8. MANAGEMENT OF RADIOACTIVE WASTE IN CANADA

Low level waste (Carter 1987)

Canada's low-level radioactive wastes (LLW) have a wide range of physical forms and radionuclides, and are currently managed either by producers or by the Atomic Energy of Canada's Chalk River (AECL) Nuclear Laboratories (CRNL), which operates a national collection and management service for small producers. The processing and storage methods are generally well established. Substantial research and development is in progress for a gradual transition to disposal methods, including a shallow land burial (SLB) demonstration facility at CRNL. With a federal policy that encourages producers to propose disposal methods, the stage is now set for a transition from the current interim methods to long-term methods of LLW management.

Low-level radioactive wastes (LLW) generated in Canada broadly fall into (a) those produced by the Canadian nuclear industry (in the uranium fuel production and power generating stages of the nuclear fuel cycle; electric utilities with nuclear generating stations in Ontario, Quebec and New Brunswick; uranium refiners; fuel fabricators; which account for the major portion of the low-level wastes in Canada and (b) those produced by a large number (5100) of licensed radio-isotope users such as hospitals and laboratories. AECL's Chalk River Nuclear Laboratories provide a national fee-based radioactive waste collection and storage service for those institutions that produce only small volumes of wastes, (licensed users of radioisotopes and nuclear research and radioisotope processing facilities); and (c) a number of non-nuclear industries dealing with naturally radioactive feedstocks in their operations (abrasives manufacturing, specialty metal alloy production, etc.). Not included here, are the uranium mine and mill tailings, which are locally managed by the mining industry.

Table 50 Canada's Low-Level Waste Volume Projections to year 2025.

Canadian nuclear industry	(m^3)	%
Refining	65,000	18
Fuel fabrication	14,800	4
Utilities	156,500	42
Isotopes and research	61,200	16
Licensed users	12,900	3
Industries using naturally radioactive feedstocks	57,100	15
Total	367,500	100

These exclude about 1.2 million m^3 of wastes, primarily contaminated soils at several 'historic' sites, CRNL site and waste management sites of Eldorado Resources Limited at Welcome and Port Granby, Ontario. Some compaction of the wastes at the source is assumed, as is carried out by the producers normally. The Low Level Radioactive Waste Management Office (LLRWMO) of AECL is

spearheading analysis of the need and alternatives for establishing disposal facilities in Canada. LLW management in the Canadian nuclear industry has reached maturity in two important phases: in the interim management of the diverse waste sources; and in the technological research and development in support of plans for disposal of LLW.

Sources of low-level wastes

Technologies used in the various phases of LLW management share the common objective of safe containment of radioactivity. Waste properties differ widely across the industry and generally have been well characterized. The fuel production stages of the nuclear fuel cycle, which include uranium refining and fuel fabrication processes, yield uranium-contaminated materials and residues. Eldorado Resources Ltd, the federally owned refiner, produces the major component of these wastes, which are currently managed in storage facilities near the plant at Port Hope, Ontario. Wastes from Canada's two fuel fabricators, Canadian General Electric and Westinghouse Canada, are sent to CRNL for storage. Wastes in the power generating stages of the nuclear fuel cycle make up the major ongoing volume component of nuclear industry wastes. Ontario Hydro, which has a committed nuclear program of 13,600 MWe, is by far the major producer of these wastes; the other contributors are the provincial electric utilities of Quebec and New Brunswick. The wastes are classified as low and intermediate level wastes. Both these subcategories are non-heat-generating, and are hence 'low-level,' although intermediate-level wastes require shielding. Low- and intermediate-level wastes consist, essentially, of all radioactive wastes produced in CANDU nuclear generating stations (NGS), other than those contained in the irradiated fuel. These wastes primarily consist of housekeeping wastes, such as paper and plastic sheeting, temporary floor coverings, used protective clothing, rubber gloves and plastic suits, mopheads, rags and other cleaning materials, and contaminated hardware; spent ion exchange resins and filters from purification systems; and large irradiated and contaminated core components, arising from rehabilitation and retubing of reactors. These wastes are mostly contaminated with short-lived radionuclides, such as Co-60, Cs-137, Sr-90, and H-3, with a particular segment of the waste (resins) containing C-14, a radionuclide with a half-life of 5,730 years.

The nuclear research laboratories at the CRNL in Ontario, the Whiteshell in Manitoba, and AECL's radioisotope processing facility in Ottawa are the major contributors of the remaining wastes from the Canadian Nuclear Industry. These consist of contaminated materials from laboratories, maintenance and purification wastes from research reactors, and wastes from isotope processing. These are not altogether different from the utility wastes in radiological character.

Institutional and industrial wastes consist of a wide range of radionuclide materials, such as sealed sources used in industrial equipment such as gauges, industrial radiography cameras, and

static electricity eliminators; contaminated materials (i. e., animal carcasses, scintillation vials, liquids, filters, syringes, wipes and gloves from medical applications of radioisotopes); and residues from abrasives manufacturing or speciality metal alloy industries, which process raw materials containing naturally occurring (incidental) radionuclides. While the institutional wastes are handled by CRNL's national collection and storage service, incidental wastes from the industries are generally managed by the producers themselves.

Waste management technology

The major technologies in the management of LLW include processing, transportation, storage and disposal. Producers segregate wastes 'at the source' taking into consideration the physical and radiological properties of the waste, to facilitate the application of the above technologies.

Processing of wastes is undertaken to reduce the volume and/or produce a waste form more suitable for packaging, storage, and eventual disposal. Some 90 % of LLW is processible, either by mechanical compaction or incineration. Compaction results in a volume reduction ratio of about six, while incineration provides a ratio of about 75. Processing of LLW by incineration and baling has been adopted by Ontario Hydro and CRNL, the two major producers in the Canadian Nuclear Industry. Ontario Hydro has been operating a Waste Volume Reduction Facility (WWRF) at the Bruce Nuclear Power Development (BNPD) since 1977.

With waste sources that rapidly increased in number in the 1970s, due to an expanding nuclear program, Ontario Hydro put into service in-station waste management systems for collection, segregation, and packaging of wastes, as well as a centralized waste management site at the BNPD consisting of an incinerator, baler/compactor system, and a central maintenance facility that carries out laundering, decontamination, and other 'active' maintenance operations in support of nuclear stations. AECL has constructed a Waste Treatment Centre (WTC) to process and condition CRNL's LLW. The WTC is composed of an incinerator and baler for solid wastes, an ultrafiltration and reverse-osmosis system for the concentration of aqueous wastes, and equipment for immobilizing the ash and solids from the waste concentrates into a bitumen matrix. The goal is to produce a final-conditioned waste which is in a stable, compact, and leach-resistant form suitable for both storage and disposal. By combining several processes in a full-scale integrated system, the WTC serves to develop waste conditioning methods, improve the management of CRNL site wastes, demonstrate waste processing technologies, and generate performance and cost data for other Canadian nuclear facility owners.

Incineration

Ontario Hydro's nuclear program currently generates about 6000 m^3 unprocessed low-level waste per year, and this quantity is

expected to increase to over 8500 m^3/y by 1992. Approximately 65 % of this volume is classified as incinerable. The Ontario Hydro system, like the CRNL system, utilizes a controlled air batch-pyrolysis technique, in which the combustion air quantity is starved in the primary chamber to about 30 to 50 % stoichiometric. The pyrolysis effluent from the combustion chamber is then fully oxidized in an afterburner. The dry off-gas cleanup system consists of an off-gas cooling stage and a one-step filtration stage in a baghouse; no polishing filtration is employed.

Although the Ontario Hydro (F 1a) incinerator is a working prototype that has required modifications during its operating life, it has, nevertheless, become one of the most productive incineration systems in the nuclear industry. To the end of 1985, over 20,000 m^3 of LLW has been processed in over 55,000 operating hours. Waste with a contact dose rate of up to 0.6 mSv/h is incinerated. Typically, solid waste with a specific gross gamma activity of 0.02 to 0.08 GBq/m^3 has been processed. Incinerator ash, which has a specific activity ranging from 0.08 to 8 GBq/m^3 is 'dumped' into 2.5 m^3 rectangular galvanized steel containers, which are then placed in the storage structures. Contact fields on most of the ash containers are between 0.1 to 0.2 mSv/h. Radioactive emission experience with the incinerator has been very satisfactory, with particulate gamma activity on the order of 70 kBq released through the stack for each m^3 of waste burned.

CRNL's incinerator (F 1b), which also uses a starved-air batch pyrolysis process, is a more advanced version of the production unit operated by Ontario Hydro. It has improvements in control, process versatility, and the use of corrosion-resistant materials. It is designed to process batches of up to about 1,300 kg of solid waste in a nominal 24-h cycle. Particulate beta-gamma stack releases have remained less than 37 kBq per burn.

Transportation

Transportation of low-level waste is carried out in accordance with IAEA transportation regulations enforced by the Atomic Energy Control Board. Most wastes, such as the bulk LLW, contaminated soils, etc., qualify - depending on their radioactivity - either as LSA (low specific activity) or type A wastes. Waste materials with higher concentration of radioactive contaminants, such as intermediate level wastes, require transportation in accident resistant type B packages. The classification of transportation packages (as LSA, type A, or type B) is carried out in accordance with transportation regulations. The infrastructure is now available in the Canadian nuclear industry to design, test, and commission transportation packages for low-level wastes, and for radioactive materials with higher levels of radioactivity such as irradiated fuel and cobalt-60.

Storage

Two Canadian utilities (Hydro Quebec and New Brunswick Power) have local sites for management of LLWs. These utilities employ designs similar to the engineered storage facilities of Ontario Hydro and CRNL. Eldorado Resources Limited, the major refining industry, operates its own storage facilities a few miles from its Port Hope plants. These facilities primarily consist of above ground waste emplacement schemes or shallow burial. Industries using materials in production processes that are incidentally radioactive (e.g., abrasives industry) generally store the waste materials at the plant sites.

Ontario Hydro experience

Currently, all radioactive waste materials are stored at BNPD, in a retrievable manner, in facilities having design lifetime of 50 years. No radioactive materials are placed directly in soil; either in-ground or above ground engineered structures are used. The storage site consists of 19 acres (0.8 km^2) and a variety of storage facilities built on relatively impermeable glacial till deposits. Ontario Hydro has been developing the BNPD Radioactive Waste Operations Site for the last fifteen years. To date, 37,000 curies (as stored) of radioactive wastes are estimated to be stored at the site. Among the storage facilities are reinforced concrete trenches used for the storage of the low-level wastes, in-ground structures, called 'tile holes' (used to store filters and ion exchange resins that contain a higher level of radioactivity), including newer versions that employ borehole augering technology to allow faster construction, lower costs, and greater depths; two above-ground prefabricated, prestressed concrete superstructures, called 'low-level storage buildings', now being used for storage of low-level wastes with radiation fields less than 10 mSv/h and double-walled, above-ground reinforced concrete structures, called 'quadricells,' used primarily to store intermediate-level resins, with a secondary role of storing highly radioactive core components.

AECL experience

The CRNL facilities are located in elevated and well drained deposits of sand. The radioactive waste is generally placed above the water table, to reduce the likelihood of contact with water. Close to 100,000 m^3 of solid radioactive wastes are stored or buried at the CRNL property. Eighty % is LLW, 15 % is MLW and 5 % is HLW. The LLW is generally buried unprotected in sand trenches, well above the water table. Solid wastes with higher radioactivity are stored, retrievably, above the water table in engineered concrete structures, ranging in diameter from 0.15 to 6.0 m, and in depths of up to 5 m. Each structure is fitted with a removable, weatherproof shielding cap, and protrudes less than a metre above grade.

Future disposal facilities

The above methods of storage are considered interim in that at least some of the wastes will be radioactive beyond the time frame of storage and will require disposal. The CRNL have taken the lead in developing and demonstrating a disposal capability for LLWs in Canada. Three concepts selected for study by CRNL include 'improved sand trench' (IST) for wastes that need isolation up to about 150 years, intrusion-resistant 'shallow land burial' (SLB) for wastes that require isolation up to about 500 years and 'shallow rock cavity' (SRC) for wastes that need isolation for more than 500 years. Based on knowledge of the radiological characteristics of the stored wastes, it is anticipated that the bulk of the waste could be disposed of in the SLB Facility (F 2). The other two concepts are considered potential complements to SLB. The SLB is about 100 m long by 20 m wide by less than 10 m deep, with the top of the wall near the surface and the bottom above the water table. Once filled it will be covered with a self-supporting, water-shedding, concrete roof (and perhaps other water-shedding barriers), then buried under a relatively thick ground-cover to prevent erosion, and thus stabilize the topography. Continued engineered storage of LLW wastes is considered the essential ingredient in Ontario Hydro's plans.

Eldorado Resources Limited (ERL) have been evaluating disposal facilities for their currently stored refinery wastes and for their ongoing production of LLW. Near surface burial in glacial till, and intermediate-depth burial concepts in the local limestone geology, have been researched for application in the regions surrounding their Port Hope refining operations.

Responsibilities

Although the responsibilities of the provincial and federal governments in the area of low level-waste management is still a subject for discussion, some of the jurisdictional aspects are becoming clearer in Canada. The federal government has established the LLRWMO in Ottawa, as the agency to discharge federal responsibilities in the area. The federal government accepts residual responsibility for LLW, i.e., responsibility for the wastes for which no person or company can be held responsible. It has adopted the principle (Federal Policy on LLW, 1986) that the primary responsibility for the management of radioactive wastes, including disposal, must rest with the producers of such wastes, and that the costs of waste management should be borne by those benefitting from the activities responsible for the generation of wastes. One of the tasks undertaken by the LLRWMO is to establish, or to ensure the establishment of, low-level radioactive waste disposal facilities that could be used by institutions, such as universities and hospitals (small producers), on an ongoing basis. These low-volume producers are those who would otherwise be unable to establish their own facilities. The benefits from the nuclear industry are diffused throughout society, while the perceived detriments from waste facilities are local to host communities. The Federal Policy on LLW management recognizes that the ideal

democratic principle - that preference should be given to courses of action resulting in greater good for the greater number of people - is not widely accepted by residents who live near a proposed waste facility. Recent opposition from potential recipient (host) communities to relocation of contaminated materials/soils from past operations are cases in point. Although many factors (such as human health and safety, environmental protection, and general societal concerns) are taken into consideration by any proponent, it is absolutely essential that co-operation and participation of the public, and local and senior levels of government be sought in the necessary decision-making processes. In some cases, it is anticipated that an area that hosts a disposal facility may obtain 'offsets' for accommodating the facilities.

The producers are accountable for ensuring that the wastes are properly isolated over their hazardous lifetime. This could include the development of sole- or joint-use disposal facilities and sites. The federal government may accept residual responsibility as in the case of cleanup and disposal of historic wastes, wastes from small producers, or companies no longer in business and as in the long-term stewardship of disposal sites after they have been closed and the producer's responsibility has been terminated.

Storage of irradiated fuel (Frost 1985)

The characteristics of Ontario Hydro's fuel and at-reactor irradiated fuel storage water pools (or irradiated fuel bays, IFB) are described. With on-power fuelling of reactors, each reactor of >500 MW(e) net discharges an average of 10 or more irradiated fuel bundles to bay storage every full power day. The logistics of handling such large quantities of irradiated fuel bundles present a formidable challenge. The development of high density fuel storage containers and remote handling mechanisms and the use of several irradiated fuel bays at each reactor site have all contributed to the safe handling of the large quantities of irradiated fuel (IF). Routine operation of the irradiated fuel bays over a period of more than 20 years and some unusual events in the bay operation are described. It is concluded that the operation of Ontario Hydro's irradiated fuel storage bays has been relatively trouble-free despite the large quantity of fuel involved, and wet storage provides safe, reliable storage of irradiated fuel. Evidence indicates that there will be no significant change in irradiated fuel integrity over a 50 year wet storage period.

Description

Data on the type, liner material, size, fuel capacity and estimated fill date for the IFB's at Ontario Hydro's nuclear generating stations (NGS) are given in Table bellow.

Table 51 Irradiated fuel bays at Ontario Hydro's NGS.

Station	Type	Dimensions - m			C	ISD	BFD	LM
		Width	Length	Depth				
Pickering A/B	PIFB	16.3	29.3	8.1	93/158	1972/83	1994/95	E
	AIFB	17	34	8.1	214	1978	1994	E
Bruce A/B	PIFB	10	41	6	21/36	1977/83	1994/02	SS+E
	AIFB	18	46	9	352/330	1979/87	1994/02	SS+E
Darlington	PIFB	9.7	20.6	5	212	1987	1996	SS

C = capacity 1000's bundles, ISD = in-service date, BFD = bay fill date, LM = liner material (SS = Stainless steel, E = Epoxy)

The earliest stations, NPD and Douglas Point, had sufficient IFB storage capacity for the station life. The other stations (Pickering A, Pickering B, Bruce A and Bruce B) will need additional storage capacity beyond existing IFB's starting in the mid 1990's; Darlington will also need additional IF storage capacity in 1996. This paper will focus mainly on the Pickering and Bruce sites, as they alone account for over 90% of all irradiated fuel presently stored at Ontario Hydro's stations. The on-site IFBs are of two types: Primary bays (PIFBs) and Auxiliary or secondary bays (AIFBs).

Irradiated fuel is discharged directly from Ontario Hydro's reactors to the primary irradiated fuel bays for initial storage and cooling. The primary IFBs consist of two compartments, separated by a hydraulically operated gate. The two compartments are the receiving bay to which IF is discharged from the reactor directly [In this bay the IF is stacked in storage containers (F 2), possibly inspected, and later transferred to the second storage compartment known as the storage bay. There are facilities for canning defected IF, if required] and the storage bay [where the IF is stored in stainless steel storage containers called baskets, trays or modules (F 2)]. The receiving and storage bays generally have separate cooling and purification systems.

The basket is the container used to initially store irradiated fuel bundles in the Pickering A and Pickering B PIFB'S. The tray is used to stack IF bundles in the Bruce A and Bruce B PIFB's (and the Bruce A AIFB). The module is a newer container designed to store the IF at about 1.5 times the storage density in the IFB compared to baskets i.e., 2189 kg U/m^3 (for the module) and 1393 kg U/m^3 (for the basket). The module not only provides for a higher storage density but has also been designed as an IF container for irradiated fuel transportation, which reduces double-handling of the bundles. Thus, all Pickering A and B IF bundles will eventually be transferred from baskets to module storage to optimize the IFB storage capacity.

The AIFBS, consisting of a single compartment, are very similar to the PIFBs in function and operation. They are designed to receive and store fuel after its initial cooling in PIFBS, and provide additional storage capacity as needed. The AIFB's also have provision for receiving IF transportation casks. Because of the reduced radioactivity of IF bundles when transferred to the AIFB'S,

the bundles need less water shielding. Thus in the AIFB'S, the IF can be stacked closer to the water surface.

The IFB walls and floor are steel-reinforced concrete about two metres thick, and are either in-ground or above-ground structures. All inner IFB walls and floors are lined with either stainless steel or a fibreglass-reinforced epoxy compound, to form a watertight liner. In all the bays, water is circulated through cooling and purification circuits, which are described below. Methods used to control water purity are a combination of ion exchange columns, filters and skimmers. Ontario Hydro's IFB's use various liners and water purification systems. The choice of these components has been made on the basis of economics for the particular nuclear generating station concerned.

Cooling and purification systems

Cooling of bay water is achieved by tube and shell heat exchangers, with demineralized IFB water on the tube size and raw lake or river water on the shell side. As the irradiated fuel in the AIFB's has been stored for at least three months in the PIFBS, the AIFB cooling system capacity is proportionally smaller than that needed for the PIFBS. All IFB purification systems are designed to remove suspended and dissolved solids (both of which may be radioactive). The IFB purification system components and flow capacity for Pickering A and B, Bruce A and B and Darlington are:

Table 52 Irradiated fuel bay purification system capacity.

Station	Type	FR-l/s	E
Pickering A/B	PIFB	12/64	IX
	AIFB	65	F+IX
Bruce A/B	PIFB	76/76	IX
	AIFB	38	IX
Darlington	PIFB	92	F+IX

FR = flow rate, E = equipment (F = filters, IX = ion exchange)

In addition, water flows continuously through skimmers located at the water surface at intervals around the bay walls to remove any floating solids. Vacuum system type equipment is used at a frequency of once every 2 or more years to remove solids deposited on the bay floor and ledges. The AIFB purification system capacity in general is proportionally less than that of the PIFB purification system, because any leaching of radioisotopes from clad crud and defected fuel is at a reduced rate.

Chemical control is maintained in order to minimize corrosion of metal surfaces, e.g. fuel clad, stainless steel bay liner, storage containers, stacking frames, and handling tools, to minimize the level of radioisotopes in the water, and as a result reduce the radiation fields and radioiodine levels in the bay area, and to maintain clarity of the bay water for ease of bay operation. The water purity is maintained by using only demineralized make-up water and close chemistry control based on pH (5.5 - 9),

conductivity (<0.2 mS/m) and for the Pickering and Bruce bays, chloride concentration (<0.3 mg/kg). The temperature of the bay water is maintained at <32 C. This specification has been selected to prevent excessive stresses in the bay walls which could eventually lead to cracking of the concrete. Such a bay water specification also provides comfortable working conditions (i.e. air temperature and humidity) for personnel in the IFB vicinity.

Routine operation

The early operating experience gained at NPD and Douglas Point stations has provided a basis for the successful operation of the irradiated fuel bays at Pickering and Bruce sites. The early experience and the development of high density storage containers, inter bay fuel transfers, and remote handling mechanisms have all contributed towards meeting the logistics challenge of handling large quantities of IF bundles in an economical and safe manner.

Fuel handling

The fuel arrives underwater at the receiving bay of the primary bay, in pairs by conveyor (PNGS-A) or by a port (BNGS-A) mounted with a discharge mechanism. At PNGS-A, each pair of bundles is pushed via a ram into a basket. Once the basket is full, the bay gantry crane moves the basket to the storage area of the bay where it is stacked vertically on stacking frames (F 3) no more than six baskets high in order to maintain an effective water barrier for shielding. These stacking frames maintain a clearance of 45 cm between the bottom of the filled baskets and the floor to ensure that the flow of cooling water is uninterrupted and that the epoxy liner has adequate water shielding for radiation protection. Baskets are loaded in a similar fashion in the PNGS-B primary bay. However, once the baskets are filled, the bundles are transferred from baskets to the higher density module containers. The modules are then placed onto a stacking frame six modules high.

The auxiliary bay provides an interim storage facility to handle the irradiated fuel volume which is in excess of the capacity of the primary irradiated fuel bay.The PNGS-A primary bay is connected to the auxiliary bay by an enclosed corridor. For each irradiated fuel transfer operation, eight baskets of at least 4-year old irradiated fuel are selected from the PNGS-A primary bay and loaded underwater into the on-site cask. The bundle age restriction ensures acceptable radiation fields from the on-site shipping cask during transfer operations. After washing down, the cask is loaded onto the transfer vehicle (F 5) and moved through the enclosed corridor (200 m distance) to the auxiliary bay. The maximum rate of travel for the transfer vehicle is 0.25 m/s. Once the cask is lowered into the auxiliary bay and unloaded, a basket-to-module transfer is carried out. The modules are then stacked seven high. PNGS-B does not have an auxiliary bay.

At BNGS-A, the discharge mechanism lowers each pair of bundles onto racks which are placed on an indexing mechanism located below

the irradiated fuel discharge port. The crane operator transfers the bundles from the racks onto the storage trays. The full trays are then moved to the storage section of the bay where they are stacked 15 high. The trays of irradiated fuel remain in the primary bay for a minimum of 3 months. Approximately every 4 months, roughly 300 trays are transferred to the auxiliary bay. Trays are transferred two at a time on a cart which travels through a water-filled tunnel connecting the two bays. The trays are then stored in stacking frames. A program is currently underway to increase the storage capacity of the auxiliary bay by approximately 3.5 station years arisings of fuel. This involves the installation of new stacking frames which allow for closer spacing coupled with higher stacks, i.e. 37 trays high compared with 32 trays high with the previous configuration.

Cooling and purification

The normal operating temperature of the PNGS-A primary bay is 23 to 32 with two heat exchangers on-line. If the temperature exceeds the specified maximum value of 32 C, a third heat exchanger is valved in. Some fouling of the shell side of the Pickering and Bruce PIFB heat exchanger tubing has necessitated periodic chemical cleaning of the heat exchangers to restore their cooling capability. The frequency of cleaning is from one to five years.

Good chemical control has been achieved in both PNGS-A and BNGS-A irradiated fuel bays. A survey covering the 1978 to 1982 period indicated that the pH, Cl-, and conductivity levels have remained within specification most of the time, the only exception being a single conductivity measurement made in the PNGS-A PIFB which was 30% higher than specified. A high conductivity reading on the outlet of an ion exchange column indicates when the resin is spent. For the Pickering and Bruce PIFB'S, this occurs about once per year. With this close chemical control, the effect of bay water contamination on the long-term integrity of IF clad and other bay metal surfaces is considered to be insignificant.

Handling of defected fuel

Since the CANLUB fuel design has been in use, (i.e. since 1974), the overall PNGS-A and BNGS-A IF defect rate has been low, i.e. <0.1% (a total of 221 bundles have defected). During early operation, the canning (i.e. the storing of a bundle in a sealed cylinder) of defected fuel was carried out. As more operating experience was gained, canning of defected fuel has become a contingency rather than a routine operation, due to the minimal release of fission products from most defected bundles.

At Bruce NGS-A, an on-power defect detection system serves to identify reactor fuel channels containing defected fuel. Once identified, fuel from such a channel is removed at the earliest possible date. Each bundle pair is pushed into the discharge mechanism and kept there while air from the mechanism cavity is purged past a gamma detector to identify the defect bundles. The

suspect bundles are then transferred to a tray in the normal fashion. This tray is segregated from the rest of the irradiated fuel until the suspect bundles on it can be inspected. After inspection, all intact bundles are returned to normal storage. Defected bundles are stored in a special location in the bay and, depending upon the severity of the defect, some may be canned.

Pickering-A PIFB has IF canning facilities but with the excellent fuel performance, no IF bundles have been canned since 1974. The plan for PNGS-B is not to send any known defected fuel to the IFB until the defected bundle has had 2 to 3 days to cool and allow fission products to decay while held temporarily in the fuel handling systems.

Unusual events

In light of the excellent overall performance of underwater IF storage, operational problems experienced at the IFB's have been minimal. Two unusual events which have occurred are described below.

Heat exchanger tube fouling

Pickering NGS-A : During the summer months of each year from 1975, it has been difficult to provide sufficient cooling of primary bay water to maintain the temperature in the 23 C to 32 C operating range, even with two heat exchangers, HX1 and HX2, on fine. If the bay temperature exceeds 32 C on a regular basis, there is a risk of minor damage to the concrete walls. In 1979, a third heat exchanger was installed to allow inspection of HX1 and HX2. The latter were both found to be seriously fouled. Chemical cleaning of HX1 and HX2 with 10% formic acid resulted in the removal of 50 kg of deposit from each heat exchanger (the tubing area is about 365 m^2 per heat exchanger). The deposit fouling the heat exchangers on the shell side was a mixture of calcium carbonate, iron oxides and silica, with an approximate thickness of 1.5 mm. However, a post-cleaning inspection revealed that although the straight legs of the tube bundle were effectively cleaned, the U-bend region was not. Formic acid cleaning was used again during 1981 and 1982 with similar results. Although most of the calcium carbonate was removed, silt and mud deposits still remained in the U-bend region. Laboratory tests to identify a more effective cleaning solvent resulted in a recommendation to use ammoniated citric acid solution. This method will be incorporated in the next heat exchanger cleaning operation.

Bruce NGS-A: In late June 1980, the primary bay water temperature rose to approximately 37 C. With HX (heat exchanger) 2 and HX3 operating with maximum cooling water flow, HX1 was valved in to cool the PIFB back to below 32 C. Fibre optics inspection of the shell side (raw lake water) of HX3 showed the tube nest to be solidly blocked with deposits. The composition of the deposits consisted of calcium carbonate, iron oxide, and silica. HX1 was found to be similarly fouled. In October 1980, HX3 and HX1 were

chemically cleaned with inhibited 10% formic acid followed by a neutralizing solution. After cleaning, the tubes were visually inspected revealing that only a very thin deposit remained. Thus the cleaning method was successful. A total of 140 kg of calcium carbonate and 63 kg of iron compounds were removed from the two heat exchangers (the shell side tube area is about 555 m^2 per heat exchanger). The tube surface of HX2 was inspected and found to be clean. Presently, there has been no further need to repeat the cleaning. However, a program to routinely monitor the cooling capability of the heat exchangers has been implemented.

Use of hydrazine to reduce volatile iodine levels

During 1972, PNGS A experienced a high fuel defect rate caused by the initial fuel management scheme. Upon discharge to the primary IFB, the defected fuel released sufficient quantities of iodine to generate high airborne iodine activity. There were no incidents of high radiation exposure of personnel. Tests indicated that the addition of hydrazine to the IFB water reduced the oxidized forms of radioiodine and led to a significant reduction in airborne iodine activity. It was also observed that hydrazine effectively reduced the release of radioiodine under transient conditions when fresh defected fuel bundles were discharged into the bay.

Actual tests conducted in IFB water indicated that a decrease of airborne I-131 activity by a factor greater than seven was observed 15 minutes after hydrazine was added to the receiving bay (to give 125 mg/kg hydrazine) and to the storage bay water (to give 5 mg/kg hydrazine). It was also confirmed that hydrazine is not rapidly decomposed by atmospheric and dissolved oxygen at the temperature and chemical conditions in the bay water. It took about 48 hours for almost all the hydrazine in the bay water to be decomposed. Hydrazine also has an advantage over many other chemicals in that its main reaction with oxygen results in the formation of water and nitrogen which do not effect bay operation. To avoid eluting any ions from the IFB purification system ion exchange columns, the latter are valved out prior to hydrazine addition and not valved in until the hydrazine concentration falls to <1 mg/kg. It has not been necessary to use hydrazine addition to the PNGS A PIFB since 1972 due to the excellent reliability of the irradiated fuel.

Long-term irradiated fuel integrity in wet storage

A key element in irradiated fuel management is to ensure the IF integrity during the various phases of its handling and management, including IFB storage. Thus, Ontario Hydro and AECL have a program, initiated in 1977, to examine irradiated fuel stored in IFBs for possible deterioration. Nineteen bundles from the Douglas Point, Pickering and NPD generating stations and the AECL Chalk River NRU prototype reactor are being examined. The oldest bundles have been in wet storage since 1962.

Seven destructive and non-destructive tests have been selected to characterize the elements initially and in subsequent re-examinations after further wet storage periods. The tests used to determine if there is any deterioration of either the uranium dioxide fuel (with defected cladding) or the Zircalloy cladding, are as follows: neutron radiography, fission gas analysis, hydrogen and deuterium analysis, ring tensile tests, visual examination, metallographic examination and torque tests.

Post-irradiation (i.e. from the time when they were first discharged from the reactor) data from such tests is available for many of the bundles for comparison with recent examination results. The original re-examination period was every five years starting in 1978. However, since no IF deterioration was detected, this period has been increased to ten years. All seven of the tests described above will be repeated for each re-examination.

The results of the characterization tests and the first set of re-examination tests show no apparent irradiated fuel deterioration of either the uranium dioxide fuel matrix (for defected fuel) or Zircalloy cladding due to storage in IFB's for a time period up to 17 years. Based on results to date, irradiated fuel should maintain its integrity during fifty years of underwater IFB storage. With future characterization results, this predicted period may be extended.

Conclusions

Ontario Hydro has gained considerable experience in the design, construction and operation of irradiated fuel storage facilities. Water-filled bays at the reactor sites have been designed with capacities ranging from about 700 Mg to 7,000 Mg of irradiated fuel. Auxiliary irradiated fuel bay storage facilities have also been constructed at the reactor sites. Irradiated fuel is being successfully transferred from the primary storage bays to these auxiliary bays of means of on-site flask/vehicle systems and conveyor systems. A new irradiated fuel storage container, the module, has been designed to provide a higher density fuel bay storage. The module has also been designed as the irradiated fuel container for off-site transportation, thus minimizing fuel handling operations at the storage/transportation interface.

Routine operation over a period of more than 20 years of the Ontario Hydro-operated irradiated fuel bays has been relatively trouble-free, and the bays have provided safe, reliable interim storage of irradiated fuel bundles. Tests on irradiated fuel after wet storage for periods up to 17 years indicate no fuel deterioration, whether it is defected (i.e. with a through-wall defect in the clad) or not. All evidence to date suggests there will be no significant change in irradiated fuel bundle integrity over a 50 year wet storage period whether or not there are any fuel clad through-wall defects.

High level waste (AECL 1994)

In 1978, the governments of Canada and Ontario established the Nuclear Fuel Waste Management Program "to assure the safe permanent disposal" of nuclear fuel waste. AECL was made responsible for research and development on "disposal in a deep underground repository in intrusive igneous rock". Ontario Hydro was made responsible for studies on the interim storage and transportation of used fuel and has contributed to research and development on disposal. In 1981, a further joint Canada-Ontario statement confirmed support for the Nuclear Fuel Waste Management Program, announced "the process by which acceptance of the disposal concept will be undertaken" and deferred the decision on allocation of "the responsibility for disposal site selection and subsequent operation until after concept acceptance."

Disposal concept

The disposal concept is a proposed method for the geological disposal of nuclear fuel waste in which the waste form would be either used CANDU fuel or solidified highly radioactive reprocessing waste; the waste form would be sealed in a container designed to last at least 500 years and possibly much longer; the containers of waste would be emplaced in rooms in a disposal vault or in boreholes drilled from the rooms; the vault would be nominally 500 to 1000 m deep; the geological medium would be the plutonic rock of the Canadian Shield; each waste container would be surrounded by a buffer; each room would be sealed with backfill and other vault seals; and all tunnels, shafts, and exploration boreholes would ultimately be sealed so that the disposal facility would be passively safe, that is, long-term safety would not depend on institutional controls.

Disposal facility

A specification vas developed in 1984 as the basis for a conceptual design study of a Used-Fuel Disposal Centre (UFDC). The disposal vault and waste emplacement alternatives selected were a single level, room-and-pillar disposal vault with in-floor emplacement of individual disposal containers. It was assumed that the disposal centre is self contained and located on a suitable plutonic rock body of the Canadian Shield.

Description

The disposal centre includes a disposal vault (F-4) excavated into the rock body at a depth of 1000 m, and surface facilities for the receipt and packaging of used fuel in disposal containers (F-5 and F-6). This conceptual design at depth of 1000 m provides for the longest construction times, the longest operation-cycle times, and the largest excavation and sealing-material volumes relative to a design for a disposal vault at a depth of 500 m. The disposal centre is designed to receive, package and dispose of about 191 000 Mg of uranium in the form of 10.1 million used-fuel bundles. The

disposal vault is essentially square in plan with an area of about 4 km^2. The used-fuel bundles are assumed to have been out of reactor for 10 a. Conceptual designs are presented for the primary facilities and equipment and for the operations for receiving, packaging and disposing of the used fuel.

The waste form

The function of the waste form would be to retain radioactive and chemically toxic contaminants under expected vault conditions. While the container remained intact, no water could reach the waste and no contaminants could leave the container. If the container failed, its contents would slowly dissolve in the groundwater, and contaminants would be released from the waste.

Used fuel

Activity

Figure 2-5 shows how the activity of the used fuel specified for the case studies decreases with time. It shows the contributions of the fission products, the uranium and activation products in the fuel pellets, and the activation products in the zirconium alloy bundle components. For comparison, it also shows the activity of natural uranium and its associated daughter products. More than 95% of the activation products are in the fuel pellets. The remaining activation products (less than 5% by mass) are in the zirconium bundle. Some care is required in interpreting F 2-5, because both the activity and time scales are logarithmic, reflecting the exponential nature of radioactive decay. During the first year, the overall activity decreases to about 1% of its initial value; within 10 years it decreases to about 0.1 %; within 100 years it decreases to about 0.01%, and within 1000 years it decreases to less than 0.001% of its initial value. The rate of decrease becomes less after about 1000 years. Thus the activity decreases very rapidly to start with, but some activity persists even after very long periods of time.

Heat output

Much of the radiation is absorbed in the fuel, causing it to heat up. As the activity decreases, so does the heat generated. F 2-7 shows how the heat from the used fuel specified for the case studies decreases with time. Immediately after being removed from a power reactor, a bundle of the used fuel specified for the case studies gives off about 37 000 W of heat. The initial decrease is rapid, to about 73 W after 1 year, about 5 W after 10 years, and about 1 W after 100 years. In the next 100 years, the decrease is only about 0.5 W. The rate of heat generation is one of the factors that would affect the design of a disposal vault. The containers of waste would be spread out within the vault to limit the heat generated in a given volume of rock, and thus limit the temperature

of the containers and the materials surrounding them. The reason for limiting these temperatures would be to limit the corrosion of the disposal containers, the thermal alteration of the material around the containers, and the thermal disturbance and thermally induced stresses in the surrounding rock. Thus the rate of heat generation would have an important influence on the distribution of waste in a disposal vault, and hence on the size of the vault.

Because the activity, heat output, and composition of used fuel change with time after the fuel is removed from the reactor, it is convenient to assume for the preclosure and postclosure assessments the same amount of out-of-reactor time (10 years). We believe that the vast majority, if indeed not all of the used fuel will be out of the reactor for more than 10 years.

Release of contaminants

The fuel in CANDU fuel bundles is in the form of high-density, high-purity ceramic pellets. The pellets are polycrystalline, with an average grain size of 1E-6 m. The pellets are se cracking under high temperatures and temperature gradients in the reactor, particularly at the centre of the pellets where temperatures during irradiation can be as high as 2000 C. The location of the new species in the fuel bundle depends on their physical behaviour and where they were produced. The radionuclides in a used-fuel bundle are located in the fuel pellets and in the zirconium alloy bundle components. During dissolution of a used-fuel bundle in water, the release of a given radionuclide or chemically toxic element would be governed by its location in the bundle, its solubility, and the extent or rate of dissolution of the UO_2 pellets and the zirconium alloy tubes in which the pellets are encased (F 2-1). The majority of the new radionuclides produced while the fuel is in the reactor are within the lattice of uranium and oxygen atoms in the fuel pellet. Laboratory studies show that the rate of release is controlled by the rate of dissolution of the UO_2 matrix. Thus radionuclides such as 239-Pu and 237-Np would be released from the fuel pellet only as the UO_2 dissolved, and the proportion released would be the same as the proportion of UO_2 dissolved. Extensive laboratory studies have been carried out to understand the factors that control the dissolution of UO_2. The results of these laboratory studies have been complemented by studies of natural analogues. Uranium oxides occur in uranium ore bodies, such as the one at Cigar Lake in northern Saskatchewan. At Cigar Lake, UO_2 ore has experienced very little dissolution during the billion years since its formation. Dissolution of UO_2 is strongly influenced by the oxidation reduction conditions at the surface of the material. In oxygen-free groundwater, which would be expected at depth on the Canadian Shield, the conditions are reducing and dissolution is very slow. Microbially mediated dissolution of used fuel is not expected to occur in a disposal vault. Microbes that are capable of dissolving uranium ores (in fact, they are used in bio-mining of uranium ores) do so indirectly by creating acidic conditions. In a disposal vault, however, the strong pH buffering of clay-based

sealing materials would maintain the pH in the 7 to 9 range. Moreover, these microbes thrive only under oxidizing conditions, whereas reducing conditions would be expected in the vault at the time of container failure. In a disposal vault, the alpha decay of activation products might cause radiolysis of water, that is its decomposition into hydrogen and oxygen and radicals. The effect of radiolysis of water on the dissolution of the fuel is expected to be small, because experimental studies indicate that the extent of oxidation of the used fuel by the products of radiolysis would be limited. The used-fuel surface in contact with the groundwater is not expected to oxidize beyond U_4O_9, an oxide composition that retains essentially the same crystallographic structure as UO_2 and has a low solubility. Small amounts of some of the new radionuclides produced while the fuel is in the reactor move out of the lattice of uranium and oxygen atoms in the pellet. Radionuclides that had moved to cracks in the pellets and to the gap between the pellets and the fuel sheath would be released immediately upon failure of the container and fuel sheath. Radionuclides that had moved to the grain boundaries would be released more slowly. Of the gaseous or somewhat volatile species that move out of the UO_2 lattice, 129-I is of particular importance. It has a long half-life (1.6E+7 years) and is present in anionic form, so it would be mobile in groundwater; that is, its movement would not be retarded by sorption. Cesium is present in cationic form, so it would sorb strongly on the surrounding clays and rock; that is, its movement would be retarded by sorption. Of the non-volatile species that move out of the UO_2 lattice, 14-C and 99-Tc are of particular importance, again because of long half-life and potentially high mobility in groundwater. The activation products in the zirconium alloy bundle components are expected to be distributed uniformly in the zirconium alloy. It is assumed that they would be released in proportion to the amount of zirconium alloy dissolved. Their release would be retarded by a thin zirconium oxide film that makes zirconium alloy highly resistant to uniform corrosion. Thus used fuel would be an excellent waste form because the majority of the contaminants would be released only as the UO_2 dissolved, which means they would be released very slowly.

Solidified high-level waste

Radionuclides and chemically toxic elements in the solidified high-level waste from reprocessing would, in general, be released congruently as the waste form dissolved, although preferential release might occur during the first few weeks that the surface of the waste form was exposed to water. In the late 1950s, AECL scientists began an underground experiment to investigate the behaviour of glass incorporating the high-level waste from reprocessing. They immobilized the waste in 2-kg aluminosilicate glass blocks and buried the unprotected blocks in flowing groundwater in sandy soil at the Chalk River Laboratories. The groundwater flow rate was significantly higher than would be expected in a disposal vault. Subsequent studies indicate that the

glass used is extremely resistant to dissolution: under the test conditions, it would take over 20 million years to completely dissolve one of the blocks. Thus aluminosilicate glass would make a suitable waste form for disposal of the high-level waste from reprocessing. AECL scientists have performed laboratory experiments to study the behaviour of glass and glass-ceramic waste forms in groundwaters typical of those expected at depths of 500 to 1 000 m in plutonic rock. The glass-ceramics are composed of both glass and crystalline material, and if used as a waste form, each of these materials would contain some high-level waste. Glass and glass-ceramic waste forms are not significantly affected by other components of the disposal system, such as a metal container, clay buffer, or plutonic rock; by gamma and beta radiation; or by radiolysis. However, alpha radiation can increase the dissolution rate by a factor of up to about five. Laboratory tests on borosilicate glasses and titanosilicate glass-ceramics have shown that they can have durabilities that are comparable to that of the aluminosilicate glass. Naturally occurring deposits of volcanic glasses, both on continents and in the seabed, have demonstrated the ability of glass to survive many millions of years in the natural environment. Titanosilicate minerals, similar to the crystalline phase of the glass-ceramic, exist in weathered granite deposits; therefore both the glass and crystalline phases are known to be very stable in the natural environment. Thus the rate of release of radioactive material from aluminosilicate, titanosilicate, and borosilicate waste forms would be extremely low. Experiments to characterize the physical properties of glass and glass-ceramics indicate that in the absence of water they would remain physically stable at vault temperatures for tens of millions of years. Burial tests using glass and glass-ceramic waste forms are being conducted in the Waste Immobilization Pilot Project in the salt dome at Carlsbad, New Mexico. Samples of aluminosilicate glass and titanosilicate glass-ceramic from Canada and borosilicate glass from other countries are being evaluated for performance under disposal conditions in salt. The results will be relevant to disposal in the highly saline, chloride-rich groundwaters of the Canadian Shield.

Gas generation

The production of hydrogen gas as a result of alpha radiolysis of water and corrosion of zirconium alloy bundle components has been examined for the period of 10 000 years. It has been shown that the rates of production of hydrogen gas would be low enough that the hydrogen would dissolve in the groundwater. The inert gases xenon, krypton, and helium, all in the form of stable nuclides, would have been produced by nuclear processes in the reactor. The release rates in the vault would be low enough that the gases would dissolve in the groundwater.

Container

The disposal container is a packed-particulate used-fuel disposal container (F-2), fabricated from ASME Grade-2 titanium, which holds 72 used fuel bundles. Therefore, the vault capacity is about 140000 containers. The annual throughput in the conceptual design is about 250000 used-fuel bundles, the assumed capacity of the used-fuel transportation system. This capacity is 3471 disposal containers per year, giving a disposal vault operating duration of about 40 a. The waste form would be sealed in a container designed to last at least 500 years follow emplacement in a disposal vault. The functions of the container would be to facilitate handling of the waste form and to isolate it from the groundwater for the desired minimum time. Container designs and fabrication and inspection techniques that would meet these objectives have been developed. Moreover, containers could be designed so that most would last for at least 10 thousands of years under the conditions expected in a disposal vault. If the disposal concept were implemented, a disposal container suitable for the waste site characteristics would be designed. Several options for container design could be considered. The selection of the container material would be part of the container design process.

Design considerations

The container would be designed to withstand external pressure and to resist corrosion under the conditions of temperature, pressure, and chemistry in a disposal vault for the design lifetime. Other factors that must be taken into consideration when designing containers include the geometry of the waste form and the excavated openings in the vault, the chemical toxicity of the container materials, and cost. External pressure would be exerted on a container by the groundwater and the buffer material around the container. The groundwater pressure increases with depth and is about 10 MPa at a depth of 1000 m. The additional pressure from the swelling of the buffer material around the container would depend on the characteristics of the buffer material and could be limited to 3 MPa. Neither seismic loading nor pressure from the overlying rock is expected to contribute to the external pressure. We have sufficient knowledge and understanding to design a container that would not fail because of structural overload or long-term material deformation. If necessary, the container could be designed to withstand the increase in structural load caused by a glacier. With more understanding and knowledge, conservatism in design could be decreased.

It is expected that container failures would occur only as a consequence of corrosion. Our development work has focused on achieving a minimum design lifetime of 500 years, during which time the activity of the radioactive waste would decrease by a factor of over 200 000 because of the decay of the fission products. However, the lifetime would almost certainly be much longer. For example, based on pessimistic assumptions, our estimates of time until failure by corrosion range from 1000 to 6000 years for 6.35-mm-thick titanium containers and from 30 000 to a million

years for 25-mm-thick copper containers; our more realistic estimates range from tens of thousands to hundreds of thousands of years for the titanium containers and are more than a million years for the copper containers. Germany has specified a minimum design lifetime of 500 years, Switzerland and the United States have specified 1000 years, and Sweden has designed containers for much longer lifetimes.

The container dimensions would have to be suitable for the geometry of the waste form and the geometry of the vault. For example, it would have to be possible to remove the waste form from the container if the container were found to be defective when it was inspected before emplacement. It would also have to be possible to retrieve the container from the vault if necessary. The vault geometry might restrict the capacity of equipment to be used for handling and transport, such as hoists and underground vehicles, and might thereby restrict the container size and weight. The cost of containers would depend on several factors, including the amounts and types of materials required, the present and future availability of those materials, the energy required for container fabrication, and the ease of container fabrication and inspection. Thus the design lifetime and cost would be related. For example, increasing the container thickness would increase the lifetime, but it would also increase the amount of material required and the difficulty of fabrication and inspection, and thus the cost. We estimate that a copper container shell 25 mm thick would cost twice as much as a titanium container shell 6.35 mm thick. The fabrication of enough 25-mm-thick copper containers to dispose of about 250 000 bundles of used fuel per year would require about 1% of the production of refined copper in Canada. The fabrication of the containers required to dispose of about 10 million bundles of used fuel would require 168 000 Mg of copper. For comparison, the Canadian reserves are 14 300 000 Mg, the world reserves are 200 000 000 to 360 000 000 Mg, and world resources are 10 to 100 times greater than the reserves. Reserves are mineral deposits that are recoverable with existing technology and under present economic conditions, whereas resources comprise reserves, known deposits that are not technologically and economically recoverable at present, and unknown deposits that may be inferred to exist.

The fabrication of enough 6.35-mm-thick titanium containers to dispose of about 250 000 bundles of used fuel per year would require about 2.25% of the production of titanium in the United States in 1988. The fabrication of the containers required to dispose of about 10 million bundles of used fuel would require 20 000 Mg of titanium. For comparison, the world reserves are 260 000 000 Mg and resources are substantially larger. Although Canada is one of the principal world suppliers of the ore for titanium, no titanium production capacity exists in Canada. A container would not need to provide self-shielding. Workers could be protected by other methods of shielding, and experiments have indicated that self-shielding to reduce gamma radiolysis would not reduce the rate of container corrosion. Because the characteristics of the site and disposal facility would need to be taken into consideration when

designing the disposal container, such design would need to be done as part of concept implementation. However, AECL and Ontario Hydro have investigated several options that would be suitable for Canada. Researchers in other countries are also developing container designs, which differ from ours for many reasons: their waste forms are different in size, shape, composition, activity, and heat output; some countries are considering different disposal media, such as salt and volcanic tuff, and some countries have different objectives for container lifetime and the radiation shielding provided by the container.

Corrosion

The corrosion of a disposal container would depend on the the oxidation-reduction conditions within the vault, the temperature in the vault, the groundwater composition and the characteristics of the container material. The important oxidants are dissolved oxygen, water radiolysis products, and possibly sulphide ions. Oxygen is not expected to be present in significant amounts because it would be consumed by reaction with ferrous minerals in the backfill. The production of gamma-radiolysis products of water would depend on the activity of the radioactive waste and the radiation shielding provided by the container. Corrosion rates increase with temperature. The temperature would depend on the heat output of the waste form, the rate at which that heat was transferred to the container shell and the surrounding buffer material, and the spacing of the containers in the vault. Salinity is the major concern because it can cause local corrosion. The elevated temperature and high level of radiation within tens of centimetres of the disposal container would likely create a zone in which microbes could not survive initially. Later, when conditions might permit the survival of microbes, the extremely mall pore size of the compacted buffer would likely limit microbial growth and movement ward the container.

Titanium

Titanium is unstable in air and water, but is well protected by the spontaneous formation of a protective oxide film on its surface, which gives this material its excellent corrosion resistance. Rates of uniform corrosion have been measured under a variety of conditions for ASTM Grades 2, 7 and 12 titanium. The results indicate that, irrespective of the grade used, a 6.35-mm-thick titanium container would last at least a million years under granitic vault conditions if uniform corrosion were the only degradation process.

If a crevice were initiated, crevice corrosion might occur. ASTM Grade-2 titanium is used for high-temperature seawater service because of its superior resistance to chloride corrosion. However, in industrial applications where temperatures are higher, crevice corrosion occurs at a rate many times greater than the uniform corrosion rate. AECL's electrochemical experiments on artificially

creviced samples show that the extent of crevice propagation of Grade-2 titanium is controlled by the oxygen supply. If crevice corrosion were initiated in a disposal vault, it would eventually stop because of the decrease in oxidants. All our experimental results with Grade-2 titanium suggest that the effect of radiation is to repassivate the crevices; that is, the crevice corrosion would stop. We have no evidence to suggest that radiation accelerates the crevice propagation rate.

If hydrogen pickup led to hydride formation and subsequently to general embrittlement or concentration of hydride at a crack tip, hydrogen-induced cracking might occur. On the basis of studies by AECL, we concluded that extensive hydrogen pickup leading to failure by hydrogen embrittlement would be highly unlikely. In acidified crevices, where the protective oxide on titanium is destroyed, significant hydrogen pickup and hydride formation have been observed. Delayed cracking, caused by the concentration of hydride at a crack tip, has been observed in some titanium alloys but not in Grade-2 titanium. Experimental evidence on the behaviour of titanium alloys leads to the conclusion that disposal containers would fail only by hydrogen induced cracking after cooling to temperatures below 200 C, and only if sufficient hydrogen pickup occurred during crevice corrosion prior to or during cooling. For the site conditions and design specified for the postclosure assessment case study, the decrease of temperature at the container surface with time is given in F 4-6. The mechanisms for crevice corrosion of titanium are shown in F 4-11. Radiation embrittlement, microbially induced corrosion, and gas production have also been considered. Studies indicate that radiation embrittlement would not occur, because radiation embrittlement of metal occurs only in the presence of a neutron flux about 10 orders of magnitude greater than the flux at the container surface, with fuel 10 years out of the reactor. Titanium appears to be immune to microbially induced corrosion. The quantity of hydrogen produced upon uniform corrosion of titanium would be small enough to be absorbed in the pore water in the buffer and backfill for well over 10 000 years. Grade-12 titanium alloys are preferable to Grade-2 alloys for industrial applications involving hot saline solutions. Our experiments show that the extent of crevice propagation of Grade-12 titanium is controlled by the composition and microstructure of the alloy. Repassivation eventually occurs, even in the presence of substantial concentrations of oxygen. The extent of crevice corrosion of Grade-12 titanium is limited compared with that of Grade-2 titanium, and container failure by crevice corrosion would be extremely unlikely; instead, it would almost certainly be by hydrogen-induced cracking. At present, we are unable to model container failure caused by the latter mechanism. Thus we cannot, as yet, recommend the use of Grade-12 titanium, despite its apparent advantages over Grade-2 titanium. Based on pessimistic assumptions, our estimates of the time until failure by corrosion range from 1000 to 6000 years for 6.35-mm-thick containers of Grade-2 titanium; our more realistic estimates range from tens of thousands to hundreds of thousands of years.

Internally supported shells

For a metal-shell container, one option is for the shell to be supported internally. The wall thickness would be sufficient to resist corrosion for the required period, but the container would utilize some form of internal support to resist the external pressure. AECL and Ontario Hydro have studied the following types of internally supported containers (F 4-12): a packed particulate container, in which a particulate, such as glass beads or sand, would be compacted into the empty space that remained in the container after used-fuel bundles or the solidified high-level waste from reprocessing were placed in it; a metal-matrix container, in which a molten, low-melting-point metal, such as lead, would be cast to fill the empty space with a solid metal matrix; and a structurally supported container, in which internal structural support, such as an array of carbon-steel tubes, would act in combination with a compacted particulate. Potential particulate materials were compared on the basis of strength, compaction characteristics, availability, cost, and chemical compatibility with other container components and with saline groundwaters. From an initial list of twelve candidate materials, it was determined that industrial glass beads offered the best combination of desirable properties. Potential metal-matrix materials were compared on the basis of chemical compatibility with the other container components and with used fuel, casting temperature, and matrix-solidification characteristics. From an initial list of six candidate materials, pure lead was determined to offer the best combination of desirable properties. The amount of lead required for about 3500 containers per year would represent about 10% of the production of refined lead in Canada. A potential material for internal structural support would be carbon steel. The corrosion of carbon steel would produce substantial quantities of hydrogen gas, whose effects on the performance of a disposal system have not been studied extensively. A series of short-term structural performance tests was conducted on the three types of internally supported containers, all with titanium shells: a full-scale prototype of a packed-particulate container, four half-scale models of a metal-matrix container, and a full-scale prototype of a structurally supported container. These tests were performed in the Hydrostatic Test Facility at AECL's Whiteshell Laboratories (F 4-13). In these tests, the external pressure was increased to 10 MPa, and the temperature was increased to 150 C. This condition was maintained for about a week, usually until all indications of further container-shell deformation had ceased. All three types of internally supported container performed well, even when severe, deliberate manufacturing defects were introduced into the supporting material. Although the shells deformed during some tests, they were not breached by rupture or tearing. In general, the experimental results compared reasonably well with those predicted by computer models. Where discrepancies occurred, their causes were identified. Research is now directed at modelling the long-term structural behaviour of titanium and copper containers.

The three types of internally supported containers were compared on the basis of the factors considered important in container design, and the packed-particulate container was found to be the most favourable. On the basis of this comparison of internally supported containers, the packed-particulate container design was selected for the preclosure and postclosure assessment case studies. Swedish researchers are studying a 60-mm-thick copper container internally supported by a lead matrix.

Self-supported shells

For a metal-shell container, an alternative to the internally supported shell is for the shell(s) to be self-supported, that is, sufficiently thick to resist the external pressure of the groundwater and the buffer without any internal support. We call this option the stressed-shell container. A stressed-shell container of titanium would need to be about 70 mm thick to prevent failure by creep-induced buckling for at least 500 years. A stressed-shell container of copper would need to be even thicker. Such thick container walls would make fabrication difficult and inspection uncertain, particularly in a mass-production environment. Therefore, we decided that neither titanium nor copper would be suitable for providing long-term structural strength. Structural strength could be provided by a stronger material such as steel. A full-scale prototype made of stainless steel was tested by raising the pressure on the shell until buckling collapse occurred. The observed deformational behaviour was consistent with that predicted by our structural-performance computer model. Several designs have been studied in which one of two or more container shells is made of steel. Ontario Hydro has studied a dual-wall design employing a 50-mm-thick carbon steel inner shell to provide structural strength against external pressure and a 6.35-mm-thick titanium outer shell to provide corrosion resistance for at least 500 years. German researchers are studying a multiple-wall design that includes a 150-mm-thick layer of steel to provide corrosion resistance for at least 500 years and a 200-mm-thick layer of cast iron to provide radiation shielding. Swedish researchers are studying a dual-wall container with a 50-mm-thick internal carbon steel shell surrounded by a 50-mm-thick copper shell (F 4-14). Swiss researchers are studying a cast steel container with a minimum thickness of 150 mm, in order to meet their minimum container lifetime objective of 1000 years, to provide built-in radiation shielding for workers, and to minimize radiolysis of groundwater. Stainless steel would not resist corrosion in the saline groundwater conditions expected on the Canadian Shield, and the corrosion of carbon steel would produce substantial quantities of hydrogen gas, whose effects on the performance of a disposal system have not been studied extensively.

Fabrication and inspection

AECL has investigated methods to weld both titanium and copper containers. The closure weld would need to be done remotely and in the presence of radiation because of the radioactive waste in the container. Both pulsed-current gas tungsten-arc welding and resistance-heated diffusion bonding are suitable for the production of high-quality joints in titanium. Electron-beam welding is considered the most feasible technique for copper. We have also investigated methods to inspect the closure weld, remotely and in the presence of radiation. Ultrasonic techniques could be used for volumetric inspection, that is, inspection of the interior of a material. Both gas-tungsten-arc welds and diffusion bonds in titanium have been inspected successfully, and deliberate defects with widths less than 0.13 mm in diffusion bonds have been detected. Although copper is less amenable to ultrasonic inspection, electron-beam welds on 200-mm-diameter copper containers with 25-mm- thick walls have been inspected successfully. Significant advances have been made in performing ultrasonic inspection with a programmed robot. Volumetric inspection of the final closure weld would be followed by leak-testing of the entire container. Of the procedures available, helium leak-testing is the most sensitive, and this widely used industrial procedure could be adapted for container inspection. If a container failed either the volumetric inspection or the leak test, it would be repaired or, if necessary, the waste would be removed and placed in a different container. We have demonstrated the methods for producing high-quality castings of lead in a metal-matrix container. We have also investigated methods for inspecting cast-lead matrices. While ultrasonic inspection appears feasible for the detection of defects between the container shell and the cast matrix, shrinkage-related flaws deeper into the matrix cannot be detected with certainty by this method because the lead absorbs sound waves. However, a technique based on 14-MeV neutron attenuation has been demonstrated at a laboratory scale. In the case of a packed particulate such as glass beads, a measured quantity of particulate would be placed and compacted around the used fuel in a container to ensure that the container was completely full, so that the particulate would provide the maximum support to the outer shell. Nuclear fuel waste would be loaded into the container remotely, using well-established methods for handling radioactive material.

Monitoring and contingency plans

Any monitoring of containers of waste emplaced in disposal rooms or in boreholes extending from the rooms would be invasive and could impair the long-term performance of the disposal vault. The AECB (1985) requires that safety must not be compromised by any provisions that may be made for monitoring. However, some waste could be placed in special test areas of the vault, where container performance could be studied until the beginning of the decommissioning stage. At or before that time, the containers would be retrieved and examined. The data from monitoring and from

examination of the retrieved containers could be used to support a request for a decommissioning licence. Should the data indicate that the performance was poorer than assumed in the container lifetime model, large-scale retrieval of emplaced waste would be an option. However, given the pessimistic assumptions used for the model, this is considered very unlikely. The possibility of container failures during the preclosure phase has been examined, and possible releases have been estimated. Contingency plans would include provisions for retrieval.

Vault

The disposal vault would be a network of horizontal tunnels and disposal rooms excavated deep in the rock, with vertical shafts extending from the surface to the tunnels. Rooms and tunnels could be excavated on more than one level. The vault would be designed to accommodate the rock structure and other subsurface conditions at the chosen site. The disposal container and vault seals would also be designed to accommodate the subsurface conditions at the chosen site. The disposal concept and its implementation constitute the proposed disposal strategy. After the disposal facility was closed, there would be multiple barriers to protect humans and the environment from both radioactive and chemically toxic contaminants in the waste: the container; the waste form; the buffer, backfill, and other vault seals; and the geosphere.

The disposal vault is located at a depth of 1000 m. The maximum temperature at the container outer surface and throughout the buffer material must not exceed 100 C. For thermal calculations, the reference used fuel has the average burnup of used fuel from the Bruce Nuclear Generating Station and a cooling period of 10 a after discharge from the reactor. The near-surface extension zone, the layer of rock at the ground surface overlaying the disposal vault that could experience a loss of horizontal confining stresses, and the potential opening of vertical fractures must not extend more than 100 m below the ground surface. The average strength-to-stress ratio is two or greater for the interroom pillars and, where applicable, the rock webs around the waste emplacement boreholes. As well, where applicable, the extraction ratio on the emplacement horizon is about 0.25. The disposal vault will use shafts for access and will be arranged in a room and pillar configuration. The emplacement configuration will be in-floor borehole emplacement with a single disposal container in each borehole.

Thermal, mechanical and coupled thermal-mechanical analyses were done for the disposal rooms and emplacement boreholes. An analytical code was used initially to analyze the temperature distribution for a vault at a depth of 1000 m to select the borehole-to-borehole spacing that satisfied the 100 C maximum temperature limit. This spacing was 2.1 m between borehole centres, with three boreholes across a room and 94 boreholes along the length of the room. Making allowance for the space required for the operation of equipment and for the sealing of the disposal room

resulted in a disposal room that was 230 m long with a cross section that was 8 m wide and 5.5 m high.

The stability of this room was analyzed under excavation (ambient temperature) conditions and under sealed (heated) conditions. Two cases were analyzed for the in situ stress conditions assumed at a depth of 1000 m with rooms excavated perpendicular to the maximum horizontal stress direction:
- a disposal room with a reference flat floor and with boreholes spaced at 2.1 a across and along the room, and
- a disposal room with a curved floor similar to the crown of the room and with boreholes spaced at 2.1 m across and 3.0 m along the room. For both cases, the analyses of the excavation conditions indicated zones of yielding in the floor of the disposal room and along the emplacement borehole walls. We judged these zones to be larger than would be desired based on the Underground Research Laboratory studies of rock response to excavation. A similar analysis was done for the in situ stress conditions assumed at a depth of 500 m and a borehole spacing of 2.1 m across and along the room. The results indicated that the stability of the excavation boundaries would be acceptable and the disposal vault could be designed to meet the near-field thermal-mechanical specifications.

The specific borehole-emplacement configuration for the assumed in situ stress, room orientation and rock strength (or failure) criteria is suitable only for depths shallower than 1000 m. The in-room emplacement configuration may be preferable under higher in situ stress conditions. Analyses were also done to assess the potential for shear displacement on a subhorizontal and a subvertical fault zone near a disposal vault at a depth of 1000 m under the influence of heat from the nuclear fuel waste. No shear displacements along a subhorizontal fault are expected below a depth of 100 m from the ground surface.

Design assumptions

In developing the conceptual design of a used-fuel disposal centre, several assumptions were made regarding the characteristics of the disposal system components and the properties of the natural system or site. The assumed characteristics of the disposal system components include the selection of used fuel as the waste form, a titanium packed-particulate used-fuel container design, a room-and-pillar disposal vault arrangement, and borehole emplacement of individual disposal containers. The thermal and mechanical properties and structural characteristics were assumed for the natural system surrounding the disposal vault. The basis for these assumptions is discussed. One assumption that warrants specific mention is the quantity of used fuel, which affects the size of the reference disposal vault. We assumed that 10.1 million CANDU fuel bundles (F-1) irradiated to an average burnup of 685 GJ/kg U and cooled for 10 a after their discharge from a nuclear power reactor would be accumulated for disposal by 2035. This was based on a 3% annual growth in nuclear electric generation and a replacement of all operating reactors in kind at the end of their

operating life. More recently, projections of used-fuel arisings are of the order of 5 million used-fuel bundles. A reduction in the amount of used fuel has no effect on either the fundamental aspects of the facility design and operation or on the technical feasibility of nuclear fuel waste disposal. It does affect the overall size of the disposal vault, the inventory of radionuclides for the postclosure safety assessment, and the total and unit costs of disposal.

Regulatory requirements

In Canada, regulations specifically applicable to the operation of nuclear facilities and the control of radioactive materials are promulgated under the Atomic Energy Control Act (AECA) by Government of Canada in 1985. Besides meeting the requirements of all applicable regulations under the Act, a nuclear fuel waste disposal facility would comply with all applicable legislation and regulations of the Canadian government and of the province and municipality in which the facility would be built. As well, it would comply with the transportation regulations of any municipality, province, or country through which the waste would be shipped (e.g., barge transportation of used fuel through waters controlled by the United States in the Great Lakes). It is recognized that the legislation will evolve over time. An example is an amendment to the Atomic Energy Control Regulations being proposed that would reduce the cumulative effective dose limits for atomic radiation workers and for the public (AECB 1991).

Project schedule

A nuclear fuel waste disposal project would be subdivided into smaller elements for planning and control. One approach would be to establish stages and activities where the project stages are sequential, and to incorporate the major blocks of effort necessary to achieve nuclear fuel waste disposal. The activities may occur concurrently, and generally span more than one stage. The Siting Stage would involve developing the siting process, and site screening and site evaluation substages to identify suitable site(s) for waste disposal. Data would be gathered during site evaluation to develop an understanding of the surface and underground physical and chemical conditions in and around the site(s) to confirm their potential for safe disposal. During the siting stage, preliminary disposal facility designs would be prepared for each site being evaluated. A specific design for the preferred site would be completed and approved prior to deciding to proceed with underground evaluation. The end point of the siting stage would be a design based on the results obtained from the surface and underground evaluation studies, and approved for construction at the site selected for a disposal facility.

The Construction Stage would involve constructing the infrastructure and surface facilities needed to transport and dispose of nuclear fuel waste, the underground accesses and service

areas, and a portion of the underground disposal rooms. The Operation Stage would involve receiving nuclear fuel waste transported to the disposal facility, sealing it in corrosion resistant containers, sealing the containers in disposal rooms, and constructing additional disposal rooms, as necessary. The Extended Monitoring Stages would involve monitoring conditions in the vault, geosphere, and biosphere between the operation and decommissioning stages and/or between the decommissioning and closure stages. The Decommissioning Stage would involve the decontamination and removal of the surface and subsurface facilities; the sealing of the tunnels, underground service areas, shafts, and underground exploration boreholes; and the return of the site to a state suitable for public use.

The Closure Stage would involve the removal of monitoring instruments from any boreholes that could compromise the safety of the disposal vault, the sealing of those boreholes, and the return of the site to a state where safety would not depend on institutional controls (i.e., to a passively safe state). Monitoring could continue beyond closure if desired by the regulatory authorities or the public, provided that such monitoring did not compromise the long-term passive safety of the sealed disposal vault.

The major activities associated with the implementation of nuclear fuel waste disposal - public involvement, characterization, design, monitoring, component testing, performance assessment and construction (F-3) - span two or more stages. The assumptions used in estimating work schedules and costs for the various stages in the life cycle of the Used-Fuel Disposal Centre (F-3) are: the siting (23 y, site screening : 5 y, site evaluation : 18 y), construction (7 y), decommissioning (16 y) and closure (2 y) stages are estimated based on operation 24 h/d, 7 d/week; The operation stage (41 y) is estimated based on a 5-d week with two 8-h shifts per day, except for security fire fighting and essential site services, which are staffed 24 h/d, 7 d/week; The work schedule for the operation stage was selected based on an assumed used-fuel transportation rate and packaging plant throughput, and provides significant reserve capability for adjusting the disposal centre annual capacity.

Operation

Used fuel is received at the packaging plant of the disposal centre in either a road or rail transportation cask that contains the used-fuel bundles in storage/shipping modules. The modules are unloaded from the casks in a module-handling cell. The modules may be held temporarily in a receiving surge-storage pool or they may be transferred directly to the used-fuel packaging cell. In the packaging cell, the fuel bundles are transferred from the shipping modules to the disposal container fuel baskets, 72 bundles to a basket, and each fuel basket is installed within a disposal container. Each bundle and container is monitored for nuclear material safeguards purposes during the transfer operations.

The reference disposal container shell and end closures assumed in this conceptual design are fabricated of 6.35-mm-thick ASTM Grade-2 titanium. The loaded container is filled with a particulate, such as glass beads, which is compacted vibrationally to fill all the void space, allowing the container to withstand the expected external loads. A top head is pressed into the container, and the top head and container shell flanges are diffusion-bonded. The assumed quantity of used fuel requires about 140 000 disposal containers, each having a mass of about 2800 kg. When initially sealed in the disposal container, the 72 used-fuel bundles produce about 300 W of heat.

Testing

Following nondestructive testing (i.e., ultrasonic bond inspection and a helium leak test) to establish the integrity of the sealed container, each disposal container is loaded into a shielding container cask. Each full cask is transferred to the disposal vault using the cage in a dedicated waste shaft. When removed from the cage, the cask is moved by crane to an underground storage area or by truck directly to a disposal room. In this conceptual design, each disposal room is about 8 m wide, 5 to 5.5 m high and 230 m long. Up to 282 vertical emplacement boreholes are drilled in the floor of each disposal room, and each borehole is prepared to receive the disposal container. The emplacement boreholes are 1.24 m in diameter, and 5 m deep, and are spaced about 2.1 m apart, three across the room and 94 along the room, as required to keep the maximum temperature of the container shell below 100 C. Before a container (F 8) cask is received in the disposal room, a clay-based buffer material (i.e., 50% sodium-bentonite clay and 50% silica sand by mass) is compacted into the emplacement borehole and a hole is centrally augered into the buffer to receive the container. When the container has been emplaced, the radial gap between the container and the buffer is filled with dry silica sand to improve heat transfer, and additional buffer material is then placed and compacted over the container to the floor level of the disposal room.

Closure

When all the emplacement boreholes in a room have been filled, the room is backfilled by placing and compacting a mixture of 25% glacial-lake clay and 75% crushed granite, by mass, to fill the lower 3.5 m of the room. The upper portion of the room is filled by spray-compacting into place an upper backfill material similar in composition to the buffer material. A concrete bulkhead is constructed at, and grouted into, the room entrance to seal the room and to withstand the buffer and backfill swelling and the groundwater pressures. A safeguards seal may be incorporated into the bulkhead to detect unauthorized entry. The operational sequence in the conceptual design, consisting of disposal room excavation by the drill-and-blast method, emplacement-borehole drilling and

preparation, waste emplacement, borehole sealing and room backfilling and sealing, continues throughout the operating period of the disposal vault. The disposal rooms are developed and filled in sequence, moving from the upcast shaft complex toward the service shaft complex (F 4) to control access, potential contamination, and potential radiation doses to personnel. When the vault has been filled with waste, the monitoring data have been assessed to show compliance with the regulatory and design criteria, and the regulators have approved the decommissioning and closure plan for the centre, the access tunnels and shafts will be backfilled and sealed, the surface facilities will be decommissioned and disassembled, and the site will permanently marked and returned to a state suitable to allow public use of the surface.

Resource requirements

The resource estimates far the Used-Fuel Disposal Centre (UFDC) were developed on the basis of conceptual design information and on assumptions on siting and on the extent of equipment engineering necessary to do certain operations. It is judged that the nominal cost estimates may be as much as 15% high or 40% low for the engineered barriers assumed in this report. These cost estimates could change significantly if different engineered barriers are selected and/or the disposal vault arrangement becomes more complicated to account for local site conditions. The costs are given in constant 1991 Canadian dollars excluding any financing costs. The cost for the specific disposal centre to dispose of 10.1 million used fuel bundles at a depth of 1000 m is estimated to be about $13.32 billion from the beginning of the siting stage through to the end of the decommissioning and closure stage, a period of 89 a. The UFDC would provide about 62 200 person-years of direct on-site employment. The total cost might range from $11.32 billion to $18.65 billion for the assumptions noted above. The corresponding lifetime labour requirement might range from 52 800 to 87 000 person-years.

Table 53 Used-fuel centre life-cycle cost and labour requirements.

	Cost - 1991 M$			Labour - person*years		
Estimate	low	nominal	high	low	nominal	high
Siting (23 a)	1850	2180	3050	6880	8100	11330
Construction (7 a)	1540	1810	2530	6240	7340	10280
Operation (41 a)	6850	8060	11280	33880	39850	55800
Decommissioning (16 a)	1060	1250	1750	5720	6730	9430
Closure (2 a)	30	30	40	120	150	200
Total	11320	13320	18650	52840	62170	87040

Note : The values in the columns do not necessarily add up to the total shown because of rounding.

The cost of a disposal facility will be sensitive to changes in a wide range of parameters. Examples of the sensitivity of the

disposal facility schedule and nominal costs to the quantity of used fuel for disposal and to the depth of disposal are:

Table 54 Scaled nominal cost (M$ 1991) and Duration (D in years) estimates for disposal vault capacities of 5, 7.5 and 10.1 Million used-fuel bundles at depth of 1000 m.

Million of bundles	5		7.5		10.1	
	D	Cost	D	Cost	D	Cost
Siting	23	2140	23	2160	23	2180
Construction	5	1520	6	1630	7	1810
Operation	20	4060	30	6040	41	8060
Decommissioning	13	940	15	1090	16	1250
Closure	2	30	2	30	2	30
Total	63	8680	76	10950	89	13320

Table 55 Comparison of nominal cost (M$ 1991) and schedule durations (D in years) for a disposal centre with a vault at depths of 500 and 1000 m (Capacity = 10.1 million used-fuel bundles).

Depth =	500 m		1000 m	
	D	Cost	D	Cost
Siting	22	2110	23	2180
Construction	7	1780	7	1810
Operation	41	8060	41	8060
Decommissioning	14	1130	16	1250
Closure	2	30	2	30
Total	86	13110	89	13320

Feasibility

It is feasible to design, build, operate and seal a nuclear fuel waste disposal facility with existing technologies, or with reasonable extensions of these technologies. The work presented in this report is based on over 15 a of study by AECL, Ontario Hydro, government departments, universities and private-sector consulting groups. Although a nuclear fuel waste disposal vault will be a unique underground facility, its design, construction, operation and management are similar to many other major underground civil engineering projects. These include the Churchill Falls hydroelectric power house in Labrador, the NORAD defence facility in North Bay, Ontario, and the La Grande hydroelectric generating station near James Bay, Quebec, which have been engineered and constructed in the Canadian Shield. These facilities have been designed for, and constructed in, remote places, and have operated safely and within design specifications for many decades. Based on the presented cost estimates, the cost of disposing Canada's nuclear fuel waste is a small fraction of the cost of electricity derived from nuclear power (less than $0.001/(kWh)).

The biosphere model (AECL 1993)

In this reference, we describe the BIOTRAC (BIOsphere TRansport And Consequence) model used to simulate the transport of

nuclides through the biosphere in the postclosure phase of the disposal concept when the vault is full, and has been decommissioned and closed. Under the direction of the executive code SYVAC3 (SYstems Variability Analysis Code - Generation 3), BIOTRAC is coupled with models of transport in the vault and geosphere to provide probabilistic estimates of nuclide concentrations in the environment, and of radiological doses to humans and other organisms.

BIOTRAC was developed specifically to evaluate the postclosure environmental and health impacts of the concept for the disposal of Canada's nuclear fuel waste. The model is applicable for up to 10 000 a into the future, the period of quantitative assessment of the concept specified by the Atomic Energy Control Board in 1987. This period will likely be free from continental glaciation. BIOTRAC and its numerous parameter values are based on a vast amount of literature data. Since 1978, the model has also been supported by an extensive research program involving a variety of field and laboratory studies to fill in gaps in knowledge on the transport of nuclides in-the biosphere and their effects on biota. Most of this research was focused on the Canadian Shield, and has involved interactions with researchers from nuclear waste disposal programs in several other countries.

The important processes and pathways associated with transport from an underground source over very long times were identified through a rigorous scenario analysis procedure that involved literature reviews, brainstorming sessions and the use of expert opinion. These processes and pathways are modelled probabilistically using a systems variability analysis approach. The values of many model parameters are represented by probability density functions (PDFs), which allow for uncertainty in model structure, variability and uncertainty in parameter values, and natural variability in spatial and temporal aspects of the biosphere. Moreover, in the absence of a specific site for the vault, the distributed parameter values allow the assessment to be conducted generically as far as the biosphere is concerned. The PDFs chosen for the biosphere parameters encompass the full range of values that might be encountered on the Ontario portion of the Canadian Shield. These values are also representative of much of the Shield in general. At the beginning of each computer simulation or run, SYVAC3 selects a possible state of the system by randomly sampling a value for each parameter from its specified PDF.

This set of values is used to calculate nuclide concentrations and doses for that state. The procedure is repeated, typically several thousand times, to provide the full range of possible consequences and their frequency of occurrence, from which the uncertainty in the results can be estimated. To ensure that computer requirements do not become impractical, the various transport processes are modelled as simply and efficiently as possible, consistent with the accuracy and the detail needed in the results. In areas where knowledge is limited, and where realistic models cannot be formulated or validated, we make conservative

assumptions to ensure that environmental concentrations and doses are not underestimated.

Human radiation doses are calculated by BIOTRAC for individuals belonging to a group of people receiving the greatest exposure because of its location and lifestyle. We assume that this all-inclusive critical group is composed of a sequence of self-sufficient rural households living near where nuclides would discharge to the biosphere, and where nuclide dilution and dispersion are at a minimum. The lifestyle of the critical group is based on present human behaviour using conservative, yet reasonable, assumptions. For example, members are assumed to live their entire lives at the discharge zone, having access only to those parts of the biosphere that are potentially contaminated. They are assumed to be entirely self sufficient, drawing all their resources, including food, water, air, heating fuels and building materials, from the local environment. For dose prediction purposes, the group is assumed to be represented by reference man, as defined by the International Commission on Radiological Protection in 1975. Predicted doses can be compared with regulatory requirements established by the AECB in 1987.

Radiation doses for non-human biota are predicted by BIOTRAC for a set of generic target organisms at the discharge zone, including a plant, a mammal, a bird and a fish. Doses to these organisms can be compared with the detrimental effects known to occur at various dose levels. Humans also serve as an indicator species for evaluating radiological effects on other biota. Furthermore, we show that radiological and chemical protection of the biota and their habitat in general can be evaluated by comparing the concentrations of nuclides in surface water, soil and air, predicted by BIOTRAC with various regulatory criteria and guidelines, and with existing environmental baseline concentrations and their variability. With these assessment methodologies, we aim to achieve the protection of plant and animal populations and also, indirectly, of higher ecological levels - communities and ecosystems.

The geosphere/biosphere interface

BIOTRAC is driven by the output of the geosphere model (F-2), which for the postclosure assessment is a site-specific model based on data obtained at the Whiteshell Research Area (WRA). For modelling purposes, a hypothetical vault is located at a depth of 500 m in the region of the WRA. The topography of the area and a conceptual model of the subsurface structure suggest that groundwater carrying nuclides from the vault would reach the surface at three distinct discharge zones in or near a water body known as Boggy Lake. Although the discharge would occur primarily to the lake itself, we assume that a small portion of each zone underlies a terrestrial area that is suitable for terrestrial biota and for farming by the critical group. Permanent and temporary, or seasonal, wetlands are considered through discharge to the lake and terrestrial areas respectively. In some model simulations, a final

point of nuclide discharge to the biosphere is a domestic bedrock well drilled into the contaminated groundwater plume.

Because the geosphere model is site-specific, its parameters have values representative of the WRA. A few of these parameters also appear in the biosphere model. To ensure consistency between the two-models, the values of the common parameters were not set independently in BIOTRAC, but were set equal to the values assigned in the geosphere model. The interfaces between the geosphere and the biosphere occur at the top of the compacted layer of sediment beneath the lake, at the bottom of the unsaturated soil zone, or water table, and at the well. In the compacted sediments, the nuclide load from the vault is assumed to arise entirely through sorption from upward moving groundwater. Concentrations in compacted sediments are calculated on the assumption that the flow through them is advection dominated, and that nuclides in the flow are partitioned between the solid and liquid phases.

The biosphere model is driven by the total nuclide flow out of the geosphere, including flows to aquatic and terrestrial areas and to the well. For aquatic areas, nuclides released from the geosphere are discharged directly into the lake from compacted sediment. For terrestrial areas, the biosphere model is driven by the nuclide concentration in the pore-water of the lowest soil layer, which is calculated from a mass balance equation that takes into account advection into and out of the layer, and ingrowth through radioactive decay and decay of the radionuclide itself. Finally, well-water concentrations are calculated using a two-dimensional analytical model that is part of the geosphere model.

The biosphere submodels

Nuclide transport in the biosphere is modelled with four separate but closely linked submodels representing surface waters, unsaturated soils, the atmosphere and food chains (F 2). The surface water body is assumed to be a typical Canadian Shield lake, and is modelled as a two-compartment system, one compartment representing the water column and the other compartment representing recently deposited mixed sediments that overlie the compacted sediments that are part of the geosphere model. Nuclides from the geosphere are released directly into the water column, from where they may be transferred to the mixed sediments. This system is described by coupled mass balance equations that take into account hydrological flushing, dilution, mixing, sedimentation, gaseous evasion, and radioactive decay and ingrowth. Nuclide inputs to the lake, caused by runoff and atmospheric deposition, and the resuspension of nuclides from the sediments to the water column, are treated implicitly. The model output includes time-dependent nuclide concentrations in the water column and in the mixed sediment.

The prediction of soil concentration is based on a mechanistic soil model, SCEMR1 (Soil Chemical Exchange and Migration of Radionuclides Model, Revision 1). This model can provide the

detailed treatment of processes and the fine time and space resolutions necessary to simulate nuclide migration through the soil profile. SCEMR1 is a one-dimensional, time dependent model that uses detailed meteorological data, together with the Darcy equation and the equation of continuity, to calculate water flows between four soil layers on a daily basis. Nuclides introduced into this system from groundwater below or from aerial irrigation water above may be advected downward by leaching or upward by capillary rise. Concentrations in a given soil layer are calculated from a simple mass balance equation involving the flows into and out of the layer, assuming that the nuclides are mixed instantaneously and uniformly within each layer. Nuclides are partitioned between solid and liquid phases using the soil solid/liquid partition coefficient. SCEMR1 is driven by the nuclide concentration in the pore-water of the soil layer that receives the contaminant input; these concentrations are also calculated using a mass balance approach. The output of SCEMR1 is the time-dependent nuclide concentration in the soil root zone for each of three contamination pathways - groundwater discharge, aerial irrigation and atmospheric deposition.

Because SCEMR1 is a detailed research model, it requires too much computer time to be of practical use in a long-term, probabilistic assessment. Accordingly, a more efficient model was derived for inclusion in BIOTRAC. This model is based on a statistical summary, in regression equation form, of the steady-state root-zone concentrations, designated by CSS, and the times to steady state, designated by TSS, predicted by SCEMR1 for a constant source term and a wide range of values of the important model parameters. The root-zone concentrations were successfully approximated as a function of time by a simple analytical expression involving CSS and TSS. This expression was used to write a mass balance equation for the root zone to allow for a time-dependent nuclide source term, ingrowth of daughter nuclides, and nuclide losses resulting from gaseous evasion, cropping and radioactive decay. In this way, root soil concentrations can be calculated for any contaminant source in a few seconds of computer time.

In each BIOTRAC simulation, we calculate nuclide concentrations in the soils of three distinct fields: a garden, which supplies all the plant food eaten by the critical group; a forage field, which provides the feed required by their livestock; and a woodlot, which supplies the wood needed to build and heat their home. Non-human organisms also live on these fields and depend on the fields for food and shelter. We model a fourth field with the characteristics of a peat bog for simulations involving an organic soil and when the critical group heats its home with peat. The transport equations defining the surface water and soil submodels are solved by a response function/convolution approach that is used throughout SYVAC3 to treat time-dependent systems.

Nuclides reach the atmosphere as a result of suspension from contaminated water bodies, soils and vegetation. The atmosphere submodel treats a variety of suspension mechanisms, both natural

and anthropogenic, including the suspension of particulate nuclides from terrestrial and aquatic sources, the evasion of gases from terrestrial and aquatic sources, and the release of nuclides when biomass is burned. Once in the air, the nuclides undergo dispersion and deposition back to the underlying surface. Additional processes can raise indoor air concentrations above outdoor levels. We model the diffusion of volatile nuclides from the soil into buildings and the release of nuclides from water used inside the home of the critical group.

The models we use to simulate the suspension mechanisms vary considerably in complexity, depending on our theoretical understanding of the process and on the amount and quality of the available data. In some cases, simple mass loading parameters are used to calculate air concentrations directly from the nuclide concentration in the source compartment, such as soil. This approach allows a number of suspension mechanisms to be modelled collectively, and also accounts for the effects of atmospheric dispersion. For other mechanisms, nuclide fluxes to the atmosphere can be predicted and combined with a dispersion model to calculate air concentrations. For a ground-level area source, such as a contaminated field or water body, we based our dispersion model on the trajectory simulation approach. In all cases, the models are equilibrium models, in that air concentrations are assumed to adjust instantaneously to changes in the concentration of the source compartment. Total air concentrations are calculated by summing the contributions from the individual suspension mechanisms. Indoor and outdoor concentrations are calculated separately for each nuclide.

The rate at which nuclides are deposited from the air to soil and vegetation is also predicted by the atmosphere submodel. Deposition velocities are used to model the dry deposition process and washout ratios are used to model wet deposition. The food-chain submodel, CALDOS (CALculation of DOSe food-chain and dose model), traces nuclide movement from the physical compartments of the biosphere, i.e., surface water, soil and air, through the food chain to humans and other organisms, and calculates radiological doses from both internal and external exposure pathways. Transfer is predicted using simple multiplicative chain equations that assume the nuclide uptake by plants and animals, and doses, are directly proportional to nuclide concentrations in the source compartment. The model is therefore a steady-state, equilibrium model.

The internal exposure pathways considered in CALDOS are the ingestion of contaminated plants, terrestrial animals, water and soil by humans; the ingestion of terrestrial animals and fish that have consumed contaminated plants, water or soil; and the inhalation of air by humans. In treating these pathways, CALDOS accounts for processes such as root uptake, contamination of plant surfaces by irrigation and atmospheric deposition, losses from plant surfaces as a result of environmental processes, transfer to animals and humans, and radioactive decay and ingrowth. The external pathways treated are immersion in contaminated air and

water, and exposure to contaminated soil and building materials. The total dose to a member of the critical group and other organisms is found by summing the individual doses from all nuclides and exposure pathways.

Internal doses depend on the amount of contaminated food, water and air taken into the body. For humans, CALDOS calculates these amounts in an integrated way from the total energy need, the diet, and the nutritional content of the diet. For modelling purposes, the diet is assumed to consist of five general food types: terrestrial plant foods, mammalian meats, milk and dairy products, poultry and eggs, and freshwater fish. Some of these food types are also used as representative organisms for evaluating doses to non-human organisms.

A few nuclides exhibit special properties that require alternative approaches to transport modelling and dose calculation. A specific activity model is used to predict internal doses to humans from tritium (3-H), which is very mobile in the environment. A limited specific-activity model is also used for 129-I because internal iodine doses are dominated by the thyroid gland and the Iodine content of the thyroid is regulated metabolically. The specific activity of 129-I in the thyroid and of 14-C in the body are not allowed to exceed the specific activity of these nuclides in the groundwater discharging from the geosphere to the biosphere. The transport and exposure pathways can be greatly simplified for the noble gases, which do not accumulate and disperse rapidly in the biosphere, but special attention has been paid to radon (222-Rn) inhalation. Short-lived daughter radionuclides with half-lives less than one day are assumed to be in secular equilibrium with their precursors throughout the biosphere, and are not modelled explicitly. The contribution of these nuclides to dose is accounted for through their precursors.

In addition to the four submodels, BIOTRAC also includes a model for predicting radiological doses to non-human biota. This model is similar to CALDOS, and the two models share many parameter values and PDFS. The model considers four generic target organisms for dose prediction - a fish, a plant, a mammal and a bird. These organisms broadly represent Canadian Shield biota in terms of exposure situations and parameter values.

The model focuses on the three nuclides that are potentially most important, i.e., 14-C, 129-I and 99-Tc, and it considers both internal and external exposure, largely in terms of whole-body exposure. Internal exposure is based on food-chain transfer, which includes food, water and soil ingestion. External exposure includes water immersion, air immersion, soil or sediment immersion, and immersion in vegetation. Depending on the exact target organism, several of these exposures are modelled simultaneously to include the diverse habits of Canadian Shield organisms, particularly animals. We also show how radiological doses can be calculated for specific species rather than for generic target organisms.

Values and PDFs for the various BIOTRAC parameters were selected only after a careful appraisal of the available data. Most of the values were drawn from the literature, but some were

supplied by our own research programs. Where possible, the values used were annual averages based on data from the Canadian Shield. Where the data were numerous, a quantitative statistical analysis was used to assign a distribution type and attributes to a given parameter; otherwise the PDF was set subjectively on the basis of all the available information. Truncations of the PDF and correlations between parameters were used to avoid unreasonable values or combinations of values. For each parameter, we show how appropriate values and PDFs were derived from the data.

The integrated model

The four BIOTRAC submodels, the geosphere/biosphere interface model and the model for non-human biota that make up BIOTRAC, although distinct, were designed to interface smoothly with each other to provide a cohesive description of nuclide transport through the biosphere as a whole. The output of one model serves as input to the next. The order in which the calculations are done is chosen to ensure that the information required at each point in BIOTRAC is available from previous calculations. A step-by-step walk through of a typical BIOTRAC simulation, focusing on human dose prediction, is presented to illustrate how the model works and to put the various exposure pathways into perspective.

The resources required by the critical group are calculated internally by BIOTRAC in a consistent manner, given the number of people in the household under consideration. The number of animals needed by the household is computed from the food yield of each animal and the quantity consumed by household members. Similarly, the size of the garden is calculated as the area needed to grow the terrestrial plant foods required by the group. The size of the forage field is calculated by considering the area required to grow the feed needed by the livestock. The area required to provide sufficient wood or peat to heat the household is calculated on the basis of energy needs and the energy content of the fuel. The amount of water used by the household is found by adding the water required for domestic purposes (drinking, cooking, bathing, laundry, etc.), the drinking water needed by the domestic animals, and the water used to irrigate the garden or forage field, if irrigation is practised.

Nuclide mass is conserved within each of the four submodels of BIOTRAC, but not necessarily when nuclides are transferred between compartments. The inventories of donor compartments are often not depleted when nuclides migrate to a new compartment. For example, soil inventories are usually not reduced when nuclides are suspended into the atmosphere. Although this type of assumption results in a generation of nuclide mass within the model, it allows complex processes such as runoff, recycling and atmospheric suspension and deposition to be handled very simply. Furthermore, it results in conservative predictions of nuclide masses for both donor and receptor compartments. In all cases where source inventories are not depleted, the nuclide flux out of the source compartment is small compared with other loss terms. The amount of

nuclide mass created is small, and does not greatly increase predicted environmental concentrations and doses.

We performed a sensitivity analysis of BIOTRAC to quantify its response to changes in input parameters, and to identify the nuclides and pathways that are important in determining doses to the critical group. The submodels were analyzed first, using unit inputs. These results were then used to guide the analysis of BIOTRAC as a whole, which was done with a simplified but realistic input from the geosphere. The results show that 129-I causes, by far, most of the dose to the critical group, with 14-C contributing almost all the remainder. Most of the 129-I dose occurs through ingestion of terrestrial plant foods contaminated by root uptake from soil irrigated with well water. The next most important exposure pathway is the ingestion of plants contaminated through atmospheric deposition. The parameters to which the total dose to man is most sensitive are, in order of importance, the source of domestic water (well or lake), the parameter describing 129-I evasion from the lake to the atmosphere, and the evasion rate of 129-I from soil to the atmosphere.

Environmental changes

BIOTRAC was developed to provide predictions over a period of about 10 000 a, during which time current interglacial climatic conditions are assumed to persist. Because the parameter values sampled at the beginning of each simulation are held constant throughout that simulation, the state of the biosphere is assumed to remain unchanged with time, instead of exhibiting its characteristic fluctuations. However, the effects of such fluctuations are incorporated implicitly through the use of distributed parameter values, assuming that nuclide concentrations depend primarily on environmental conditions at the time of interest, and not on conditions prior to that time. Our parameter distributions likely account for all the temporal changes that could occur at a specific site during interstadial conditions of the glacial cycle since they reflect today's very large spatial variability across the Canadian Shield.

Many geological processes will affect the Canadian Shield on time scales longer than 10 000 a. However, the majority of these processes need not be considered in detail because their potential to influence nuclide migration through the biosphere is small. We believe that only continental glaciation, including glacially induced faulting and succession in a glacial regime, could affect consequence predictions significantly. We assessed glaciation by using a modified version of BIOTRAC to calculate radiological doses to humans for conditions representative of a cold interstadial climate, and by qualitatively evaluating a number of glacially induced pathways. The results demonstrated that glaciation will not cause doses to rise appreciably above those predicted for current interglacial conditions. Furthermore, we conclude that the only aspect of succession that must be considered is the gradual filling in of surface water bodies. Even here, the important effects on

dose occur in the final phase of succession when the bottom sediments become exposed and are used for agriculture. The use of sediment as soil has been explicitly included in BIOTRAC.

Long-term changes in human culture and technology are impossible to predict, and we have made no attempt to account for their effects in the model. The use of the critical-group concept overcomes many of the difficulties in defining appropriate exposure pathways to humans far in the future. However, we have explicitly considered human intrusion. We calculate the consequences of a bedrock well drilled into the groundwater plume from the vault. We also discuss the impacts of other intrusion scenarios, including exploratory drilling and mining.

Postclosure assessment

The system-model is implemented using the computer program SYVAC (SYstems Variability Analysis Code). SYVAC is an executive program developed to quantify the effect of variability in the parameter values. For each parameter, the computer program accepts a distribution of values that the parameter may have. SYVAC has undergone significant development and refinement over the past decade, and we are currently using the third generation, SYVAC3. To simulate the behaviour of the disposal system, SYVAC3 uses the computer programs representing the vault, geosphere, and biosphere models described above. The combined computer program is designated SYVAC3-CC3.

For each simulation, SYVAC3-CC3 selects a value for each parameter by sampling from its distribution. The resulting set of parameter values is then used in the vault, geosphere, and biosphere models. The output from the vault model is the estimated time-dependent rate of movement of contaminants out of the 12 vault sectors. These rates are passed to the geosphere model. The output from the geosphere model is the estimated time dependent rate of movement of contaminants that reach the discharge areas in the biosphere. They are passed to the biosphere model. The outputs from the biosphere model are the concentrations of contaminants in water, soil, and air, and the radiation dose to an individual of the critical group and to non-human biota.

To quantify the effects of uncertainty represented by the parameter-value distributions, SYVAC3-CC3 repeats the selection and simulation steps, typically many thousands of times; it thus makes many estimates of dose versus time (F 7-6). The distribution of these estimates reflects the variability associated with modelling the disposal system. The arithmetic average of the estimates of dose to an individual of the critical group is calculated for comparison with the AECB criterion for protection of human health and the environment. For every parameter in SYVAC3-CC3, a group of AECL researchers in the relevant disciplines has evaluated and reviewed pertinent information to determine the appropriate probability distribution. The distribution for each parameter takes one of several forms, such as the normal, lognormal, or uniform distributions. Some parameters have fixed values; that is, their

values do not vary from one simulation to the next. Such parameters are called constants in the following text.

Some model parameters, called "switches," are used to select from among two or more alternative physical situations. For example, the critical group may obtain its drinking water from either a lake or a well. A "water-source" parameter in the model, which can take on one of two values whose probabilities are based on information on well usage by inhabitants of the Canadian Shield, determines which of the two sources is assumed for a given simulation. As a second example, the system model permits the garden soil of the critical group to be any one of four possible types: sand, loam, clay, or organic. This is achieved by defining a "soil-type" parameter, which can take on one of these four values with a probability based on information on soil types on the Canadian Shield.

The system model deals with a very large number of different physical situations. The two examples above cover eight alternatives, four possible soil types each matched with two possible water sources. Additional switches determine, for example, whether lake sediments are used as garden soil, whether or not the garden is irrigated (and whether the irrigation water is taken from surface water or the well), whether the critical group uses wood or peat as a source of fuel, and whether the critical group uses local organic or inorganic building materials. We may regard the alternative situations determined by the switch settings as scenarios within the system model. The probability that a scenario would occur in a simulation would be determined by the combined probabilities for the switches to take their particular settings. Thus the probabilities of the scenarios occurring are automatically taken into account in the probabilistic analysis using SYVAC3-CC3.

SYVAC3-CC3 was used to produce many thousands of estimates of the radiological dose rate to an individual of the critical group as a function of time after closure of the disposal facility. The arithmetic averages of these estimates (the estimated mean dose rate), for times ranging from 0 to 100 000 years, was then used to calculate the radiological risk for times to 100 000 years. To indicate the significance of the estimated effects, the resultant risk curve was compared with the risk criterion specified by the AECB for times up to 10 000 years.

SYVAC3-CC3 was also used to determine the potential effects on the natural environment. Mean concentrations of contaminants in water, soil, and air were calculated, as were the mean dose rates to four representative hypothetical organisms. To indicate the significance of the estimated effects, concentrations in water and soil were compared with available criteria, guidelines, and standards and with adopted guidelines, and doses were compared with background levels and with doses known to cause harm.

In addition to the probabilistic analysis, use was made of individual simulations, each with a single set of preselected parameter values. Such individual simulations were used to study details of the system model, such as the details of contaminant

movement through the system. They were also used to study details of how changes in a single parameter affect the results.

Table 56 Percentage of contaminants present in different compartments at 1E+4 a.

Amount remaining in	I-129	C-14	Tc-99	U-238
Containers	96.06	28.0	91.0	99.99
Backfill + Buffer	3.85	1.9	5.79	2E-7
Geosphere	0.07	0.02	0	0
Released to biosphere	0.02	0	0	0

Estimated radiological effects on human health

SYVAC3-CC3 was used to produce many thousands of estimates of the radiological dose rate to an individual of the critical group. The dose rate is given as the sum, over one year, of the effective dose equivalent resulting from external exposure and the 50-year committed effective dose equivalent from that year's intake of radionuclides.

The dose rate to an individual of the critical group was estimated for times ranging from the time of closure of the disposal facility to 100 000 years thereafter (F 7-7). At 10 000 years, the estimated mean dose rate is only 1E-8 mSv/a, which is 6 orders of magnitude below the AECB criterion. At times preceding 10 000 years, it is smaller still. For all times up to 100 000 years, the estimated mean dose rate is less than 3% of the AECB risk criterion and less than 0.05% of the dose rate from natural background radiation (about 3 mSv/a). The distribution (F 7-8) of 40 000 estimates of dose rate at 10 000 years is highly skewed: 99.8% of the estimated dose rates are less than 9E-7 mSv/a, and all are less than 3.7E-5 mSv/a.

Throughout the 100 000-year simulation time, 129-I gives the largest mean dose rate, 1.4E-3 mSv/a, followed by 14-C with 1.4E-5 mSv/a. The mean dose rate attributable to any other radionuclide is less than 1E-7 mSv/a. The contributions (F 7-9) to the estimated mean dose rate and risk from 129-I and 14-C are functions of time. The mean dose rate from 14-C, which has a half-life of 5730 years, reaches a maximum near 40 000 years, because radioactive decay is effective in reducing the amount of 14-C that could arrive at the biosphere at longer times. The mean dose rate from 129-I, which has a half-life of 1.57E+7 years, would reach its maximum value after 100 000 years.

Contaminant concentrations in the biosphere

For each of the radionuclides and chemically toxic elements released from the disposal vault, we used the SYVAC3-CC3 results to identify the maximum concentrations in water, soil, and air over

100 000 years for each simulation. We then calculated the arithmetic mean of these maximum concentrations. These estimated contaminant concentrations apply to the small region of the biosphere occupied by the critical group and other biota. To evaluate the potential for the chemically toxic elements to cause environmental effects, we compared their concentrations in water and soil with the available criteria, guidelines, and standards and found that none were exceeded. However, quantitative criteria, guidelines, or standards for protection of the natural environment have not been established for all the radionuclides and chemically toxic elements in the used fuel. Therefore, in some cases we established our own guidelines for indicating the significance of estimated effects on the natural environment.

Environmental increment, is defined as the additional amount of nuclide that can be added to the background level without exceeding the natural, local, spatial variation in concentration. This value is sufficiently stringent so that if the additional contribution from a vault is less than this, the presence of an underground vault would not likely cause detectable environmental effects. We have determined baseline concentrations and environmental increments for the radionuclides and chemically toxic elements in the used fuel. When available, data from the Canadian Shield were used. For most naturally occurring nuclides, the environmental increment is based on one standard deviation of the mean of the existing concentration. For example, concentrations of bromine in soils on the Canadian Shield typically vary between 5 and 40 mg/kg, with an average background concentration of about 10 mg/kg and a standard deviation of 2 mg/kg. On the basis of these data, we take its environmental increment to be 2 mg/kg.

We believe that environmental increments can be used in a rigorous and demanding test to identify contaminants that could have significant effects. If an increase in concentration of a nuclide is less than the environmental increment, the stresses imposed should be within the natural variability of stresses, provided natural concentrations are not at a toxicological threshold. If an increase is much less than the environmental increment, then any effect is unlikely to be significant compared with the existing effects. Even if an increase exceeds the environmental increment, there may still be no significant effect on the ecosystem, or on individuals of species within the ecosystem. However, if the estimated increase in environmental concentration of a nuclide exceeds its environmental increment, the radiological doses to representative hypothetical organisms are estimated, as described below. The estimated mean concentrations of contaminants exceed the environmental increments only for 129-I and 14-C. For all other contaminants, the estimated mean concentrations are much less than the environmental increments.

Table 57 Mean concentrations (MC) of contaminants in soil and water and their environmental increments (EI).

		129-I		14-C	
Medium		MC	EI	MC	EI
Soil	Bq/kg	2	1E-5	9E-3	9E-3
Water	Bq/L	3E-3	4E-8	5E-4	2E-5

Environmental increments for 14-C depend on the stable carbon content in the environment, and the values reported here use typical values for stable carbon in soil and water. Chemical toxicity effects from 129-I are expected to be insignificant, because the estimated concentration of 129-I from the disposal vault is negligible when compared with existing concentrations of iodine in the environment. Iodine (principally as 127-I) is a common element in the biosphere; with a median concentration of 4E-5 mol/kg in dry soil, which is about 5 orders of magnitude greater than the maximum estimated concentration of 129-I from the vault. Chemical toxicity effects from 14-C are also expected to be insignificant. Carbon is generally not regarded as a chemically toxic element. Moreover, carbon (principally as 12-C) is much more abundant in the environment than 14-C from the disposal-vault would be.

We conclude that there would be no significant chemical toxicity effects on the natural environment from the small increases in concentration of 129-I, 14-C, or any other contaminant potentially released from the disposal-vault over 100 000 years. Because the estimated increases in the environmental concentrations of 129-I and 14-C exceed their environmental increments, we estimated the radiological doses to representative hypothetical organisms. We conclude that no other radionuclides would cause significant radiological effects on the natural environment over 100 000 years, because the estimated increases in their concentrations are much less than their environmental increments.

Radiation doses to non-human biota

We used SYVAC3-CC3 to estimate the dose rates to four representative hypothetical organisms: a plant, a mammal, a bird, and a fish. The characteristics, habitat, and lifestyles of these biota are expected to represent a wide range of organisms. From 1000 randomly sampled simulations, we calculated the mean dose rates to these organisms arising from 129-I and 14-C.

Table 58 Arithmetic mean of the maximum doses to four hypothetical organisms estimated in 1000 simulations for a 100 000-year simulation time (mGy/a).

Nuclide	Plant	Fish	Mammal	Bird
129-I	4E-3	3E-3	1E-2	5E-2
14-C	2E-4	2E-2	5E-4	5E-4
Total	4E-3	2E-2	1E-2	5E-2

All estimated mean dose rates are less than 0.1 mGy/a. For comparison, the lower end of the range of dose rate resulting from natural background radiation is 1 mGy/a. Thus the estimated dose rates for times up to 100 000 years are at least 1 order of magnitude below the dose rates to biota from background radiation in the environment, and even further below doses known to cause harm. Moreover, our estimates of doses to these four hypothetical organisms are based on many pessimistic assumptions. Therefore, we have concluded that there would be no significant radiological effects to nonhuman biota.

Sensitivity analyses for I-129

Sensitivity analysis involves changing the values of parameters in the system-model and determining the effects of these changes on the model estimates. Such analysis increases our understanding of the assessment results and provides insight into the factors that are most import to the safety of the disposal system. We summarize here the results of sensitivity analysis on the system model. Sensitivity analyses were also performed on the individual vault, geosphere, and biosphere models. The results from the system model show that 129-I is the major contributor to risk for times up to 100 000 years. The following discussion therefore deals with this radionuclide.

Median-value calculations

To examine the sensitivity of the system-model results to the values of its parameters, a two stage process was adopted. First, we screened the many parameters in the model, selecting for further study those that had the largest influence on estimated dose from 129-I. Next, we studied the effect of varying each of these influential parameters independently over its entire range of feasible values, while all other parameter values were fixed. Switch settings were based on judgment, taking into account the probability of the switch setting as well as the effect of the switch setting on the results. Other variable parameters were fixed at their median values. This median-value sensitivity analysis does not consider simultaneously the full range of uncertainty of all parameter values. However, it does provide in-depth information on many subtle effects and interactions that are frequently less apparent in the sensitivity analysis of the randomly sampled simulations.

The tortuosity of the lower rock zone had the largest effect on the maximum dose rate over 100 000 years. Tortuosity in rock is a measure of the winding nature of the interconnected water-saturated pathway within the rock. Larger values of tortuosity correspond to longer distances a contaminant must diffuse over when moving between two points in the rock. Changing its value from the minimum to the maximum of its range (2 to 8) caused the estimated maximum dose rate to decrease by a factor of 3600. All other parameters had less significant effects.

The switch that most influenced the estimated dose rate to an individual in the critical group was the well-usage parameter, which specifies whether the domestic water used by the critical group is well or lake water. Estimated doses were about 100 times larger when well water was used than when lake water was used. Other switches, for example the switch specifying whether or not the garden is irrigated, were less influential; estimated doses are only about three times larger when the garden is irrigated than when it is not.

Probabilistic calculations

As for the median-value analysis, parameters were screened to select for further study those that have the largest influence on the estimated mean dose rate. The screening method used, iterated fractional factorial design permits examination of the effects of each parameter over its range of feasible values. Once the most influential parameters were identified, the sensitivity of the model results to changes in the values of these parameters was studied. The tortuosity of the lower rock zone was once again identified as the most important parameter. Results show that larger values of tortuosity generally yield smaller estimates of dose, the same trend as in the median-value simulation. Other parameters also had a significant influence on the estimated mean dose rate, although their effects were much less significant than the effects of tortuosity. For example, the well-usage switch was again an influential parameter, but its relative effects were a fraction of the effects of tortuosity.

Supporting studies

Studies were carried out to support the conclusions of the sensitivity analysis and to confirm that all of the influential parameters (including the switch parameters) had been identified in the sensitivity analysis. One of the supporting studies examined high-dose and low-dose simulations. The high-dose and low-dose simulations are those yielding the largest and smallest estimates of maximum dose rate for times up to 100 000 years. From the first 9000 randomly sampled simulations, we selected for study the 20 simulations with the largest and the 20 simulations with the smallest estimates of maximum dose rate. We then examined the values used in these simulations for influential parameters. There is a strong correlation between the values of the tortuosity of the lower rock zone and the estimated dose rate to a member of the critical group. The values of tortuosity in the high-dose simulations were all small, generally less than its 0.10 quantile value, whereas its values in the low-dose simulations were all large, greater than its 0.90 quantile value. These trends conform with the sensitivity analyses on the effects of the tortuosity of the lower rock zone. Similar strong correlations were also observed for the other influential parameters, such as the well usage switch; for example, the well was selected as the source of

domestic water used by the critical group in 19 of the 20 high-dose simulations.

Use of sensitivity analysis studies

The information provided by these analyses could be very valuable to the siting and design of a waste disposal facility. There are some parameters, such as disposal depth and the distance of waste from fracture zones, over which the implementing organization would have some degree of control, and some parameters, such as those defining the diet of the critical group, over which the implementing organization would have little or no control. The information from these analyses about which controllable parameters are important could be used to modify the location and design of the vault to improve safety margins. The information could also help determine whether further studies on a parameter would help reduce the uncertainty of the assessment results.

Analysis of potential barrier effectiveness

The vault, geosphere, and biosphere models were used to examine the potential effectiveness of the engineered and natural barriers that contribute to the safety of the disposal-system. We refer to "potential" effectiveness, because we analyzed the effectiveness independently of the behaviour of other barriers in the system. The actual effectiveness of a particular barrier would depend on the performance of all components of the system. In the analysis, we assumed the values of all relevant inputs were set at the values they had in the median-value sensitivity analysis. We examined the potential effectiveness of the used fuel, the titanium container, the buffer/backfill, and the rock in the waste exclusion distance (about 50 m of low-permeability rock between a fracture zone and the nearest vault room containing waste). For each of these barriers, we calculated the %age of each nuclide released over 100 000 years. The smaller the %age released, the more effective the barrier.

Table 59 Percentage of a nuclide released by a barrier over 100 000 years.

Nuclide	T - a	Fuel	Container	Vault	Rock
3-H	12.4	30	<<0.001	<<0.001	<<0.001
90-Sr	29.1	0.05	<<0.001	1	<<0.001
39-Ar	269	8	0.08	<<0.001	<<0.001
14-C	5730	6	60	0.8	0.007
239-Pu	2.41E+4	<<0.001	100	<<0.001	<<0.001
99-Tc	2.13E+5	6	100	<<0.001	0.1
129-I	1.57E+7	6	100	10	5
Br	stable	6	100	10	5
Sb	stable	<<0.001	100	0.003	5

The cumulative fraction released over 100 000 years (F 7-10) is given by multiplying the fractions released by each of the barriers over 100 000 years, and is thus an overestimate. Table above illustrate (F 7-10) that the potential effectiveness of a barrier depends on the nuclide being considered. For example, the used fuel is a very effective barrier for 139-Pu and Sb, because they are located in the UO_2 matrix and are only released slowly during dissolution of the UO_2 matrix. The used fuel releases very much less than 0.001 % of the mass of these nuclides over 100 000 years. On the other hand, the used fuel is a less effective barrier for 3-H, 90-Sr, 39-Ar, 14-C, 99-Tc, 129-I, and Br, because a significant proportion of these nuclides is located in cracks in the fuel pellets, in gaps between the pellets and the fuel sheath, and at the grain boundaries and is assumed to be released instantly upon contact with groundwater.

More than one barrier may be effective in limiting the release of a particular nuclide. For example, there are three very effective barriers for 239-Pu: the used fuel, the buffer/backfill, and the low permeability rock within the exclusion distance. Each of these barriers releases very much less than 0.001 % of the mass of this nuclide over 100 000 years. Similarly, there are three very effective barriers for 3-H and two for 90-Sr and 39-Ar. For 14-C, 99-Tc, 129-I, Br, and Sb, there are three barriers that are either effective (release no more than 10% of the mass of these nuclides over 100 000 years) or very effective: the used fuel, the buffer/backfill, and the low-permeability rock. The container is an effective barrier only for radionuclides with half-lives that are short in comparison with the container lifetime. The examples given in the table are 3-H, 90-Sr, and 39-Ar.

Human intrusion scenarios

Inadvertent human intrusion could be initiated by drilling a water supply well; this event was included in the system-model. It could also be initiated by a drilling operation that penetrated the waste in a sealed disposal vault and brought it to the surface; this event was included in four significant scenarios that were not incorporated in the system-model, but whose risk was estimated as described below. Deliberate human intrusion was not analyzed. We believe that it should be considered the responsibility of any society that takes this action. The four scenarios analyzed are as follows: a member of a drilling crew is exposed to undispersed waste extracted in a drilling operation (drilling scenario); a laboratory technician examining drill core is exposed to undispersed waste extracted with the core (core examination scenario); a construction worker is exposed to waste that was previously brought up in a drilling operation and dispersed at the construction site (construction scenario) and a resident is exposed to waste that was previously brought up in a drilling operation and dispersed at the site where the resident now lives (resident scenario).

An event tree methodology was used as a framework for defining probabilities of occurrence for each of these inadvertent human intrusion scenarios. Each scenario implies a series of events, such as selecting a drilling site, missing controls or warnings about the vault, and drilling to disposal depth. The events and their associated probabilities were based on the judgements of experts in relevant technologies and social sciences. For example, the drilling and core examination scenarios include events such as the following: a proposal is made to drill a borehole on the disposal site to the disposal depth; institutional controls, such as surveillance for safeguards continuing after closure of the facility, do not stop the drilling (institutional controls are assumed to gradually become ineffective over a period of 500 years); passive measures, such as long-lasting site markers, records, and archives, and societal memory do not stop the drilling (passive measures and societal memory are assumed to gradually become ineffective over a period of 2000 years); detection of the disposal vault during predrilling investigations does not stop the drilling; and the borehole intersects a container.

The probability of occurrence of an inadvertent human intrusion scenario is given by the product of the probabilities of its constituent events. The construction and resident scenarios take into consideration the cumulative probability that waste may have been extracted by drilling in prior years. Estimated doses are largest at earlier times, when the waste is most radioactive. The important pathways and radionuclides vary with time and the scenario considered.

Using the doses and probabilities, we estimate the risk as follows: For dose rates up to 1 Sv/a, the risk associated with each of these scenarios is estimated as specified by the AECB: the product of the probability of the scenario, the magnitude of the resultant radiological dose, and the probability of a health effect per unit dose (0.02/Sv). For higher dose rates, it is assumed that a deterministic health effect would occur; that is, the risk is numerically equal to the probability of the scenario. Immediately after facility closure, when it is assumed that institutional controls, passive measures, and societal memory would almost certainly prevent intrusion, the estimated risk is very small. Thereafter, the estimated risk increases to a maximum and then decreases. The maximum estimated risk and the time of its occurrence depend on the scenario, as shown in Table bellow. The risks calculated for these scenarios are 3 to 6 orders of magnitude below the risk criterion (1E-6 per year) specified by the AECB.

Table 60 Maximum Estimated Risk (MER) and Time of Occurrence (TO) from four human intrusion scenarios.

Scenario	MER/y	TO - y
Drilling	3E-10	40
Core Examination	9E-11	500
Construction	4E-13	3000
Resident	3E-10	150

Very long-term effects

In R-104, the AECB states that, where the predicted risk to an individual does not peak before 10 000 years: There must be reasoned argument leading to the conclusion that beyond 10 000 years sudden and dramatic increases in the rate of release to the environment will not occur, acute doses will not be encountered by individuals and that major impacts will not be imposed on the biosphere. We interpret acute dose to mean the dose at which serious deterministic health effects may occur, which according to the ICRP (1993) is 1 Sv. We use 1 Sv/a for comparison. For the postclosure assessment case study, the estimated dose rate does not peak before 10 000 years and is still increasing at 100 000 years. Therefore, reasoned argument is required regarding potential effects in the very long term.

Table 61 Amounts of contaminants (in mol) present in different compartments at 1E+5 a.

Amount in	Br	C-14	I-129	Kr-81	Pu-239	U-238
Inventory*	11000	3000	56000	0.011	19E+5	67E+7
Containers	9900	0.015	52000	0.0072	1E+5	67E+7
Buffer	0	0	0	7E-5	5E-4	4.2
Backfill	590	16E-4	3100	5E-4	2E-4	8E-3
Vault	11000	16E-3	55000	8E-3	1E+5	67E+7
Released**	2E-2	81E-8	0.28	1E-8	0	0

* initial, ** to biosphere.

Projected trends from the system-model analysis

The estimates of the system-model covering the first 100 000 years after closure exhibit trends that are reasonable from what we know about the very long-term processes likely to affect the vault and geosphere, provided the vault, geosphere, and biosphere do not experience major changes that disrupt the barriers to contaminant movement or render the natural environment unusually sensitive to the effects of the contaminants.

The only radionuclides that contribute significantly to the estimated dose within the first 100 000 years are 129-I and 14-C. The estimated mean dose rate resulting from 14-C peaks prior to 100 000 years, and is always many orders of magnitude below the acute dose rate of 1 Sv/a. The mean dose rate from 129-I, which has a half-life of 1.57E+7 years, would reach its maximum value after 100 000 years. Iodine-129 cannot, however, lead to an acute internal dose to an individual. Because of the way that iodine behaves in the human body, 129-I can give an internal dose that is at most about 39 mSv/a. This figure would, in general, be reduced by

isotopic dilution with stable iodine occurring naturally in the environment (typically by a factor of about 100 on the Canadian Shield).

Radionuclides such as 135-Cs, 79-Se and 99-Tc, although they do not contribute significantly to the estimated dose over 100 000 years, have the potential to continue reaching the biosphere at times greater than 100 000 years. However, these radionuclides interact strongly with one or more of the media between the containers and biosphere, which retards their movement and disperses them. This is why they do not have an earlier effect. This media-waste interaction would be expected to continue retarding and dispersing the radionuclides at times beyond 100 000 years.

From these considerations of an undisturbed disposal system beyond 10 000 years after closure, we conclude that any radionuclide releases would be gradual (rather than sudden and dramatic), that radiation doses would be of the level now present in nature (far below acute doses), and that major effects would not be imposed on the biosphere.

Analogy to a uranium ore deposit

We compared the potential health hazards of used fuel and uranium ore. The comparison indicates that 10 000 years after used fuel is removed from a reactor, it represents a health hazard (if ingested) similar to that from a rich uranium ore deposit containing the same amount of uranium. The hazards will continue to be comparable thereafter. At very long times, the principal constituents of the used fuel will be 238-U and other long-lived heavy radionuclides, which, if the chemical environment of the vault remained similar to that assumed in the system model, would be released to the groundwater only very slowly as the UO_2 matrix dissolved. An analogy for the very low solubility of the UO_2 matrix is found in the low solubility of uraninite ore in the 1.3-billion-year-old deposit at Cigar Lake. Uraninite ore is essentially UO_2 and is similar to the composition of used fuel. There is no direct evidence at the surface above the Cigar Lake deposit that a high-grade ore deposit exists at a depth of about 450 m. Studies of this deposit have focused on the reasons for these and related observations. The results, from the Cigar Lake and other studies, show that uraninite undergoes extremely slow dissolution under reducing conditions and retains most of its radionuclides with no significant movement for millions of years. The chemical conditions in the deposit are reducing, as expected in a disposal vault based on the proposed disposal concept.

The contribution to environmental radiation by 238-U in old uranium ore deposits arises mostly through the effects of its radioactive decay products, such as radioactive isotopes of radium and radon, which can contribute significantly to natural background radiation, particularly indoors. Similar effects would be expected from the relatively large amount of 238-U in the undisturbed disposal-vault at very long times after closure. The presence of

the disposal vault might cause a fractional increase in local groundwater radioactivity, through the slow dissolution of the used-fuel matrix, and thereby the release of heavy radionuclides and their decay products. The analogy with a rich uranium ore deposit suggests that any effect of a disposal vault would be gradual, rather than sudden and dramatic, and would fall within the range of current background radiation, well below levels that could produce acute doses.

Biosphere processes

Significant climatic changes are expected within the next million years, including a number of periods of glaciation. Depression and uplift of the ground surface are expected with each glaciation, as well as erosion, redistribution, and deposition of sediments. However, the potential to amplify the relief by glacial erosion is limited because most of the easily erodible rock has already been removed. We expect that, following future glaciations, the landscape would be similar to the present landscape, although the detailed distribution of rock, sediments, and water bodies would change. This could change the number and locations of groundwater discharges from a disposal vault. There might also be substantial changes in biotic communities.

During glaciation, there could be a buildup of contaminants from a disposal vault in the more permeable rock and overburden because they would not be flushed to the surface while the glacier covered the discharge locations. The extent of this buildup might depend on permafrost conditions under the ice sheet. Trapped contaminants could be released when the glacier retreated and groundwater flow to the surface was re-established. The potential effect of this pulse of water containing a higher concentration of contaminants would likely be offset by the large volumes of glacial meltwater and proglacial lake water available at the surface for dilution during the first few hundred years after glacial retreat. In addition, it would be unlikely that humans could permanently resettle in the vicinity of the locations of contaminant discharge from the disposal vault until the proglacial lakes drained.

We concluded that the changes with the most potential for altering the biosphere in a way that would affect exposures of humans to contaminants from the vault were likely to be caused by human activities and glaciation. We have evaluated changes in climatic conditions and the effects on the Canadian Shield biosphere caused by the glacial cycle. Evaluation of movement of contaminants through the biosphere for discrete glacial states did not indicate significant increases in exposures over those obtained for the interglacial conditions included in the system model.

The local and global effects of human activities, including ozone depletion and global warming, are uncertain; however, we expect that these effects are reflected in the wide ranges of biosphere parameter values used in the system model. Provided the geosphere does not experience major changes that disrupt the barriers to contaminant movement, we do not expect future changes

in the biosphere to lead to sudden and dramatic increases in releases of contaminants, to acute doses, or to major effects on the natural environment as a result of a disposal vault.