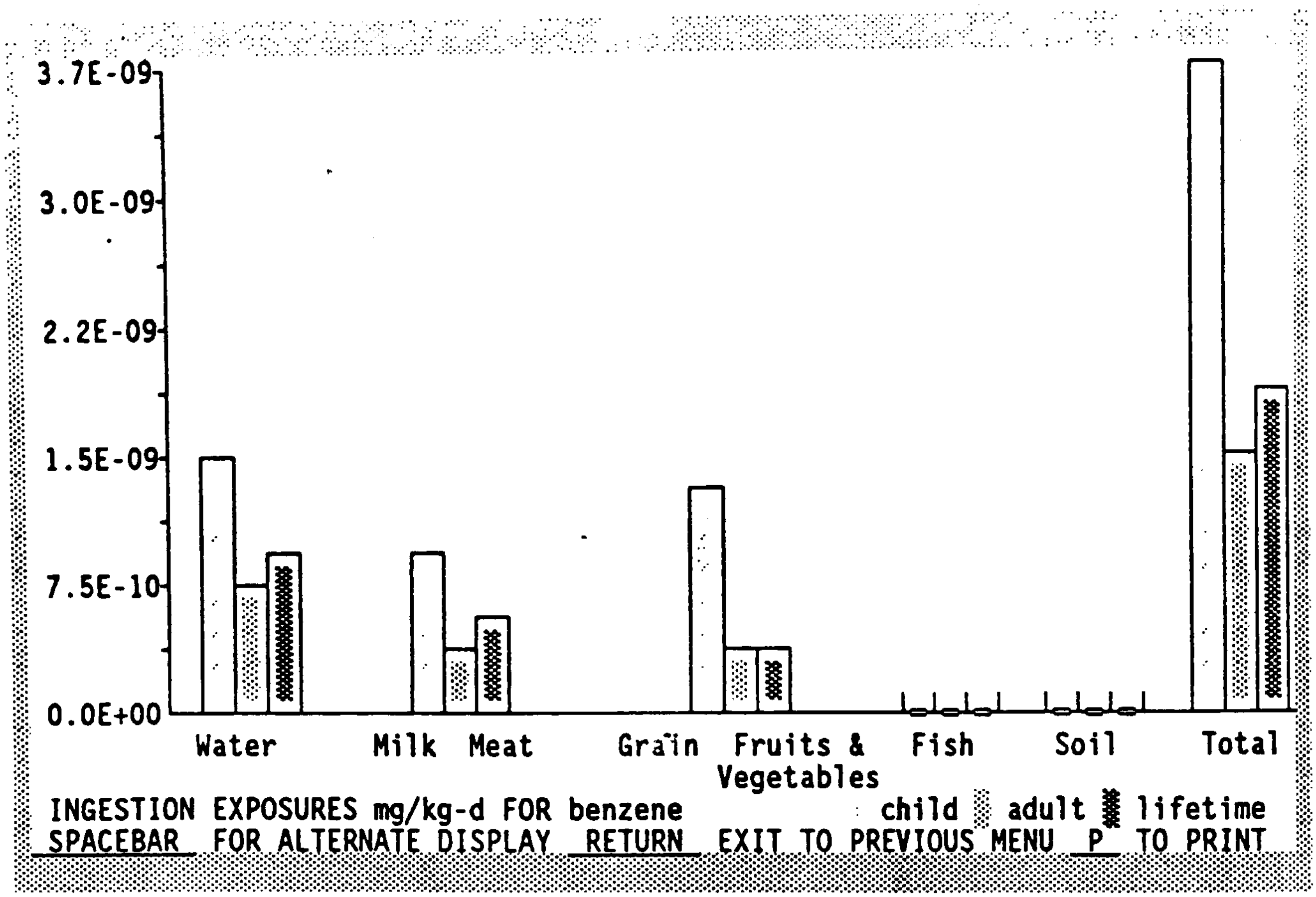


Figure B-10. Ingestion exposure to benzene from the sample problem as displayed on the PC screen by the program EXPOSE.

1.000E+00: arsenic

1.000E+00: benzene

For each compound, enter the reference safe dose (RSD) (in mg/kg-d) upon which the relative ranking will be based. For example, for a carcinogen, the reference safe dose corresponding to a lifetime risk of 1 in a million is RSD = 1/(10^6 x q) where q is the potency in kg-d/mg.

When all the boxes have nonzero entries, push PgDn to see the figure comparing their relative ranks.

To compare relative exposure enter a '1' in each box.

USE ARROWS TO PICK CONTAMINANT PgDn TO SHOW RANKS PgUp FOR MAIN MENU

Figure B-11. Display used to compare the contaminants based on exposure.

6.700E-08:arsenic

1.000E-03:benzene

For each compound, enter the reference safe dose (RSD) (in mg/kg-d) upon which the relative ranking will be based. For example, for a carcinogen the reference safe dose corresponding to a lifetime risk of 1 in a million is RSD = 1/(10^6 x q) where q is the potency in kg-d/mg.

When all the boxes have nonzero entries, push PgDn to see the figure comparing their relative ranks.

To compare relative exposure enter a '1' in each box.

USE ARROWS TO PICK CONTAMINANT PgDn TO SHOW RANKS PgUp FOR MAIN MENU

Figure B-12. Display used to compare the contaminants based on potential risk.

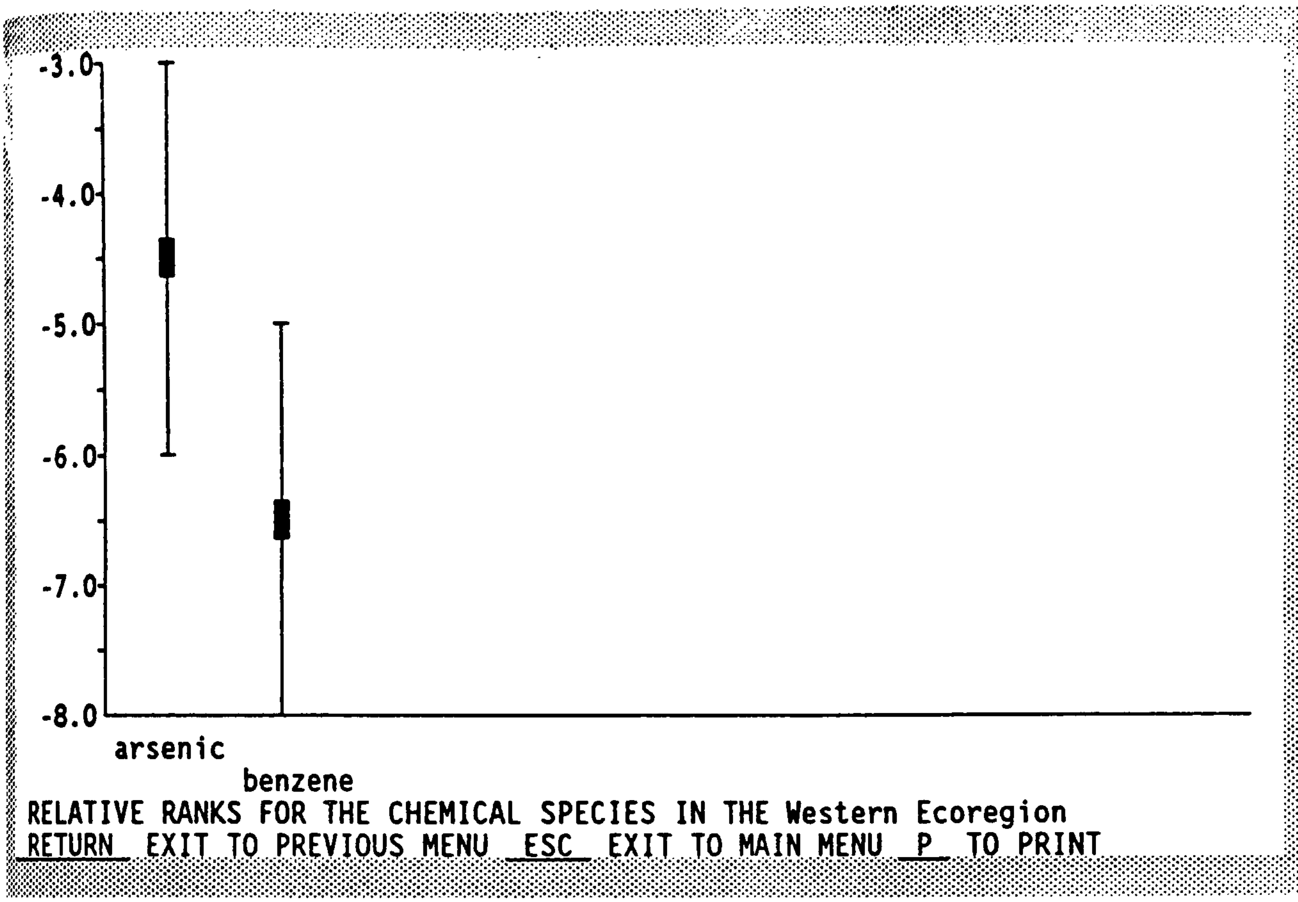


Figure B-13. Display comparing exposure between the two contaminants.

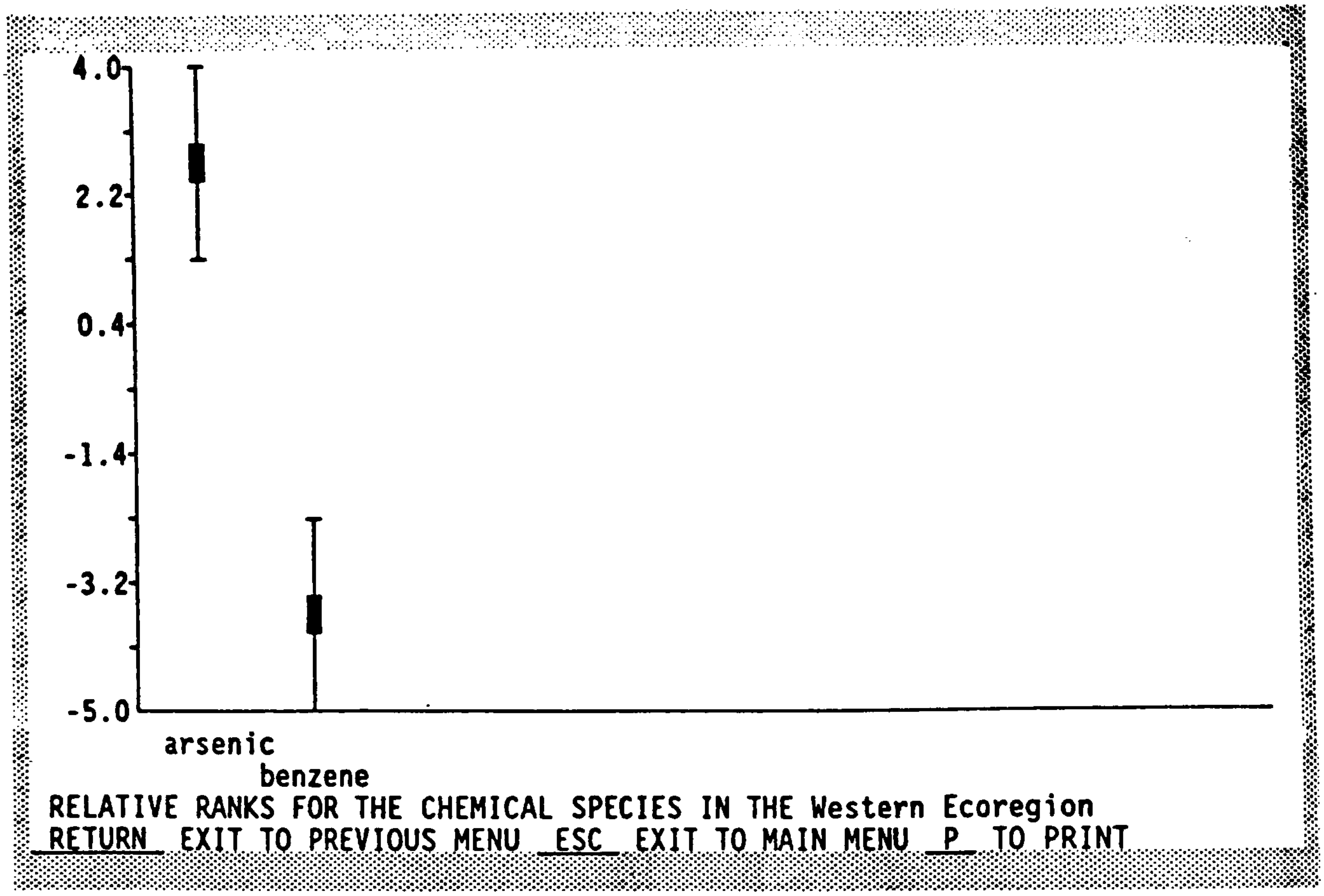


Figure B-14. Display comparing potential risks between the two contaminants.

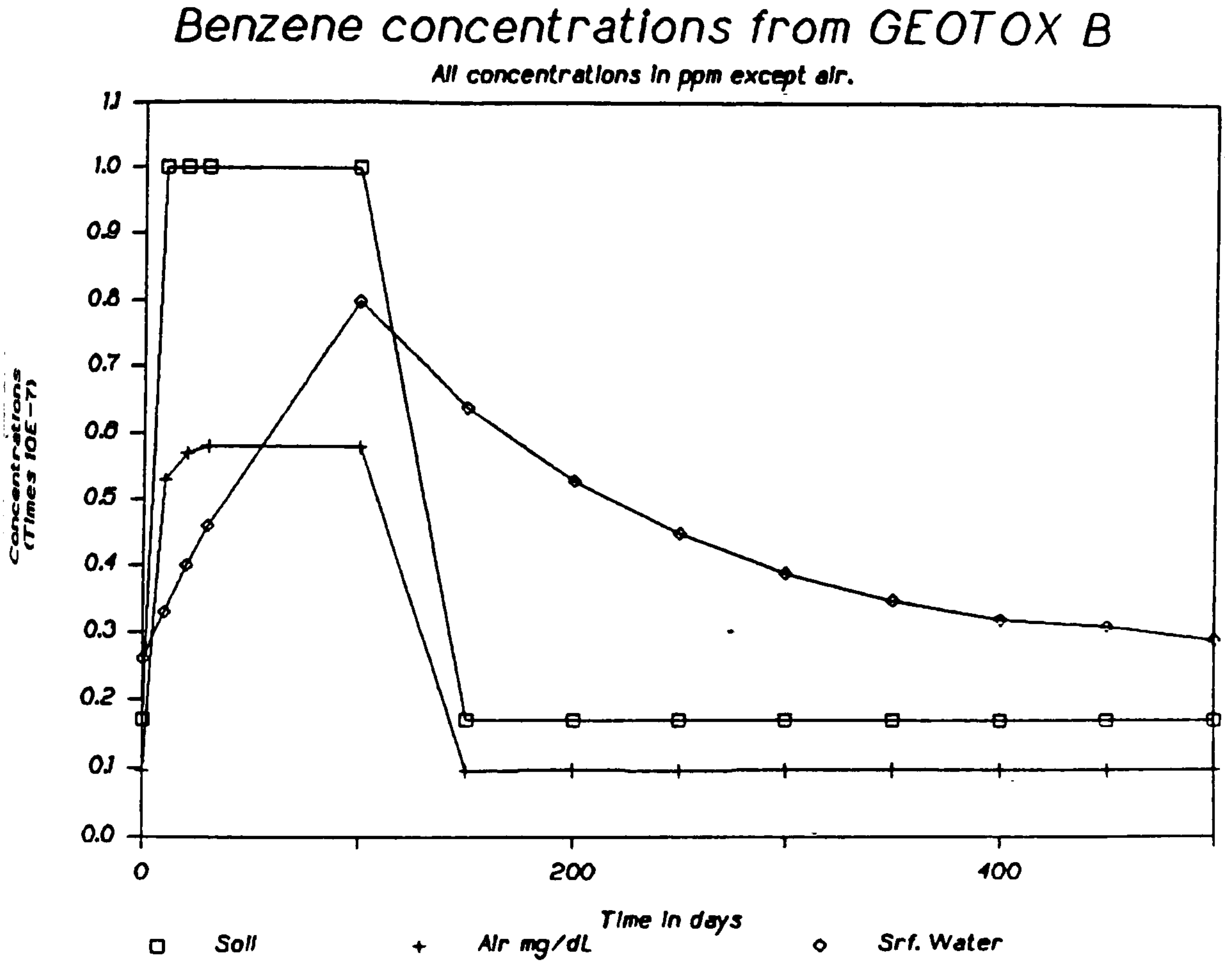


Figure B-15. Output from GEOTOX plotted using Lotus 123(R).

APPENDIX C

MODEL REVIEWS - OCEAN

IDENTIFICATION

1.1 *Name:*	Bryan-Semtner-Cox (BSC) Model
1.2 *Prepared by:*	Geophysical Fluid Dynamics Laboratory, Princeton
1.3 *Prepared for:*	NOAA
1.4 *Report Title:*	A Primitive Equation, 3-Dimensional Model of the Ocean
1.5 *Report Number:*	GFDL Ocean Group Tech. Rep. No. 1
1.6 *Report Date:*	1984
1.7 *Availability:*	Available from: GFDL, Princeton, NJ
1.8 *Purpose and Scope:*	A 3-dimensional numerical ocean model designed for studying the most basic aspects of large-scale, baroclinic ocean circulation. Highly dissipative and slow running, it was intended to simulate the upper ocean circulation, not the deep flow. Does not accurately handle the vertical circulation or the heat-driven flow.

2 SUMMARY OF FINDINGS

The Bryan-Semtner-Cox (BSC) model (M.D. Cox, "A Primitive Equation 3-Dimensional Model of the Ocean," GFDL Ocean Group Technical Report No. 1, 1984) is acclaimed as "the principal tool for modeling ocean circulation in irregular domains having realistic coastlines and bottom topography" (A.J. Semtner and R.M. Chervin, "A Simulation of the Global Ocean Circulation with Resolved Eddies," J. Geophys. Res., 93, 15502-15522, 1988). However, the BSC has been used by the English (MAFF), the French (CEA), and the German (DHI) scientists in connnection with the SWG-POTG project, and all but the Germans dropped it in favor of other models, and the others questioned the validity of the German results.

Sandia has used the BSC extensively, including a 40-year simulation to spin up the North Atlantic circulation. Sandia noted that the BSC requires excessive dissipation parameters, beyond the levels reasonable for ocean flows, and that this has negative effects on the results. With 200 km horizontal resolution, the BSC was found to be unstable when eddy diffusion coefficients less than 109 cm2/sec were used. SOMS is stable with 106 cm2/sec eddy diffusion coefficients and 200 km horizontal resolution. We have also noted that SOMS runs more than ten times faster (per model year) using the same resolution.

Recently, the BSC was applied in a 20-year integration of the world ocean, using 1/2 degree resolution and 20 layers. This required 500 hours of dedicated time on the Cray-XMP/48 computer, with all vector processors operating nearly all the time (99.64%). SOMS would require only 150 hours cpu time on a much slower Cray 1 computer to do a similar calculation, using a 90 minute time step. This translates to a factor of ten savings based on the relative speeds of the two machines.

The BSC does not address boundary layer phenomena, which are important in environmental questions, while SOMS includes a fully three-dimensional submodel specifically for addressing the bottom boundary layer. It is fully coupled to the "free-stream" submodel, which models the overlying ocean.

Swiss scientists (Nyffeler and Zuur) are presently using SOMS to model the circulation of Lake Neuchatel (Zuur and Dietrich, 1989), and high resolution open-ocean simulations in the region of the NOAMP 5 cruise. SOMS is also being used at Florida State Univesity by Weatherly, in studying flow around a seamount and by Ezer, in his Ph.D. study of continental shelf phenomena. This is noteworthy, as SOMS was first made available to others only one year ago.

In summary, SOMS is far more capable than the BSC in addressing the important waste disposal equations, due to the BSC's large numerical dissipation, high computing cost, inaccurate vertical velocity fields, and lack of a good bottom boundary layer turbulence closure model.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

The BSC is thoroughly documented in the user's manual. A large number of papers have been published in the literature using this model.

3.2 Hardware Requirements

The BSC is best run on main frame computers.

3.3 Application

The BSC is applicable to the whole range of hydrostatic, incompressible geophysical flows, except boundary layer phenomena.

3.4 Level of Expertise Required

A working knowledge of Fortran, understanding of basic numerical modeling concepts, and understanding of modeling requirements of geophysical flows are all highly recommended.

4 TECHNICAL CRITERIA

4.1 Peer Review

The BSC is a well-established model that has been carefully reviewed and analyzed by several prominent ocean modelers.

4.2 *Verification*

The BSC has not been shown to converge (at least in the literature) as resolution increases. Such demonstration is a necessary part of verification, and has been done for SOMS. SOMS converges for a realistic geophysical prototype problem with baroclinic instability in a stratified environment (Dietrich, Roache, and Marietta, submitted for publication). Even vertical velocity converges to reasonable values, which the NEA POTG doubted would happen with the BSC using practicable resolution, in view of the unrealistic results obtained with 200 km resolution. Vertical overturnings are very important in issues involving ocean circulations.

4.3 *Uncertainty*

All models of time-dependent flow are uncertain to some degree. Accuracy depends on many things, initial and boundary conditions, model resolution, model assumptions, duration of the calculation, and the nature of the feature to be determined. However, due to its large required numerical dissipation, the BSC does not achieve much higher certainty or accuracy except by using very high resolution and associated high computing cost, especially compared to SOMS, which runs much faster for a given resolution and requires much less numerical dissipation. Further, SOMS' special treatment of the dominant Coriolis and vertical diffusion terms is much more accurate than the treatment used by the BSC.

4.4 *Sensitivity*

The sensitivity of the BSC to important model resolution, dissipation, and thermodynamic driving parameters has not been addressed in detail in the literature, in spite of its maturity.

4.5 *Required Input*

The required input is the usual geophysical modeling inputs.

4.6 *Outputs*

Contour plots and a number of numerical diagnostics are included in the output.

4.7 *Source Term*

NA.

4.8 *Scenarios*

NA.

4.9 Relationship to Regulatory Standards

NA.

5 SCIENTIFIC CRITERIA

5.1 Theory

The theory behind the BSC's basic modeling approach is a control-volume concept based on finite-volume conservation laws.

5.2 Validation

Validation of the BSC has been slow, in spite of its maturity, due to its large required numerical dissipation, which is not characteristic of geophysical flows. Recently, a partial validation (at a cost of hundreds of thousands of dollars in computer time) was made by applying it to the world ocean (A.J. Semtner and R.M. Chervin, "A Simulation of the Global Ocean Circulation with Resolved Eddies," J. Geophys. Res., 93, 15502-15522, 1988). Several numerical artifices were applied, including biharmonic diffusion and direct forcing of all but a relatively thin layer around the thermocline depth toward climatological values, and the dissipation was relaxed toward more realistic values (but still too large) only during the last two years of the simulation. Thus, this expensive calculation, although encouraging due to its exhibiting many observed features of the world ocean circulation, is at best a partial validation. In view of the artificial numerical constraints, it would have been disappointing if the results had not exhibited many features of the real ocean.

5.3 Treatment of Radioactive Decay Products

Not addressed by the BSC, but could be added.

5.4 Underlying Assumptions

The BSC uses the hydrostatic and Boussinesq approximations.

5.5 Pathways

NA.

5.6 Dose Converstion and Dose Response

NA.

IDENTIFICATION

1.1 Name:	Sandia Ocean Modeling System (SOMS)
1.2 Prepared by:	Sandia National Laboratories
1.3 Prepared for:	U.S. DOE Subseabed Program, Sandia National Laboratories
1.4 Report Title:	Dietrich, D., Marietta, M.G., and P. Roache (1987). An ocean modeling system with turbulent boundary layers and topography: numerical description.
1.5 Report Number:	Int. J. Numer. Methods in Fluids 7:833-855, 1987.
1.7 Purpose and Scope:	Multi-scale dynamical ocean circulation code. High-resolution eddy-resolving basin scale with realistic topography and attached fully turbulent boundary layers. Options include very high resolution for local open-ocean domains, separate open boundary layer model, and boundary fitted coordinates for basin model. Code was written for simulating deep ocean dispersal for waste disposal. A program is used to calculate interbox transports for driving the Mark-A box model with SOMS simulations.

2 SUMMARY OF FINDINGS

SOMS was initially developed for the U.S. DOE, under the Subseabed Waste Disposal Program, to help evaluate deep ocean circulations. SOMS is designed specifically to address bottom boundary layer flows over realistic topography, but is applicable to the whole range of geophysical flows, from small lakes to large oceans. SOMS has been applied to both high level subseabed (M.G. Marietta and W.F. Simmons, 1988, NEA Final Report) and low level sea dumping (Nyffeler and W.F. Simmons, NEA Report, 1989). SOMS has also been applied to the circulation of Lake Neuchatel (E. Zuur and D.E. Dietrich, 1989) and the North Atlantic Ocean (D.E. Dietrich, M.G. Marietta, and P.J. Roache, to be published), continental shelf phenomena (T. Ezer, personal communications, 1989), and flow around a seamount (G. Weatherly, personal communications, 1989).

Modeling the deep ocean requires much more carefully designed numerical approaches than are required to model shelf phenomena. Shelf models are not very sensitive to the numerical methods used due to the strong local turbulence and associated large local dissipation rates. On the other hand, deep ocean flows have extremely low real dissipation, and are dominated by eddies that are 10 to 100 times smaller than the real ocean scale, yet the eddy-scale based turbulent Reynolds number is very large. No turbulence closure model has successfully addressed their role in the ocean circulation (A.J. Semtner and R.M. Chervin, "A Simulation of the Global Ocean Circulation with Resolved Eddies," J. Geophys. Res., 93, 15502-15522, 1988). Thus, these eddies must be addressed directly, using the full (hydrostatic) Navier Stokes equations. It follows that numerical models must have very low total dissipation on a scale of 50 km or less in order to model ocean flows realistically. This either requires impractically high resolution (10 km or smaller grid intervals) or using numerical methods that are robust for large true cell Reynolds numbers.

To achieve the small numerical dissipation required to model deep ocean circulations, the numerical method used must be fully conservative. This translated to requiring that incompressibility be satisfied for even the smallest resolved grid elements, which requires use of an Arakawa "c" staggered grid (A. Arakawa and V.R. Lamb, "Computational Design of the Basic Dynamical Processes of the UCLA General Circulation Model," Methods in Computational Physics, Vol. 17, Academic Press, 174-265, 1977) for certain critical parts of the calculation. Deep ocean circulation models must also include full hydrostatic dynamics, including consistent determination of the barotropic mode. Proper determination of the barotropic mode requires a two-dimensional elliptic equation, either for the barotropic stream function as done by the BSC model -- see below) or a top surface pressure, as done by SOMS. Doing these things right is a non-trivial task that is not required in modeling shelf phenomena.

The Bryan-Semtner-Cox (BSC) model is "the principal tool for modeling ocean circulation in irregular domains having realistic coastlines and bottom topography" (A.J. Semtner and R.M. Chervin, "A Simulation of the Global Ocean Circulation with Resolved Eddies," J. Geophys. Res., 93, 15502-15522, 1988). The difficulty of addressing deep ocean phenomena is clearly demonstrated by recent application of this model to the world ocean circulation. This required using on the order of 500 hours of dedicated time on a Cray-XMP/48, with a fully vectorized code and all processors in use almost all of the time (99.64%). From our experience with the BSC, and also according to data reported by Semtner and Chervin, SOMS is at least ten times faster (as noted by D.E. Dietrich, M.G. Marietta, and P.J. Roache, "An Ocean Modelling System with Turbulent Boundary Layers and Topography: Numerical Description," International Journal for Numerical Methods in Fluids, 7, 833-8555, 1987), and requires much less numerical dissipation than the BSC.

SOMS' low numerical dissipation, and the need to have such low numerical dissipation in order to realistically address deep ocean circulations, is clearly demonstrated in convergence studies recently performed with SOMS (D.E. Dietrich, P.J. Roache, and M.G. Marietta, submitted for publication). In this paper, SOMS is applied to a prototype geophysical problem with baroclinic instability in a statically stable environment characteristic of intense frontal zones near western boundary current separation. Also in this paper, a new, more accurate treatment of these dominant terms, including the treatment used by the present world standard BSC, cannot match SOMS' high accuracy and corresponding low numerical dissipation. Finally, this paper shows that SOMS' low numerical dissipation leads to features not realistically addressed by previous models, such as secondary barotropic gyres, and clearly reveals the strong influence of the aforementioned thermodynamically driven eddies, which are the primary drivers of the general circulation of the deep ocean.

In summary, SOMS: is the only deep ocean model designed to address in detail the dynamics of boundary layers over realistic topography; is far less dissipative, more accurate, and runs ten times faster for a given resolution than the world standard BSC model; and is the first ocean model to clearly demonstrate convergence in a realistic geophysical prototype problem (at least, no such demonstration exists in the literature), which is a fundamental requirement in verifying a model.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

SOMS is thoroughly documented in a standard Sandia National Laboratories users' manual (D.E. Dietrich, M.G. Marietta, and P.J. Roache, 1989, that has just been completed.

3.2 Hardware Requirements

Depending on application, SOMS makes effective use of computers ranging in power from small workstation computers to super computers. For example, to model 30 days real time in a 600 km X 480 km rectangular region that is 3750 meters deep, under conditions typical of the North Atlantic Ocean, with 20 km horizontal resolution and 7 layers in the vertical, SOMS requires about 3 hours cpu using a 30 minute time step on an Everex Step 386/20 computer with an 80387 co-processor (retail price about $7,500). For a 1/2 degree resolution, 20 layer global ocean calculation, SOMS would use a 90 minute time step, which would require an estimated 150 hours on a Cray 1 computer (based on calculations performed at Sandia National Laboratories) compared to the 500 hours required by the BSC on the much faster Cray-XMP/48 computer (see Section 2 above). This translated to SOMS running ten times faster than the world standard model, using the same resolution. As noted above (see Section 2), SOMS is also much more accurate and has much less numerical dissipation.

3.3 Application

SOMS is applicable to the whole range of hydrostatic, incompressible geophysical flows, from boundary layer and lake scales to large ocean scales (see Section 2 above).

3.4 Level of Expertise Required

A working knowledge of FORTRAN, understanding of basic numerical modeling concepts such as numerical stability limitations, and understanding of modeling requirements of geophysical flows are all highly recommended before any attempt to use any general geophysical model such as SOMS. With such background, the user will find that SOMS is well documented and reasonably easy to learn to use and, in using SOMS, will find that SOMS is robust and accurate due to its good numerical design. Learning SOMS is facilitated by several built-in sample problems, including the circulation of Lake Neuchatel and the circulation of the North Atlantic Ocean, both using real topography data sets. Past experience indicates that an excellent graduate/post doctorate student can learn to use SOMS in about six months.

4 TECHNICAL CRITERIA

4.1 Peer Review

In view of its having been made available only recently, SOMS' varied applications in international university and industrial environments, noted in Section 2 above, provided a detailed code review. Users include (see Section 2 above): Professor Weatherly, Florida State Univesity; Professor Nyffeler, at the University of Neuchatel in Switzerland; and Dietrich, Marietta, Roache, and Steinberg in Albuquerque (Roache and Steinberg are working on implementing an advanced boundary-fitted coordinate scheme into SOMS).

4.2 Verification

As noted in Section 2 above, SOMS is the first model that has been shown to converge (as resolution increases) in a geophysical prototype problem, with baroclinic instability in a statically stable environment, which is a necessary part of model verification. Other verifications include: robustness under extreme modeling conditions (low dissipation); symmetry preservation in symmetric problems; and global conservation checks.

4.3 Uncertainty

All models of time-dependent flow are uncertain to some degree. Accuracy depends on many things, including initial conditions, resolution, model assumptions, duration of calculation, the nature of the feature to be determined (for example, whether it be the temperature at some point in time, or a long term average temperature in some region makes a great difference in model certainty), and when applicable, turbulence closure. How to carry out uncertainty analyses with these models and related box models for radionuclide transport is still an unsolved problem, but a reasonable approach was outlined by the POTG/SWG.

Model sensitivity to resolution and dissipation parameters is discussed in great detail by Dietrich, Roache, and Marietta (1989).

4.4 Required Input

The required input includes the usual geophysical modeling inputs, plus if the optional turbulence closure scheme is used, input data for the Mellor-Yamada level 2.5 turbulence closure scheme (Mellor and Yamada, 1982), which is used to determine vertical eddy diffusivities. Initial conditions for velocity and density must be specified, although hydrostatic and geostrophic relations can reduce the required initial data (initial pressure can be determined diagnostically directly from velocity and density). Boundary conditions for mass, momentum, and buoyancy flux must be specified at all times during a calculation. Model dissipation parameters (horizontal momentum and heat diffusivities, and nominal vertical diffusivities) must also be specified. Model geometry is specified by an input depth array. Non- homogeneous boundary conditions are all handled by a special subroutine and are controlled by input flag parameters. All input parameters are described in the SOMS users' manual (D.E. Dietrich, M.G. Marietta, and P.J. Roache, 1989).

4.5 *Output*

Contour plots as well as a number of numerical diagnostics are included in the output. Horizontal and vertical cross-sections of many different fields are contoured, and the amount of output is easily controlled by a few input parameters.

4.6 *Source Term*

Sources of heat and momentum (wind driving) are included in the top layer. Optional open boundaries and ports provide additional sources to drive the flow.

4.7 *Scenarios*

Scenarios such as sources of species from waste disposal sites have been studied (Marietta and Simmons, 1988; Nyffeler and Simmons, 1989); sources of pollutant to Lake Neuchatel are currently under investigation (Zuur and Dietrich, 1989). Dispersion of radionuclides has not been added to SOMS yet, and must be reinforced by the companion box model.

5 SCIENTIFIC CRITERIA

5.1 *Theory*

The theory behind SOMS basic modeling approach is a control-volume concept (P.J. Roache, Computational Fluid Dynamics, Hermosa Publishers) based on finite-volume conservation laws.

Other theories are associated with the two-parameter transportive turbulence closure model and the application of implicit numerical integration schemes.

5.2 *Validation*

As noted in Section 2 above, SOMS has been successfully applied to real ocean and lake flows, to continental shelf phenomena, and compares favorably with both theory (such as wind driven Sverdrup flow and boundary layer effects) and observations (including the circulation of Lake Neuchatel and western boundary current measurements in real oceans). Other model verifications include: accurate internal wave phase and group velocity; dominant eddy size corresponds to theoretical most unstable wavelength; realistic quasi-geostrophic flows; realistic turbulent boundary layer structures; and realistic deep ocean vertical velocities (less than one millimeter per second).

5.3 *Treatment of Radioactive Decay Products*

NA.

5.4 Underlying Assumptions

As with nearly all large-scale geophysical hydrodynamic models, SOMS uses the hydrostatic and Boussinesq approximations; and SOMS ignores secondary sphericity effects. It should be noted, however, that non-hydrostatic effects can easily be added (as described by D.E. Dietrich, M.G. Marietta, and P.J. Roache, 1987).

5.5 Pathways

NA.

5.6 Dose Conversion and Dose Response

NA.

IDENTIFICATION

1.1 Name: Holland Model

1.2 Prepared by: National Center for Atmospheric Research (NCAR), Boulder

1.3 Prepared for: NSF and ONR

1.4 Report Title: None

1.5 Report Number: None

1.6 Report Date: None

1.7 Availability: Not publicly available.

1.8 Purpose and Scope: A 3-dimensional quasi-geostrophic model designed for studying high-resolution basin-scale eddy- resolving ocean circulations.

2 SUMMARY OF FINDINGS

The Holland model has had some noted success in modeling ocean flows (W.J. Schmitz and W.R. Holland, "Observed and Modeled Mesoscale Variability Near the Gulf Stream and Kuroshio Extension," J. Geophys. Res., 91, 9624-9638, 1986; W.R. Holland and W.J. Schmitz, "Zonal Penetration Scale of Model Midlatitude Jets," J. Phys. Oceanogr., 1859-1875, December 1985). It is limited to free-ocean calculations. Due to its vorticity-streamfunction formulation, it is inapplicable to flow over realistic topography, which must be addressed by primitive equations, or to turbulent boundary layer flows.

A natural question is therefore why not use the more general primitive equations in the first place? It appears that the main motivation for using the Holland model is that its numerical efficiency is aided by not having to resolve internal waves in time. However, this is largely compensated by needing to solve Poisson equations, one for each layer at each time step.

Primitive equation models (in particular, SOMS) are thus generally competitive with the Holland model in terms of computational cost for problems to which both are applicable, while having a much broader range of applications. They can even avoid the need to resolve internal waves, by using implicit numerical approaches whose overhead can be less costly than the multiple Poisson solutions used by the Holland model.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

The Holland model is not publicly available.

3.2 *Hardware Requirements*

The Holland model is best run on main frame computers, but can also be run on mini-computers and supermicros.

3.3 *Application*

The Holland model is limited to free-ocean calculations with uniformly stratified thermal structures. It is not applicable to regions of significant topography, or to boundary layer phenomena. It is also not applicable to flows in small bains where deviations from non-divergent horizontal flow can be important.

3.4 *Level of Expertise Required*

A working knowledge of Fortran, understanding of basic numerical modeling concepts, and understanding of modeling requirements of geophysical flows are all highly recommended.

4 TECHNICAL CRITERIA

4.1 *Peer Review*

The Holland model is a well-established model, whose results have been reviewed and analyzed by several ocean modelers. The code has not been used or reviewed outside NCAR.

4.2 *Verification*

Important convergence tests have not been published.

4.3 *Uncertainty*

The Holland model has additional uncertainty compared to primitive equations models. This is associated with the horizontally non-divergent assumption and with the horizontal stratification assumption.

4.4 *Sensitivity*

No sensitivity studies have been published to date.

4.5 *Required Input*

The Holland model requires pressure specification at some level, and density everywhere for initial conditions. Buoyancy flux must be specified at inflow boundary points.

4.6 *Output*

Contour plots and a number of numerical diagnostics are included in the output.

4.7 Source Term

Internal heat and momentum sources are not included.

4.8 Scenarios

NA.

4.9 Relationship to Regulatory Standards

NA.

5 SCIENTIFIC CRITERIA

5.1 Theory

The Holland model is based on quasi-geostrophic flow theory, and uses a conservative difference form for the vorticity equation.

5.2 Validation

A limited validation experiment was performed against the measured distribution of potential and eddy kinetic energy with reasonable results.

5.3 Treatment of Radioactive Decay Products

Not addressed, but could be added.

5.4 Underlying Assumptions

The Holland model uses the quasi-geostrophic (more generally, quasi-non-divergent), hydrostatic, and Boussinesq approximations.

5.5 Pathways

NA.

5.6 Dose Conversion and Dose Response

NA.

IDENTIFICATION

1.1	*Name:*	Harvard Model
1.2	*Prepared by:*	Harvard University
1.3	*Prepared for:*	ONR
1.4	*Report Title:*	A draft user's manual exists, but is not publicly available.
1.5	*Report Number:*	None, but see 4.2
1.6	*Report Date:*	None, but see 4.2
1.7	*Availability:*	Available from: Harvard, Cambridge, MA
1.8	*Purpose and Scope:*	A baroclinic regional-scale eddy-resolving open-ocean numerical model designed for studying quasi-geostrophic ocean circulations. Theoretically, it is not applicable around topography although bottom bathymetry has been added through the bottom heat distribution and seems to perform well for gentle slopes. Model is part of the ocean prediction forecasting system presently being put into operational phase by the Navy.

2 SUMMARY OF FINDINGS

The Harvard model (Miller, R.N., A.R. Robinson, and D.B Haidvogel, "A Baroclinic Quasigeostrophic Open Ocean Model," J. Comp. Phys., 50, 38-70, 1983) is limited to free-ocean calculations with strongly stratified thermal structures. Due to its vorticity-streamfunction formulation, it is inapplicable to flow over realistic topography, which must be addressed by primitive equations, or to turbulent boundary layer flows.

A natural question is therefore why not use the more general primitive equations in the first place? It appears that the main motivation for using the Harvard model is that its numerical efficiency is aided by not having to resolve internal waves in time. However, this is largely compensated by needing to solve Poisson equations, one for each layer each time step, and to transform between real and normal mode space, which can dominate the calculation if the vertical resolution is greater than O(10) layers.

Primitive equation models (in particular, SOMS) are thus generally competitive with the Harvard model in terms of computational cost for problems to which both are applicable, while having a much broader range of applications. They can even avoid the need to resolve internal waves, by using implicit numerical approaches whose overhead can be less costly than the multiple Poisson solutions and normal mode transformations used by the Harvard model.

The Harvard model has some nice theory behind it, and can be used as a teaching and research tool in spite of its practical limitations (although its value as a research tool is limited).

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

The Harvard model is well doeumented by an internal user's manual. It is not publicly available.

3.2 Hardware Requirements

The Harvard model is best run on main frame computers, but can also be run on mini-computers and supermicros.

3.3 Application

The Harvard model is limited to free-ocean calculations with strongly stratified thermal structures. It is not applicable to regions of significant topography, or to boundary layer phenomena. It is also not applicable to flows in lakes where deviations from non-divergent horizontal flow can be important.

3.4 Level of Expertise Required

A working knowledge of Fortran, understanding of basic numerical modeling concepts, and understanding of modeling requirements of geophysical flows are all highly recommended.

4 TECHNICAL CRITERIA

4.1 Peer Review

The Harvard model is a well-established model that has been reviewed and analyzed by several ocean modelers. It is not considered to have much practical applicability, but it is used for ocean forecasting by the Navy.

4.2 Verification

The Harvard model is one of the two ocean models which has undergone extensive numerical and physical parameter studies (the other is SOMS). These studies are well reported in both the scientific literature (Haidvogel et al., 1980; Miller et al., 1983) and SDP reports (Marietta and Simmons, 1987).

Haidvogel, D.B., A.R. Robinson, and F.F. Schuman (1980). The accuracy, efficiency, and stability of three ocean models with application to open ocean problems. J. Comp. Phys., 34:1-53.

Marietta, M.G. and W.F. Simmons (1987). SDP Annnual Report: Modeling Studies. SAND86-0929. Sandia National Laboratories, Albuquerque, NM.

Miller, R.N., A.R. Robinson, and D.B. Haidvogel (1983). A baroclinic quasigeostrophic open ocean model. J. Comp. Phys., 50:38-70.

Marietta, M.G. and A.R. Robinson (1986). Status and Outlook of Ocean Modeling, Research, Dispersion and Related Applications. SAND85-2806, Sandia National Laboratories, Albuquerque, NM.

4.3 *Uncertainty*

The Harvard model has additional uncertainty compared to primitive equations models. This is associated with the horizontally non-divergent assumption and with the horizontal stratification assumption used in the determination of normal modes.

4.4 *Sensitivity*

See 5.2.

4.5 *Required Input*

The Harvard model requires pressure specification at some level, and density everywhere for initial conditions. Pressure must be specified at some level for all time, which is a major limitation. Buoyancy flux must be specified at inflow boundary points.

4.6 *Output*

Contour plots and a number of numerical diagnostics are included in the output.

4.7 *Source Term*

Internal heat and momentum sources are not included.

4.8 *Scenarios*

NA.

4.9 *Relationship to Regulatory Standards*

5 SCIENTIFIC CRITERIA

5.1 *Theory*

The Harvard model is based on quasi-geostrophic flow theory, and uses a conservative difference form for the vorticity equation.

5.2 *Validation*

The Harvard model was the first model used for ocean forecasting. Early work was carried out in the California Current System with the Naval Postgraduate School (Robinson et al., 1984). Ocean data from AXBTs and XBTs were transmitted to shore and assimilated in near real time into model simulations that forecast future fields. These forecast fields were compared to later observed fields for accuracy. This work led to the development of the Ocean Descriptive Prediction System (ODPS) for forecasting in real time using shipboard computers (PDP-11/73s and micro-VAX IIs). The first such at sea forecasts were carried out on the Nares Abyssal Plain as a joint project between Harvard and Sandia (Marietta and Simmons, 1987). Validation was only partially successful. The first real at sea validation was performed by Sandia using the Harvard model at the NOAMP site working jointly with the DHI/FRT and Swiss/Neuchatel through the NEA/CRESP. SOMS was also used successfully during this work (Kupferman et al., 1986; Marietta and Simmons, 1988). The Harvard model is currently used by the Navy in the ODPS framework for at sea forecasting for the fleet. The model's limitations are well known and a number of questionable tricks have been employed to improve model performance in flow regimes beyond the model's capability defined by extensive sensitivity studies, e.g., empirically embedding flow features observed in satellite pictures, including topography through the bottom heat distribution instead of explicitly. Still this model has had many successes, primarily because before SOMS there were no primitive equation models that performed well in real ocean validations.

Robinson et al., 1984. A real time forecast of ocean synoptic/mesoscale eddies. Nature, 309(5971). 781-883.

Kupferman et al., 1986. An intense cold-core eddy in the North-East Atlantic. Nature, 319(6053), 474-477.

5.3 Treatment of Radioactive Decay Products

Not addressed, but could be added.

5.4 Underlying Assumptions

The Harvard model uses the quasi-geostrophic (more generally. quasi-non-divergent), hydrostatic, and Boussinesq approximations.

5.5 Pathways

NA.

1 IDENTIFICATION

1.1 Name:	NRPB 91-Box Model (91BXM)
1.2 Prepared by:	National Radiological Protection Board (NRPB), Oxfordshire and Ministry of Agriculture, Food and Fisheries, Lowestoft, U.K.
1.3 Prepared for:	NEA Site Suitability Review (SSR) of the Northeast Atlantic Dumpsite
1.4 Report Title:	Review of the Continued Suitability of the Dumping Site for Radioactive Waste in the North-East Atlantic
1.5 Report Number:	NEA Report, no number
1.6 Report Data:	1985
1.7 Availability:	Available from: National Radiological Protection Board, Chilton, Didcot, Oxon OX11 ORQ, England
1.8 Purpose and Scope:	91BXM is a coarse-grid box model which calculates radionuclide transport based on an ocean circulation that was subjectively assembled from an extensive literature review. It is not coupled with a dynamical circulation model although like all box models, it could be driven by simulated circulations. 91BXM has been used for risk assessments of high-level waste disposal in the subseabed and assessments of low-level sea dumping at the North-East Atlantic NEA site.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

See 1.4

3.2 Hardware Requirements

91BXM has been used on the Harwell CDC machines and the NRPB VAX 786. It uses an ODE solver developed at Harwell which is available at a significant cost.

3.3 Application

Code is not user friendly. Interbox transports must be obtained either by a subsidiary calculation from a high resolution simulation of the basin circulation or from expert descriptive oceanographers. Ocean data are sparse so considerable subjective judgement is required. If simulated ocean circulations are used, interbox transports must be hand calculated and entered. Interbox transports for nested boxes which must be site-specific have to be obtained from subjective expert estimates. Problem setup for a new site requires considerable recoding.

3.4 *Level of Expertise*

Running the code is simple. Setting up new problems or performing sensitivity analyses on physical oceanographic parameters wouldn't be worth the effort. Box model codes should be simple. This one is not, and the code could probably be run only by one of the authors.

4 TECHNICAL CRITERIA

4.1 *Peer Review*

As part of the review of the North-East Atlantic Dumpsite, NEA conducted expert review workshops on the 91BXM and data bases. Two review sessions were held at NEA headquarters. The first group of experts rejected this model because of a theoretical problem. NRPB was asked to change their modeling approach from one that relied on an inverse technique to one relying on a subjectively determined circulation. As a result the coding of this model remains messy. Model performance using the subjective circulation was judged by the second workshop as adequate for the suitability review although problems still remain, as they must, since the driving circulation is based on very sparse data in key regions of the ocean.

4.2 *Verification*

91BXM was a participant code in a model comparison study conducted by the NEA CRESP Modeling Task Group. Mobbs et al. (1986). A preliminary comparison of models for the dispersion of radionuclides released into the deep ocean. NRPB-R194.

4.3 *Uncertainty*

No uncertainty analyses have been carried out with 91BXM although NRPB has the LISA systems code for this purpose. Uncertainty analyses using a box model with too many boxes would not be sensible. Consequently a simpler version, MINIBOX, was developed for uncertainty analysis.

4.4 *Sensitivity*

Sensitivity studies on various pathway and physical parameters were performed for the NEA workshops and are discussed in the NEA SSR Report.

4.5 *Input*

Interbox transports (an ocean circulation). Geochemical parameters.

4.6 *Output*

Box concentrations of radionuclides and maximum individual dose estimates.

4.7 Source Term

release rates of each radionuclide from bottom source at repository location.

4.8 Scenarios

Scenarios for ocean disposal have been defined and analyzed but require different model setups.

4.9 Reg. Standards

There are no ocean standards

5 SCIENTIFIC CRITERIA

5.1 Theory

The compartment model covers the area of the Atlantic Ocean from 50 S to 65 N and uses observed isopycnal (density) surfaces to define the vertical box structure, since mixing and movement in the ocean are believed to occur primarily along isopycnal surfaces. Exchanges with other oceans are also included, since radionuclides entering the Atlantic will eventually disperse throughout the world's oceans. These exchanges are with the Arctic Ocean through the Norwegian and Greenland Seas to the north, with the Mediterranean Sea to the east and with the Pacific and Indian Oceans, via the Antarctic Circumpolar Current, to the south.

The Atlantic Ocean is divided into eight areas (Figure 6.37). The central dividing line corresponds to the Mid-Atlantic Ridge and the areas extend out to the edge of the continental shelf. Areas 1 and 3 are based on areas chosen by Worthington (1976) on the basis of bottom topography. Area 2 is chosen to represent adequately the current flow patterns in the north. The southern boundary of Area 4 (20 N) corresponds approximately to the position of the Cape Verde Islands. Areas 5 and 6 cover the equatorial waters and their southern boundaries (both at 25 S) and correspond to the Rio Grande Rise and Walvis Ridge, respectively. Areas 7 and 8 cover the remainder of the South Atlantic and extend to 50 S, which is approximately the latitude of the Falkland Islands in the southwest. The present low-level dump site is in Area 4, relatively close to the northern boundary.

It is possible to identify well-defined water masses at specific depths throughout the Atlantic Ocean. Therefore, the eight ocean area were divided into layers so that each compartment would represent a separate water mass. The following water types were identified: (1) Antarctic Bottom Water (levels 9 and 10, only in Areas 6, 7 and 8); (2) North Atlantic Deep Water (levels 7 and 8); (3) North Atlantic Central Water (levels 5 and 6); (4) MODE Water (levels 3 and 4); (5) surface water (levels 1 and 2). The boundaries of the compartments were obtained by tracing the neutral (density) surfaces through the ocean, using the method of Ivers (1975) and data taken from the GEOSECS survey (Bainbridge, 1972) with additions from the International Geophysical Year (IGY) surveys (Fuglister, 1960). Figures 6.38 and 6.39 show these levels for the east and west Atlantic, respectively. Initially, sic levels were chosen in the vertical, but it was thought that a better resolution of the water masses would be obtained using up to 10 levels. This gives rise to 67 boxes in the Atlantic and a further 24 boxes (defined by the same density levels) for the other oceans, which have much less spatial resolution, to give a total of 91 boxes (Figure 6.40). Each box is allowed to communicate with each of its neighbors, unless topography interferes, resulting in 442 transfer coefficients to be determined. Instead of describing these transfer coefficients in terms of an advective flow and a diffusive mixing among boxes, they are given as a transfer from A to B and another from B to A. The difference between the two transfers is the advective flow, and a weighted sum of them can be interpreted as the eddy diffusivity (Fiadeiro, 1975).

The model structure described above assumes that the oceans are zonally well mixed and that it is possible to describe the isopycnal surfaces of the oceans from meridional sections. To define these surfaces, the method of Ivers (1975) was used in which the neutral surface is defined to be normal to the gradient in potential density, referred to the pressure at the point in question. Ivers used this definition to discuss the behavior of five neutral surfaces in the North Atlantic. The shallowest outcrops in the Labrador Sea and descends to 900 m at 30 N, and the deepest outcrops in the Norwegian-Greenland Sea and descends to 3000 m at 40 N. Ivers found that most features in salinity could be explained by lateral flow and mixing, except in regions of sills and outcroppings, where vertical mixing becomes important.

The GEOSECS results (Bainbridge, 1972) represent a good quality data set which provides meridional sections required for the model description, except for the northeast Atlantic. In this area, the continuation of eastern Atlantic GEOSECS section was compiled from IGY data (Fuglister, 1960) and was constructed to run as near as possible along the deep water of the northeast basin and to the west Rockall Bank to the Norwegian Sea.

The neutral surfaces were constructed by starting at the southern end of each section with and values, and following them north from station to station on the sections. These surfaces tilt steeply near the polar ends of each section, while remaining almost horizontal in the deep central ocean. Some surfaces intersect the seafloor, whereas others outcrop. The geographical position of the outcropping varies from season to season (Levitus, 1982).

Similar methods were used to calculate the neutral surface depths for the Arctic, Antarctic, and Pacific Oceans from the GEOSECS data. However, data for the Mediterranean Sea are taken from Sankey (1973) and Wust (1961), with some adjustments made to obtain consistent values for inflow and outflow. Only two levels are used in the Mediterranean because the shallow sill depth in the Straits of Gibralter effectively isolates any deeper layers from communicating directly with the Atlantic. Similarly, the Denmark Strait restricts communication between the Greenland Sea and Labrador basin. The flow pattern described above gives only the advective exchanges between the boxes, whereas two-way exchanges are needed to express advection and mixing. An upwind differencing scheme was used to include eddy diffusivities of 1 cm /s diapycnally and 10 cm /S laterally, and the resulting transfers are listed in Table C-2. Where box dimensions were similar, the mean dimension was used to calculate gradients needed to produce the transfers, but when one box was much larger than the other, the smaller dimension was used.

5.2 Validation

Box models including the 92BXM have not been shown to compare favorably with any ocean tracer distribution. The NEA SSR panel of experts and the NEA SWG POTG. Following this reversal the NRPB/MAFF joined the POT approach using dynamical simulations for driving circulations in an attempt to validate these ocean models against natural and introduced tracer distributions. A total of eight different tracers were studied: salinity, temperature C-12, Th-230, R-226, Th-232,R-228, and Th-228.

5.3 Treatment of radioactive decay products

If data are available, simplified chains could be handled.

5..4 Underlying assumptions

See 5.1

5.5 Pathways

Pathways are the IAEA pathways described under Mark-A

5.6 Dose conversion and dose response

Dose is estimated to critical individuals and the collective dose commitment. Dose response is not included. Requires code coupling.

Table C-1. NOMINAL DENSITIES OF THE TEN LEVELS

Level	Density		
1 (top)			<26
2	26	<	<26.75
3	26.75	<	<27
4	27	<	<27.3
5	27.3	<	< 45.5
6	45.5	<	<45.8
7	45.8	<	<45.9
8	45.9	<	<46
9	46	<	<46.1
10 (bottom)			<46.1

Table C-2. WATER TRANSFERS BETWEEN BOXES IN SVERDRUPS

AH = 10000000. cm*cm/s and K=1. cm*cm/s

		Transfer	
Box A	Box B	A to B	B to A
1	2	.358	4.358
1	9	.060	3.060
1	16	8.581	.581
1	24	.091	.091
1	68	.025	1.025
2	3	.208	3.208
2	10	.587	.587
2	17	3.274	.374
2	25	.654	.654
2	69	.008	4.008
3	4	.362	1.362
3	11	2.331	2.331
3	18	.772	.772
3	26	1.665	1.665
3	70	.081	2.081
4	5	4.066	4.066
4	12	.232	.232
4	19	.098	1.098
4	27	.198	.198
4	71	.012	.012
5	6	9.701	7.701
5	13	.241	1.241
5	20	.108	1.108
5	28	.243	.243
5	72	.035	.035
6	7	6.739	4.739
6	14	.143	.143
6	21	.139	.139
6	29	.138	.138
7	8	3.753	22.753
7	15	.112	.112
7	22	.193	3.193
7	30	24.171	.171
8	24	.085	19.085
8	31	.108	.108
9	10	.047	1.047
9	24	.258	1.258
9	68	.010	1.010
10	11	.0631	1.063
10	25	1.313	.313
10	69	.038	1.038

1 IDENTIFICATION

1.1 Name: TASC Box Model, MARINRAD

1.2 Prepared by: The Analytic Sciences Corporation under contract from Sandia National Laboratories

1.3 Prepared for: U.S. DOE Subseabed Program, Sandia National Laboratories

1.4 Report Title: User's Guide to MARINRAD: Model for Assessing the Consequences of Release of Radioactive Material into the Oceans.

1.5 Report Number: SAND83-7104

1.6 Report Date: 1984

1.7 Availability: Available from: Sandia National Laboratories, Albuquerque, NM

1.8 Purpose and Scope: MARINRAD is a coarse-grid box model which calculates radionuclide transport based on an ocean circulation that was subjectively assembled from historical literature. It is not coupled with a dynamical circulation model although like all box models, it could be driven by simulated circulations. MARINRAD has been used for risk assessments of high-level waste disposal in the subseabed of both the Atlantic and Pacific and assessments of low-level sea dumping at the North-East Atlantic NEA site.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

Koplik, C.M., M.F. Kaplan, J.T. Nalbandian, J.H. Simonson, and P.G. Clark, (1984). User's Guide to MARINRAD. SAND83-7104. Sandia National Laboratories, Albuquerque, NM.

Ensminger, D.A., C.M. Koplik, and R.D. Klett, (1984). Prelilminary Pre-Emplacement Safety Analysis of the Subseabd Disposal of High-Level Nuclear Waste. SAND83-7105, Sandia National Laboratories, NM.

Kaplan, M.F., C.M. Koplik, and R.D. Klett, (1984). Preliminary Post-emplacement Saftey Analysis of the Subseabed Disposal of High-Level Nuclear Waste. SAND83-7106, Sandia National Laboratories, Albuquerque, NM.

Kaplan, M.F. (1984). Biological and Physical Sensitivity Analyses for Subseabed Disposal of High-Level Waste. SAND83-7107, Sandia Natinal Laboratories, Albuquerque, NM.

3.2 Hardware Requirements

MARINRAD has been used on CDC and DEC VAX machines.

3.3 Application

Code is user friendly. Interbox transports must be entered as input like the other box models, but there are many fewer boxes. Ocean data are sparse so considerable subjective judgement is still required. If simulated ocean circulations are used, interbox transports must be hand calculated and entered. Interbox transports for nested boxes which must be site-spectific have to be obtained from subjective expert estimates. Problem setup for a new site requires considerable work with the descriptive oceanographic literature but little recoding.

3.4 Level of Expertise

Running the code is simple. Setting up new problems is not. See, therefore, MARK A.

4 TECHNICAL CRITERIA

4.1 Peer Review

MARINRAD has never been peer reviewed, but it was part of the NEA CRESP comparison study (see 4.2 Verification).

4.2 Verification

MARINRAD was a participant code in a model comparison study conducted by the NEA CRESP Modeling Task Group. Mobbs et al. (1986). A preliminary comparison of models for the dispersion of radionuclides released into the deep ocean. Report No. NRPB-R194.

4.3 Uncertainty

No uncertainty analyses have been carried out with MARINRAD although they could be if coupled with a dynamical circulation model. Mark-A would be a better candidate since it is a full basin model, not a half basin like MARINRAD.

4.4 Sensitivity

Sensitivity studies on various pathway and physical parameters were completed (see documentation). The adjoint method was also used for sensitivity studies with MARINRAD.

4.5 Input

Interbox transports (an ocean circulation). Geochemical parameters.

4.6 *Output*

Box concentrations of radionuclides and maximum individual dose estimates.

4.7 *Source Term*

Release rates of each radionuclide from bottom source at repository location.

4.8 *Scenarios*

Scenarios for ocean disposal have been defined and analyzed with MARINRAD but require different model setups.

4.9 *Reg. Standards*

There are no ocean standards.

5 SCIENTIFIC CRITERIA

5.1 *Geometry and Transports for the MPG-s Model*

A hypothetical north-central Pacific (MPG-1) site is located about 10^3 km north of Hawaii in roughly 6000 m of water. The vertical compartmentalization of the model is in three levels. A southward flowing middepth water mass, 2000 m in thickness, is formed from a mixture of northward flowing deep and abyssal waters, 1000 m in thickness, and weakly southward flowing surface waters, 1000 m in thickness. A 10-cm deep bioturbated sediment layer underlies all the deep boxes. Near-shore sediment boxes with higher sedimentation rates underlie the North and South Pacific surface boxes. They are taken to be 19.5% of the bottom area of their associated water boxes.

The initially imposed water mass balance is derived from work on South Pacific Ocean/Circumpolar Current exchanges and deep vertical South Pacific advection rates. Specifically, 14.5 Sv flow into and out of the South Pacific from the Circumpolar Current. Of that, 12 Sv of Circumpolar Bottom Water enter the western boundary undercurrent. South Pacific Bottom Water advects vertically to the Deep Water at a rate of 8 Sv. South Pacific Deep Water returns to the Circumpolar Current at a rate of 7.5 Sv and also advects to the surface water at 3 Sv. The remaining circulation is adjusted to conserve water mass as required by the above transports and to allow for vertical advection at roughly 3 m/yr.

Within the Bottom North Pacific box is a set of nested process boxes. The site region is taken as a 100 x 100 km square. It is overlain by a turbulent bottom mixed layer (BML) of height H 100 m, taken to be the height of the site box. The vertical eddy diffusivity in the BML is A 100 cm /sec. Thus, the time scale for vertical mixing in the BML is t = H /4A 3 days. Using a bottom mean velocity of 3 cm/sec, the site box is flushed in 100 km/3 cm/sec 39 days. Thus, the flushing scale is much geater than the mixing scale and therefore a box model approach is acceptable.

The eddy box which circumscribes the site box is taken to be 200 x 200 square so that, at most, four close-packed 100-km diameter deep eddies can occupy it. This is roughly in keeping with observed eddy length scales. The physical idea here is tht the BML disperses the contaminant to horizontal and vertical scales which are large enough for the eddies to work on. They in turn disperse the contaminant to still greater scales which are large enough to be dispersed efficiently by the general circulation. This idea, although correct in principle, is stretched to i ts limits of applicability by a 200 x 200 km eddy mixing box. One would prefer to see at least a 5 x 5 and preferably a 10 x 10 eddy box. This problem is addressed in the Mark A model. The time required for a patch to grow to eddy size is roughly 20 days (i.e., 100 km/5 cm/sec). At this time the patch is eddy-sized, but may be streaky. It has diffused vertically upward from the BML by an amount (4 x 0.6 cm^2/sec x 20 days)$^{0.5}$ 20 m, where the vertical eddy diffusivity in the deep water A_2 = 0.6 cm /sec. Thus, the height of the eddy box is taken as 20 m, i.e., the height of the BML plus the diffusive spreading height.

The next box in the nest is a basin-scale geographical box, where mixing scales come from horizontal diffusivity of the order of AH 5 x 10^6 cm^2/sec and flushing scales from the general circulation, i.e., the volume of the boxes divided by th bulk flow rates through them. Large differences between these times are difficult to establish, i.e., the ocean is not well-mixed.

Geometry and Transports for the NAP Site

A hypothetical western North Atlantic site is located about 1000 km south of Bermuda on the Nares Abyssal Plain in roughly 5000 m of water. Because deep east-west communication is essentially prohibited by the mid-Atlantic Ridge, the eastern North Atlantic is not separately compartmentalized but rather is grouped with Remaining Ocean Waters.

Vertical compartmentalization of the western North Atlantic is into two layers divided roughly at the 4^oC isotherm, and corresponding to southward flowing North Atlantic Dep Water and northward flowing North Atlantic Surface Water. The lower boxes are roughly 2810 m in height. Sediment boxes are treated as they were in the Pacific site.

The imposed water mass balance allows for a 3 m/yr vertical advection in the North American and Guiana Basins and absorbs transports to and from the Norwegian Sea as an internal circulation in the Labrador Sea box. The essential features are (i) a 6-Sv transport from the deep North American Basin carried by the Western Boundary Undercurrent, (ii) a 3-Sv inflow to the Deep Labrador Basin, (iii) a 2.5 Sv inflow to the Surface Guiana Basin (reduced from Worthington's 4-Sv value in order to allow for vertical advection at 3 m/yr while retaining 4 Sv inflow to the Surface Larador Basin). The other features are determined by (i) an assumed 3 m/yr vertical advection and (ii) mass continuity box-by-box.

Process boxes in the Deep North American Basins are divided geographically to account for the strong gyre circulatin induced by the Gulf Stream to the north and the strong eddy field to the south. As in the Pacific, the burial site is taken to be a 100 x 100 km^2 area. Since the BML height is 50 m and the vertical boundary layer eddy diffusivity is 10 cm^2 /sec, the internal boundary layer mixing time becomes t = $H^2/4A$ or 7 days.

Typical boundry layer velocities at the site are 4 cm/sec, so the flushing time is 100 km/4 cm/s or about 29 days, roughly four times the mixing time. Thus, scale separation in the Atlantic model is not quite as strong as in the Pacific model, but it is acceptable. The eddy box is again a minimally-sized 200 x 200 km square. The scales for (i) mixing and (ii) variability are taken as the time for horizontal diffusion (at AH 10^7 cm^2/s) (i) from the site to the southern edge of the Gyre box and (ii) double that distance. The times work out to 9.6 yrs and 29 yrs, respectively, which, again, is close but acceptabale. The gyre box has a characteristic circulation time of about 2.5 yr and a mixing time which is probably less than a month, since the eddy field in this region is quite intense (AH 5×10^7 cm^2/sec).

The height of the eddy and gyre boxes is taken as the sum of the BML height plus the advective and diffusive (A_z 2 cm^2/sec) heights acquired while the contaminant spreads across the two boxes, i.e., during roughly 12 yr. This works out to 50 m + 36 m + 390 m or 475 m.

Because AH = 5×10^6 cm^2/sec in the Remaining Ocean Waters box, there are three horizontal diffusivities in the model. In the numerical work, the smaller diffusivity was used when there was a choice. A brief sensitivity analysis suggested small dependencies on this value. Sedimentation was treated as in the Pacific.

5.2 Validation

Box models including MARINRAD have not been shown to compare favorably with any ocean trracer distribution.

5.3 Treatment of Radioactive Decay Products

If data are available, simplified chains can be handled.

5.4 Underlying Assumptions

See 5.1

5.5 Pathways

Pathways are the IAEA pathways described under Mark-A.

5.6 Dose Conversion and Dose Response

Dose is estimated to critical individuals. Dose response is not included. Requires code coupling.

1 IDENTIFICATION

1.1 Name:	Mark-A Box Model
1.2 Prepared By:	Physical Oceanography Task Group, Seabed Working Group, NEA
1.3 Prepared For:	NEA
1.4 Report Title:	Research, Progress and the Mark A Box Model for Physical, Biological and Chemical Transports
1.5 Report Number:	SAND84-0646
1.6 Report Date:	1985
1.7 Availability:	Available from: Ecoles des Mines, Fountainebleau, France; Commission of European Communities; Joint Research Center, Ispra, Varese, Italy.
1.8 Purpose and Scope:	The Mark-A is a coarse-grid box model which calculates radionuclide transport using an ocean circulation as input. It must be coupled with a dynamical circulation model. Mark-A has been used for risk assessments of high-level waste disposal in the subseabed for two different locations.

3 ADMINISTRATIVE CRITERIA

3.1 Documentation

1. Robinson, A.R. and M.G. Marietta, (1985). Research, Progress and the Mark-A Box Model for Physical, Biological and Chemical Transports. SAND84-0646. 2. Marietta, M.G. and W.F. Simmons, (1988). Feasibility of Disposal of High-Level Radioactive Wastes into the Seabed: Dispersal of Radionuclides in the Oceans: Models, Data Sets, and Regional Descriptions. SAND87-0753 (Also Feasibility of Disposal of High-Level Radioactive Wastes into the Seabed. Volume 5. (1989) NEA, Paris). 3. de Marsily, G. et al. (1989). Feasibility of Disposal of High-Level Radioactive Wastes into the Seabed. Volume 2. Radiological Assessment. NEA, Paris.

3.2 Hardware Requirements

Mark-a has been used on DEC PDP11's, VAXs, and IBM PCs. It uses an ODE solver, and GEARB is provided with it.

3.3 Application

Code is not user friendly. Interbox transports must be obtained via a subsidiary calculation from a high resolution simulation of the basin circulation. Interbox transports for nested boxes which must be site-specific have to be obtained from local eddy-resolving simulations of the site region. Problem setup for a new site involves understanding the dynamical ocean simulations from which the transports are extracted.

3.4 Level of Expertise Required

Running the code is simple. Setting up new problems or performing sensitivity analyses on physical oceanographic parameters is not. Ocean modeling experience and a knowledge of descriptive data bases is required.

4 TECHNICAL CRITERIA

4.1 Peer Review

Code was developed by an NEA working group comprised of national experts from C.E.C., F.R.G., France, Switzerland, U.K., and U.S. It has been exercised and intercompared at Sandia, Ecoles des Mines de Paris, and C.E.C. (Ispra). The various experts who served on the working group and the staff at these institutions have reviewed the model.

4.2 Verification

In addition to peer review comments, Mark-A was a participant code in a model comparison study conducted by the NEA CRESP Modeling Task Group. Nyffeler, F. and W.F. Simmons, 1989. Interim Oceanographic Description of the Northeast Atlantic Dumpsite III. NEA, Paris.

4.3 Uncertainty

Uncertainty analyses were carried out by C.E.C. using a Monte Carlo sampling approach by coupling the Mark-A with a systems called LISA. LISA is a C.E.C. version of the Canadian systems code, SYVAC which was also used in an earlier study on the Mark-A. In both studies using LISA and SYVAC insufficient data were available to do a meaningful uncertainty analysis. Parameter ranges and distributions that were randomly sampled were guessed in order to demonstrate the capability for doing uncertainty analysis when sufficient data become available. Therefore, results of these studies were not considered realistic.

4.4 Sensitivity

Sensitivity studies on various box configurations and physical and numerical parameters have been carried out by the NEA SWG POTG and are contained in the POTG reports.

4.5 Input

Interbox transports (an ocean circulation). Geochemical parameters.

4.6 Output

Box concentrations of radionuclides and maximum individual dose estimates.

4.7 Source Term

Release rates of each radionuclides from bottom source at repository location.

4.8 Scenarios

Scenarios for ocean disposal have been defined and analyzed but require different model set-ups.

4.9 Regulatory Standards

There are no ocean standards.

5 SCIENTIFIC CRITERIA

5.1 Theory

For the Mark-A box configuration, East Atlantic and West Atlantic boxes were placed side by side to constitute a four-zone North Atlantic model. This choice was motivated primarily by basin geometry, underlying topography, site locations, local mixing times, and anticipated resolution. The east-west division of the basin was motivated by the Mid-Atlantic ridge, and the north-south division by the different senses of flow typifying the northern and southern parts of the North Atlantic gyre. Simple two-layer vertical resolution is sufficient to separate upper water from deep water, but because the deep water is bounded intermittently by the Mid-Atlantic ridge, three layers are necessary in the vertical. Boxes for the Arctic and South Atlantic Oceans are also included, as is a remaining ocean waters box. These boxes all have two layers in the vertical. This geographic box arrangement, is thought to be the simplest coarse grid arrangement consistent with realistic results.

The source basin box includes two nested boxes: a site box over the source and a larger, eddy-mixing box sized from the local mean eddy diameter. Their size and configuration take into account the projected mixing times of the region and desired resolution of the results. Benthic sediment layer boxes, described in the geochemistry section, are underneath the water boxes. A collection of shelf/slope boxes is a subsidiary calculation, but does not have a significant effect on the distribution of tracers in a basin or gyre-sized box, except for some sidewall scavengers.

The geochemical scavenging model described herein for the Mark-A model is essentially that developed by GESAMP (1983) which was taken from the ideas formulated primarily by Cochran (Robinson and Kupferman, 1985; Marietta and Robinson, 1981). The geochemical boxes are of higher resolution than the physical boxes. Geochemical boxes were chosen on the basis of their ability to describe concentration distributions (natural trace fields and releases) dicatated by the dominant geochemical processes. These differ significantly from the dominant physical processes and, correspondingly, Mark-A geochemical box geometry differs from the physical box geometry.

Two types of particles are considered: large, biogenic particles (>50 m) and small, inorganic (<50 m) particles, both of which interact with dissolved contaminants. The large-particle class includes two types of biogenic particles: fecal pellets (agglomerations of fine particles rich in organic matter) and hard parts such as the shells of plankton. Both types of large particles disintegrate during settling. The organic-rich fecal pellets break down through organic matter oxidation and the hard parts (tests) dissolve. The inorganic small particles sink slowly and interact with dissolved contaminants throughout the water column. Small particles can be transformed into large particles by grazing of bathypelagic organisms.

Particle-solution interactions are considered first-order interactions with respect to concentration of contaminants on particles and in solution. The use of a sorption coefficient, Kd (forward reaction rate/backward reaction rate), to parameterize these interactions seems appropriate at this stage of the effort. The sorption coefficient refers only to water column particles, not sediments.

The bottom sediments are represented by a sell-mixed bioturbated layer about 10-cm thick in which particle-associated contaminants can reach sorption equilibrium with the pore water and can diffuse into the overlying water of the benthic boundary layer. Net removal from the water column and the sediment mixed layer is by sediment accumulation (i.e., sedimentary burial).

5.2 Validation

Box models including the Mark-A have not been shown to compare varoably with any ocean tracer distribution. Coupled with a robust ocean circulation model, the Mark-A has been shown to reporude model distributions. Further work is required with both the dynamical ocean model and related box model to reproduce both natural and introduced tracers in the ocean (Marietta and Simmons, 1988).

5.3 Treatment of Radioactive Decay Products

If data are available, simplified chains could be handled.

5.4 Underlying Assumptions

See 5.1.

5.5 Pathways

Compliance with the system of dose limitation recommended by the ICRP and embodied in the IAEA Basic Safety Standards is checked by assessing the dose to the critical group of members of the public and ensuring that this is less than the appropriate dose limit. In many cases it is not possible to asses the doses directly, so that a derived limit is used. The derived limit is directly related to the basic dose limit by a model or models: the models and parameter values are chosen to maximize the dose, so that compliance with the derived limit ensures virtual certainty of compliance with the basic dose limit. The models used are ones describing the redistribution of the radionuclides in the environment and specific pathways by which man may be irradiated. Site-specific assessments, applicable to wastes of known composition, and optimization and/or an apportionment of a fraction of the dose limit may lead national authorities to limit dumping rates well below the dumping rate limits.

The choice of the critical pathway is not straightgorward and for generic assessments of this type it is necessary to postulate a number of pathways which may be critical. Calculations are then carried out for all the postulated pathways; the one which leads to the highest doses from a given radionuclide is the critical pathway for the radionuclide.

The pathways selected are listed in Table C-3 and include many which are known to exist as well as some that could occur in the foreseeable future due to changes in the use of ocean resources. Other pathways have been investigated, the existence of which is possible given the current knowledge of the marine environment, but whose proability of leading to exposures is very low. These are listed in Table C-4 as hypothetical pathways. Even though these pathways are viewed as improbable, they are identified so that due account can be taken in assessing results for the purpose of setting limits. This use of many general pathways should ensure that the discovery or postulation of a new pathway will not involve changes to the limit unless it is clearly not already covered by a pathway. The parameters selected for the various pathways are intended to be sufficiently general as to cover critical groups in all areas of the world. In addition to considering each separate pathway it is also necessary to determine the manner and extent to which the doses through the various pathways should be summed in determining overall dumping rate limits. Bearing in mind current known habits from experience and dietary habits in North American, European and Asiatic countries, and guided also by effective limitations of protein intake, it was concluded that a total consumption of 600 $g.d^{-1}$ of seafood would be reasonable and should be used in summing doses via actual pathways.

Both of the hypothetical pathways, deep-sea fish and plankton, should be considered separately and the doses not added to those derived for the actual seafood pathway in determining dumping rate limits wihout careful consideration of such a step.

The dumping rate limit for a combination of pathways irradiating a single critical group is obtained conservatively from

$$I,_{comb} = \frac{1}{L_i}$$

where:

L_{comb} is the dumping rate limit or N pathways combined

i refers to the pathway

N is the number of pathways leading to exposure of the same critical group

L_i is the dumping rate limit for pathway i alone.

Conservative assumptions and parameter values are used throughout the models leading to the critical group dose assessment. This procedure obviates the need for specific additional safety factors to be applied to the resultant dumping rate limits. Lower limits might, however, be set by introducing safety factors, or through the process of optimization, and these would then correspond to the 'authorized limits' set by national authorities.

See text.

5.7 Ingestion Pathways

The limiting dumping rate for any radionuclide from ingestion is dervied from the appropriate concentration in sea water as follows

$$L_{ij} = \frac{A}{K_{ij}\ G_i\ Q_i}\ \mathrm{Bq.a^{-1}}$$

where i refers to the pathway and j to the raionuclide

and K_{ij} is the radionuclide concentration in sea water corresponding to the unit dumping rates of the radionuclide ($\mathrm{Bq.L^{-1}}$ per $\mathrm{Bq.a^{-1}}$)

Q^i is the consumption rate ($\mathrm{kg.a^{-1}}$)

G_i is the concentration factor in the ingested material for the radionuclide ($\mathrm{L.kg^{-1}}$)

A is the ALI for the radionuclide ($\mathrm{Bq.a^{-1}}$).

(1) Seafood pathways. Consumption rates for seafood were chosen on the basis of experience of high rate values in a number of countries in which seafood forms a large part of the diet of communities which are often involved with seafood production. These consumption rates and the appropriate concentration factors are derived primarily from knowledge of coastal and continental shelf products. The fish pathway covers all mobile fish species, whether pelagic or benthic. No deep-sea fish pathway exists at present and thus there is no direct evidence on which to derive consumption rates. The deep-sea fish pathway is still considered hypothetical and assessment is retained primarily for completeness.

The molluscan pathway is representative of sessile filter feeders and grazers and is characterized by relatively high concentration factors for non-conservative nuclides. Seaweed is eaten directly and may also be used to produce food additives. The crustacean consumption pathway is intended to embrace similar organisms and could include krill.

(2) Other ingestion pathways. Certain other ingestion pathways may be identified. These pathways may be subdivided into two categories: those of a more primary (direct intake) nature, and those related to indirect intake. The second group is related to such activities as the use of seaweed and other marine products as fertilizers, rainfall on to soil and crops and the use of rain as a source of drinking water. The primary pathways are the desalination of sea water for use as drinking water and the evaporation of

sea water to produce salt for human consumption. In addition, the ingestion of beach sediment was considered as a pathway.

For some groups, desalinated sea water may form the only source of drinking water and an intake rate of 2 $kg.d^{-1}$ has been adopted. The concentration factors in this situation are taken as unity for tritium and 10^{-3} for all other radionuclides, except those with compounds of a volatile nature in sea water (e.g. C, Sc, I, Po) for which a concentration factor of 4 x 10^{-3} is assumed. These numbers are based upon at least a two-stage desalination plant where a decontamination factor of $3x10^{-2}$ and $6x10^{-2}$ per stage exists for non-volatile and volatile radionuclides, respectively.

For the sea-salt consumption pathway an intake rate of 3 $g.d^{-1}$ has been taken. The concentration factor for tritium in this pathway is assumed to be unity, since tritiated water will be evaporated in salt manufacture. Note that a very small amount of tritium will remain in the water of crystallization of sea-salt. For non-volatile and volatile radionuclides, the concentration factor is taken to be 3 x 10^{1}. It is recognized that some volatile radionuclides will be lost in the evaporation; however, this has been ignored to retain a conservative approach.

(3) Other pathways investigated. An evaluation was made of the importance of the ingestion of beach sediment, particularly by children, as an important pathway. Ingestion of such sediments is expected to become more important than the inhalation of resuspended sediment if the ingestion rate exceeds 2.1 $g.a^{-1}$. In addition, the quantity of sediment that would have to be ingested for the beach ingestion pathway to attain the same radiological significance as the adult seafood pathway was determined. Sand concentration factors were deduced from the available sediment K_d values (which relate to fine sediment) on the basis that the K_d values for sand are generally only 10 of those for fine sediment. The calculations were done for several radionuclides with long radioactive half-lives and emphasis was placed on those exhibiting high K_d but low concentration factors. The necessary sand ingestion rate varies as much as from 10^5 $g.d^{-1}$ for ^{99}Tc and 700 $g.d^{-1}$ for ^{239}Pu down to 12 $g.d^{-1}$ for ^{232}Th. the conclusion was that the ingestion of sand does not rank as a pathway comparable to the consumption of seafoods, with the possible exception of thorium.

A second problem considered was that of an ingestiong pathway for sediments that may occur when areas of marine sediment are reclaimed for agriculture; K_d values for fine sediment were adopted for this purpose. A comparison was therfore derived between the dose at unit concentration in sea water between the combined seafood pathway and a vegetarian diet of 500 g.d green vegetables. It was assumed that there was no change in concentration as marine sediment was converted to soil: this provides a safety margin which strengthens the conclusion that this pathway ranks much lower than marine seafood consumption.

Calculations were performed for Pu, Tc and Cs. Consumption rates for vegetables as opposed to seafoods by factors of at least more than 1000, 200 and 20, respectively, would be required for the two pathways to attain equal importance.

5.8 Pathways Involving External Exposure

A second group of pathways that sould be considered is that involving external exposures. The dumping rate limit based on external exposure is given by

$$Lij = \frac{D}{K_{ij}E_{ij}T_iF_i}\ Bq.a^{-1}$$

where i refers to the pathway and j refers to the radionuclide

and D_j is the appropriate dose limit ($Sv.a^{-1}$)

K_{ij} is the radionuclide concentration in sea water or on the sediment corresponding to the unit dumping rate of the radionuclide ($Bq.L^{-1}$ per $Bq.a^{-1}$)

E_{ij} is the dose factor which gives the dose rate as the result of unit radionuclide concentration in sea water or on the sediment ($Sv.h^{-1}$ per $Bq.L^{-1}$)

T_i is the occupancy time ($h.a^{-1}$)

F_i is a geometrical modifying factor

The inclusion of a large number of external exposure pathways, all with different occupancy times and geometric factors, would over-complicate the calculations. Only one sediment-related external exposure pathway was defined, the BEACH pathway, for which both whole body external doses and doses to the skin would be calculated. The following pathways are postulated:

(1) Swimming in contaminated sea water (SWIM)

For a critical group a total exposure time of 300 $h.a^{-1}$ seems a reasonable assumption. Calculation of a dumping rate limit should be done for total immersion in the sea water.

(2) Sailing on contaminated sea water (BOAT)

The contaminated sea can be considered as an infinte surface source. It seems prudent to consider enrichment of radionuclides in the surface microlayer and any consequent increase in the gamma radiation field, although this is likely to be minor. People working at sea may spend considerable time on board a ship or on a platform. An occupancy time of 5000 $h.a^{-1}$ should therefore be used. A geometrical modifying factor of 0.2 may be used to account for the fact that these people are generally well above the surface source and are shielded by the vessel or platform.

(3) Sediment-related pathways (BEACH)

The occupancy time selected for the BEACH pathway is 2000 $h.a^{-1}$, and concentration factors appropriate to silt are to be used in the dose calculations. It is unlikely that fishing gear (nets) and other marine tools such as anchors, drilling pipes, etc. will have a contamination level higher than the surface level. Therefore, the exposure due to handling these materials contaminated with gamma-emitting radionuclides can never be a critical pathway compared with the situation described in item (2). If sediments are contaminated with radionculides originally dumped into the deep sea, the handling of these sediments could result in an exposure because of dredging activities. This exposure will mainly come from the transportation of dredged spoils contained in the disposal vessel. The spoils can be regarded as a surface source. Man can consequently be irradiated by standing at the edge of this surface source. Operations of this kind as well as other sediment-related pathways such as lying on the beach and living on contaminated sediment are included in the BEACH pathway.

(4) Mining of deep-sea sediments (MINE)

It is assumed that the operations would mine for 250 $d.a^{-1}$ at a depth of 3000 to 4500 meters. Estimates for the size of a mine site are from 18,000 to 55,000 km^2 to be used for 20 years; 1.5 to 3 million tonnes of nodules would be processed per year. The nodules are assumed to be in a wet slurry. Since manganese nodule mining is expected to be a highly mechanized and remote operation, an occupancy time of 500 $h.a^{-1}$ is assumed. As with the deep-sea fish pathway, a lateral distance from the source is needed.

5.9 Inhalation Pathways

Inhalation pathways are to be dealt with on the basis of the following assumptions:

(1) The rate of human air respiration is 23 $m^3.d^{-1}$

(2) The concentration of atmospherically borne particles in the coastal environment is 10 $g.m^{-3}$

(3) The atmospherically borne particles comprise 0.25 $g.m^{-3}$ fine coastal sediment particles, 3.3 $g.m^{-3}$ dried sea-salt particles, and 6.6 $g.m^{-3}$ particle-associated water

(4) The concentration of atmospheric vapour in coastal air is 10 $g.m^{-3}$. In order to include inhalation pathways involving each of these atmospheric constituents, the following approach is used:

(a) Inhalation of radionuclides associated with sediment particles should be calculated using K_d values for coastal sediments.

(b) Inhalation of radionuclides associated with dry sea-salt particles should be calculated using a common enrichment factor of 3×10^1

for all elements. It is clear that for some subpathways this will introduce some inherent conservatism for volatile elements in sea water, but the magnitude of this is unlikely to exceed a factor of 2. The enrichment factor should be used with the surface sea water radionuclide concentration in the far-field model and the average surface concentration in the near-field model.

(c) Inhalation of radionuclides associated with the aqueous phase of atmospheric particles should be ignored. Enrichment factors (EF) for nuclides would have to exceed 10^6 for the aerosol water inhalation pathway to rival the magnitude of the atmospheric vapour inhalation pathway. Estimated enrichment factors are less than 10.

(d) Inhalation of atmospheric vapour should be calculated on the basis of the enrichment factor for single phase distillation of 3×10^{-2} for most elements. Exceptions to the use of this enrichment factor are the following elements:

(i) ^{3}H, which should have an EF of 1

(ii) ^{14}C, which should have an EF based upon equilibrium between CO_2 in sea water and atmospheric CO_2

(iii) Cl, Kr, Ar, Xe, I, Tc, Se, Te, Po and Hg, which should have EFs corresponding to equilibrium between atmospheric vapour and sea waer at STP.

In addition to the inhalation pathways considered above, a further pathway was identified and considered. This involves the resuspension of biological material (e.g. algae) from the sea surface into the atmosphere that has been observed in the Marshall Islands. However, transport and subsequent inhalation of radionuclides by this pathway would be covered by the resuspension and inhalation of dry sediment particles, with higher concentration factors for the radionuclides compared to algae. It was concluded that there was no need to consider this inhalation pathway separately, assuming that the mass concentration of dry sediment particles in the atmosphere ($0.25\ g.m^{-3}$) was sufficiently large to encompass the additional concentration of algae resuspended from the sea surface.

5.10 Ocean Regions of Water Concentration Calculations

To carry out dumping rate calculations it is necessary to specify the regions of the ocean in which water concentrations are to be calculated for each exposure pathway.

5.11 Calculations for Actual Pathways

For actual pathways, with the exceptions of mid-depth fish and mining, it is appropriate to use only surface water concentrations of radionuclides. This is because the mollusca, crustacea, seaweed and most of the fish currently consumed all originate from surface waters. In addition, all the external exposure pathways and inhalation pathways, except for mining, are related to man's immediate environment. In the case of mid-depth fish, it is appropriate to calculate a water concentration for a depth typical of those in which the fish (which are caught at present) live. A depth of 1500 m was therefore selected. For the mining pathway, the ocean region needs to be selected on the basis of the areas in which mineral resources are located. Since dumping sites are chosen outside areas with significant resources, it is not appropriate to calculate extreme near-field (i.e. dumpsite) bottom water concentrations for this pathway.

5.12 Calculations for Hypothetical Pathways

Plankton is, at present, viewed as a hypothetical pathway only, and no firm basis could be found for representing it within the assessment of dumping rate limits. Because of this the results of calculations should be dealt with separately. If they should indicate a limiting constraint such as to influence decisions on grouping, further consideration will be necessary before any binding decision is taken to adopt them. Surface water concentrations as such suffice for these calculations.

It is clear that for the deep-sea fish pathway, bottom water concentrations should be used. However, specifying the lateral distance from the dumpsite at which the concentrations should be calculated is a more difficult matter. The minimum region required to supply food to sustain a yield of one demersal fish per day has been estimated to be of the order of hundreds of square kiometre of the deep-ocean floor. For a sustainable fishery the area would be very much larger and would be greater than the dumpsite area assumed in the ocean models. Thus, while it is not inconceivable that fish living immediately above a dumpsite could be caught and consumed, it is extremely unlikely that members of critical groups would obtain all their annual intake of deep-sea fish from such a region.

5.13 Dose Conversion and Dose Response

Dose is estimated to critical individuals. Dose response is not included. Requires code coupling.

TABLE C-3. PATHWAYS AND MODES OF EXPOSURE

Pathway	Symbol	Mode of Exposure
Actual Pathways		
Surface fish consumption	FISH-S	Ingestion
Mid-depth fish consumption	FISH-M	
Crustacea consumption	CRUST	
Mollusc consumption	MOLL	
Seaweed consumption	WEED	
Salt consumption	SALT	
Desalinated sea water consumption	DESAL	
Suspended airborne sediments	SED	Inhalation
Marine aerosols	EVAP	
Boating	BOAT	External irradiation
Swimming	SWIM	
Beach sediments	BEACH	
Deep-sea mining	MINE	External irradiation/inhalation
Hypothetical Pathways		
Deep-sea fish consumption	FISH-D	Ingestion
Plankton consumption	PLANK	

APPENDIX D

DECISION-AIDING METHODS FOR COMPARING DISPOSAL OPTIONS

1 INTRODUCTION

Formal decision-aiding approaches are orderly, systematic procedures for making comparisons among complex alternatives. They clarify the structure and sequence of a decision process or a comparison by disaggregating it into smaller sub-decisions or evaluations that are easier to make.

Decision-making theory distinguishes between descriptive theory, which is concerned with how people actually reach decisions, and normative or prescriptive theory, which describes how people should reach decisions under assumptions about rational behavior. This section is devoted to the latter, although some understanding of the former is useful in predicting public responses (mostly objections) to formal approaches.

Any basic discussion of decision making should include mention of an important difference between good decisions and good outcomes. Writers on decision-aiding methods make a strong distinction between the two.[1] A good outcome is a state of the world that we value above others. A good decision is one that is logically consistent with the alternatives we know about, the information we have, and our preferences for the things involved. Under uncertainty, a good decision can lead to a bad outcome, and visa versa.

It is failure to understand this distinction that causes decisions to become bogged down in "but what if?" types of questions. No decision-aiding method can assure the quality of an outcome. All we can do is play the odds and attempt to include in our decisions resilience for responding to unfavorable outcomes. Holling cautions:[2]

> [Analytical decision-aiding techniques] "are typical of the search for an optimal solution. It usually comes as a shock to those nurtured in this perspective that complex living systems have not organized themselves in accordance with this principle,...ecological systems sacrifice efficiency for resilience....Evolution shows that their fail-safe strategy is eminently suited to a world which is inherently unpredictable at certain times."

Wildavsky extends these thoughts:[3]

> "In the past we relied on resilience. We anticipated the worst and we relied on a general capacity to be flexible to respond. By attempting a politics of anticipation, we destroy any potential we have for growth and for group accommodation."

There can also be large differences between "good" decisions based on some set of technical criteria and "satisfying" decisions, ones that makes people feel they have done the right thing. This arises in part from fundamental inabilities of formal approaches to deal adequately with subjective responses. Subjective criteria may be left out because they are not understood, are thought not to be important or appropriate, or for lack of a way of including them that is consistent with the formal method. Or they may be included, but evaluated in ways that produce the "wrong" answer, one that fails to capture the proper values -- either because of inadequacies in the elicitation or because peoples' subjective probabilities or values are based on misinformation and lack of understanding, so that the "real" answer doesn't match their understanding.

There may also be feelings that some things should not be subjected to this kind of analysis. Human lives, for example, should not be for sale. But a comprehensive analysis might show that the cost of saving a human life in a particular circumstance is unreasonably high given other lives that could be saved with the money. The formality of the decision process therefore might force people to face what seems in this context to be a reasonable judgment that they otherwise think should not be made.

2 DECISION-MAKING THEORIES

Merkofer outlines the three distinct, internally consistent theories of normative decision making that are most widely accepted: cost-benefit theory, decision theory, and social welfare theory.[4] Other, less accepted theories are available.[5] The following overview of these theories is abstracted directly from Merkofer. Major characteristics of the theories are summarized in Table D-1.

Cost-benefit theory is based on the premise that alternatives should be selected through a systematic comparisons of advantages (benefits) and disadvantages (costs) arising from estimated consequences of the choice. The theory does not include a decision maker with special responsibility for deciding what is "best." Instead, "best" is defined in terms of an "efficiency criterion" calculated as a function of total costs and benefits. Incommensurable costs and benefits are aggregated as dollars through the concept of individual "willingness to pay" to avoid or incur them. In the absence of other information, it is assumed that market prices reflect willingness to pay. Thus cost-benefit theory attempts to maximize social utility by maximizing the aggregate economic value of goods and services consumed by all individuals.

Although cost-benefit theory is appealing, Merkofer points out that it has several troublesome implications. Most important is that use of aggregate net benefit means that cost-benefit theory considers only total societal welfare, without regard for the distribution of that welfare within the society. Thus, through a higher willingness to pay by wealthier individuals, the theory can produce a redistribution of resources from the poor to the rich, so long as the rich gain more than the poor lose.

The general concept of willingness to pay also has deficiencies. Market prices are not always good indicators of willingness to pay, and the prices reflect only the preferences of the purchasers of the product. In addition, quantifying willingness to pay for impacts that are not traded in the marketplace is exceedingly difficult and highly uncertain.

Other miscellaneous assumptions are also troublesome. The relationships between money and welfare (utility), for example, is assumed to be linear and the same for everyone, and the current distribution of income is assumed to be an equitable distribution of power.

Decision theory describes how individuals should make decisions in the face of uncertainty. The axioms of decision theory are based on principles that define "rational behavior" in mathematical terms. These lead to two results that are central to the method. First, that a decision maker's preferences can be encoded in terms of a "utility function" representing a scaling of the values assigned to outcomes, including capturing attitudes toward accepting risk (see von Neuman and Morganstern, 1947[6]). Second, that a decision maker's preferences for alternatives can be measured by the expected utility of their outcomes (the sum of the utilities of all outcomes weighted by their probabilities). The preferred alternative must have the highest expected utility.

Decision theory contains a number of implied assumptions that may be questionable. Most important of these are:

- The decision maker accepts the axioms of rationality on which the theory is based;
- All possible events and all significant consequences can be enumerated in advance;
- Meaningful subjective probabilities and utilities can be obtained and assigned to them; and
- Disparate outcomes of concern to a decision maker can somehow be made comparable.

The first assumption is demonstrably incorrect in many circumstances, especially when decisions involve large amounts of money or differences among probabilities are small. Others may be correct under ideal circumstances but not under more complicated ones.

Finally, decision theory is focused on individual decision making. It does not lend itself well to collective decisions. How individual decisions should properly be extended to collective problems remains an open question.

Social welfare theory is based on the premise that societal decisions should arise from a rational synthesis of the preferences of all affected individuals. It is therefore concerned with finding decision rules or procedures by which individual specified preferences can be incorporated into the decision process. There are various lines of research in the field; the relevant line for this report attempts to derive ways of aggregating individual preferences into a societal utility function.

Individual preferences can be expressed cardinally or ordinally. Cardinal preferences contain quantitative statements about the relative strength of preference of one outcome over another (e.g., A is twice as good as B); ordinal preferences contain qualitative statements about the direction of preference (e.g. A is better than or equal to B). Because cardinal expressions of preference are especially vulnerable to subversion through strategic misrepresentation, early efforts focused on whether or not ordinal expressions of preference can be combined. It has now been shown that it is not possible to quantify a group's preference structure using ordinal preferences in a way consistent with some fundamental assumptions of rationality. And even if it were possible to find an optimal social decision-making rule, it is not clear that individuals would reveal their true preferences without strategic bias, so decisions would inevitably be suboptimal because they would be based on incorrect information.

Researchers have therefore been forced to rely on cardinal preferences, despite the difficulty in assessing them. These utility functions are of the kind used in conventional economics, which measure preferences under certainty, or those used in decision analysis as described above, which measure preferences under uncertainty .

Several sets of postulated properties for aggregating individual preferences have been explored. Most have specifically been aimed at justifying use of a linear combination of individual preferences. That is, group functions should be weighted averages of individual functions. Harsanyi obtains this result by assuming that group preferences satisfy the same axioms of rationality as individual preferences and that the group will be indifferent between alternatives if all individuals are indifferent.[7,8] Harsanyi also obtains it by assuming "impersonally situated" rational decision makers who face equal changes of winding up in the shoes of anyone affected by the decisions.[9] Kirkwood obtains it by adopting Pareto-efficiency for aggregating individual preferences -- that is, if one alternative is better than another for at least one individual and no worse for any other, then that alternative must be preferred by the group.[10] Keeney and Kirkwood find three assumptions required to provide a linear group utility function are:[11]

- Group utilities depend only on individual utilities;
- If everyone has the same utility function, then the group function must be a scaled version of the individual function; and
- If all but one of a group are indifferent between alternatives, the group preference should match the preferences of the individual who is not indifferent.

When Keeney and Kirkwood revise the second assumption to state that group preferences between alternatives should depend only on the preferences of individuals who are not indifferent, then they obtain a multiplicative form for the group function.

A major problem with social welfare theory is the need to assess, or assume, individual utility functions. First, utility functions in general assume that individuals are "rational" in the sense that they try to behave in accord with a set of axioms defining rationality. In general, this in not the case. Even when individuals try to be systematic in their thinking, they tend to engage in strategic behaviors intended to influence decisions in their favor. So it is unlikely that individual utility functions can successfully be measured for the large number of persons required. It is also unlikely that all persons involved could be identified.

Another problem is aggregation of individual utility functions. The aggregate function will be some weighted combination of individual functions. How should the weights be assigned? All opinions are probably not equally good. For technical problems that are not well understood by all, it is especially likely that there will be large differences between quality of opinion (level of information and understanding) and political power. Should weights then be based on political power alone? This can produce happier people and technically weak decisions.

Merkoffer presents an example of an application of social welfare theory in which eight different aggregation and weighting schemes were used to test the effects of this stage of analysis on results.

3 DECISION-AIDING APPROACHES.

Practical applications of the decision-aiding theories include a wide variety of procedures for dealing with all aspects of analysis. Most are sufficiently flexible to accommodate a great many variations as required. Merkofer describes several classes or types of applications that are commonly discussed in the literature.

The two basic types of cost-benefit analyses are a Paretian approach and a decision-maker approach. The Paretian approach attempts to produce a purely objective decision model. It is relatively uncompromising in maintaining objectivity and using market values for goods and services. Some practitioners evaluate only those consequences having market values, stating that results represent purely economic efficiency and are regarded as only one of many social welfare inputs necessary for decision making. The decision-maker approach attempts to include all issues of importance, not all of which have market prices. Non-market goods and services are included as personal values and judgments of the decision makers. Although, in theory, the decision-maker approach should be based on values and judgments of decision makers, Merkofer states that in practice, most have not involved the decision makers at all, but have relied on assumed value functions generated by the analysts.

Decision analysis methods applied to social decisions can diverge somewhat from the basic theory, which rests on an assumption of a single decision maker. The simplest application assumes that a single individual, or a group with consistent views, has responsibility and authority for making public decisions. In such a case, the group decision reduces to an individual decision and no modifications are required. A variation on this is assumption of a "supradecision-maker," a benevolent dictator who synthesizes a social utility function from the preferences of the individuals affected rather than his own preferences. On occasion, one or a few members of a group are taken as representative of the whole.

An alternative approach is to base the analysis directly on group judgments and preferences using special schemes for obtaining probabilities and utilities that represent society as a whole (social decision analysis). The analyst creates an explicit model of the decision including preferences of different groups, often called "stakeholders" and "players."[12,13]. Stakeholders are persons with a direct interest in the outcome of a decision; players are persons interested in influencing a decision. Note that players need not be large stakeholders in the ordinary sense, but may instead have interests arising out of other, related agendas.

Either the supradecision-maker or the group judgment approach can be applied as separate preference models that are combined into a societal judgment by a weighting function that expresses the relative importance of the various groups. The weighting function is supplied by the decision maker, and can be based on any criterion he considers appropriate, including size, political power, ability to slow the project through opposition, level of knowledge and skill, etc., depending on the objectives of the analysis.

In practice, in an open decision it may be politically difficult to use weights based on anything other than group size or group need, although for many formulations of project objectives, other weighting schemes make more sense. This is particularly true if the decision maker has some mandated responsibility that conflicts with public desires. Regulatory control of cigarettes is a particularly good example in which health effects are relatively clear (compared to other environmental problems), the government's responsibility to protect public health is clear, yet legislation is constrained by the political power of smokers and the tobacco industry.

All decision analysis approaches rely heavily on interviews with decision makers and technical experts to obtain the necessary information on probabilities and values. An important characteristic if these interviews is posing of choices in which respondents much indicate preference or indifference among clearly defined alternatives. The process tends to be dynamic, changing with the nature of the responses, and can require hundreds of judgments and decisions from each individual.

Because of this reliance on interviewing, the probabilities used on decision analysis are often subjective. That is, they are individual beliefs about the likelihood of events rather than observed frequencies. If the probabilities are elicited from technical experts, they may be observed probabilities, generalizations of objective probabilities (i.e. mental syntheses of observations), or best estimates of objective probabilities in the absence of specific data (guesses; experts call it "engineering judgment", which means a guess by someone who claims to know more). If probabilities are elicited from the decision makers, who may or may not also be experts, then they represent part of a model of the decision as seen by the decision makers. Depending on the knowledge and skill of the decision makers, the two need not necessarily be different.

Social decision analysis tends to rely more on normative arguments and less on interviewing individuals than classical decision analysis.

Applied social welfare approaches all attempt to identify decisions that maximize the welfare of a group as a whole through some aggregate social welfare function. Uncertainty in the consequences of a decision are dealt with through a centralized or decentralized process. In the centralized process, individuals provide value functions describing preferences for different time streams of outcomes or utility functions describing preferences for specified probability distributions over those time streams without regard for the actual outcomes of the decision. Estimation of consequences is done centrally using models. This separates the value judgments from estimation of consequences, reducing possibilities for strategic behavior. In the decentralized process, each individual is asked to specify his preference for each alternative outcome directly. The social welfare function then aggregates the preferences without regard for uncertainty in the consequences.

Although the basic theories presented above are relative distinct, in practice the applications borrow heavily from one another. The decision-maker approach to cost-benefit analysis often uses procedures from decision analysis or social welfare theory for representing preferences. Social decision analysis uses willingness-to-pay and efficiency arguments from cost-benefit theory to establish a social welfare function. Social welfare theory often use elicitation procedures from decision analysis to encode probabilities and utilities. This borrowing of procedures tends to blur the philosophical differences among the methods. Merkofer provides a "map" showing relationships among theories and overlapping of rationales for decision-making approaches (Figure D-1). With the exception of paretian cost-benefit analysis and clinical decision analysis, all have significant overlaps of perspective. Only decision-maker cost-benefit analysis has significant overlap in all three perspectives.

4 APPLICATIONS

All of the decision-aiding methods described above involve steps that separate information available on the implications of a decision from the values and preferences that are used for evaluating them (Figure D-2). The core of the process is the decision basis, a formal disaggregation of the pieces of the problem, including alternative choices at all stages, available information that connects decisions and outcomes, and preferences of the decision makers with respect to important characteristics of outcomes, all connected by feedback loops that redefine the statement of the problem in light of preliminary results.[14] Each part of the decision basis is narrowly focused and can be completed relatively simply. This helps to prevent decision makers from becoming overwhelmed with the enormity and complexity of the whole decision problem. And it makes the basis for the resulting decision transparent; all of the pieces -- the information and the judgments -- are open to examination.

Alternatives. The most important step in this process is determining the structure of the problem. Often, careful specification of the structure of a problem is can yield sufficient understanding to show the solution. Little further analysis need be done.[15] The first stage of determining the structure of the problem is specifying and characterizing alternatives. Note that the alternatives need not be end decisions or actions; they might start with selection of an approach to decision making. One can envision cases in which selection of a specific decision-making method would essentially complete the decision, in that all following actions and outcomes are prescribed by the method and the characteristics of the problem. A decision to select an alternative on the basis of cost as analyzed in a particular way might often be such an example, especially if there are large and uncontroversial differences in the costs of alternatives. Diversity of the alternatives included in an analysis depends largely on analysts' understanding of the problem, the structuring of the problem in general, and the effort devoted to creation of options.

von Winterfeld cautions that the initial stages of problem formulation are critical to the success of the whole.[16] Many problems can be viewed more than one way; some can be viewed many ways. Improper formulation can lead to sophisticated solutions to the wrong problem.

In general, because of greater interaction between decision makers and analysts, the decision analysis method generates a broader range of alternatives than other techniques. These often include contingency plans for responses to future information. Thus decision analysis can include desirability of one action over another caused by greater flexibility to respond appropriately to future information.

The alternatives to be considered can be readily apparent or derived as part of formulation of the problem using special tools developed by practitioners of decision analysis. The first stage of selecting alternatives is framing the problem, often the most difficult stage of all. Framing is difficult because of fundamental characteristics of human nature that tend to bury or obscure real problems under a protective facade of face-saving or decision-simplifying pseudo-problems. Also, in the absence of a systematic approach to developing statements of problems, preconceptions tend to interfere by generating subproblems based on preliminary, intuitive evaluations (blinders). So the first stage, deciding what must be decided, becomes much more complex than might initially be thought. Few formalized methods are available to assist in this stage. Mostly it requires understanding of the way people think about decisions.

Decision trees are fundamental to structuring a decision process, providing understanding of the relationships among decisions and outcomes and the flow of the process. These describe the sequence of decisions and possible random outcomes of decisions from beginning to end (Figure D-3). Note that the structure of a decision tree is not fixed by the problem. It is determined by decision makers' understanding of the problem and the way it should be approached -- the framing.

Decision Criteria. Cost-benefit approaches normally attempt to maximize net present value (the current value of a time stream of benefits minus costs). If probabilities are included in the analysis, then the decision criterion is expected net present value. Closely related criteria are: (1) maximizing internal rate of return, which is the value of the discount rate that makes the net present value equal to zero; and (2) allocating resources among alternatives so as to equalize marginal rates of return.

All decision analyses attempt to maximize expected utility. Whose utility is to be maximized depends on the approach; it may be a single decision maker representing himself or his organization, a supradecision-maker representing a group, or several supradecision-makers representing different stakeholder groups.

Social welfare approaches maximize group aggregate utility derived without reference to a decision maker. Social decision analysis attempts to integrate the approaches of cost-benefit theory, decision theory, and social welfare theory by appealing to efficiency arguments to justify net present value as a reasonable measure of social value. It is a value function, not a utility function in the sense of von Neuman and Morganstern, however, so is incapable of accounting for attitudes towards risk. Occasionally risk attitudes are included by converting the value function to a utility function using an assumed functional form and a "risk aversion coefficient."

Merkhofer describes an application of social welfare theory by Dyer and Miles[17,18] that is particularly appropriate to the problem of comparing different views on disposal of low-level wastes. The problem was to select a pair of trajectories past Jupiter and Saturn for two Voyager spacecraft given the requirements of 80 scientists in 11 independent experimental teams, each having different optimal trajectories. Rather than decide on a single approach, outcomes were compared when the problem was formulated as:

- A problem of collective choice based on rank sum (ordinal) judgments;
- A bargaining problem;
- A social welfare maximization problem; and
- A decision for a supradecision-maker concerned with maximizing group preferences.

Rank sums are one of the oldest and most widely used decision criteria for collective judgments. Each group assigns a preference rank to each alternative and the ranks are added to produce a total score for each alternative. This approach cannot satisfy Arrow's axioms for group decision making (see above), and it does not account for differences in the strengths of preferences among alternatives and groups.

The bargaining model was based on a formulation by Nash[19] and generalized to n players by Harsanyi.[20] In this formulation players may trade goods but not make side payments; they must bargain with what they have. This can be shown to produce a solution that is:

- Invariant with respect to utility transformations;
- Pareto optimal;
- Independent of irrelevant alternatives; and
- Symmetric with respect to players roles.

Following some manipulations and simplifying assumptions, the decision criterion was found to be to maximize the product of the teams' utilities for each alternative. This formulation does not necessarily have a unique solution and it may not be symmetric.

The social welfare model used an additive formulation in which total welfare was quantified as a weighted sum of team utilities for each alternative. The team utility functions were scaled so that the differences between the best and worst alternatives were the same for each team. The weights were then taken to be the relative importance of each experiment (team). Participants agreed that all experiments were not equally important, but were not willing to quantify the differences, so weights were all assigned to be 1.0 and a sensitivity analysis was done by assigning 2.0 to six experiments that all agreed were more important.

The supradecision-maker model used the preferences of the project manager who attempted to maximize group utility. A linear utility function was assumed for the whole, based on the utilities of each team for each alternative. The manager selected weights representing his evaluation of the quality (bias) of each team's estimate of utility and their relative strengths with respect to the best and worst alternatives, and weights representing the relative importance of each experiment to the project as a whole.

A comparison of the results of these four models showed substantial agreement. All models yielded the same three top-ranked alternatives, but differed with respect to their order. In spite of the agreement of the four models, selection was complicated by the fact that the individual teams disagreed significantly with the collective preferences. Only one team was in complete agreement with the collective judgment; one team gave all of the top three alternatives its lowest possible ranking.

Thus, regardless of the apparent agreement of the collective judgments, the actual selection remained highly controversial. This is characteristic of group decisions of this kind and illustrates the importance of using methods that are able to characterize and display the differences of opinion and preference that can generate strong political forces. No such method can produce a "perfect" solution.

The final decision in this case was reached by discussion and compromise, essentially through a sense of fairness that produced a willingness of the "high-weighted" winning groups to give up a little in order that the losing group would not be left with nothing. Such equity considerations were not included in the original formulation of the problem. The alternative selected was adjusted to the extent possible to regain some lost characteristics without giving up important gained characteristics.

Variables. All of the decision-aiding methods included here require some means of selecting and quantifying magnitudes of the outcomes on which the decision will be based and on which decision makers must place values. Special procedures for generating lists of variables have been developed to ensure they are complete, appropriate, and free of redundancies.

In decision analysis, Keeney and Raiffa,[21] for example, recommend that objectives be established first through questioning decision makers. These are then organized in a hierarchy that distinguishes their relative importance and generality. The resulting objectives hierarchy defines general objectives (ensure human health and safety) in terms of more specific, lower-level objectives (low incidence of accidents, exposure, etc.). The lowest level of the hierarchy is then used to establish quantitative measures of success in meeting objectives (predicted mortality, etc.).

In cost-benefit analysis, procedures are much less formal, relying on general literature and discussion with decision makers. Partly this is because of problems with placing monetary values on nonmarket costs and benefits and resulting confusion about what can and cannot be included in the analysis.

Values. Merkoffer divides the many valuation procedures used in decision-aiding methods into those based on market values and those based on utilities. Market-value-based procedures use established market prices or measures of willingness-to-pay, either expressed in responses to questions or "revealed" through analyses of behavior. These procedures assume that efficient markets insure that prices represent aggregate willingness to pay. If the decision under analysis will change prices, then the costs and benefits must be adjusted appropriately through estimated changes in aggregate consumer surplus -- the difference between what people paid on average and what they would have been willing to pay, but did not have to.

Utility-based procedures rely on direct questioning of individuals about relative preferences. The method normally uses exacting and time-consuming procedures involving assigning numbers on a scale of desirability, stating relative preferences for a series of conditions, or evaluating tradeoffs between increases and decreases in pairs of conditions. Value functions are assessed for consequences that are certain; von Neumann-Morgenstern utility functions are assessed for consequences that are uncertain. Utility functions include assessment of attitudes toward risk by having decision makers evaluate preferences between sure outcomes and lotteries having two outcomes with known probabilities.

The elicitation methods are difficult to design and administer, and the responses required of decision makers are often difficult to understand or to relate to familiar circumstances. If more than a few conditions (attributes) must be evaluated at the same time, it is nearly impossible to make all of the comparative judgments necessary. Analysts must, therefore, assume independence of preferences among attributes, which requires care in definition of attributes to ensure that the assumption is reasonable.

Because of these difficulties, there is not uniform agreement that the results so obtained are always meaningful representations of individuals' preferences.[22] At a minimum, elicitation of preferences must be done with great care by experts.

Value of Time. Because cost-benefit approaches tend to rely on dollar values, the value of time is treated in the standard way of discounting by opportunity cost. But other approaches include impacts lacking market prices, and the cost-benefit approach can assign dollar values to nonmarket goods and services for which the time discounting function may be significantly different. For example, many people are reluctant to transfer environmental problems to future generations that are not parties to current decisions. Others assume that future generations will have improved ways of solving these problems, so society is better off to defer them to the future. The first implies a negative discount rate; the second a positive rate. Provisions must be made for dealing with these different perspectives on the value of time.

5 CRITICISMS

Merkofer describes a broad range of criticisms of these methods that have appeared in the literature, ranging from fundamental inadequacies of the basic logic and assumptions to deficiencies of specific applications. The following summarizes those that are specifically related to social rather than individual decisions.

The most fundamental criticism arises from the impossibility of finding a socially optimal decision rule. If individual preferences are expressed as rankings (as in voting, for example) without an indication of the strength of their preferences, then there is no consistent decision-making rule that is not either dictatorial or completely arbitrary (Arrow's impossibility theorem).[23] Most approaches attempt to avoid the impossibility theorem by incorporation measures of strengths of preferences in social utility functions. These must either be estimated by a supradecision-maker or measured by asking a "representative" sample. Asking people what they want may not produce good answers, because what they say does not always agree with what they do. Also, the process is subject to subversion -- strategic misrepresentation or "second guessing" in an attempt to influence the decision in a favorable direction. Such misrepresentations are common. Even if preferences are measured accurately, there are problems that have insoluble structures because of an imbalance between willingness to pay for a good and willingness to accept payment for a bad. People are often not willing to pay to get what they want enough to compensate others for giving up what they have.

The basic approach of all decision-aiding methods is to decompose decision problems into manageable subproblems or subdecisions. Fundamental to decomposition is separation of information and preference. But there is no philosophical distinction between facts and values; normally facts shape values. Also, focusing on the decomposed details of a problem commonly oversimplifies it and may divert attention from more creative approaches to problem solving. Certainly decomposition requires relatively unnatural decisions that are outside most peoples' experience, and people think most naturally by analogy with past experiences. Some argue that the results are therefore spurious.

All decision-aiding methods have limited capability to account for irreversibility. Because the significance of an irreversible outcome depends on future events, and it is impossible to foresee and establish probabilities for all future events, the methods necessarily undervalue irreversibility. A decision that is good now can become suboptimal in the future as circumstances change and available information improves. Because of this, recoverability from mistakes often has relatively high priority among the general public (see discussion of resilience, above). All methods are weak in this area.

Sources of errors in application of decision-aiding methods arise from

- Omissions and inaccuracies;
- Difficulty of measuring costs and benefits;

- Bias in assessment, modeling, and analysis;
- Interference by analysts;
- Susceptibility to manipulation;
- Susceptibility to misuse and misinterpretation;

Decision-aiding methods can also have basic incompatibilities with established institutional, political, and social norms. There is a feeling, for example, that it is more important for the decision-making process to conform with democratic principles of visibility, participation, and fairness that it is to be logically "correct." The American system of regulatory and judicial review is based on an adversarial process that seeks to resolve conflict without violence, not to achieve objective truth. A formal approach that attempts to produce an integrated, comprehensive, balanced presentation of issues holds little appeal to advocates. Both sides tend to fear that introduction of such evidence will reduce the strength of their subjective arguments. In addition, most approaches concentrate power in the person or persons who provide value or utility functions to an extent that may be seen as undesirable by advocates.

It is also clear that decision-aiding methods do not account for institutional structures and existing governmental decision-making processes, which tend to be based on bargaining and responses to emergencies. Decision-aiding methods tend to become locked into a predefined "ritual" that lacks flexibility to adjust to changing circumstances.

Some argue that formal decision-aiding approaches are incompatible with existing social attitudes, preferences, and practices, so regardless of their potential value, will not gain real acceptance. Central to this attitude is a belief that society should not quantify environmental and health impacts, especially in such a subjective and inconclusive way. Although people routinely make such judgments implicitly by their actions, they do not want to be faced with the moral dilemma of an èxplicit judgment. They also dislike dealing explicitly with uncertainty, preferring, certain statements that are incorrect to uncertain statements that fail to support specific decisions. Ethical concerns include those about equity of distribution of costs and benefits across peoples and generations, and promotion of anthropocentric values. There is considerable question about who has a reasonable right to inflict how much on whom, including mankind on the natural environment.

6 EVALUATION

There is no formal decision-aiding approach for social decisions that is free from criticism. And the available approaches have different strengths and weaknesses. Merkofer and others claim that no formal decision-aiding approach is intended to replace decision makers. Instead, they present useful information in ways that can help decision makers. Choosing a decision-aiding approach, therefore, is a matter of choosing which presentation will be most helpful.

This view emphasizes the importance of the analyst - a facilitator whose skill and capabilities are of prime importance in determining the success achieved. Emphasis is shifted away from results toward the process, and how development of the structure and characteristics of the decision can help a decision maker understand the nature of his problem.

Merkofer characterizes the basic decision-aiding approaches with respect to five characteristics.

Logical Soundness. The soundness of the approaches depends on the persuasiveness of their underlying behavioral assumptions. Cost-benefit analysis will be less attractive for situations having considerable differences in the quality of relevant information. Decision analysis is less useful if there is no decision maker willing to provide or delegate authority for providing subjective value judgments. Social welfare theory has little appeal if the strengths of preferences of affected parties is not a central issue.

Completeness. Formal decision-aiding approaches have few restrictive assumptions on how problems should be defined, so completeness is largely determined by the available information and the skill of the decision makers, experts, and analysts. Cost-benefit analysis is less effective when little or no data are available for quantifying important uncertainties or outcomes. Decision analysis is less useful if decision makers define problems in habitual ways that overlook options or overemphasize certain factors. Social decision analysis, because of its broader representation of experts, can provide a more complete accounting of available information. Supradecision analysis and social welfare theory provide a more complete representation of social preferences through integration of preferences of stakeholders. Standard cost-benefit analysis provides no information on distributional equity. In general, decision analysis approaches allow a more comprehensive accounting of factors, because they are able to use data when they are available and subjective judgments when they are not. Social decision analysis is especially flexible in this respect.

Accuracy. Cost-benefit approaches are most accurate in assessing well-defined projects, but are less useful when there are significant market distortions or when information or preferences are changing rapidly. Decision analysis approaches rely heavily on subjective judgments which are necessarily less accurate than more objective measures. Social welfare theory and supradecision analysis approaches are subject to inaccuracies from motivational biases and strategic misrepresentation. Iteration and peer review, important components of any analysis, can help to increase accuracy.

Practicality. None of the decision-aiding methods discussed here was originally designed for application to social decisions involving risk. Each therefore shows some strain when forced to accommodate aspects of risk that are unlike those of the more traditional problems for which they were designed. Cost-benefit analysis is best suited to go/no-go decisions with immediate, predictable consequences and for which responsive markets exist and decision makers are well informed. When attempts are made to apply cost-benefit analysis to other kinds of problems, data requirements, especially placing dollar values on nonmarket goods and services, cause great difficulty. Decision analysis approaches have information requirements that, in theory, are more easily obtained, but they assume decision makers willing and able to represent society. And if such persons can be found, considerable time and effort is required of them.

There is a tradeoff in all methods between cost and accuracy. The more the analysis is restricted by cost, the greater reliance must be placed on the skill of experts and analysts. Consequently the less representative are the results of overall societal preferences.

Acceptability. Critics of formal decision-aiding methods worry that they place too much power in the hands of a technical elite. Cost-benefit analysis is the worst in this respect, providing little opportunity for stakeholders to contribute other than in defining the problem. It does, however, avoid a need for subjective judgments, which makes it seem more value-free, although in fact it is not. Methods that accept subjective judgments are more amenable to public participation, because anyone's perspective can be represented. In the end, however, only one person's perspective is represented, even if he is a supradecision maker representing the good of the whole. Decision analysis approaches highlight risk and uncertainty, which may make them less attractive to decision makers reluctant to admit the subjectivity and uncertainty inherent in their decisions. These approaches are also subject to suspicions that analysts are playing sophisticated number games.

7 SELECTION

Based on the above selection criteria it would appear that cost-benefit analysis is inappropriate to the comparison of land and ocean disposal of LLW because of its inability to deal easily with nonmarket goods and services and with large differences of opinion about their values. Classical social welfare methods are inappropriate because of difficulties in adequately quantifying group utility functions and inability to deal with large individual differences of opinion about values. So we are left with some form of decision analysis or a social welfare formulation that includes stakeholder groups. The alternatives available are:

- Classical decision analysis;
- Supradecision analysis; or
- Suprastakeholders.

The third alternative is intermediate between the supradecision-maker approach and the social welfare approach. Suprastakeholders' preferences must be combined by one of the methods for individuals in classical social welfare theory.

Classical decision analysis is appropriate to the comparison of land and ocean disposal of LLW only if there is a decision maker in EPA who can represent the agency and who's preferences can be quantified. This must be determined early in the process.

Supradecision makers can be selected to represent the country/world, or stakeholder groups in the country/world. Selection from among these alternatives depends on the extent to which EPA decision makers want to include preferences of others and to understand the political implications of alternatives in the comparison. But the success a single individual or small group could have in quantifying the preferences of the broad range of values represented in this decision problem is unclear.

Much depends on the nature of the alternatives that must be compared. The more one alternative tends to dominate others with respect to characteristics the decision makers care about, the less important are differences among decision makers' preferences. In such a case, the solution is "robust," remaining best over a broad range of preferences.[24] The larger the tradeoffs that must be made among characteristics decision makers consider important to the comparison, the greater the sensitivity of the solution to differences in preferences. And the greater the need for more politically sensitive decision-making methods.

For this reason, the suprastakeholder approach has greatest potential to illuminate the nature of the comparison, not only with respect to the technical issues involved, but also with respect to the political implications of differences of opinions and values.

Note that suprastakeholders not only have different values, they may also see (and frame) the comparison differently. So using a suprastakeholder approach may not be just a matter of eliciting preferences of various groups for a particular set of outcomes. Instead, the comparison should be examined from beginning to end for each group as represented by its supradecision maker. The results of each group analysis should then be combined in some way by an overall decision maker, perhaps through one of the methods from social welfare theory with supradecision makers substituted for individuals.

A distinction must be made by the overall decision maker among:

- How stakeholders would respond to a process and its outcome from the outside;

- How they would respond to a process as part of the team; and

- What outcome they would produce were they, alone the decision makers.

Stakeholders who are not part of the process engage in gamesmanship intended to place political pressure on the decision makers, and so to enter the process from outside. Their preferences and the decisions they make under this condition have goals that may be unrelated to the decision problem under evaluation. They tend to be aimed more at influencing the process than the outcome. Some care must be exercised in formulating a multiple-stakeholder decision problem to ensure that the proper perspective is maintained for each group. What that perspective should be depends on the overall decision maker's view of their position in the decision -- co-decision makers or exogenous political forces to which he must respond.

This stage of the decision process must address the problem of how to deal with differences in framing by stakeholder groups. If suprastakeholders are vastly different in their views of the problem, they will probably also be much different in the way they frame it. One group might, for example, reject out of hand particular classes of options at the initial stages of identifying candidates for consideration. Candidate options that might score well under other framings or value systems (other suprastakeholders), would not be included in the restricted subset. Some care must therefore be exercised to ensure that the comparisons made by stakeholder groups have a sufficiently common basis that the results are comparable among them. Overall decision makers can only compare the relative merits of the alternatives with respect to stakeholders' values if all of the alternatives are evaluated by all of the stakeholder groups (or suprastakeholders).

Even if the preferences and framings of the stakeholder groups are irreconcilably different, it nevertheless is important for understanding their political responses to try to see the problem from their point of view. An overall decision maker need not accept that point of view as valid (or valued) in the process, but it helps to know from where the responses come.

REFERENCES

1. Howard, R.A. Decision analysis: practice and promise. Management Science, 34:679-695 (1988).

2. Holling, C.S. The curious behavior of complex systems: lessons from ecology. In H.A. Linstone and W.H.C. Simmonds, eds. Futures Research: New Directions. Addison Wesley Press, Readding, MA, 1977. Quoted in Merkoffer, 1987.

3. Wildavsky, A. Statement of Professor AAron Wildavsky. Department of Political Science, University of California at Berkeley, Berkeley, California. In Risk/Benefit Analysis in the Legislative Process. U.S. Senate Committee on Science and Technology, No. 71. U.S. Government Printing Office, Washington, DC, 1980. Quoted in Merkoffer, 1987.

4. Merkofer, M.W. Decision Science and Social Risk Management. D. Reidel Publishing Co., Boston, MA, 1987.

5. Arrow, K.J. Alternative approaches to the theory of choice in risk-taking situations. Econometrics, 19:404-437 (1951).

6. von Neumann, J., and O. Morgenstern. Theories of Games and Economic Behavior. Princeton University Press, Princeton, NJ, 1947.

7. Harsanyi, J.C. Cardinal welfare, individualistic ethics, and interpersonal comparisons of utility. J. Political Economy, 63:309-321 (1955).

8. Harsanyi, J.C. Nonlinear social welfare functions. Theory and Decision, 6:311-322 (1975).

9. Hirsanyi, J.C. Mortality and the theory of rational behavior. Social Research, 44: 623-667 (1977).

10. Kirkwood, C.W. Pareto optimality and equity in social decision analysis. IEEE Transactions on Systems, Man, and Cybernetics, SMC-9(2):89-91 (1979).

11. Keeney, R.L., and C. Kirkwood. Group decision making using cardinal social welfare functions. Management Science, 22:430-437 (1975).

12. von Winterfeldt, D., and W. Edwards. Decision Analysis and Behavioral Research. Cambridge University Press, New York, NY, 1986.

13. Procter, M. Probing the core of controversy: issue management in the Office of Technology Assessment. Presented at the European Congress on Technology Assessment, Amsterdam, The Netherlands, February 2-4, 1987.

14. Howard, R.A., 1988.

15. von Winterfeldt, D., and W. Edwards. 1986.

16. von Winterfeldt, D., and W. Edwards. 1986.

17. Dyer, J.S., and R.F. Miles, Jr. An actual application of collective choice theory to the selection of trajectories for the Mariner Jupiter/Saturn Project. Operations Research 24:220-244 (1976).

18. Dyer, J.S., and R.F. Miles, Jr. A Critique and Application of Collective Choice Theory: Trajectory Selection for the Mariner Jupiter/Satern 1977 Project. JPL Internal Document 900-777. Jet Propulsion Laboratory, Pasadena, CA. February 1977.

19. Nash, J.F. The bargaining problem. Econometrica, 18:155-162 (1950).

20. Harsanyi, J.C. A bargaining model for the cooperative n-person game. In A.W. Tucker and R.D. Luce (Eds.), Contributions to the Theory of Games, 40:325-355 (1959).

21. Keeney, R.L., and H. Raiffa. Decisions with Multiple Objectives: Preferences and Value Tradeoffs. John Wiley & Sons, New York, NY, 1976.

22. Rowe, M.D., and B.L. Pierce. Sensitivity of the weighting summation decision method to incorrect application. Socio-Econ. Plan. Sci., 16:173-177 (1982).

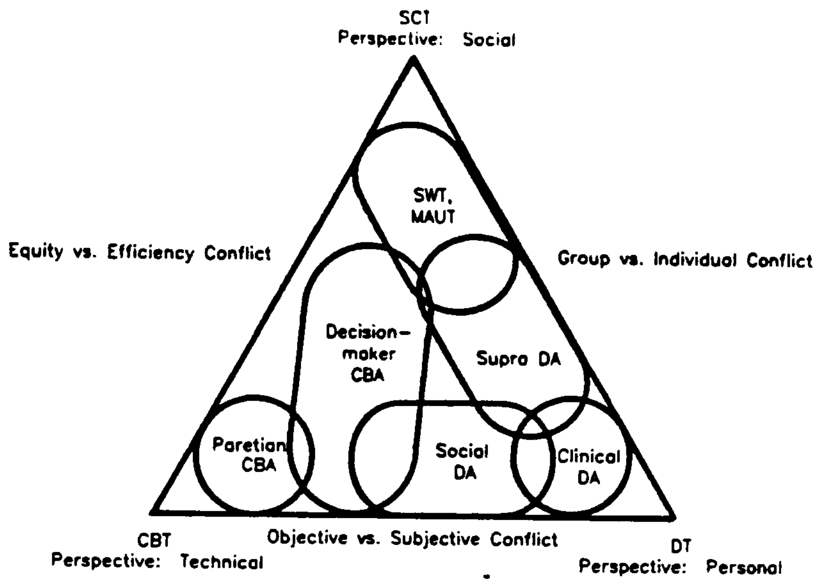
23. Arrow, 1951.

24. Rowe, M.D., and B.L. Pierce. 1982.

Table D-1. Comparison of decision-making theories.[a]

	THEORY		
	Cost-benefit	Decision	Social welfare
Intellectual roots	Engineering economics	Engineering, psychology, management science, economics	Welfare economics
Conceptual basis	Economic efficiency	Axioms of individual choice	Axioms of social choice
Method of analysis	Comparison of aggregate value of estimated consequences	Determination of logical implications of alternatives, information, and preferences of decision maker	Derivation of group decision from acceptable mechanisms for incorporating individual preferences
Perspective on value	Total monetary equivalent as determined by economic actors in a free market	Responsibility of decision maker, objective is consistency	Social preference derived from "equitable" synthesis of preferences of impacted parties
View of uncertainty	Objective characterization of environment	Subjective beliefs of the individual	Product of individuals coping with erratic environment

[a] Merkofer, M.W. Decision Science and Social Risk Management. D. Reidel Publishing Co., Boston, MA, 1987.



KEY:

SWT = Social welfare theory
MAUT = Multi-attribute utility theory
CBA = Cost-benefit analysis
DA = Decision analysis

Figure D-1. Implicit overlaps of rationales and perspective conflicts inherent in decision-aiding methods.
(Source: Merkofer, M.W. Decision Science and Social Risk Management. D. Reidel Publishing Co., Boston, MA, 1987.)

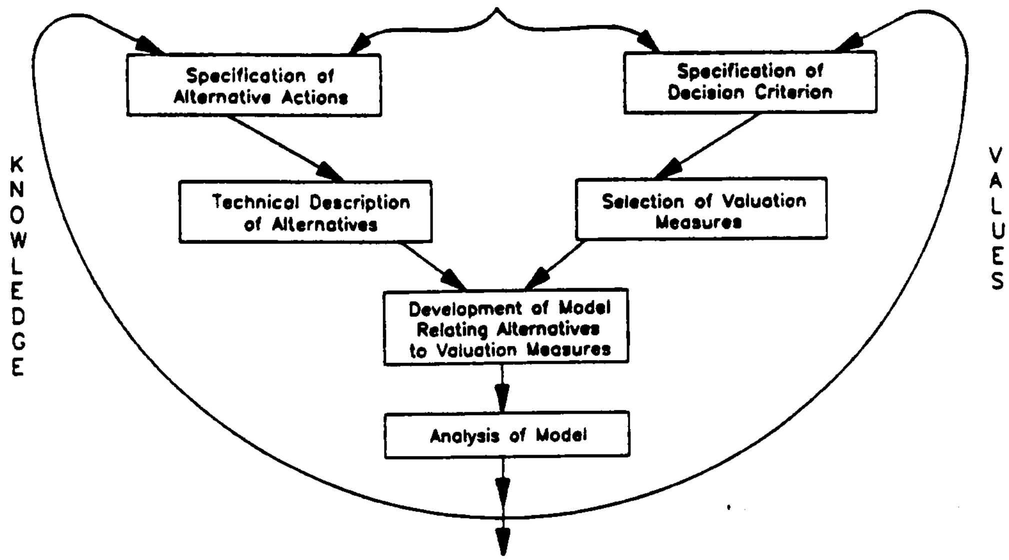


Figure D-2. Partitioning of decision-aiding methods.
(Source: Merkofer, M.W. Decision Science and Social Risk Management, D. Reidel Publishing Co., Boston, MA, 1987.)

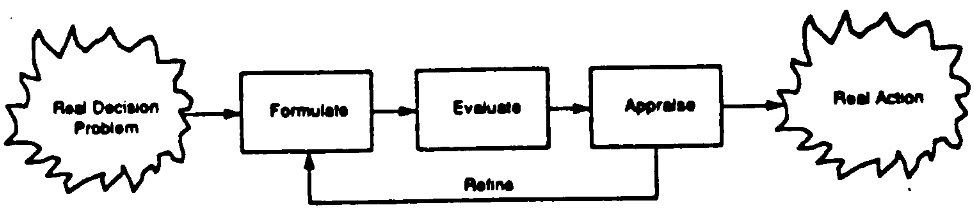


Figure D-3. The decision analysis process.
(Source: Howard, R.A. Decision analysis: practice and promise. Management Science, 34:679-695 (1988).